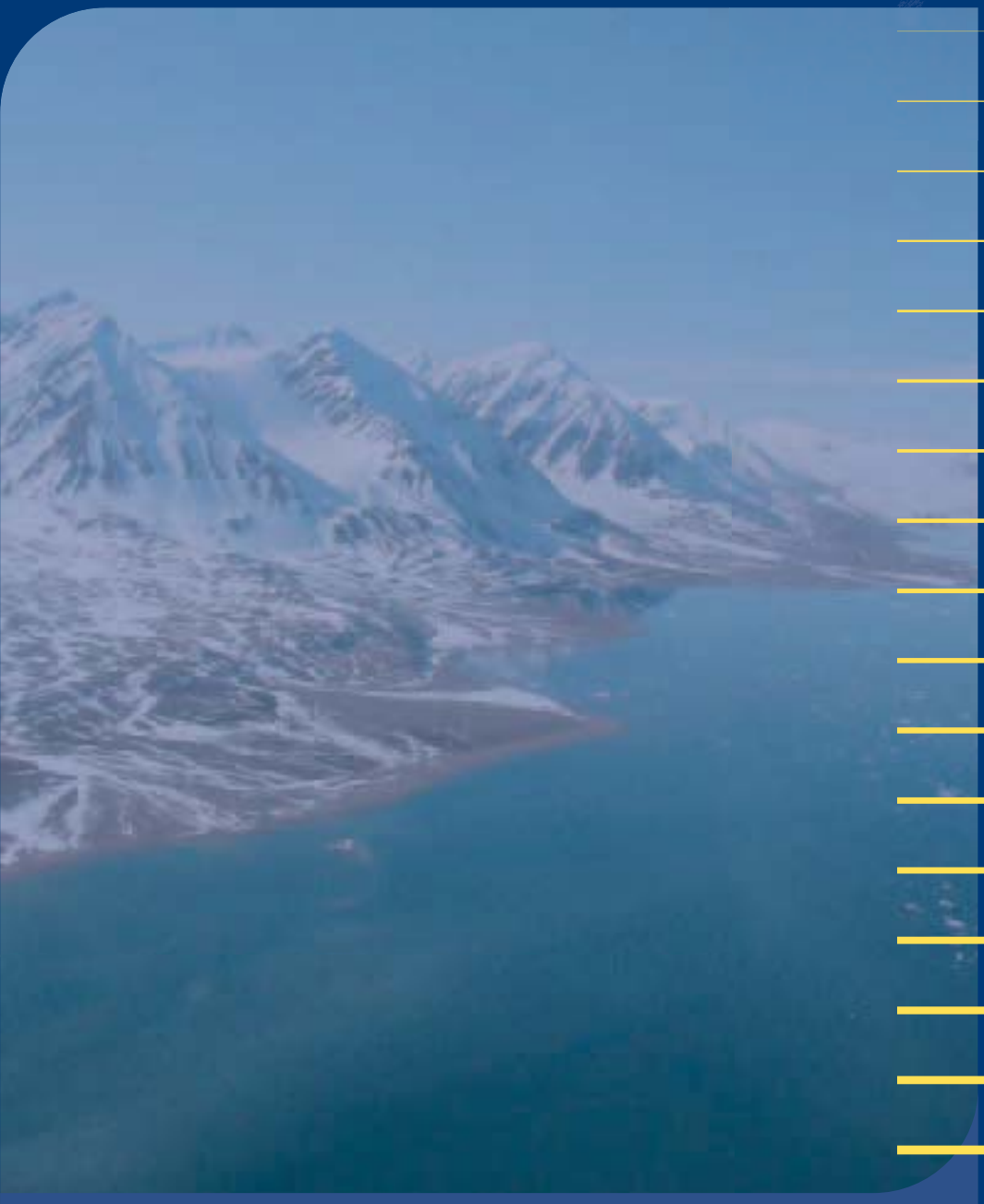
Strålevern Rapport 2004:2

The Radiological Environment of Svalbard



Statens strålevern

Norwegian Radiation Protection Authority
Postboks 55
N-1332 Østerås
Norway

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Key words:
Arctic, radioactivity, Svalbard

Abstract:
This report details the monitoring of radioactivity by the Norwegian Radiation Protection Authority on the Svalbard archipelago. Results indicate contamination of both the terrestrial and marine environment with a variety of isotopes from a variety of sources and highlight the continued contamination of Svalbard with radioactive materials from European nuclear facilities. The role of Arctic specific processes in the occurrence and behaviour of contaminant radionuclides is also investigated.

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Emneord:
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Resymé:
Denne rapporten redegjør for Statens stråleverns overvåkning av radioaktivitet på Svalbardøygruppen. Resultater gir uttrykk for at både det terrestriske og det marine miljøet har blitt forurenset med en rekke isotoper fra forskjellige kilder og framhever den fortsatte forurensningen av Svalbard med radioaktive materialer fra europeiske kjernefysiske anlegg. Rollen til prosesser som er spesielle for Arktis når det gjelder forekomst og oppførsel til forurensende radionuklider blir også undersøkt.

Head of project: Tone Bergan.

Approved:

Per Strand, Director, Department for Emergency Preparedness and Environmental Radioactivity.

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Norwegian Radiation Protection Authority, P.O. Box 55, N-1332 Østerås, Norway.
Telephone +47 67 16 25 00, fax + 47 67 14 74 07.
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1. Introduction

The common perception of the Arctic as a pristine wilderness and the vulnerability of its ecosystems have contributed to an increase in the amount of interest and attention focused on these regions. Much of this attention has been in relation to pollutant levels in the Arctic and its ecosystems, with radioactive contaminants being one of the most often discussed suite of pollutants. A number of factors have influenced the current focus on radioactivity within the Arctic regions, as evidenced by the radiological components of both the International Arctic Environmental Protection Strategy (IAEPS) and the Arctic Monitoring and Assessment Programme (AMAP, 1993). These include the relatively large number of sources of both actual and potential nuclear contamination that exist in the Arctic, the particular vulnerability of Arctic ecosystems to nuclear contamination (Wright *et al.*, 1997) and the relative lack of knowledge with respect to both the occurrence and behaviour of radioactive contaminants in High Arctic ecosystems and the effects of even low levels of radioactivity on High Arctic biota.

Although historically, the emphasis of radiological protection has been directed towards health impacts on humans, the need for developing a system for assessment of radiological impacts to both flora and fauna has come to the fore in recent times. This shift in emphasis has largely been due to the realisation that the previous philosophy of environmental radioprotection, encapsulated in the phrase "*if man is protected, then the environment is protected*", is not sufficient to ensure the protection of biota from the effects of radiation (Pentreath, 1999). Related to this change in focus is the concept that the assessment of the impacts of anthropogenic radioactive contaminants on the environment in general, and the Arctic in particular, can only be conducted with reference to the intrinsic natural dose commitment upon which any further anthropogenic doses are subsequently superimposed. Fundamental to both of these concepts is a need for an improvement in the amount and the quality of information relating to the current level of radionuclides in High Arctic environmental components and a focussing of attention towards site-specific processes influencing the behaviour and occurrence of radionuclides in constituent matrices therein. Of added importance in the consideration of the impact of radioactivity on the Arctic environment is the acute sensitivity of public perception to levels of radioactive contamination. Given the productivity of the Arctic marine environment and the importance of that productivity to a number of industries, it is imperative to ensure public confidence with respect to levels of radioactive contamination in the Arctic.

Current models pertaining to global climate change indicate that global warming will be most acute in Polar Regions. This change in climate will most likely result in changes in the extent of sea ice, increased thawing of permafrost and melting of polar ice masses. The impact of predicted changes in climatic conditions on the transport of radionuclides to, from and within the Arctic and their behaviour within pertinent ecosystem components is relatively unclear at present (McDonald *et al.*, 2003). However, a significant impact is likely to occur with reference to the radiological dose received by Arctic residents and biota from the naturally occurring radionuclides in the ^{226}Ra decay chain that results in the production of ^{222}Rn, ^{210}Pb and ^{210}Po. Changes in the snow and ice cover of the terrestrial Arctic environment and concomitant changes in permafrost may result in an increase in the ambient dose due to these nuclides. Predicted increase in the dose from increased exhalation of ^{222}Rn due to melting of permafrost is of the order of a factor of 2 – 3 (McDonald *et al.*, 2003).

Figure 1. Although often considered to be a pristine wilderness, the Arctic exhibits a variety of pollutants including a number of radioactive isotopes. Arctic ecosystems are especially vulnerable to radioactive contamination due to both the large number of actual and potential sources of such contaminants and the efficiency with which radioactive contamination can be accumulated and transferred within Arctic food chains.

The uncertainty surrounding the impact of climate change on the radioecological situation in the Arctic warrants more extensive monitoring and research to further elucidate how climate change and its effects will manifest themselves with respect to the occurrence and behaviour of radionuclides in the Arctic environment.

With a view towards bridging the information gaps pertaining to radioactive contamination within the Norwegian Arctic, the Norwegian Radiation Protection Authority (NRPA) has undertaken research and monitoring programmes to establish levels of radioactive contamination and examine the effects of natural environmental cycling and the impacts of current and legacy anthropogenic processes on the radiological landscape of the Svalbard archipelago. These programmes serve to elucidate current contamination levels and the impacts thereof, to increase knowledge pertaining to radioecologically vulnerable facets of the Svalbard environment and to establish improved systems for the monitoring and protection of the Norwegian Arctic in general with respect to radioactive contamination.

As part of the national terrestrial and marine surveillance programme undertaken by the NRPA on behalf of the Ministry of the Environment, this report presents data concerning levels of anthropogenic (technetium-99, cesium-137, plutonium-238, plutonium-239+240 and americium-241) and natural (uranium-238, radium-266, thorium-232 and potassium-40) radionuclides from both the marine and the terrestrial environment in and around the Svalbard archipelago, which shall be followed by a separate report dealing with levels of radionuclides in Arctic faunal species.

Svalbard, in comparison with the Norwegian mainland, has received little attention with regard to the levels and behaviour of radionuclides in the marine and terrestrial environments. A search of the available literature reveals only a limited number of previous studies that report levels of radionuclides in Svalbard marine and terrestrial matrices. This is especially true for the terrestrial environment, for which little information exists on current contamination levels and on the behaviour of radionuclides within this High Arctic environment. Of the previous studies that have been reported, the majority were conducted in the 1980's and were severely limited both in their spatial representativeness, the number and type of samples examined and the nuclides reported. For the marine environment, there is a greater wealth of available data, both on a spatial and temporal scale. However, there is an overwhelming bias in the data for the abiotic compartments (i.e. seawater and sediments), with limited studies of radionuclide contamination in marine biota within the Svalbard region. That Svalbard has previously been overlooked with regard to a more intensive radiological investigation is an oversight, particularly in consideration of its geographical location, lying in close proximity to sites of previous atmospheric weapon testing and in the path of continued and potential oceanic long-range exposure to sources of radionuclides from the European mainland. Given the paucity of

available and, more importantly, contemporary data, Svalbard has warranted a more thorough investigation into radionuclide levels and transport pathways to assess with greater accuracy the ramifications of past, current and future contamination.

1.1 The Svalbard Archipelago

The Svalbard archipelago encompasses 63000 km² of islands between 74 and 81°N and between 10 and 35°E and seas inside a territorial boundary out to 4 nautical miles. The archipelago contains the major islands Spitsbergen, Nordaustlandet, Barentsøya, Edgeøya, Kong Karls Land, Hopen, Prins Karls Forland and Bjørnøya as well as numerous smaller islets. The landscape of Svalbard is dominated by ice, with two thirds of the landmass being permanently covered by ice and glaciers and with less than 30 % of the ice-free areas being covered by vegetation.

Figure 2. Geographical location of the Svalbard Archipelago.

Svalbard is underlain by permafrost that penetrates between 200 and 300 m below the soil surface (Hanssen-Bauer *et al.*, 1990), depending on the thermal forcing at the surface. During summer months, the soil surface thaws, permitting plant and animal life in the upper 1 to 2 m of the soil (Putkonen, 1998). Svalbard has a rich geology dominated by four major geologic units, local Tertiary basin sediments, Carboniferous through cretaceous platform cover sequences, Devonian basin sediments and metamorphic basement rocks. The youngest rocks from the Tertiary are found in central and southern parts of Spitsbergen, surrounded by formations from the Triassic, Jurassic and Cretaceous. Older rocks from the Precambrian are found in northern parts of Nordaustland, North Eastern Spitsbergen and along the western coast of Spitsbergen. The islands within the archipelago display distinct ice and water erosional features, forming terraced plateaus intersected by deep fjords, of which Isfjorden is the largest.

Spitsbergen, the largest island (39044 km²), contains the highest mountain of the group (Newtontoppen, 1720 m), the administrative centre, Longyearbyen and the main settlements

Figure 3. The Svalbard Archipelago.

of Ny-Ålesund, Barentsburg, and Pyramiden. In all there are a total of 9 settlements in Svalbard, including the permanently manned meteorological stations on Hopen and Bjørnøya. The majority (~55 %) of Svalbard's small population of 2868 (July 2002 est.) are Norwegian and 44 % are Russian.

Svalbard is surrounded by a shallow shelf sea with an average depth in the Barents Sea of 230 m while west and north of Svalbard, the Norwegian Sea and the Arctic Ocean descend down to more than 2000 m. The fjords and sea areas to the north and east of Svalbard are covered with ice for 8 – 9 months of the year, while the fjords on the west side of Spitsbergen can be ice-free for long periods of the winter.

Though defined as a High Arctic environment, the climate on Svalbard is milder than in other areas at similar latitudes due to frequent low-pressure passages and northerly flowing Gulf Stream water. Mean temperatures vary from -14 ^{o}C in the winter to +6 ^{o}C in the summer, with maximum and minimum temperatures recorded of -47 ^{o}C and 21 ^{o}C in Longyearbyen. Svalbard can be characterized as an arctic semi-desert, with an annual precipitation of around 20 cm (Hanssen-Bauer *et al.*, 1990), although greater climatic differences do occur across the archipelago. In Longyearbyen, the midnight sun lasts from April 20 to August 23 whilst the polar night exists between October 26 and February 15.

Figure 4. The terrestrial environment of Svalbard exhibits characteristics typical of its High Arctic location.

The terrestrial environment of Svalbard is, in the main, typical of a High Arctic environment. Mountains display features characteristic of the effects of ice erosion, slopes are scree covered and glacial deposits often cover lowland areas. The soils of Svalbard can vary greatly with respect to both type and depth and can range from gravelly post-glacial marine terraces with little or no organic matter (Orvin, 1934) to highly organic soils in low lying saturated areas.

Terrestrial primary production in Svalbard is constrained by the short growing season, the reduced availability of nutrients and the presence of permafrost with around 165 species of arctic flora to be found near the coast and on patches of interior tundra. Increased abundance of vegetation can be found near seabird colonies, as a result of nutrient enrichment through guano deposition.

Major terrestrial fauna include large waterfowl and seabird populations (the largest bird colony in the North Atlantic is on Svalbard), commonly eider, fulmars, auks and kittiwakes. Other terrestrial fauna includes the Svalbard reindeer, Arctic foxes and land locked populations of Arctic char. Marine fauna of note are the marine mammals, such as polar bears, walrus and various seal and whale species.

Svalbard's social and economic history centred around whaling and fur trading throughout the 17 and 18th century, before the discovery of coal at the end of the 19th century. Norway, Russia, and Sweden disputed the sovereignty of Svalbard, until a treaty signed in 1920, ensuring recognition of claims of other countries to parts of the coalfields, awarded Svalbard to Norway, which took formal possession of the archipelago in 1925. Today, the chief wealth of the islands is derived from their mineral resources, most notably coal, though deposits of asbestos, copper, gypsum, iron, marble, zinc and phosphate exist.

1.2 Principal Sites of Investigation

Terrestrial and marine samples were principally collected from and around Spitsbergen during NRPA field campaigns in 2000, 2001 and 2002, by the R.V. Lance in 2001 and the R.V. Polarstern research cruise in 2000 to the Fram Strait. Investigations on the islands of Hopen and Bjørnøya were conducted during the K.V. Senja research cruise in 2000. Additional samples were taken at locations around Spitsbergen by the Norwegian Polar Institute during various field campaigns in 2001 and 2002. As part of NRPA's ongoing radionuclide marine monitoring programme (RAME), funded by the Ministry of the Environment, stations for the routine collection of seawater samples to monitor the levels of ^{99}Tc were established in 2001 at the Ny Ålesund Settlement in Kongsfjorden, Spitsbergen and at the Norwegian Meteorological Institute's base stations on Bjørnøya and Hopen.

1.2.1 Kongsfjorden

The focus of the investigations by the NRPA on Svalbard has been centred on, but not limited to, the area surrounding Ny Ålesund (78°56'N 11°56'E), Kongsfjorden, on the west coast of Spitsbergen.

Kongsfjorden, a glacial fjord, is approximately 20 km long and 4 to 10 km wide, varying in depth from less than 100 m to greater than 300 m and has an estimated total volume of 29.4 km^3 (Ito and Kudoh, 1997). Through lack of an entrance sill, the outer fjord is influenced by the prevailing oceanographic conditions whilst the presence of large tidal glaciers (Kronebreen, Kongsvegen, Conwaybreen and Blomstrandbreen) dominates the inner fjørd (Svendsen *et al.*, 2002).

The Kongsfjorden terrestrial environment encompasses a total land area of 1428 km^2 of which 77 % is covered by glaciers (Svendsen *et al.*, 2002), while the ice-free terrain is composed of Arctic tundra with continuous permafrost and mountain slopes with scree.

Figure 5. Map of the marine and terrestrial environment of Kongsfjorden. Base map © Norwegian Polar Institute

Figure 6. The inner fjord of Kongsfjorden is dominated by large tidal glaciers.

The geology of the area (Brøgger peninsula) is composed of sedimentary lithologies from the Middle and Upper Carboniferous and Permian age (Hjelle, 1993), through the presence of sandstones, limestone and dolostone along the peninsula, whilst schist's, shale's and quartzites can be found near the inner part of the fjord.

The overburden on the Brøgger peninsula is of variable thickness, the low-lying parts of the peninsula being covered with interglacial pelagic deposits (Forman and Miller, 1984). Raised sea-terraces exist along the peninsula and these are covered with young soils (9000 to 12000 BP, Mann *et al.*, 1986) at an early developmental stage and exhibit features typical of a polar desert soil zone (Plichta, 1977). Tundra mires are present in low-lying areas that are waterlogged during the summer season. The vegetation in the area belongs to the mid- and the high arctic regions, detailed documentation of types and occurrence being found in Elvebakk and Prestrud (1996).

1.2.2 Hopen and Bjørnøya

Hopen (76°30'N 25°1'E), the easternmost manned Norwegian station in the Arctic, is a long and narrow rocky island of 46 km^2, with no lakes or rivers and low mountains of 150 to 370 m. The island is usually surrounded by ice during November/December to June/July, but there can be large variations from year to year. Composed of alternating strata of slate and limestone, Hopen contains some thin seams of coal, although test drilling found no evidence of oil or gas. Vegetation is sparse, limited to around 30 species, mainly represented by moss and a number of other hardy plants. There are few resident fauna, but the island receives seasonal influxes of Polar Bears, Arctic foxes and seabirds, particularly auks. The meteorological station on the southeast side of Hopen, was first established by the Norwegian Meteorological Institute in 1947 and was the focal point for collection of samples during the NRPA's visit in 2000.

Bjørnøya (74°30'N 19°1'E) lies approximately halfway between mainland Norway and Spitsbergen, at the southernmost extent of winter pack ice in the Norwegian Sea. The island (176 km^2) contains some six to seven hundred small lakes and is mostly flat with mountainous areas in the south (360 to 440 m) and a highest peak on the east coast of 563 m. Represented by many geological periods, Bjørnøya no longer has any glaciers and permafrost only descends to 50 to 75 m below the surface. Large colonies of seabirds occupy the southern coastal cliffs, whilst populations of Arctic Char are to be found in a number of lakes. In addition there is a small resident population of Arctic foxes with occasional appearances of polar bears when the pack ice reaches the island. The only present residents occupy the meteorological station on the north coast that was first established in 1918, although previously coal and lead ore were mined from 1916 to 1925. As was the case for Hopen, collection of samples on Bjørnøya by the NRPA in 2000 centred on the immediate area surrounding the Meteorological station.

Figure 7. Map of selected potential and actual sources of nuclear contamination to the European Arctic region.

1.3 Sources of Radionuclides to the Svalbard Environment

There are several actual and potential sources of radionuclides to the wider European Arctic region of which a number are of pertinent importance to the terrestrial and marine environments of Svalbard. These include: historical atmospheric testing of nuclear weapons, discharges from reprocessing of nuclear material in Western Europe (Aarkrog *et al.*, 1987), the Chernobyl Accident, discharges from the Techa, Ob and Yenisey River systems (Academy of Science, 1991), dumping of nuclear material at Novaya Zemlya and in the Kara Sea, sunken nuclear submarines, civilian and military nuclear facilities on the Kola Peninsula, the use of Radionuclide Thermoelectric Generators (RTGs) and sundry accidents involving nuclear materials. Furthermore, the possible production of technologically enhanced naturally occurring radioactive material (TENORM), as a by-product of current and historical extractive industries conducted or to be conducted either on Svalbard or in areas adjacent to the archipelago should be considered as an additional source.

1.3.1 Fallout from Nuclear Weapon Testing

Global fallout from atmospheric nuclear weapons testing is one of the main sources of radioactive contamination in the Arctic region. Of the 543 atmospheric weapons tests conducted globally, 91 of these were carried out in the Arctic region by the Former Soviet Union (FSU) at Novaya Zemlya with a total yield of 239.6 Mt (UNSCEAR, 2000). Aarkrog (1993) estimated a level of fallout in the Arctic region of 20 PBq of ^{90}Sr and 30 PBq of ^{137}Cs from 87 of these tests alone. Within the 70 to 80°N latitude band, UNSCEAR (2000) calculated mean integrated deposition densities as a result of atmospheric testing of 0.68 kBq/m^2 of ^{90}Sr and 1.09 kBq/m^2 ^{137}Cs, the latter in agreement with the lower end of an estimated ^{137}Cs cumulative deposition range of 1 to 5 kBq/m^2 made by AMAP (1997). Levels of atmospheric contamination reaching the Arctic dropped significantly in the years following the signing of the Limited Test Ban Treaty in 1963 (Barrie *et al.*, 1992), with a resultant but slower reduction in levels in terrestrial matrices.

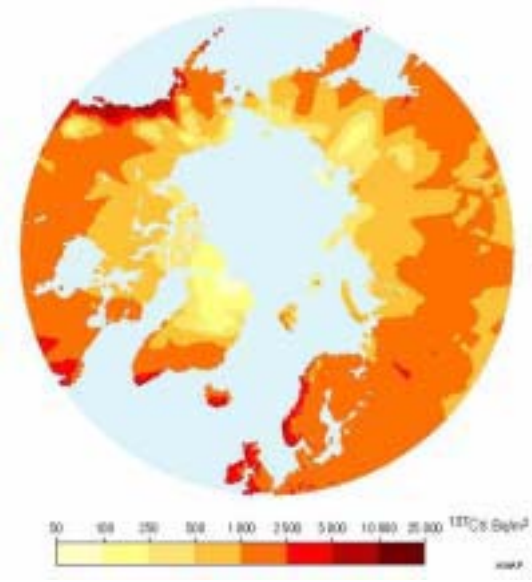

Figure 8. Estimated Arctic ground deposition of ^{137}Cs from nuclear weapon fallout, decay corrected to 1995 (AMAP, 1998).

In the early 1960's, underwater testing in Chernaya Fjord on the south west coast of Novaya Zemlya, resulted in contamination of the bay's benthic environment with elevated levels of plutonium and radioactive cesium, as well as other radioactive isotopes. However, the mobility of radionuclides in sediment is low and may at present only cause insignificant exposure for people. Exposure of biota is unknown. Today, the inventory of plutonium in Chernaya Bay is similar to other sites of major plutonium contamination, such as the most contaminated areas of Bylot Sound (where a B-52 bomber crashed) and the Irish Sea in the vicinity of the Sellafield reprocessing plant. In addition, underground detonations were carried out in the period 1963 to 1989 at Novaya Zemlya and at Amchitka Island, Alaska. While fission products have been identified in air after underground nuclear detonations, (Bjurman *et al.*, 1990), such detonations are assumed to have no significant impact on the level of radioactive contamination in the marine environment.

The predominant expression of fallout from atmospheric weapons testing in the environment of Svalbard today is primarily via the long lived isotopes of plutonium (^{238}Pu, ^{239}Pu, ^{240}Pu) with contributions from ^{137}Cs and ^{90}Sr that have diminished since time of deposition due to relatively short half-lives. ^{241}Am, a daughter of ^{240}Pu, is also present due to ingrowth over the years since the deposition of its parent.

1.3.2 Discharges from European Reprocessing Plants

A past and continuing source of anthropogenic radionuclides to the Arctic marine environment are the major nuclear fuel reprocessing facilities in Europe. These facilities include Sellafield in the United Kingdom, Dounreay in Scotland, and Cap la Hague in France. The highest discharges of nuclear waste have occurred from the reprocessing facilities at Sellafield, UK. From the late sixties until the mid-eighties, releases of

radiocesium (^{137}Cs) from Sellafield were a factor of 100 higher than the releases from Dounreay and Cap la Hague, peaking in the mid seventies at around 5 PBq/a. Due to stronger regulatory controls and plant improvements that have been implemented since this time, releases of several of the main radionuclides, including ^{137}Cs and the actinides plutonium ($^{239+240}$Pu) and americium (^{241}Am), have declined markedly over the ensuing years. In 1994 the Enhanced Actinide Removal Plant (EARP) and the Thermal Oxide Reprocessing Plant (THORP) began operations.

While the operation of these plants resulted in reduced discharges of some radionuclides such as $^{239+240}$Pu and ^{241}Am, plant operations

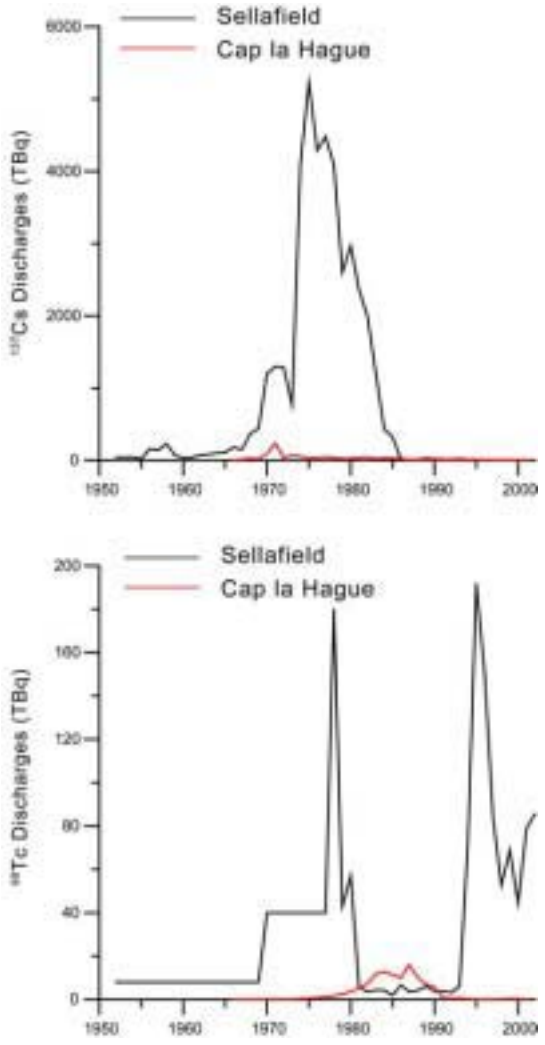

Figure 9. Discharges of ^{99}Tc and ^{137}Cs from Sellafield and Cap la Hague

increased the discharges of others, especially technetium (^{99}Tc) but also ^{137}Cs, ^{60}Co and ^{90}Sr.

Throughout the 1980's and early 1990's (1981 to 1993), ^{99}Tc was discharged from Sellafield at a rate of 1.9 to 6.6 TBq/a, following an earlier peak release of 180 TBq/a in 1978. Commencement of EARP operation saw a step increase in discharges to a level close to the authorised limit (200 TBq/a) of 72 to 190 TBq/a during the period 1994 to 1996. A reduction in the discharge of ^{99}Tc has occurred in recent years to levels under the current revised authorised limit of 90 TBq/a.

The reprocessing facility at Cap la Hague has operated since 1965 and although the total discharges are much less than for Sellafield, releases of ^{99}Tc during the period 1981 to 1990 exceeded those of Sellafield. ^{137}Cs discharges from la Hague peaked in 1971 at 240 TBq/a, but have been appreciably lower than for Sellafield over the duration of its operations as is the case for all discharges from Dounreay. The major discharges from Dounreay occurred in the 1960's and early 1970's, with smaller peaks in 1968 and 1973 from plant cleaning and decontamination procedures. The aggregate activity discharged from Dounreay up to 1986 was in the region of 10 PBq with ^{95}Zr/^{95}Nb, ^{144}Ce and ^{106}Ru representing 55 %, 17 % and 10 % of this activity respectively (CEC, 1990).

1.3.3 Chernobyl Accident

The most serious accident involving nuclear reactor operations occurred at the Chernobyl nuclear power plant in the Ukraine in April of 1986. Large amounts of radionuclides were released to the atmosphere contaminating both the local environment and areas across the Scandinavian countries of Finland, Sweden and Norway and much of Western Europe. It is estimated that as a result of the Chernobyl accident approximately 131 PBq of radiocesium (^{134}Cs and ^{137}Cs), 8 PBq of ^{90}Sr and 0.1 PBq of plutonium isotopes were released to the

environment (AMAP, 1997). In addition to direct fallout from the atmosphere, the Arctic marine environment may also be contaminated by transport of contamination from the North Sea and the Baltic Sea, the catchments of both of which received considerably more radionuclides from Chernobyl than Arctic regions. Calculations based on $^{134}Cs/^{137}Cs$ ratios in the Kara Sea in 1992, suggest that some 30 % of the ^{137}Cs contamination in the Kara Sea is derived from the Chernobyl accident (Strand *et al.*, 1993).

1.3.4 Other Actual and Potential Anthropogenic Sources

Several major nuclear facilities of various kinds are located in, and discharge to, the drainage basins of the large Russian rivers, the Ob and the Yenisey. Discharges or accidental releases from these facilities combined with surface run-off of radioactive fallout from atmospheric nuclear weapons testing may enter the river drainage basins and be transported downstream to the Kara Sea and further into the wider Arctic region.

Figure 10. Location of Russian nuclear facilities within the Ob and Yenisey watersheds.

The Mayak plant built in 1948 to produce ^{239}Pu for nuclear weapons discharged large quantities of radioactive waste into the nearby Techa River from 1949 to 1956. Current releases from the Mayak plant are considerably reduced, since the last of the reactors producing weapons-grade plutonium was shut down in 1990. In addition to intentional discharges, the area around Mayak (~20000 km^2) was contaminated by the Kyshtim accident in 1957, when a storage tank of highly radioactive material exploded. Between 1951 and 1966, a system of dams along the upper parts of the Techa was constructed in an attempt to retain most of the radioactive material, creating several artificial lakes along the river course. Failure of the dam system would results in a large scale discharge of radionuclides into the already contaminated Asanow Swamp and into the Techa and Ob river systems. Other large-scale releases would occur if the Asanow swamp were to dry out, after which spring floods could wash remobilised radionuclides into the river systems. Lake Karachay, which received Mayak discharges from 1951 onwards, has no outlet but caused contamination of the surrounding region during a dry period in 1967, when lake sediments were exposed and transported by winds. The lake bottom is now covered with blocks of concrete to prevent further resuspension, but radionuclides might leach from the sediments into groundwater and eventually into the Techa/Ob river system.

The Siberian Chemical Combine at Seversk near Tomsk is one of the largest nuclear weapons production facilities in the world. Since 1956, the plant has released contaminated cooling water into the river Tom, which ultimately drains into the Ob. The plant came to international attention in April 1993, when a chemical reaction caused an explosion in a tank containing fission products and uranium nitrate solution, contaminating an area of about 90 km^2. A recent report from the Russian Federation Security Council has stated that large amounts of radioactive wastes were stored within the

industrial zone, some of which are retained in reservoirs, leading to fears over contamination of groundwater.

The Krasnoyarsk Mining and Chemical Combine, recently renamed Zheleznogorsk, reprocesses spent nuclear fuel for the production of plutonium. Prior to 1992, contaminated cooling water from two reactors was routinely discharged into the Yenisey River. At present liquid waste is stored in reservoirs or injected into deep holding wells, which may contaminate groundwater migrating into the Yenisey.

It is difficult to estimate the impact on the Barents Sea from these sources both because the transport routes and the magnitude of transport are not sufficiently documented and because the data regarding discharges and radioactive inventory in the open literature contains inconsistencies. Observations conducted by Roshydromet (Vakulovsky *et al.*, 1993) indicate that about 1 PBq of ^{90}Sr and 0.1 PBq of ^{137}Cs were transported by the Ob and Yenisey rivers during 1961 to 1989. In addition, approximately 200 TBq of ^{137}Cs have been transported to the Barents Sea by the rivers Pechora, Onega and Severnaya Dvina (Vakulovsky *et al.*, 1993).

Figure 11. Sites in the Arctic used by the FSU for dumping radioactive waste.

Figure 12. Dumped containers in Stepovogo Bay, Novaya Zemlya (AMAP).

The Former Soviet Union (FSU) dumped high, intermediate, and low level radioactive waste in the Arctic Seas during the years 1959 to 1991, including six nuclear submarine reactors and a shielding assembly from an icebreaker reactor containing spent fuel. The solid waste and the nuclear reactors were dumped in the Kara Sea and in the fjords of Novaya Zemlya at depths of 12 to 135 m, and in the Novaya Zemlya trough at a depth of 300 m.

The liquid, low-level waste was dumped into the open Barents and Kara Seas. At the time of dumping, experts estimated that the spent nuclear fuel represented a total activity of 8.5 x 10^{16} Bq. From 1992 to 1994, a joint Norwegian-Russian expert group has used sonar and a remotely operated vehicle in an attempt to find and examine the waste. The exploratory cruises also took samples of water, sediments, and biota in the area. The results show that there is no significant contamination of the Kara Sea. In fact, the levels of radionuclides in the water are lower than in many other marine areas, such as the Irish, Baltic, and North Seas. However, higher levels of radioactivity in the immediate vicinity of the waste show that there is local contamination at the dumpsites, whilst the major risks are for the long term, after the containment systems corrode.

Figure 13. Location of submarine accidents in the Arctic region.

In 1989 the Soviet nuclear submarine Komsomolets caught fire and sank to the southeast of Bjørnøya. The submarine contained a nuclear reactor with a radionuclide inventory including 2.7 PBq of ^{90}Sr and 3.0 PBq of ^{137}Cs and two nuclear warheads containing approximately 16 TBq of ^{239}Pu (CCMS, 1995). Underwater monitoring has shown elevated levels of ^{137}Cs and also indicated ^{134}Cs near the reactor section, indicating some leakage of radionuclides has occurred (Kolstad, 1995). However, conservative modelling of the possible releases of ^{137}Cs indicates that radionuclide concentrations in seawater and fish caused by past, present and future releases from Komsomolets are at least a factor 100 lower than current concentrations in these media (CCMS, 1995).

In 2000, the Kursk, a Russian Oscar II class attack submarine containing two 190 MW pressurized water reactors and 22 Granit cruise missiles sank in international waters in the Barents Sea. No indication of leakage from the submarine was observed in any dose rate readings or any of the measurements on environmental samples taken close to the Kursk immediately after the sinking, during subsequent operations at the site or during its final raising (Amundsen *et al.*, 2001). The fact that no elevated radioactivity levels were observed indicates that the reactors had been shut down, as stated by the Russian authorities and that the reactor compartment was not flooded with contaminated water. The section of the Kursk containing the two reactors and the cruise missiles was raised in 2001.

On the 30th August 2003, the K-159, a decommissioned Russian November class attack submarine, while being towed on pontoons to the Polyarny shipyard for dismantlement, sank 5 km north west of the Kildin Islands in Russian territorial waters to a depth of 240 m. The submarine contained two 70 MW pressurised water reactors which were shut down in 1989, but no nuclear missiles. The cores of the reactors have an estimated 800 kg of spent nuclear fuel with an activity of between 2 – 4 PBq. A decision on whether to raise the submarine is expected to be made in 2004.

On the Kola Peninsula, the Russian Northern Fleet and the civilian nuclear icebreaker fleet have their main bases. Approximately 100 nuclear vessels operate from these bases, containing together about 200 nuclear reactors. Besides the operating vessels, about 80 decommissioned nuclear submarines waiting to be dismantled are stationed at the military bases. Most of these submarines still contain their nuclear reactors and fuel. Plans exist to decommission 125 submarines during the next several years. However, many challenges exist in performing the dismantling work minimising the risks of accidents and doses to the workers. Disposing of the nuclear materials poses a further challenge as storage facilities for spent nuclear fuel in the Kola Peninsula are already filled to capacity (NEFCO, 1996). The operations involved in the decommissioning of these submarines and related activities pose some risk to the Arctic environment should an accident occur or discharge of radioactivity occur.

The Kola Nuclear Power Plant (NPP) represents a potential major source of radionuclide contamination to the Arctic due to its high inventory and lack of containment. The Kola NPP has 4 pressurized water reactors in operation from the early 1970's and 80's with an estimated total inventory of fission products of about 10^{19} Bq (Stokke, 1997; JRNEG, 2002). According to Larsen et al. (1999) a worst case accident scenario involving a large loss of cooling could result in the release of 26.7 PBq of ^{134}Cs and ^{137}Cs and 1.7 PBq of ^{90}Sr.

The disintegration of the Cosmos 954 satellite over the Canadian Northwest Territories on January 24, 1978, spread radioactive material over a thousand kilometre long swathe to the northeast of the Great Slave Lake. Part of the radionuclide inventory, containing ^{90}Sr and ^{137}Cs, was volatilised and dispersed as fine particles high in the upper atmosphere and deposited over a period of years. In 1980, the deposited activity of ^{90}Sr in the northern hemisphere from this source was estimated to be 3 x 10^{12} Bq compared to a total deposition of 1.1 x 10^{15} Bq (UNSCEAR, 1982).

The remoteness of the Arctic and the difficulty of civil and military operations in the environment have led to nuclear materials being introduced as power sources for a variety of installations. Radionuclide Thermoelectric Generators (RTGs) are often used as power supplies, utilising radioactive decay, of typically ^{90}Sr, as a heat source. RTGs are located throughout Alaska and the Eurasian arctic region and are generally considered to pose little risk of contamination, although some RTGs have been reported as lost or stolen. Any releases from RTGs are unlikely to impact the Svalbard environment unless release should occur to the marine environment.

1.3.5 Naturally Occurring Radioactive Materials (NORM)

Naturally occurring radioactive materials on Svalbard are, as with all regions of the globe, constituted by non-series nuclides (^{87}Rb, ^{40}K, ^{14}C and ^{3}H) and series-nuclides (those derived from the three natural decay chains of ^{238}U, ^{235}U and ^{232}Th). These nuclides are present in all environmental matrices and biota (marine and terrestrial) as a result of natural weathering and uptake processes and constitute the primary dose contribution via ^{210}Pb and ^{210}Po which arise from the decay of the gaseous nuclide ^{222}Rn which is itself a daughter of ^{226}Ra. Various natural physical and chemical processes can lead to locally enhanced levels of these nuclides, in excess of what may be considered "background" values with a related increase in the resultant dose. Knowledge of these processes and the extent to which concentration of NORM may occur is important as the significance of superimposed anthropogenic dose contributions is largely determined by the intrinsic dose due to NORM concentrations. Information on NORM levels on Svalbard is lacking and more research is required to assess the peculiarities of NORM behaviour in this Arctic environment in order to afford better assessment of the radiological sensitivity of various environmental components.

1.3.6 Technologically Enhanced Naturally Occurring Radioactive Material (TENORM)

Mineral extraction activities, such as those conducted by oil, gas and coal industries, are widespread throughout the Arctic region. Waste products of these activities can result in significant contributions to the radioactive burden of the surrounding environment due to the increased concentration of naturally occurring radioactive materials (NORM) to levels that would not normally be found in the

environment. Coal, mined commercially on Svalbard since the early 1900's, contains radionuclides of the uranium and thorium series as well as ^{40}K and extraction and processing of coal can result in releases of these radionuclides to the broader environment. Typical concentrations of these nuclides in a variety of coals have been collated by Beck *et al.* (1980) and indicate that coal may contain between < 2.4 – 304 Bq/kg ^{238}U and from 2.5 – 191 Bq/kg ^{232}Th depending on the origin of the coal. Of greater significance is that the activity concentrations of NORM in ash, produced from the combustion of coal, can be 10 to 80 fold higher than in the raw coal itself (Baxter, 1993). Commercial coal mining began on Svalbard with mines in and around Longyearbyen and at Sveagruva at the head of Van Mijenfjord and has been conducted at a variety of locations on the archipelago since. Mining of coal in Kongsfjorden at Ny Ålesund was conducted from 1917 to 1962, with an estimated total export of 1.43 million tonnes (Hjelle *et al.*, 1999). Evidence of these past operations at Ny Ålesund remains to this day, in the form of machinery, equipment and pronounced piles of waste materials. Contemporary coal mining operations, are principally conducted by the Store Norske Group at two mines, Mine 7, near Longyearbyen, Svea Nord and by other mine operators at the Russian mining towns of Barentsburg, 45 km to the west of Longyearbyen and Pyramiden in Billefjorden. Store Norske's main operation is in Svea, roughly 60 km South-East of Longyearbyen, with an annual production of between 1.6 and 1.9 million tonnes.

Although extraction of oil and natural gas can result in the production of waste materials with high levels of natural radioactivity, the nature and location of these activities means that, at present, they are unlikely to contribute significantly to radioactivity levels in the Svalbard region. The potential for future TENORM contamination of the Svalbard region is dependant on the initiation of new extractive

Figure 14. Coal mining wastes near Ny Ålesund.

industrial ventures or the expansion of existing operations. Information about current levels of NORM in the Svalbard region is therefore required in order to fully assess the radiological implications of any new extractive ventures that may be conducted on or near Svalbard.

1.4 The Vulnerability of the Svalbard Terrestrial and Marine Environments to Radionuclide Contamination

Although a region of relatively undisturbed wilderness, Svalbard remains vulnerable to a wide variety of pollutants, not least of these being radioactive contaminants. Due to its geographical location and the nature of environmental processes that are specific to the High Arctic, it is important to understand the behaviour, redistribution and accumulation of radionuclides within Svalbard terrestrial and marine environments.

In the terrestrial environment, the principal factors that govern the transport and biotic uptake of radionuclides (persistence, solubility, nutrient competition and location/trophic level of biota within ecosystems) are all affected to varying degrees by arctic specific processes. In the marine environment, the same can be said of the controls and the mechanisms by which

radionuclides are transported to the Svalbard area from within the Arctic itself or from further afield.

Effects of climate on both the moisture content and organic input into Arctic soils, largely determines the retention, mobilisation and behaviour of radionuclides within this matrix. In many arctic regions, ^{137}Cs penetration is minimal, despite the main deposition outside of Chernobyl affected areas occurring in the 1950 and 60's. Typically ^{137}Cs is detected predominantly within the upper soil layers (0 to 5 cm), in association with organic material rather than the underlying mineral horizons (e.g. Taylor *et al.*, 1988; Baskaran *et al.*, 1991; Stranberg, 1997). Arctic freeze-thaw cycles can have physical and chemical effects on the vertical distribution of radionuclides in these soils. These effects on soil chemistry exert some control over the movement and solubility of radionuclides between and within soils and the overlying snowpack. Such movement can promote a surge in contaminants in runoff associated with the initial spring meltwater (Johannessen and Henriksen, 1978). Soil horizons can undergo severe distortion due to high pressures generated by the freezing of soil that result in frost heave (see for example, Nakano, 1990) or by the slipping of saturated layers (solifluction).

The unique environment of the Arctic has produced many adaptations in the ecosystems that are found there, which can increase the vulnerability of the organisms involved and the environment in general to radioactive contamination. This is perhaps most evident in a consideration of Arctic terrestrial food chains, which tend to be extremely short. In this regard the role of lichens and mosses, which are highly efficient in their uptake and retention of radioisotopes, in the introduction of radionuclides to herbivores has received a great deal of attention (e.g. Gaare and Staaland, 1994). An as yet unexplored transfer mechanism that may have significance for arctic radioecology is the potential of seabirds, through faecal deposits,

Figure 15. Previously identified as playing a role in heavy metal introduction and behaviour in the Arctic, the role of seabird colonies on Arctic radioecology has been the subject of resent research attention.

to transfer radionuclides from the marine to the terrestrial environment or to condition soils, via nutrient input, such that the affected soils have a greater capacity for the adsorption of radionuclides. It has been shown that seabird faeces are the principal source of heavy metal input to arctic soils (Headley, 1996) and that transfer can occur of these metals to vegetation growing on faecal affected soils (Godzik, 1991).

The role of arctic vegetation species in the accumulation and retention of radionuclide contaminants is closely linked to the overall nutrient status of the Arctic terrestrial environment. The enhanced nutrient status of faecal affected soils subsequently causes an increase in both the diversity and quantity of vegetation growing in these areas (Eurola and Hakala, 1977) and therefore provides enhanced grazing for the herbivores of the region. The consumption of such vegetation by herbivores and further trophic transfer may result in novel or increased exposure to radionuclide contaminants.

Similarly to the terrestrial environment, contamination of the marine environment surrounding Svalbard has occurred directly through global fallout from atmospheric weapon testing, yet further contamination occurs through oceanic and ice driven long-range transport of radionuclides. Marine long-range transport from European reprocessing plants and coastal water contaminated by Chernobyl fallout is principally mediated through oceanic circulation in the North Atlantic and Arctic Seas. The general circulation pattern describes Atlantic water flowing through the Irish Sea and English Channel, via the North Sea to become incorporated with Baltic Sea outflow through the Skagerrak and forming the north flowing Norwegian Coastal Current. As the Norwegian Coastal Current continues northwards, it progressively mixes with Atlantic water from the Norwegian Atlantic Current, until the Norwegian Atlantic Current splits at the western boundary of the Barents Sea into the North Cape Current and the West Spitsbergen Current that flows north along the western coastline of Svalbard. Transit times of radionuclides from Sellafield to the Svalbard area via this route have been estimated on the order of 5 to 7 years for ^{137}Cs (Kautsky, 1987; Dahlgaard, 1995) and 4 to 5 years for ^{99}Tc (NRPA, 2001; Kershaw *et al.*, 2003).

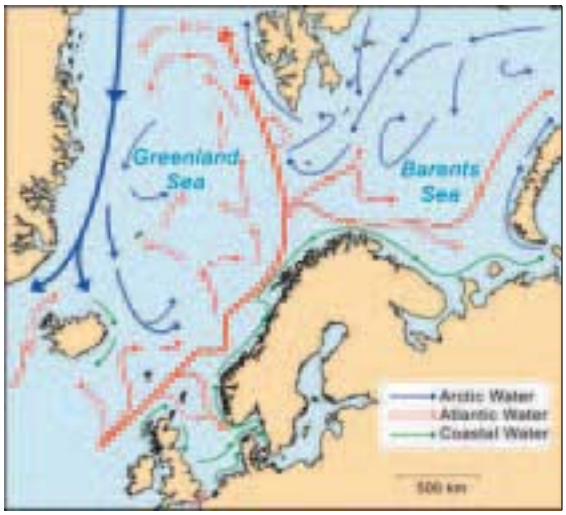

Figure 16. Oceanic circulation in the Northern Seas (adapted from Loeng (1998)).

Models of the dispersion of radionuclides from the European reprocessing plants to the Arctic have shown good overall agreement with experimentally data (e.g. Nies *et al.*, 1999; Iosjpe *et al.*, 2002; Karcher *et al.*, 2003) and in addition, have highlighted mesoscale variability in surface concentrations of radionuclides, which may have important implications for future monitoring strategies.

It has been suggested that incorporation of radionuclides as well as other pollutants, into the ice cover in the Arctic Seas, may result in significant transportation of contaminants from one area to another. Contamination of ice with radionuclides occurs from *in situ* seawater contamination, atmospheric deposition onto existing sea ice cover and through the incorporation of contaminated suspended material from terrestrial run-off and contaminated bottom sediments in seasonally formed coastal ice. The formation of seasonal coastal sea ice occurs along long swathes of the European Arctic coastline and of particular interest, in the Kara Sea, in the shallow coastal areas of the Ob and Yenisey estuaries and the coastal areas off Novaya Zemlya (e.g. Vinje and Kvambekk, 1991; Dethleff *et al.*, 1998; Landa *et al.*, 1998). Kara Sea ice has been shown to travel north round the tip of Novaya Zemlya, into the Barents Sea and southwest towards Svalbard

Figure 17. Sediment laden sea-ice off the west coast of Svalbard. Transport of radionuclides by sea-ice may have implications for the Arctic environment.

(e.g. Vinje and Kvambekk, 1991; Nurnberg *et al.*, 1994; Landa *et al.*, 1998). Levels of ^{137}Cs and $^{239+240}$Pu of sediments entrained in sea ice from across the Arctic Basin have been reported in the range of 0.2 to 78 Bq/kg and 0.02 to 1.8 Bq/kg respectively (e.g. Meese *et al.*, 1997; Landa *et al.*, 1998). During the spring and summer, contaminated sediments may be deposited during ice melting, which can occur in the ice-melting zone along the polar front in the Svalbard area (Loeng, 1991).

Furthermore, high levels of primary production associated with the polar front in the Svalbard area, may lead to increased fluxes of certain radionuclides to marine sediments through active scavenging of nuclides from the water column during the production period (Føyn and Sværen, 1997). Additional radionuclide contamination of the Svalbard marine environment can originate from Svalbard itself, through fluxes of radionuclides associated with terrestrial run-off, suspended sediments and glacial meltwater.

Radionuclides present in the marine environment may then be available for biological uptake by marine biota and possible subsequent transfer through marine and marine/terrestrial food webs. Indeed, some marine biota such as crustaceans, molluscs and marine algae exhibit very high uptake rates of certain anthropogenic and natural radionuclides (e.g. Pentreath *et al.*, 1982; Aarkrog *et al.*, 1997; Brown *et al.*, 1999) that may have important consequences for Arctic ecosystems in the event of significant levels of contamination.

1.5 Previous Radiological Investigations of Svalbard

There have been limited radiological studies of the marine and terrestrial environments in the Svalbard region. Of the two, the marine environment has received the most attention, due in part to the well-documented previous and continued discharges from the European Reprocessing Plants and the utilisation of radionuclides in these discharges as tracers of oceanic circulation. In comparison, there have been only a few radiological investigations of the Svalbard terrestrial environment, with what little data available pertaining mostly to the deposition of global fallout.

1.5.1 Radioactive Contamination of the Marine Environment of Svalbard

For the Svalbard marine environment, the temporal variation in levels of radionuclides in surface seawater has in general reflected the relative contributions of the different sources. For ^{137}Cs, the 1992 contribution from global fallout in North Atlantic surface water was reported as ~2.5 Bq/m^3 (Dahlgaard *et al.*, 1995), whilst it has been estimated that 13% of the total historical ^{137}Cs discharges from Sellafield, amounting to some 5.2 PBq, has passed through the West Spitsbergen current into the Arctic Ocean (Kershaw and Baxter, 1995). The peak ^{137}Cs discharges during the mid 1970's resulted in elevated levels in the early to mid 1980's in the Svalbard area of around 20 to 30 Bq/m^3 (Hallstadius *et al.*, 1982; Kershaw and Baxter, 1995). The decrease in discharges of ^{137}Cs from Sellafield from the late 1980's onwards, has seen a decrease in Svalbard seawater values, e.g. 8 Bq/m^3 in 1989 (Guegueniat *et al.*, 1997) and 3 to 5 Bq/m^3 in 1994 (Kershaw *et al.*, 1997), as the 1970's plume has been diluted with ingress of Atlantic Water. Levels however remain higher than 'background' due to the continued

contributions from the European reprocessing plants and run off containing fallout from the Chernobyl Accident.

The history of Svalbard seawater ^{99}Tc levels, as for ^{137}Cs, is dominated by the contributions of the European reprocessing plants and in recent years, Sellafield in particular. Dahlgaard (1996) estimated that the relative contributions to the levels of ^{99}Tc in the East Greenland Current over the period 1989 to 1990 was 85% from European Coastal discharges and 15% from global fallout, previously reported as 5 mBq/m^3 (Dahlgaard *et al.*, 1995). Following the elevated ^{99}Tc EARP associated discharges from Sellafield in 1994, increased levels of ^{99}Tc were observed in seawater and marine biota off the Norwegian Coast (Brown *et al.*, 1999; Rudjord *et al.*, 2001). Further oceanic transport of EARP associated ^{99}Tc to Svalbard waters has been observed, with levels of ^{99}Tc in the West Spitsbergen Current off the Western coast of Svalbard increasing from 0.04 to 0.06 Bq/m^3 in 1994 (pre-EARP) to 0.21 to 0.3 Bq/m^3 in 2000 (Kershaw *et al.*, 1999, in press).

Available data for $^{239+240}$Pu concentrations in surface seawater for West Svalbard showed a decrease from 17 mBq/m^3 in 1981 to 8 mBq/m^3 in 1989 (Kershaw and Baxter, 1995). Over a similar time period, Hallstadius *et al.* (1986) assigned global fallout as the dominant source north of 65°N, although ^{238}Pu/$^{239+240}$Pu activity ratios suggested a non-fallout component of ^{238}Pu in Arctic waters.

Source	^{238}Pu/$^{239+240}$Pu activity ratio
Northern hemisphere fallout	~0.035
Sellafield, historical	~0.2
Sellafield, recent	~0.3
Chernobyl fallout	~0.5

Table I. ^{238}Pu/$^{239+240}$Pu activity ratios associated with various source terms (UNSCEAR).

	$^{239+240}$Pu (mBq/m^3)	^{238}Pu/$^{239+240}$Pu
1981[1]	17.4	0.054
1983[2]	13.0	0.07 - 0.08
1985[1]	15.6	0.056
1989[1]	8.8	0.053
1994[3]	6.5	0.05

Table II. $^{239+240}$Pu concentrations and ^{238}Pu/$^{239+240}$Pu activity ratios for West Svalbard surface seawater Data from: 1. Kershaw and Baxter, 1995; 2. Hallstadius et al., 1986; 3. Kershaw et al., 1999.

More recently, concentrations of ~6.5 mBq/m^3 were reported for West Svalbard, with higher concentrations of ~8.5 mBq/m^3 off the East coast (Kershaw *et al.*, 1999).

Activity concentrations of ^{241}Am for the northern seas are relatively uniform with a reported level of 2.1 ± 0.9 Bq/m^3 (Hallstadius *et al.*, 1986) and a ^{241}Am/$^{239+240}$Pu activity ratio of 0.13 ± 0.04 (Holm *et al.*, 1983).

Inventories of ^{137}Cs and $^{239+240}$Pu in marine sediments around Svalbard have been reported in the range of 120 to 500 Bq/m^2 and 18 to 115 Bq/m^2 respectively, with ^{238}Pu/$^{239+240}$Pu ratios (0.055 to 0.065) indicating the influence of Sellafield (Roos *et al.*, 1995). The observed lower inventories in sediments east of Svalbard were ascribed to the difference in hydrographic and ice cover regimes between Western and Eastern Svalbard waters. In comparison, Føyn and Sværen (1997) and Heldal *et al.*, (2002) reported elevated levels (50 % higher than adjacent areas) of ^{137}Cs, ^{238}Pu, $^{239+240}$Pu and ^{241}Am in surface sediments west of Svalbard and in the Svalbard-Bjørnøya area. ^{238}Pu/$^{239+240}$Pu activity ratios of these sediments were consistent with that of global fallout, indicating that the source of these nuclides was probably either ice rafted debris (IRD) from the Barents and Kara Seas or run-off and IRD from Svalbard itself.

	^{137}Cs	$^{239+240}$Pu	^{99}Tc
Fucus sp.	75	8000	ND
Laminaria sp. (stipes)	45	4300	69000[1]
Laminaria sp. (blades)	70	6100	23000[1]

Table III. Concentration factors (d.w.) in marine algae from the Svalbard area Data from: 1. Rudjord et al., 2001. All other values Holm et al. (1983).

For marine biota in the Svalbard area (excluding marine mammals), information is only available for certain marine algae. Levels of ^{137}Cs and $^{239+240}$Pu in *Laminaria* sp. (stipes) reported in 1980 were 1.7 and 0.1 Bq/kg, similar to that observed in a *Fucus* sp. (Holm *et al.*, 1983; Rissanen *et al.*, 2000). Holm *et al.* (1984) determined levels of ^{99}Tc in *Fucus* spp. collected in 1980 and 1981 from the Svalbard area in the range of 8 to 23 Bq/kg and in *Laminaria* spp. and *Alaria esculenta* of between 0.5 – 2.7 Bq/kg and 0.9 – 2.4 Bq/kg respectively. More recently, concentrations of ^{99}Tc in the range 8.0 to 10.3 Bq/kg (stipes) have been reported for a *Laminaria* sp. (Rudjord *et al.*, 2001), with an average concentration factor of 69000, a factor of 10 and 1000 fold greater than concentration factors for $^{239+240}$Pu and ^{137}Cs respectively.

1.5.2 Radioactive Contamination of the Terrestrial Environment of Svalbard

Deposition on Svalbard of ^{137}Cs from atmospheric weapon testing has been estimated, on the basis of soil/lichen sampling, at 2.2 ± 0.3 kBq/m^2 (Hallstadius *et al.*, 1982), though lower estimates exist of between 200 to 540 Bq/m^2 from a survey of glacial ice cores, which showed higher deposition on the eastern side of Svalbard compared to the western side (Pinglot *et al.*, 1994). Other studies report pre-Chernobyl soil ^{137}Cs values of 1.51 and 1.59 kBq/m^2 (Kjos-Hanssen and Tørresdal, 1981), close to the higher global fallout estimate of Hallstadius *et al.* (1982), although Negoita (1997, 1999) reported typical post-Chernobyl ^{137}Cs levels in soils of between <1.5 and 35.8 Bq/kg, with one measurement of 213 Bq/kg. Svalbard is generally considered to have been relatively unaffected by fallout from the Chernobyl Accident, with the ice core survey of Pinglot *et al.* (1994) showing Chernobyl fallout levels for ^{137}Cs of only 20 Bq/m^2, a factor of 10 lower than levels from fallout due to atmospheric weapon testing.

Source	^{137}Cs	$^{239+240}$Pu
Global fallout	200-2200	13.3
Chernobyl	20	-

Table IV. Reported deposition contributions of ^{137}Cs and $^{239+240}$Pu from global fallout and the Chernobyl Accident on Svalbard (Bq/m^2).

Average fallout levels of $^{239+240}$Pu in the latitude band 70 to 80°N have been calculated at 13.3 Bq/m^2, with 0.3 Bq/m^2 of ^{238}Pu (Hardy *et al.*, 1973), compared to reported levels of 14 to 26 Bq/m^2 $^{239+240}$Pu in Svalbard surface vegetation and soil samples (Holm *et al.*, 1983; Rissanen *et al.*, 2000). Estimated ^{241}Am/$^{239+240}$Pu activity ratios for integrated fall out on Svalbard have been reported as 0.37 ± 0.05 (Holm *et al.*, 1983).

For terrestrial flora, levels of ^{137}Cs in Polar Willow (*Salix polaris*) and moss (unknown sp.) have been reported at 27 Bq/kg and 230 Bq/kg (Kjos-Hanssen and Tørresdal, 1981; Aarkrog *et al.*, 1984) with levels of ^{137}Cs and $^{239+240}$Pu in Lichen (*Cladonia* sp.) of 580 ± 20 Bq/kg and 4.5 ± 1 Bq/kg (Holm *et al.*, 1983).

2. Current Levels of Radioactive Contamination in the Environment of Svalbard

Information on the current levels of radioactive contamination in the Svalbard environment is important for a number of reasons. Such information allows for assessment of the impacts of such contamination on this vulnerable environment and for elucidation of how contaminant radionuclides behave in a High Arctic environment. The information also allows for the accurate assessment of the impact of future contamination incidents and prediction of the long-term effects of resulting contamination.

2.1 Levels of Radionuclides in the Marine Environment of Svalbard

This section presents results on the levels of the anthropogenic radionuclides ^{99}Tc, ^{137}Cs, ^{238}Pu, $^{239+241}$Pu and ^{241}Am in seawater, sediments and marine algae from the Svalbard marine environment over the period 2000 to 2002.

Figure 18. Sampling of seawater for ^{99}Tc off the west coast of Svalbard.

These results are discussed in the light of previous radiometric investigations in the Svalbard and Norwegian marine environments and in the context of past and continuing sources of radionuclides to the Svalbard area.

2.1.1 ^{99}Tc in Seawater

Activity concentrations of ^{99}Tc in seawater from the Svalbard area in 2001 ranged from 0.13 to 0.36 Bq/m^3, a similar magnitude to that reported for the Svalbard area in 2000, but a five fold increase on 1994 levels of 0.03 to 0.08 Bq/m^3 (Kershaw *et al.*, 1999, in press).

The highest concentrations of ^{99}Tc at the Svalbard monitoring stations were observed at Bjørnøya and Hopen (0.35 ± 0.05 and 0.32 ± 0.04 Bq/m^3) whilst the highest concentration at Ny Ålesund was 0.25 ± 0.03 Bq/m^3. This compares with an average concentration for 2001 of 1.24 Bq/m^3 (range 0.9 – 1.95 Bq/m^3) for the NRPA monitoring station at Hillesøy on mainland Norway (Kolstad and Lind, 2002).

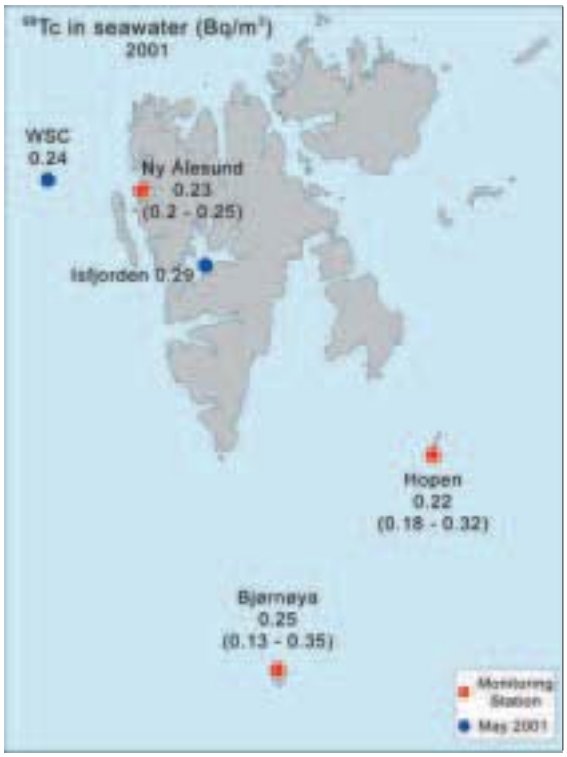

Figure 19. ^{99}Tc (Bq/m^3) in seawater in the Svalbard region in 2001.

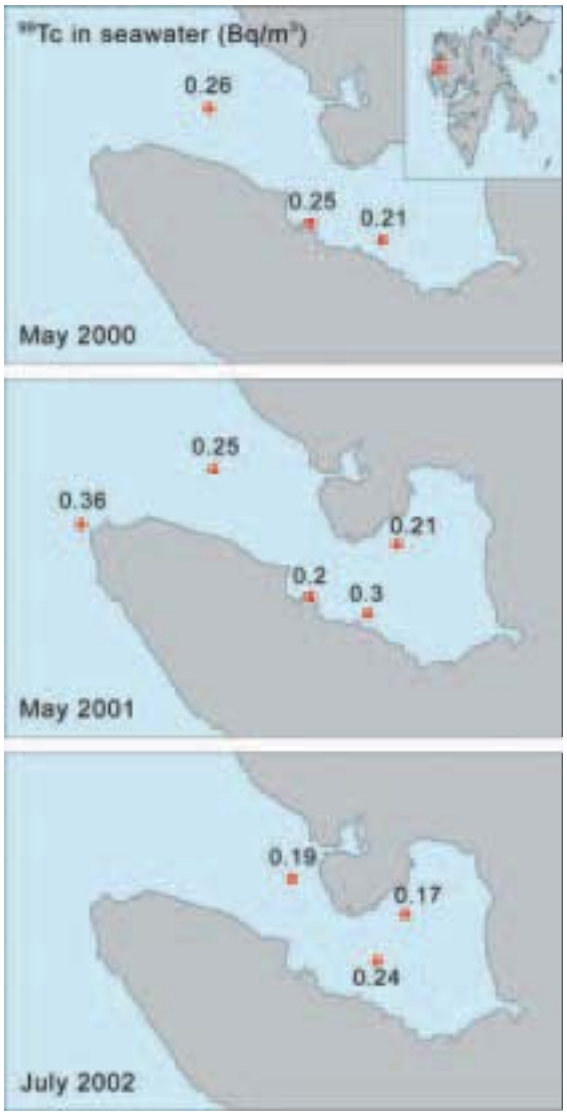

Figure 20. Seawater ^{99}Tc activity concentrations (Bq/m³) in Kongsfjorden from May 2000, May 2001 and July 2002.

These values reflect the dominant oceanic surface circulation in the Northern seas and the concomitant transfer of EARP associated ^{99}Tc into the Norwegian Arctic. That lower levels of ^{99}Tc are observed in the Svalbard area compared to Norwegian coastal waters for the same period is due to the ingress of North Atlantic Water into the Norwegian Coastal Current and the West Spitsbergen Current (WSC) with the resultant dilution of the EARP associated ^{99}Tc signal. Nevertheless, levels of ^{99}Tc in the centre of the WSC for 2000 and 2001 were recorded at 0.39 ± 0.05 and 0.24 ± 0.04 Bq/m³ respectively compared to 1994 pre-EARP levels of 0.04 Bq/m³ (Kershaw *et al.*, 1999).

The importance of the WSC influence on Svalbard can be seen in the associated increase in levels of ^{99}Tc both within the WSC and within fjords along the west coast of Spitsbergen. In 2001, the concentration of ^{99}Tc in two water samples from Isfjorden (~100km south of Kongsfjorden) was 0.28 ± 0.04 and 0.29 ± 0.05 Bq/m³ compared to a single measurement of 0.13 ± 0.03 Bq/m³ in 1999.

That ^{99}Tc levels in the inner part of Kongsfjorden are of a similar magnitude to levels observed in the outer fjord is an indication of the importance and scale of the WSC in exchange processes throughout the entirety of this fjord. In 2001, sub-surface samples taken in addition to surface samples at stations in Kongsfjorden and in the WSC, showed relatively uniform levels of ^{99}Tc.

However, sub-surface samples taken in the WSC in 2000 at greater depths (465 to 1000m) showed levels of ^{99}Tc of 0.17 and 0.04 Bq/m³, 2 to 5 fold lower than surface levels. That uniform distributions of ^{99}Tc were observed in the top 100 – 125m of Kongsfjorden and the WSC is a reflection of the conservative nature of ^{99}Tc in seawater and the apparent depth of the mixed layer at the time of sampling.

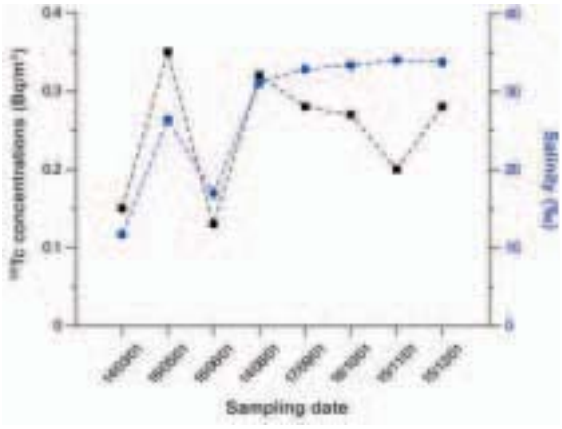

Figure 21. Activity concentrations of ^{99}Tc (Bq/m³) and salinity (‰) of surface seawater samples collected at Bjørnøya in 2001.

Location	Depth (m)	Salinity (‰)	^{99}Tc (Bq/m^3)	^{137}Cs (Bq/m^3)
Kongsfjorden	0	34.4	0.25 ±0.04	2.41 ±0.14
	125	34.6	0.29 ±0.03	2.33 ±0.11
WSC	0	34.5	0.24 ±0.04	2.3 ±0.11
	100	34.1	0.22 ±0.04	2.1 ±0.1

Table V. Activity concentrations of ^{99}Tc and ^{137}Cs in surface and sub-surface seawater samples collected from Kongsfjorden and the West Spitsbergen Current (WSC) in 2001.

The observed variation in sample salinity may partly explain the variation in surface ^{99}Tc levels both within fjords and between consecutive years. Due to the conservative behaviour of ^{99}Tc in seawater, mixing dynamics between ^{99}Tc bearing seawater from the WSC and fresh water input from terrestrial runoff and glacial meltwater should have proportional effects on both ^{99}Tc levels and sample salinity. In Kongsfjorden, average ^{99}Tc and salinity levels for May 2001 were 0.24 Bq/m^3 and 34.5 ‰, whilst average ^{99}Tc levels for July 2002 were 0.2 Bq/m^3 and 28.9 ‰. A relationship between salinity and ^{99}Tc concentrations in seawater has been observed previously at Hillesøy (Kolstad and Lind, 2002) and is further suggested by the 2001 time series for Bjørnøya.

When considered together, these observations suggest that ^{99}Tc levels may in fact be underestimated, in situations where the salinity of a seawater sample is below the normal range for that sampling location. Alternatively, fluctuations in ^{99}Tc concentrations in the short term may simply reflect variations in actual discharges from Sellafield (Brown *et al.*, 2002).

2.1.2 ^{137}Cs in Seawater and Sediments

The average seawater concentration of ^{137}Cs off Western Svalbard in May 2001 was 2.33 Bq/m^3 with a range of 2.26 – 2.43 Bq/m^3, of similar magnitude to two measurements of 2.32 ± 0.12 and 2.19 ± 0.1 Bq/m^3 from Kongsfjorden in 2000 and an average value of 3.4 Bq/m^3 from the eastern Barents Sea in 1999 (Rudjord *et al.*, 2001).

These values reflect the continuing trend of decreasing levels of ^{137}Cs in seawater in the Svalbard area from levels of 20 to 30 Bq/m^3 (Hallstadius *et al.*, 1982; Kershaw and Baxter, 1995) in the early to mid 80's, following lower discharge levels of this radionuclide from Sellafield over the last 15 to 20 years. That observed levels of ^{137}Cs are still higher than the decay corrected (2003) global background value for North Atlantic Water of ~1.94 Bq/m^3 (Dahlgaard *et al.*, 1995), points to a continual source of additional input of this radionuclide into the Norwegian Arctic. The additional input of ^{137}Cs into the Svalbard area is probably a combination of the transfer of Chernobyl affected water masses from the Baltic, Norwegian Coastal areas and Barents Sea and continued low level discharges from the European reprocessing plants.

That similar levels of ^{137}Cs were observed within Kongsfjorden and Isfjorden as in the WSC, as was the case for ^{99}Tc, is further confirmation of

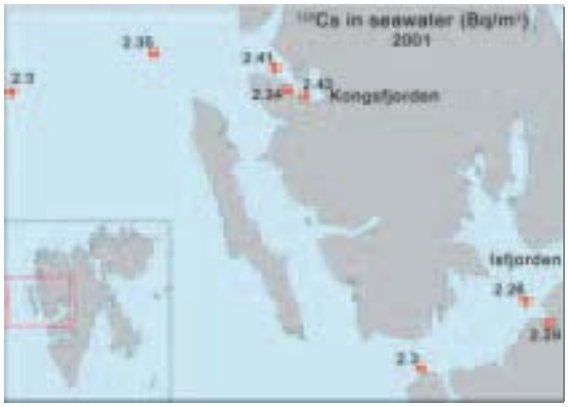

Figure 22. ^{137}Cs activity concentrations of in seawater (Bq/m^3) off Western Svalbard during May 2001.

the importance of the WSC on Western Svalbard's marine environment. As was the case with ^{99}Tc, sub-surface water samples taken in Kongsfjorden and in the WSC showed similar ^{137}Cs levels to corresponding surface samples with similar salinities. The uniformity of the vertical distribution of ^{137}Cs at these locations is a reflection of the relatively conservative behaviour of ^{137}Cs in seawater and a further indication of a deep well mixed layer at the time of sampling.

Activity concentrations in 2001 in surface sediments at two stations at depths of 107 and 246 m in Kongsfjorden were 2.9 ± 0.1 Bq/kg and 8.1 ± 0.3 Bq/kg respectively, while an additional sediment sample taken off the west coast of Svalbard at a depth of 319 m had a ^{137}Cs concentration of 1.1 Bq/kg. These values are comparable to previously reported ranges of ^{137}Cs in surface sediments off West Spitsbergen of 3.6 – 9.3 Bq/kg (Føyn and Sværen, 1997; Heldal *et al.*, 2002), but are up to two orders of magnitude lower than levels in fjords on the Norwegian mainland (Rudjord *et al.*, 2001).

Sediment cores taken at the two stations within Kongsfjorden showed ^{137}Cs sub-surface maxima of 4.4 ± 0.5 and 10.2 ± 0.6 Bq/kg at 8 to 9 cm (107m station) and 5 to 6 cm (246 m station) respectively. The observation of ^{137}Cs sub-surface maxima at these two stations, suggests a reduction over time in the flux of ^{137}Cs to marine sediments in the Svalbard area, which

Figure 23. ^{137}Cs activity concentrations in surface sediments (<5 cm) from and outside Kongsfjorden collected in 2001 (Bq/kg d.w.).

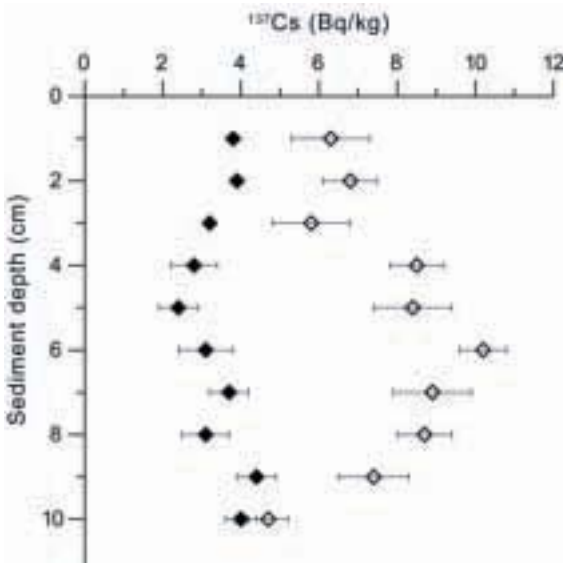

Figure 24. ^{137}Cs activity concentrations (Bq/kg d.w.) in two sediment cores collected from Kongsfjorden in 2001 at water depths of 107 m (black diamonds) and 246 m (white diamonds). The minimum detection activity (MDA) is plotted for samples below the MDA. Variations in MDA are due to variations in sample size and counting times. The activity uncertainty is shown with bars for all samples above the MDA.

would be in accordance with the observation of a reduction in ^{137}Cs levels in Svalbard seawater in recent years.

Alternatively, the sub-surface maxima may simple reflect changes in local sedimentation rates, which may, in addition, explain the observed differences in ^{137}Cs concentrations between the two stations in Kongsfjorden.

2.1.3 ^{238}Pu, $^{239+240}$Pu and ^{241}Am in Seawater

Activity concentrations of ^{238}Pu in 2001 in surface seawater off the west coast of Svalbard ranged from less than 0.3 to 0.7 ± 0.6 mBq/m^3, while average activity concentrations of $^{239+240}$Pu and ^{241}Am were 7 ± 3 mBq/m^3 (range, 5.6 – 8.9 mBq/m^3) and 1.1 ± 0.8 mBq/m^3 (range, 0.6 to 2.4 mBq/m^3) respectively. These values are similar to levels of ^{238}Pu (< 0.4 mBq/m^3), $^{239+240}$Pu (7.2 ± 1.1 mBq/m^3) and ^{241}Am (0.8 ± 0.7 mBq/m^3) in a seawater sample collected in Kongsfjorden in 2000.

Location	Depth (m)	Salinity (‰)	$^{239+240}$Pu (mBq/m^3)	^{241}Am (mBq/m^3)	^{238}Pu/$^{239+240}$Pu	^{241}Am/$^{239+240}$Pu
Kongsfjorden	0	34.4	7.1 ±1.7	0.7 ±0.3	0.1	0.1
	125	34.6	6.1 ±1.5	1.2 ±0.4	-	0.2
WSC	0	34.5	8.9 ±1.4	2.4 ±0.6	0.056	0.27
	100	34.1	12 ±2.0	2.4 ±0.6	0.125	0.2

Table VI. Activity concentrations (mBq/m³) and activity ratios of actinides in surface and sub-surface seawater samples collected from Kongsfjorden and the West Spitsbergen Current (WSC) in 2001. Compare with ^{99}Tc and ^{137}Cs activity concentrations for the identical sampling stations.

For $^{239+240}$Pu, these values are in good agreement with the range of previously reported contemporary levels for the Svalbard area and Barents Sea of 6.5 to 9.9 mBq/m^3 (Kershaw *et al.*, 1999; Rudjord *et al.*, 2001). Furthermore, the 2001 average ^{238}Pu/$^{239+240}$Pu activity ratio of 0.066 (range, 0.056 – 0.108) is comparable to previously observed ^{238}Pu/$^{239+240}$Pu activity ratios in the Svalbard area, reflecting as a source, global fallout rather than the European reprocessing plants.

The activity concentration of ^{241}Am in the WSC (2.4 ± 0.6 mBq/m^3) in 2001 is comparable to the reported average levels in the northern seas of 2.1 ± 0.9 mBq/m^3, which was ascribed to the radioactive decay of fallout ^{241}Pu and not plutonium produced at Sellafield (Hallstadius *et al.*, 1986).

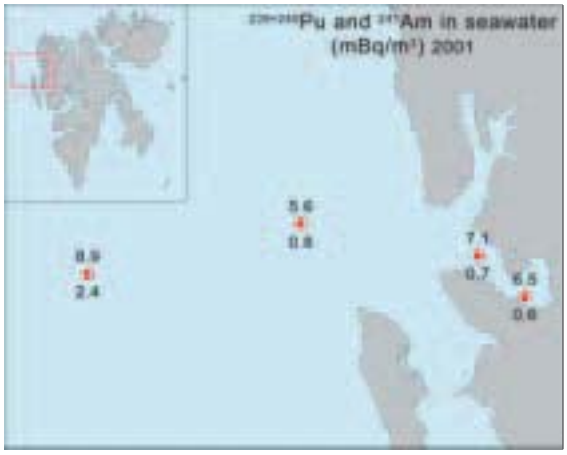

Figure 25. Activity concentrations of $^{239+240}$Pu (above) and ^{241}Am (below) in seawater off the west coast of Svalbard in 2001 (both mBq/m³).

Levels of ^{241}Am appear to decrease on approach to the Svalbard coast, which is possibly due to a greater flux of ^{241}Am to the particulate phase nearer Svalbard through association with elevated levels of terrestrially derived suspended particulate matter. ^{241}Am has been shown to deposit local to a source through a high (higher than plutonium) association with particulate matter (Holm *et al.*, 1980). The 2001 average ^{241}Am/$^{239+240}$Pu activity ratio of 0.15 (range, 0.09 – 0.27) is in close agreement with the average ratio of 0.13 ± 0.04 previously reported for the northern seas (Holm *et al.*, 1983), as indeed was the single activity ratio from 2000 of 0.11.

Activity concentrations of $^{239+240}$Pu and ^{238}Pu/$^{239+240}$Pu activity ratios were similar in sub-surface and surface seawater samples in Kongsfjorden, whereas activity concentrations of ^{241}Am and ^{241}Am/$^{239+240}$Pu activity ratios were higher in the sub-surface sample. The reverse was true for the WSC, where higher $^{239+240}$Pu and ^{238}Pu/$^{239+240}$Pu activity ratios were observed in the sub-surface sample, as has been observed previously in the Barents Sea (Rudjord *et al.*, 2001), but activity concentrations of ^{241}Am and ^{241}Am/$^{239+240}$Pu activity ratios were similar in both sub-surface and surface seawater samples.

2.1.4 ^{99}Tc and ^{137}Cs in Seaweed

Seaweed samples (brown algae only) were collected from Kongsfjorden in 2000, 2001 and 2002 and from Bjørnøya and Hopen in 2001. Data for *Fucus distichus* represents bulked samples

Figure 26. Fucus distichus on the shore in Svalbard.

while data for *Laminaria* spp. and *Alaria esculenta* represent individual specimens.

The average ^{99}Tc concentrations in different species of seaweed collected from Kongsfjorden in 2000, 2001 and 2002 are summarised in Table VII.

For each year, higher ^{99}Tc levels are generally observed in the *F. distichus* compared to *Laminaria* spp. and *A. esculenta*, as has been reported for *Fucus* spp. from mainland Norway (Kolstad and Lind, 2002). It is difficult to draw any firm conclusions with regard to any temporal trends in levels of ^{99}Tc due to the limited number of samples that were taken, although there is some evidence that levels in *F. distichus*. may be decreasing with time, whilst levels in the kelp seaweeds are increasing. The variation in ^{99}Tc concentrations between species during the sampling period are probably related to species specific metabolic processes governing uptake and depuration rates.

For seaweeds collected from Bjørnøya and Hopen in 2001 (Table VIII), higher ^{99}Tc concentrations were seen in *F. distichus* (26.6 – 58.7 Bq/kg) than *Laminaria* spp. (15.1 – 25.1 Bq/kg) as was observed for samples collected from Kongsfjorden. However, levels of ^{99}Tc were in general slightly higher compared to those observed in Kongsfjorden for the same year, especially with regard to *Laminaria* spp. This is probably a reflection of the higher levels of ^{99}Tc in the marine environment around Bjørnøya and Hopen as compared to Kongsfjorden. Levels of ^{99}Tc in *F. distichus* from the Svalbard area from 2000 and 2001 were 7 to 8 fold lower than reported average levels for the same period in *F. vesiculosus* from Hillesøy of 318 and 321 Bq/kg respectively (Kolstad and Lind, 2002). This is in keeping with the observed gradient in ^{99}Tc concentrations in seawater between the Svalbard area (lower ^{99}Tc levels) and mainland coastal sites in Norway (higher ^{99}Tc levels). However, levels of ^{99}Tc in all seaweeds collected from the Svalbard area between 2000 and 2002 were higher than previously reported

Species	2000		2001		2002	
	^{99}Tc (Bq/kg)	CF	^{99}Tc (Bq/kg)	CF	^{99}Tc (Bq/kg)	CF
Laminaria digitata	8.9 ±1.4 (n=2)	42000	7.1± 0.7 (n=1)	20000	28.9 ±2.8 (n=1)	120000
Laminaria saccharina	2.8 ±0.3 (n=1)	11000	4.4 ±0.4 (n=1)	12000	-	-
Fucus distichus	34.3 ±3.3 (n=1)	137000	42.9 ±11.2 (n=5)	156000	28.6 ±4.1 (n=2)	168000
Alaria esculenta	8.2 ±0.8 (n=1)	40000	13.2 ±1.3 (n=1)	37000	-	-

Table VII. Average activity concentrations (Bq/kg d.w.) and concentration factors (CF) of ^{99}Tc in marine algae from Kongsfjorden collected in 2000, 2001 and 2002.

Location	Species	^{137}Cs (Bq/kg)	^{99}Tc (Bq/kg)	CF (^{99}Tc)
Hopen	*Laminaria digitata* (n=1)	0.9 ±0.2	15.1 ±1.5	72000
	Fucus distichus (n=1)	< 3	58.7 ±5.8	280000
Bjørnøya	*Laminaria digitata* (n=3)	0.7 ±0.3	25.1 ±4.6	132000
	Fucus distichus (n=1)	0.8 ±0.2	26.6 ±2.6	140000

Table VIII. Average activity concentrations (Bq/kg d.w.) of ^{137}Cs and ^{99}Tc and concentration factors (CF) of ^{99}Tc in marine algae from Hopen and Bjørnøya collected in 2001.

levels from 1980 and 1981 (Holm *et al.*, 1984).

Average concentration factors (CF) for ^{99}Tc in seaweed samples from Kongsfjorden, Bjørnøya and Hopen were calculated using single or where possible, average seawater levels. Since the inferred CFs represent snapshot and not equilibrium values, care must be taken in their use. However, CFs for ^{99}Tc in *F. distichus* from the Svalbard area during the period 2000 to 2001 of 1.4 x 10^5 to 2.8 x 10^5 are in good agreement with previously reported values of 1.5 x 10^5 and 2.6 x 10^5 for *F. vesiculosus* from Norwegian waters in 1998 and 2001 respectively (Kolstad and Lind, 2002). Likewise, ^{99}Tc CFs for *Laminaria* spp. from the Svalbard area, in the sampling period are, with the exception of two samples, of the same order of magnitude compared to a 2001 value of 1.4 x 10^4 for a *Laminaria* sp. from Lista in Southern Norway (Kolstad and Lind, 2002).

Levels of ^{137}Cs in all species of seaweed sampled from Kongsfjorden in 2001 and 2002 were below the minimum detectable activity (MDA), with the exception of one sample of *F. distichus* collected in 2001 (1.3 ± 0.3 Bq/kg) and one sample of *L. digitata* collected in 2002 (0.7 ± 0.1 Bq/kg). Variations in MDA are due to variations in sample size and counting times.

Levels of ^{137}Cs in all samples of seaweed collected from Hopen and Bjørnøya in 2001 that were above the MDA, were in the range 0.4 – 1.2 Bq/kg. Svalbard seaweed ^{137}Cs values from 2000 to 2002 are comparable to levels of ^{137}Cs in *F. vesiculosus* collected from Hillesøy between 1997 and 1999 of 0.2 – 0.8 Bq/kg (Rudjord *et al.*, 2001). The similarity between levels of ^{137}Cs in seaweeds in the Svalbard area and mainland northern Norway, as opposed to the 7 to 8 fold difference in ^{99}Tc levels, is a reflection of the similar ^{137}Cs seawater concentrations and lower concentration factors for ^{137}Cs in brown algae as compared to concentration factors for ^{99}Tc.

2.2 Levels of Radionuclides in the Terrestrial Environment of Svalbard

This section presents results on the levels of the anthropogenic radionuclides ^{137}Cs, ^{238}Pu, $^{239+240}$Pu and ^{241}Am in soil and plants from the Svalbard terrestrial environment over the period 2000 to 2002. Additional information is presented on the levels of the natural radionuclides ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K in soil and other terrestrial matrices.

Figure 27. Sampling of soils near Kongsfjorden.

These results are discussed in the light of previous radiometric investigations in the Svalbard and Norwegian terrestrial environments and within the context of past and continuing sources of radionuclides to the Svalbard area. Special attention is drawn to the possible role of Arctic specific processes that may influence the behaviour and levels of radionuclides in terrestrial matrices.

2.2.1 Levels of Radionuclides in Soil

The average ^{137}Cs activity concentration in surface soils in the Svalbard area in 2001 and 2002 was 21 Bq/kg. In Spitsbergen, the average ^{137}Cs activity concentration in surface soils was 13 Bq/kg in a range of < 0.9 to 39 ± 1.2 Bq/kg inside Kongsfjorden and 33.6 Bq/kg in a range of < 0.5 to 63 ± 5.6 Bq/kg outside of the Kongsfjorden area.

^{137}Cs activity concentrations in soil samples from Hopen and Bjørnøya were in the range of 5.4 to 42 Bq/kg. These ^{137}Cs concentration ranges are in good agreement with the typical range of values previously reported for Svalbard soils by Negoita (1997, 1999) of < 1.5 to 35.8 Bq/kg. No obvious relationship was observed between levels of ^{137}Cs and latitude, suggesting a relatively uniform and widespread deposition of fallout ^{137}Cs over the entire Svalbard area.

Levels of the natural radionuclides exhibited by soil samples taken across the Svalbard area were in broad agreement with average Norwegian and global concentrations (UNSCEAR, 2000) and

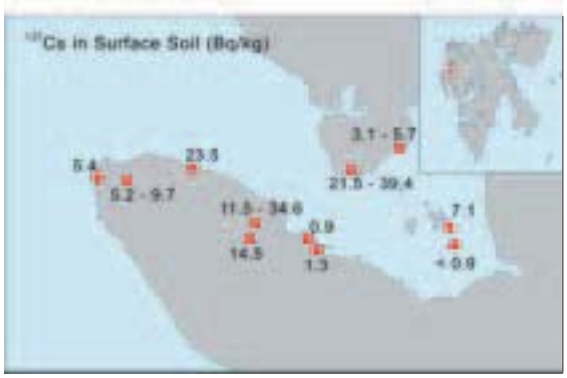

Figure 28. Activity concentrations of ^{137}Cs (Bq/kg d.w.) in surface soil samples collected from Kongsfjorden in 2001 and 2002.

displayed little variation with respect to sample location. Despite available information pertaining to the geology of the area (Hjelle, 1993), the lack of information on permafrost depth at individual sampling sites prevents interpretation of how soil natural radionuclide levels reflect the levels in the underlying lithology.

A series of soil cores taken in the Kongsfjorden area in 2001 showed that levels of the ^{137}Cs diminished rapidly with depth, with ~80% of ^{137}Cs residing within the top 0 – 3cm of soil. That this nuclide appears to be confined to the upper layers of the soil column, with no discernable evidence of any significant downward migration, is in keeping with previous assessments of the vertical distribution of ^{137}Cs in Arctic tundra (e.g. Taylor *et al.*, 1988; Baskaran *et al.*, 1991; Stranberg, 1997).

Location	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Northern Spitsbergen					
Liefdefjorden (n=3)	< 1 - 49	30 - 40	33 - 37	37 - 58	663 – 913
Woodfjorden (n=3)	< 0.5 - 63	26 - 72	28 - 49	36 - 42	536 – 949
Wijdefjorden (n=2)	11 - 59	31 - 46	41 - 52	52 - 57	727 – 818
Southern Spitsbergen					
Longyearbyen	< 0.7	34 ±8	39 ±10	39 ±0.9	706 ±23
Semmeldalen	3.4 ±0.5	47 ±5	70 ±7.6	46 ±2.3	724 ±31

Table IX. Range of activity concentrations of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K in soil (0-3cm) from Spitsbergen, collected outside of Kongsfjorden in 2001 (Bq/kg d.w.).

Location/Depth (cm)	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Hopen 0-5cm	5 ±0.3	36 ±6.3	38 ±5.0	43 ±1.2	665 ±21
Hopen 0-10cm	7 ±3.0	43 ±4.3	47 ±6.0	43 ±1.5	646 ±49
Bjørnøya 0-3cm	42 ±1.3	17 ±4.2	30 ±5.2	14 ±0.6	170 ±20

Table X Activity concentrations of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K (Bq/kg d.w.) in soil (n=1) from Hopen and Bjørnøya collected in 2001.

A ^{137}Cs ground deposition value of 1.6 kBq/m^2 was calculated from an integrated soil/vegetation core taken in Kongsfjorden in 2002. Although undoubtedly composed of contributions from both atmospheric weapon test fallout and the Chernobyl Accident, this value is of a similar magnitude to decay corrected (2003) pre-Chernobyl Svalbard soil deposition values of 0.9 to 1.3 kBq/m^2 (Kjos-Hanssen and Tørresdal, 1981; Hallstadius *et al.*, 1982). Furthermore, a ^{137}Cs value of 1.6 kBq/m^2 is in reasonable agreement with the 2003 decay corrected AMAP (1997) and UNSCEAR (2000) estimated range of 0.4 to 2.1 kBq/m^2.

In comparison, 1998 levels of ^{137}Cs deposition in soil in Northern Norway (Troms and Finmark) were measured at 0.086 to 6.2 kBq/m^2 (JRNEG, 2002). The higher values measured in Northern Norway reflect the greater contribution of fallout ^{137}Cs from the Chernobyl Accident at these lower latitudes.

Unlike ^{137}Cs, there was little variation with soil depth in the levels of the four natural radionuclides. This is most probably due to the processes that normally govern vertical redistribution of these radionuclides being either diminished or retarded by the influence of the climate of the area, the presence of permafrost below the active layer and the relatively low level of biological activity in the soils for much of the year. This hypothesis is supported by the constancy of the ^{226}Ra/^{238}U ratio with depth, the average ratio being 1.4 at the surface of the soil column and 1.0 at depths of 9 to 12 cm.

The primary processes responsible for disruption of ^{226}Ra/^{238}U secular equilibrium in surface soils are leaching, dissolution and precipitation via the action of percolating ground or surface water (Ivanovich and Harmon, 1992). The low ratios exhibited by these soils and the lack of variation with depth appear to indicate either the absence or retardation of processes that are responsible for segregation of decay chain radionuclides in more temperate climates.

Depth (cm)	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
0-3 (n=10)	14 (0.9 – 39)	33 (17 – 68)	41 (21 – 70)	26 (10 – 52)	338 (115 – 564)
3-6 (n=10)	2 (< 0.3 - 5.8)	38 (23 – 66)	40 (17 – 85)	27 (9 – 48)	342 (114 – 577)
6-9 (n=9)	1 (< 0.3 – 2)	42 (25 – 69)	44 (15 – 73)	31 (14 – 51)	387 (123 – 643)
9-12 (n=5)	1 (< 0.4 - 1.4)	40 (17 – 61)	42 (18 – 80)	29 (16 – 47)	350 (135 – 575)

Table XI. Average and range (in brackets) of activity concentrations of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K (Bq/kg d.w.) in soil cores within Kongsfiorden collected in 2001.

Location	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Ny Ålesund (Disused mine)				
Coal waste (n=5)	36 - 62	31 - 69	31 - 61	434 - 1068
Coal	< 7	< 10	3.8 ±0.4	28 ±4
Longyearbyen (Mine 7,active)				
Coal (n=2)	< 18	< 20	< 20	< 22

Table XII. Range of activity concentrations of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K (Bq/kg d.w.) in coal and coal wastes from a disused site in Ny Ålesund and a current production site in Longyearbyen collected in 2001 and 2002.

2.2.2 Coal and Coal wastes

Levels of the four natural radionuclides in coal from a disused site in Ny Ålesund and from a current production site in Longyearbyen (Mine 7) were all low and in broad agreement with the lower end of previously reported ranges for these nuclides in coal (Beck *et al.*, 1980).

Average levels of the four natural radionuclides in coal waste from Ny Ålesund were 54 Bq/kg, ^{238}U, ^{68} Bq/kg ^{226}Ra, 60 Bq/kg ^{232}Th and 914 Bq/kg ^{40}K, significantly higher than levels of these nuclides found in coal samples taken from the environs of Ny Ålesund. Furthermore the average levels of the four natural radionuclides in coal wastes were generally higher than the average levels of these nuclides in soil from the Kongsfjorden area.

There is a need for further investigations of the occurrence of radionuclides in association with these coal wastes, due in part to the lack of a detailed assessment of the nature, amount and spatial distribution of these coal wastes in the Ny Ålesund area. Given that coal burning was probably a source of power during the working life of the mines in Ny Ålesund, it would appear likely that part of the wastes at the site consist of fly ash, which can exhibit enhanced levels of natural radionuclides. Of further potential concern for the local environment is that the wastes affect a number of run-off systems, which ultimately drain into Kongsfjorden. Sampling of sediments near the point of discharge to the fjord would therefore seem expedient in order to determine the flux of certain radionuclides to the fjord itself and the marine environment in general.

2.2.3 Terrestrial Plants

Samples of mosses, lichens and flowering plants were collected from Kongsfjorden in 2001 and 2002 and from Bjørnøya and Hopen in 2001. Due to limitations of sample sizes, the reported activity concentrations for all terrestrial plants represent bulked samples.

In Kongsfjorden, the average ^{137}Cs activity concentration in all mosses sampled (excluding mosses growing near bird colonies) was 157 Bq/kg in a range of 29 0.8 (*Amphidium lapponum*) to 292 70 Bq/kg (*Racomitrium ericoides*),

Figure 29. Vegetation typical to the Svalbard area.

Species	^{137}Cs (Bq/kg)
Moss	
Racomitrium ericoides	292 70
Ditricum flexicaule	216 36
Bryum sp. (n=2)	124 - 166
Amphidium lapponum	29 0.8
Sanionia uncinata	117 4
Lichen	
Cetraria nivalis (n=2)	75 - 140
Flowering plants	
Silene acaulis (n=2)	19 - 47
Cassiope tetragona	109 3
Carex nardina	64 2.4
Dryas octopetala (n=6)	31 - 101

Table XIII. Range of activity concentrations in mosses, lichens and flowering plants from Kongsfjorden collected in 2001 and 2002 (Bq/kg d.w.).

while two samples of the lichen *Cetraria nivalis* showed ^{137}Cs levels of 75 and 140 Bq/kg. In flowering plants sampled in Kongsfjorden, the average ^{137}Cs activity concentration was 54 Bq/kg in a range of 19 0.6 (*Silene acaulis*) to 109 3 Bq/kg (*Cassiope tetragona*). No relationship was observed between levels of ^{137}Cs in any of the mosses, lichens or flowering plants that were sampled and the corresponding levels of ^{137}Cs in soils where these samples were collected.

Furthermore, no relationship was observed between the levels of ^{137}Cs in the terrestrial plants sampled and their spatial distribution within Kongsfjorden.

On Hopen and Bjørnøya the average ^{137}Cs activity concentration in 3 species of moss sampled was 99 Bq/kg in a range of 36 5.2 (*Hygrohypnum polare*) to 245 21 Bq/kg (*Aulocomnium turgidum*), while one sample of the lichen *Cladonia mitis* showed ^{137}Cs levels of 30 1.1 Bq/kg.

Species	^{137}Cs (Bq/kg)
Hopen - moss	
Hygrohypnum polare (n=2)	36 - 90
Aulocomnium turgidum	245 21
Bjørnøya - moss	
Sanionia uncinata (n=2)	49 - 74
Bjørnøya – lichen	
Cladonia mitis	30 1.1

Table XIV. Range of activity concentrations in mosses and lichens from Hopen and Bjørnøya collected in 2001 (Bq/kg d.w.).

Taken as a whole, these 2001 and 2002 ^{137}Cs levels are in reasonable agreement, allowing for decay correction and uncertainties over plant ages, with the limited number of previously reported levels of ^{137}Cs in terrestrial plants on Svalbard from the early 1980's. Additionally, the levels of ^{137}Cs in terrestrial flora on Svalbard are similar to recently reported levels of ^{137}Cs in a range of terrestrial flora collected in Finmark, Northern Norway, but up to 2 fold lower than levels reported for flora collected further south in Troms (JRNEG, 2002).

Figure 30. In general, levels of ^{137}Cs were higher in mosses and lichens than flowering plants in Svalbard.

That the observed levels of ^{137}Cs were generally higher in mosses and lichens than in flowering plants is a reflection of their ability to accumulate this radionuclide due to their large surface area and nutrient uptake characteristics which differ significantly from those of vascular plants.

Further study is needed to assess the ^{137}Cs uptake abilities of floral species in the High Arctic and how the specificities of this environment impact on the concentration of this nuclide (and others) within primary producers of High Arctic terrestrial food chains.

2.3 Localised Enrichment of Radionuclides within Kongsfjorden

Analysis of the spatial distribution of radionuclides within the terrestrial environment of the Kongsfjorden area indicated a number of areas of localised enrichment of radionuclides. These areas were found to be primarily associated with the presence of large amounts of nesting seabirds. As a result of these findings, secondary surveys were implemented to investigate levels of anthropogenic and natural radionuclides in well-drained soils, peat bogs, aggregated faecal and nesting material (AFNM) and plants associated with seabird colonies at Krykkefjellet and on the Brøgger Peninsula, in Kongsfjorden.

At the Brøgger Peninsula site, soil samples taken from below the seabird colony showed elevated levels of both ^{137}Cs and ^{238}U compared to soils remote from seabird colonies within the Svalbard area. In soils near the Krykkefjellet seabird colony, even greater levels of ^{137}Cs, ^{238}U were observed together with elevated levels of ^{226}Ra, significantly above the upper range of these radionuclides in soils observed elsewhere in Svalbard. For ^{137}Cs, levels of this radionuclide in soils below both seabird colonies were up to

Figure 31. Build up of faecal and nesting material below the Krykkefjellet seabird colony.

10 fold higher than average levels for surface soils in Kongsfjorden.

At the Krykkefjellet seabird colony, mounds of AFNM have developed directly below the seabird colony. Levels of ^{137}Cs, in this material were higher than those observed in well drained soils at this site, while levels of ^{238}U and ^{226}Ra were lower, with average activity concentrations of 103 Bq/kg ^{137}Cs, 72 Bq/kg ^{238}U and 64 Bq/kg ^{226}Ra. In addition, the levels of these radionuclides appear to increase with depth within the mounds of AFNM, especially with regard to levels of ^{137}Cs. This suggests that either the input of these radionuclides has reduced over time or that the radionuclides are being leached from surface layers by percolating water. Leaching of nutrients from this material has occurred, resulting in the formation of a peat bog at the base of the AFNM mounds. Levels of ^{137}Cs and ^{238}U in this peat soil were twice as high as were observed in the AFNM and up to 20 fold higher and 8 fold higher respectively, than average levels for typical Svalbard soils.

	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Brøgger -soil					
0-10 cm (n=2)	43 - 54	37 - 80	20 - 62	4 - 9	84 - 148
0-5 cm (n=2)	56 - 146	18 - 75	12 - 24	4 - 10	31 - 78
Krykkefjellet - soil					
0-5 cm (n=3)	48 - 123	134 - 185	133 - 248	23 - 27	319 - 441
Krykkefjellet - AFNM					
0-10 cm (n=8)	89 - 128	58 - 94	35 - 88	17 - 22	356 - 441
0-15 cm	85 ±3	65 ±8	64 ±10	19 ±0.7	394 ±29
15-30 cm	69 ±2	56 ±7	61 ±8	18 ±0.5	406 ±13
30-45 cm	80 ±2	73 ±4	55 ±7	18 ±0.4	451 ±14
45-60 cm	148 ±4	69 ±5	94 ±8	21 ±0.8	534 ±40

Table XV. Range of activity concentrations in soil from below seabird colony on the Brøgger Peninsula and in soil and aggregated faecal and nesting material (AFNM) below the Krykkefjellet seabird colony, Kongsfjorden collected in 2001 and 2002 (Bq/kg d.w.).

Average levels of these radionuclides in the underlying mineral soil (34.2 Bq/kg ^{137}Cs and 31 Bq/kg ^{238}U) were similar to levels for typical Svalbard soils, demonstrating the strong association of these radionuclides with the upper peat layers. Further credence to the leached origin of these radionuclides, is evident from the disruption of the ^{226}Ra/^{238}U secular equilibrium within the peat soil, with an average ^{226}Ra/^{238}U ratio of 0.64, compared to a ratio of ~1 for the underlying mineral layer.

Levels of actinides were also measured in samples of AFNM and soil from the peat bog below the Krykkefjellet seabird colony and compared to background levels in soil on the opposite side of Kongsfjorden. The background activity concentrations of ^{238}Pu, $^{239+240}$Pu and ^{241}Am in Kongsfjorden are in broad agreement with earlier estimated and observed integrated deposition values as previously reported by Hardy *et al.* (1973), Holm *et al.* (1983) and Rissanen *et al.* (2000), with ^{238}Pu/$^{239+240}$Pu and ^{241}Am/$^{239+240}$Pu activity ratios reflecting a global fallout signature. However, activity concentrations of ^{238}Pu, $^{239+240}$Pu and ^{241}Am in AFNM below the Krykkefjellet seabird colony were up to 33 fold higher than background soil levels, while levels in soil from the peat bog at the base of the AFNM mounds were up to 73 fold higher. Despite these elevated levels, the average ^{238}Pu/$^{239+240}$Pu and ^{241}Am/$^{239+240}$Pu activity ratios for these samples are similar to the global fallout ratios for Svalbard of 0.025 and 0.37 respectively (Hardy *et al.*, 1973; Holm *et al.*, 1983). Furthermore the $^{239+240}$Pu/^{137}Cs activity ratio in AFNM and peat soil below the Krykkefjellet seabird colony is similar to the

	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Peat soil (n=20)	167 (65 – 283)	96 (< 12 – 244)	44 (< 4 – 94)	31 (23 – 48)	518 (340 – 784)
Underlying mineral layer (n=28)	34 (0.7 – 92)	31 (20 – 56)	28 (19 – 45)	27 (25 – 31)	622 (562 – 698)

Table XVI. Average and range (in brackets) of activity concentrations in peat soil and underlying mineral layers from a peat bog at the base of AFNM mounds at the Krykkefjellet seabird colony collected in 2002 (Bq/kg d.w.).

	^{238}Pu (Bq/kg)	$^{239+240}$Pu (Bq/kg)	^{241}Am (Bq/kg)	^{238}Pu/$^{239+240}$Pu	^{241}Am/$^{239+240}$Pu	$^{239+240}$Pu/^{137}Cs
Soil (n=1)						
	0.01 ±0.007	0.2 ±0.03	0.05 ±0.02	0.056	0.27	0.035
AFNM (n=2)						
	0.14 ±0.01	3.8 ±0.3	1.6 ±0.1	0.036	0.44 ±0.01	0.028
Peat soil (n=6)						
	0.25 ±0.05	6.9 ±2.1	2.8 ±0.8	0.038 ±0.01	0.42 ±0.03	0.033 ±0.01

Table XVII. Average activity concentrations and activity ratios of actinides in peat soil and AFNM from the Krykkefjellet seabird colony and soil sampled in Kongsfjorden at a site remote from seabird colonies collected in 2002 (Bq/kg d.w.).

background soil $^{239+240}$Pu/^{137}Cs activity ratio in Kongsfjorden and to an estimated global fallout $^{239+240}$Pu/^{137}Cs activity ratio between 70 and 80°N of 0.03 (Hardy *et al.*, 1973; UNSCEAR, 2000).

The inference from these observations at the seabird colonies at Krykkefjellet and Brøgger is that there is either a greater flux of radionuclides into these terrestrial environments and/or that radionuclides are retained to a greater degree at these sites compared to terrestrial environments remote from seabird colonies. The formation of nutrient enriched soils through guano deposition at the base of seabird colonies in Arctic areas is well documented, having being observed both on Spitsbergen and on Novaya Zemlya (Goryachkin *et al.*, 1997), as is the ability of such soils to adsorb and enrich various radionuclides (Owen, 1991; Dowdall and O'Dea, 1999; Schleich *et al.*, 2000). Therefore it is perhaps

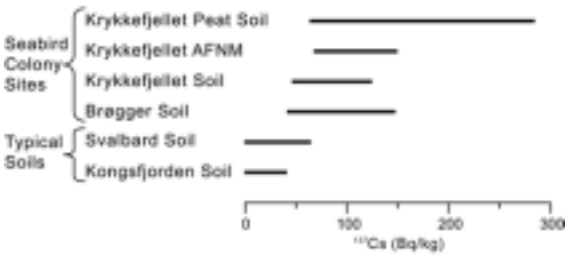

Figure 32. Range of ^{137}Cs activity concentrations in typical soils from Svalbard and matrices associated with seabird colonies.

unsurprising to find higher levels of fallout nuclides and natural radionuclides in these enriched soils as compared to the typically nutrient poor soils found elsewhere in Svalbard, especially if there is a greater flux of radionuclides into these sites. An additional source of radionuclides may arise from the seabirds themselves as Headley (1996) and Godzik (1991) previously ascribed elevated levels of heavy metals in soils near Svalbard seabird colonies, to the transfer of these elements from the marine environment via the consumption of marine organisms by seabirds and the subsequent deposition of guano. As such, it is feasible then to suggest that the elevated levels of radionuclides that were observed at these seabird colonies were introduced in a similar fashion as per heavy metals, i.e. through the deposition of seabird guano. There is no available information as to levels of radionuclides in seabird prey items in the Kongsfjorden environment although some evidence exists that the body burdens of nuclides such as ^{210}Po and ^{210}Pb in decapod shrimp and other pelagic organisms from the Atlantic Ocean can be of the order of kBq/kg (Cherry and Heyraud, 1981).

Levels of ^{137}Cs in mosses and flowering plants that were collected from beneath the Krykkefjellet seabird colony were in general lower than levels of ^{137}Cs observed in other

Species	^{137}Cs (Bq/kg)
Moss	
Bryum sp.	11 3
Dicranoweisia sp.	16 0.5
Sanionia uncinata (n=20)	9 - 76
Flowering plants	
Cochlearia groenlandica	59 2
Saxifraga cernua (n=2)	10 - 16
Deschampsia alpine (n=3)	14 - 20

Table XVIII. Range of activity concentrations in mosses and flowering plants from the Krykkefjellet seabird colony, Kongsfjorden collected in 2001 and 2002 (Bq/kg d.w.).

terrestrial flora taken from different areas on Svalbard. The average ^{137}Cs activity concentration in mosses collected from the base of the AFNM mounds and from the adjacent peat bog was 37.3 Bq/kg, while in flowering plants growing on the AFNM mounds, the average ^{137}Cs activity concentration was 21 Bq/kg.

Despite growing in an area with elevated levels of ^{137}Cs and other radionuclides, these average levels are 4 and 2 fold lower respectively, than average levels in mosses and flowering plants collected at sites in Kongsfjorden that are remote from seabird colonies. This observation may in part be an artefact due to the different species compared, but in addition may reflect, increased plant growth and associated grazing pressures which are thought to be higher at nutrient enriched seabird colony sites (Eurola and Hakala, 1977). Alternatively, the unique nature of the substrates at the bird colonies, with respect to nutrient content, may have some impact on how this nuclide is taken up by vegetation.

2.4 Ambient dose levels

The primary radiation dose delivered to biota on Svalbard is due to the natural gaseous nuclide ^{222}Rn and its daughter nuclides such as ^{210}Pb and ^{210}Po. These radionuclides and a variety of other natural and anthropogenic radionuclides, confer an internal dose to organisms that absorb, ingest and retain them. A further ambient external dose is received due to energy deposited directly in the tissues from gamma radiation emanating from radionuclides within the soil and terrestrial environment. Using methods originally devised by Hultqvist (1956) and further developed by Beck (1972), it is possible to estimate the kerma (kinetic energy released in matter) rate at a specified height above the soil surface by establishing the activity concentrations of gamma radionuclides in soils.

In addition to the naturally occurring gamma emitting radionuclides, the presence of the primary gamma emitting anthropogenic contaminant ^{137}Cs also contributes ambient dose rate. Thus, the kerma rate above soils containing elevated levels of any of these radionuclides is expected to be higher than in areas with 'normal' radionuclide activity concentrations in soils. Calculations were performed to obtain an estimate of the dose rates received at a point 1 m above the surface based on the contributions of the ^{238}U and ^{232}Th series, ^{40}K and ^{137}Cs as determined by gamma ray spectrometry. The dose rate conversion factors used were those derived by Clouvas *et al.* (2000). These factors incorporate 300 different photon emissions for each series and, for the purpose of calculation it was assumed that the density of the soil was 1.3 g/cm^3 and that the nuclides were uniformly distributed to a depth of 1 m. Secular equilibrium was assumed to exist between radionuclides and their progeny within each series. At locations where depth cores were taken, the dose was calculated on the average activities of the nuclides over all depths sampled.

Table XIX shows kerma rate estimates calculated from the data obtained on soil and substrate activity concentrations of ^{137}Cs, ^{238}U, ^{232}Th and ^{40}K. The highest estimated kerma rates occur over the coal waste piles in Ny Ålesund where rates are some 2 to 3 times higher than those calculated for locations further away from Ny Ålesund. The ^{232}Th series and ^{40}K appear to contribute most to the overall kerma rate within Ny Ålesund. The very low contribution of ^{137}Cs to the total kerma rate at all sites from which samples were obtained in this study underscores the relatively small impact on ambient radiation fields that the civil and military nuclear power and weapon programmes have had on within ecosystems in this area of the Arctic. However the ambient kerma rate is affected quite strongly by climatic conditions and can be expected to be lower during the winter months when attenuation by soil is greater due to the presence of snow and ice.

Site	Total	^{137}Cs (%)	^{238}U (%)	^{232}Th (%)	^{40}K (%)
1	335	3	35	32	30
2	448	1	34	34	31
3	497	1	27	35	37
4	262	1	47	25	27
5	255	1	42	28	29
6	166	2	48	27	23
7	353	2	23	33	42
8	264	1	29	35	35
9	315	1	22	30	47
10	439	3	26	34	37
11*	484	0	27	32	41
12*	590	0	22	39	39
13*	506	0	26	35	39
14*	770	0	27	36	37
15*	490	0	45	46	9

*Table XIX. Total kerma rate (Gy/a) and percentage contributions from ^{137}Cs, ^{238}U, ^{232}Th and ^{40}K activity concentrations in soil or substrate at different locations in Kongsfjorden. Where depth–wise samples were taken, the kerma rate was calculated from activity concentrations averaged over the sample depth. * denotes samples taken near mines or waste piles.*

3. Conclusions

Results of radiometric monitoring on the Svalbard archipelago indicate that levels of anthropogenic radioactive contamination are generally low, as per the wider Arctic region.

This study has shown that the Svalbard marine environment exhibits levels of Pu isotopes, ^{137}Cs and ^{99}Tc. Pu isotope activity ratios in seawater reflect global fallout as source, while declining seawater levels of ^{137}Cs, are probably maintained above global fallout levels, by the oceanic import of European coastal run-off containing ^{137}Cs from Chernobyl. Levels of ^{99}Tc in Svalbard seawater have increased since the early 1990's, in association with increased discharges from the Enhanced Actinide Removal Plant at Sellafield (UK), but are lower than levels reported from coastal waters surrounding the Norwegian Mainland. A similar trend is observed in the levels of ^{99}Tc in seaweed from Svalbard and the Norwegian Mainland, but contemporary levels in seaweed from Svalbard are higher than the levels in the early 1980's.

The terrestrial environment is primarily contaminated with Pu isotopes and ^{137}Cs from the historical atmospheric nuclear weapon testing programmes and to a lesser extent, ^{137}Cs from the Chernobyl accident in 1986. Typically, levels of ^{137}Cs are higher in certain Arctic flora (e.g. mosses and lichens) than in the surrounding soils.

Certain aspects of this High Arctic environment are of some concern however and require further investigation. Of particular note is the occurrence of localised enrichment of radionuclides in areas associated with high nutrient levels and concomitant productivity. Further study is required to assess the implication of such enrichments on radionuclide levels in local vegetation and any subsequent impacts on consumers of this vegetation. Attention is also required with respect to characterisation of High Arctic specific processes that may result in either the transfer of anthropogenic radioactive contaminants within and between ecosystem components in this region or that may result in elevated levels of the natural radionuclides that constitute the primary dose commitment.

Continued monitoring of the Svalbard region is required for a number of reasons. The detection of ^{99}Tc in the Svalbard environment emphasises the regions vulnerability to long-range transport of radionuclides. Only through regular monitoring can information be provided on trends in radionuclide levels and the long-term consequences of existing and future sources of radioactive contaminants to the Svalbard region.

Furthermore, impact assessments of anthropogenic contaminants are largely achieved by determining the dose delivered by the contaminant. This assessment can only be considered meaningful when conducted in the context of the natural background dose, upon which the contaminant dose is superimposed. Information and data pertaining to the background dose is somewhat lacking for High Arctic environments. This situation requires greater attention, if effective radiological protection is to be provided following any significant rise in radioactive contamination levels in the High Arctic.

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APPENDIX I

This Appendix serves to detail sampling and analytical procedures used for the monitoring of the Arctic terrestrial and marine environments as documented in this report. It should be noted that all sampling was performed with the view of reducing any impact on the environment of Svalbard itself, with the removal of the minimum amount of material as was required to conduct radiometric analysis in a practicable manner.

Sampling – Terrestrial

Sampling of soils was conducted according to the type of soil encountered, the climate and conditions in the field. Soil samples were taken primarily by digging a pit to a depth at which the soil was frozen. The pit was typically dug using a field spade (stainless steel) after breaking of the soil using a hammer and chisel where necessary. One wall of this pit was then incrementally sampled using a flat stainless steel trowel. Samples were then placed in a ziplock polythene bag and frozen as soon as possible after sampling. In all cases, effort was taken to return the sampling site to a condition closely resembling, that which existed prior to sample removal.

Figure 33. Soil sampling.

The nature of the soils in the Arctic is such that sampling of defined cross sectional areas is not always possible, due to the presence of stones and the consistency of the soils. Calculation of deposition values is therefore complicated. Typical sample masses were of the order of 500 g to 1000 g.

Vegetation sampling was conducted by collecting the above surface parts of the selected plant species by hand. Care was taken to avoid cross species contamination and to only select full-grown, healthy specimens. In some cases, sampling was conducted using a stainless steel knife or scissors. Samples consisted typically of 100 g to 200 g. Samples were then placed in ziplock polythene bags before being frozen. In all cases, samples were only taken where there would be no significant impact to the particular species and the environment.

Sampling of other terrestrial matrices was conducted in as similar a manner as for soils as was practicable. All samples were bagged in polythene ziplock bags and frozen.

Sampling – Marine

Sampling of seawater was conducted by a number of methods. These included direct abstraction of samples using clean polyethylene buckets into clean or new 50 l polyethylene canisters or the use of Niskin bottles for the collection of samples at depth. Sample sizes are typically 50 l to 100 l for ^{99}Tc and 200 l for ^{137}Cs and Pu isotopes.

Sediment cores and grab samples were obtained using box corers. Individual cores were taken using polyethylene tubes of a nominal diameter of 10 cm. The sediment in these tubes was then extruded and sliced into defined sections. The slices were then transferred to ziplock polythene bags and frozen prior to transfer to the laboratory.

Figure 34. Sediment box corer.

Samples of marine vegetation were taken from rocks in the intertidal zone at low tide, from shallow near shore waters and from greater depths by divers. Samples of individual species, consisting of fronds and stipes, were rinsed in the water they were taken from to remove adhering sand, animals or detritus and then placed in ziplock polyethylene bags before being frozen. Sample sizes were from 100 g to 500 g.

APPENDIX II

Laboratory Preparation of Soil and Vegetation Samples

Terrestrial vegetation samples were first rinsed with distilled water to remove adhering soil or detritus. Soil, terrestrial matrices, marine vegetation and terrestrial vegetation samples were dried at 105 ^{o}C in a fan-assisted oven for a period of 24 hours or to constant mass. All dried samples were homogenised using a stainless steel laboratory blender, sieved through a stainless steel sieve of 2 mm aperture size and packed to defined fill heights in standardised counting geometries of volumes between 14 - 550 ml in both simple cylinder and Marinelli configurations.

Analysis – Gamma

Gamma analysis was conducted on terrestrial samples for the measurement of gamma emitting isotopes including ^{137}Cs, ^{40}K, U and Th isotopes. The detection system used was an electrically cooled p-type coaxial high purity germanium detector constructed of low background materials. Nominal resolution and efficiency of the system was 1.9 keV at 1332 keV and 40 %. The detector was connected to an Inspector 2000 (Canberra) MCA (8k channels) utilising Genie 2000. Spectra were obtained between 50 and 2000 keV for periods between 24 and 72 hours and were corrected for a laboratory background counted for 3 months. The detector was calibrated using internationally traceable standard single isotope solutions for each geometry in an aqueous matrix. Differences between the calibration source and the samples, with respect to both density and composition, were corrected for via the calculation of efficiency correction factors using GamaTool

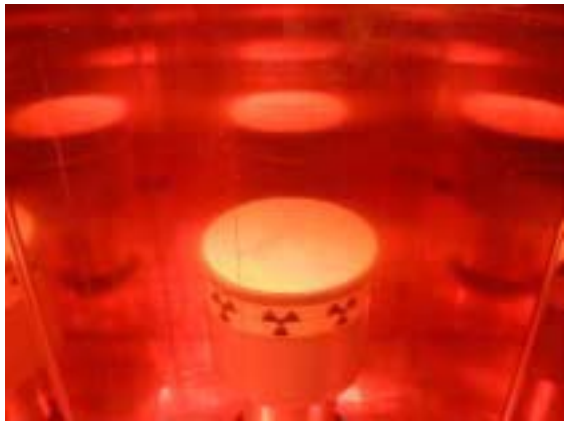

Figure 35. Analysis of a sample on a high resolution gamma ray spectrometer.

(AEA Technology) with attenuation coefficients taken from Hubbel (1987). The system is subject to the normal QA procedures of the laboratory, involving participation in international and national intercomparisons with sample batches containing splits, duplicates, blanks and spikes.

^{137}Cs was determined from its characteristic emission at 661 keV, ^{40}K via 1461 keV. Determination of ^{238}U was achieved using the daughter nuclide ^{234Th}, as ^{238}U does not exhibit appreciable gamma emissions. ^{234}Th, with its short half-life and relative immobility in the biosphere can be assumed to be in secular radioactive equilibrium with its parent ^{238}U. The 63 keV line of ^{234}Th was therefore used to infer ^{238}U activities. ^{226}Ra has a gamma line at 86.2 keV with an emission probability of 3.5 %, however ^{235}U has an interfering line at 185.7 keV which cannot be sufficiently resolved for determination. Although ^{235}U is present at much lower concentrations that ^{238}U, the strength of its emission means that it cannot be ignored. Correction of the 186 keV peak for the ^{235}U contribution was made by assessing the activity of ^{235}U using the ^{238}U activity and a ^{238}U/^{235}U activity ratio of 21.4. Once known, incorporation of the count time, emission probabilities and the efficiency of the detector at 186 keV allowed calculation of the contribution of ^{235}U to the peak and hence calculation of ^{226}Ra activities. In cases where samples had been sealed for a long time (greater than 4 months), comparison of yielded activities using this method agreed well with determined ^{214}Pb activities.

^{137}Cs: Analysis in seawater

The low level of ^{137}Cs in seawater necessitates a pre-concentration step to obtain a detectable analytical signal. This was achieved using sorbent cartridges (cotton) impregnated with a copper hexacyanoferrate, as a Cs selective exchange resin. 200 l of filtered seawater was then pumped at a speed of 5 l per minute over two cartridges in series. These filters were then air dried and

Figure 36. Pre-concentration of ^{137}Cs from seawater on hexacyanoferrate impregnated cartridges prior to gamma analysis.

ashed. The ash from each sorbent was then packed into standard fill height geometries and counted as for solid samples.

^{99}Tc: Analysis

^{99}Tc is a pure beta emitter and its quantitative analysis is conducted by its pre-concentration using ion-exchange, isolation using co-precipitation and solvent extraction and final mounting on steel planchettes using an electrolytic procedure. A schematic of the method is presented in Figure 39. Sample volumes are usually 50 l but sometimes may be as much as 100 l. The recovery of ^{99}Tc from the analytical procedure is determined via the use of

Figure 38. Electroplating of ^{99}Tc samples.

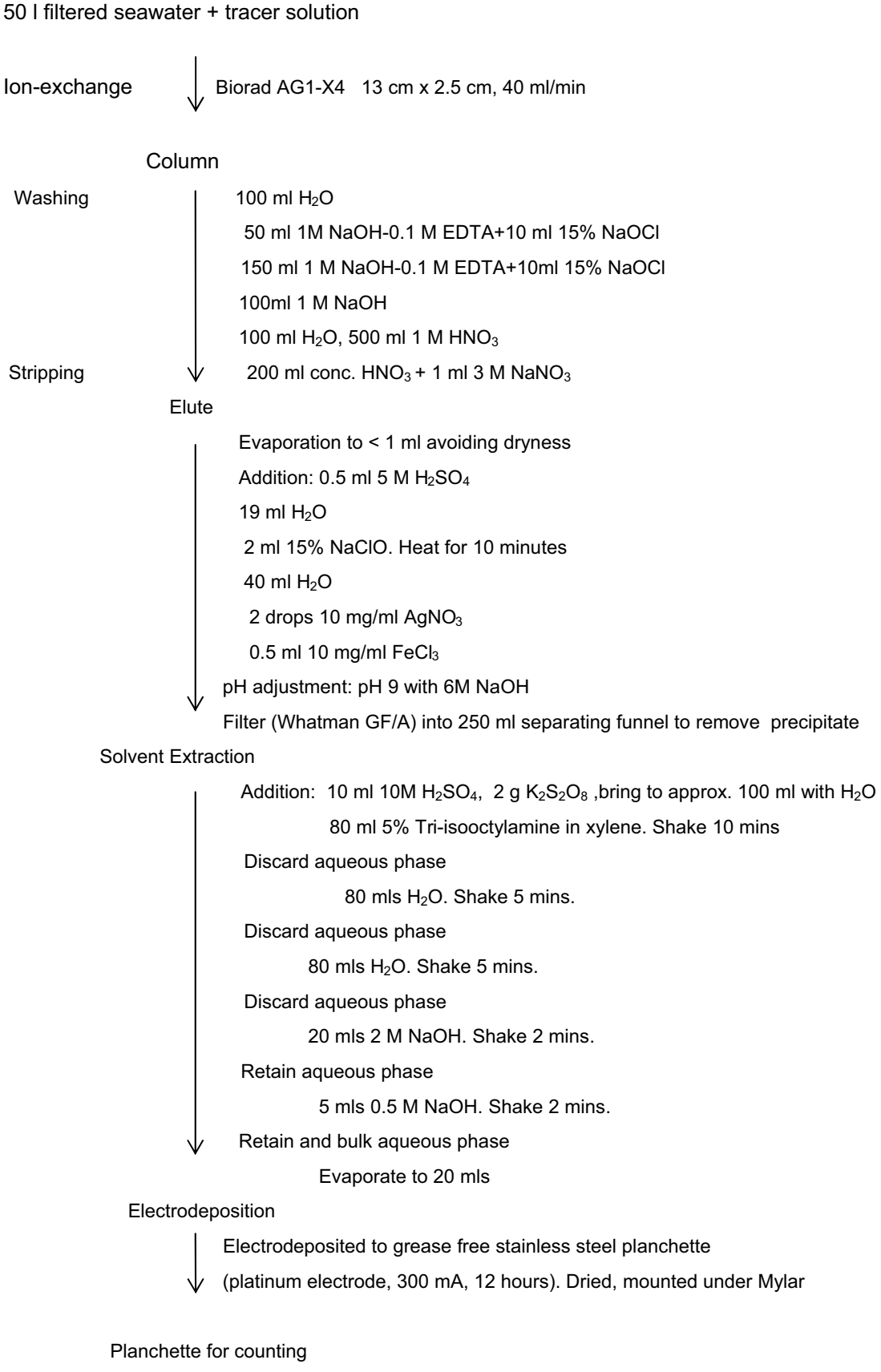

Figure 39. Radiochemical separation of ^{99}Tc from seawater

^{99m}Tc, which is added at the start of the procedure.

All samples are analysed in conjunction with a blank sample. Final analytical signal is determined using a low background anti-coincidence gas flow proportional beta counter (Risø, Denmark). Analysis of seaweed is based on an acid digestion of approximately 10 g of dried seaweed with addition of the ^{99m}Tc yield tracer during the digestion process.

Pu and Am: Analysis

Concentrations of ^{238}Pu, ^{239,240}Pu and ^{241}Am were determined using alpha spectrometry. Samples consisted of 200 l of 0.45 m filtered seawater or 10 g of dried soil. Recovery was determined by using ^{242}Pu and ^{243}Am as yield tracers. Precipitation from water samples was achieved according to the analytical procedure described by Chen *et al.*, (2003). Different radiochemical separation techniques were applied to separate plutonium and americium from other nuclides using solvent extraction with 10 % TIOA/xylene solution and ion-exchange chromatography with a BIO-RAD AG1-X4 (100 – 200 mesh) column.

Purified plutonium and americium fractions were then electroplated onto stainless steel planchettes and counted on semiconductor silicon detectors. As the resolution of such detectors is insufficient to resolve emissions from ^{239}Pu and ^{240}Pu, these nuclides are quoted as a single result.

StrålevernRapport 2004:1

Avvikshåndtering ved norske stråleterapisentre

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