Strålevern Rapport 2005:7

Radionuclides in Marine and Terrestrial Mammals of Svalbard

Norwegian Radiation Protection Authority
Postboks 55
N-1332 Østerås
Norway

Reference:
Gwynn J.P., Andersen M., Fuglei E., Lind B., Dowdall M., Lydersen C., Kovacs K. Radionuclides in Marine and Terrestrial Mammals of Svalbard. StrålevernRapport 2005:7. Østerås: Norwegian Radiation Protection Authority, 2005.

Key words:
Arctic, marine, terrestrial, mammals, radioactivity, Svalbard

Abstract:
This report details the monitoring of radioactivity by the Norwegian Radiation Protection Authority in marine and terrestrial mammals from the Svalbard archipelago. Results indicate that the contamination of both marine and terrestrial mammals with anthropogenic radionuclides is currently low. Comparisons are made to radionuclide contamination in marine and terrestrial mammals from the wider Arctic region and within mainland Norway. Impacts of diet on bioaccumulation of radionuclides and the case for biomagnification of radionuclides within Arctic food chains are discussed.

Referanse:
Gwynn J.P., Andersen M., Fuglei E., Lind B., Dowdall M., Lydersen C., Kovacs K. Radionuclides in Marine and Terrestrial Mammals of Svalbard. StrålevernRapport 2005:7. Østerås: Statens strålevern, 2005. Språk: engelsk.

Emneord:
Arktis, marint, terrestrisk, pattedyr, radioaktivitet, Svalbard

Resymé:
Denne rapporten beskriver resultater fra Strålevernets overvåkning av radioaktiv forurensning i marine og terrestre pattedyr på Svalbard. Resultatene viser at de nåværende konsentrasjonene i både marine og terrestre pattedyr er lave. De målte radioaktive nivåene blir sammenlignet med tilsvarende nivåer i marine og terrestre pattedyr fra andre arktiske områder og fra fastlands-Norge. Videre blir konsekvenser som følge av næringsinntak for bioakkumulasjon av radionuklider samt oppkonsentrering av radionuklider i arktiske næringskjeder diskutert.

Head of project: Tone Bergan.
Approved:

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

60 pages.
Published 2004-06-20.
Printed number 250 (05-06).
Cover design: Lobo Media AS.
Printed by Lobo Media AS, Oslo.

Orders to:
Norwegian Radiation Protection Authority, P.O. Box 55, N-1332 Østerås, Norway.
Telephone +47 67 16 25 00, fax + 47 67 14 74 07.
www.nrpa.no
ISSN 0804-4910

StrålevernRapport 2005:7

Radionuclides in Marine and Terrestrial Mammals of Svalbard

Justin Gwynn[1]

Magnus Andersen[2]

Eva Fuglei[2]

Bjørn Lind[1]

Mark Dowdall[1]

Christian Lydersen[2]

Kit Kovacs[2]

1-Statens strålevern/Norwegian Radiation Protection Authority

2-Norsk Polar Institutet/Norwegian Polar Institute

Norwegian Radiation
Protection Authority
P.O.Box 55
No-1332 Østerås
Norway

Contents

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 concern over bioaccumulation of part of this pollution load and possible trophic transfer through marine and terrestrial food webs. 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 an ecosystem approach for assessment of radiological impacts to both fauna and flora 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 in marine and terrestrial biotic and abiotic compartments. 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 (strontium-90, cesium-137, plutonium-238, plutonium-239+240 and americium-241) and natural (potassium-40 and polonium-210) radionuclides in marine and terrestrial mammals from the Svalbard area.

Svalbard, in comparison with the Norwegian mainland, has received little attention with regard to the levels and behaviour of radionuclides in marine and terrestrial mammals and a search of the available literature reveals only a limited number of previous studies. That the mammals of Svalbard have previously been overlooked with regard to a more intensive radiological investigation is an oversight, particularly in consideration of Svalbard's 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, the mammals of Svalbard have warranted a more thorough radiological investigation in order 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. 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 ˚C in the winter to +6 ˚C in the summer, with maximum and minimum temperatures recorded of -47 ˚C and 21 ˚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.

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 physical oceanography of the seas surrounding Svalbard is dominated by north flowing warmer Atlantic water and south flowing cooler Arctic water. Along the boundary of these water masses, the Polar front, vertical mixing results in the up welling of nutrients creating a zone of high biological production. The fjords and sea areas to the north and east of Svalbard are covered

Figure 2. Geographical location of the Svalbard Archipelago.

Figure 3. The Svalbard Archipelago.

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. The seasonal retreat of sea-ice produces stable nutrient rich surface waters allowing the development of high levels of primary production which in turn supports large numbers of fish, seabirds and mammals.

The terrestrial environment of Svalbard is, in the main, typical of a High Arctic environment. Dominated by ice, two thirds of the landmass is permanently covered by ice and glaciers, with less than 30 % of the ice-free areas covered by vegetation. Mountains display features characteristic of the effects of ice erosion, The islands within the archipelago display distinct ice and water erosional features, forming scree covered terraced plateaus, intersected by deep fjords, of which Isfjorden is the largest.

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 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. Underlying the surface soils, permafrost penetrates down to between 200 and 300 m (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). The combination of the presence of permafrost, the short growing season at these latitudes and the reduced availability of nutrients, limits the amount of primary production within the terrestrial environment. Despite these factors, around 165 species of arctic flora can be found on Svalbard, near coastal areas and on patches of interior tundra. Of particular note, is the increased abundance of vegetation that can be found near seabird colonies, as a result of nutrient enrichment through guano deposition.

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

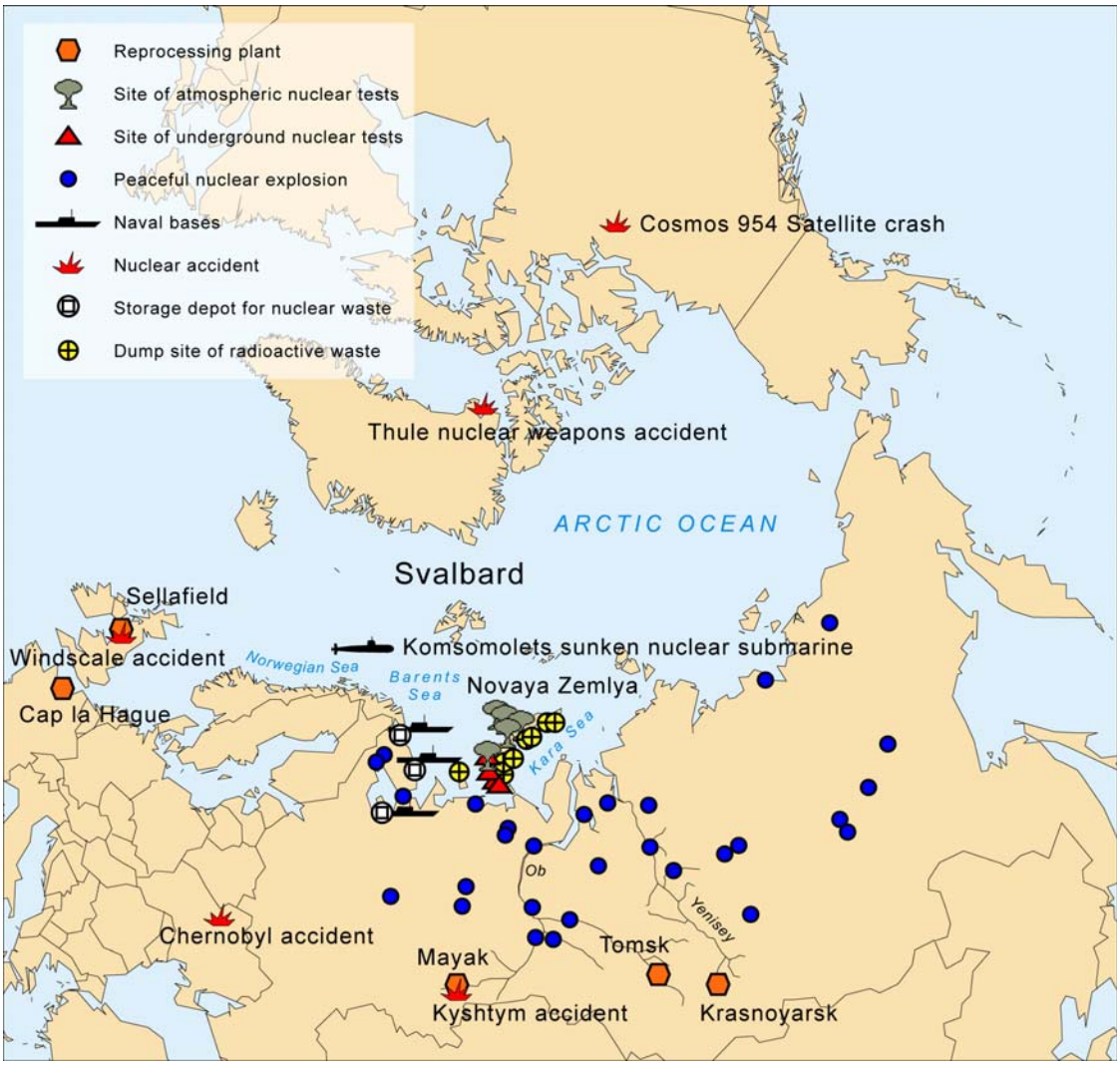

Figure 5. Map of selected potential and actual sources of nuclear contamination to the European Arctic region.

1.2 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.

1.2.1 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.

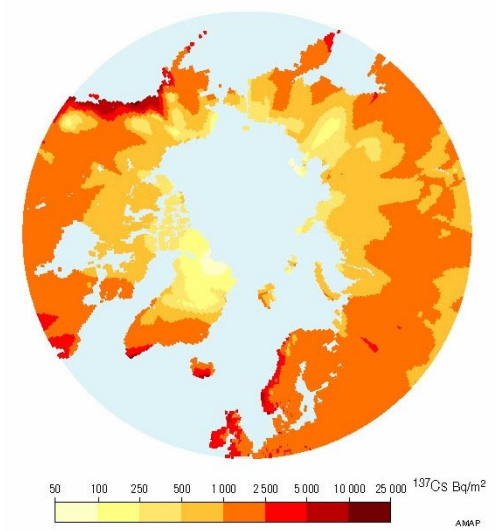

Figure 6. Estimated Arctic ground deposition of ^{137}Cs from nuclear weapon fallout, decay corrected to 1995 (AMAP, 1998).

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.

1.2.2 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) began operations, which while reducing the discharges of actinides, led to the increased discharge of other radionuclides, especially technetium (^{99}Tc).

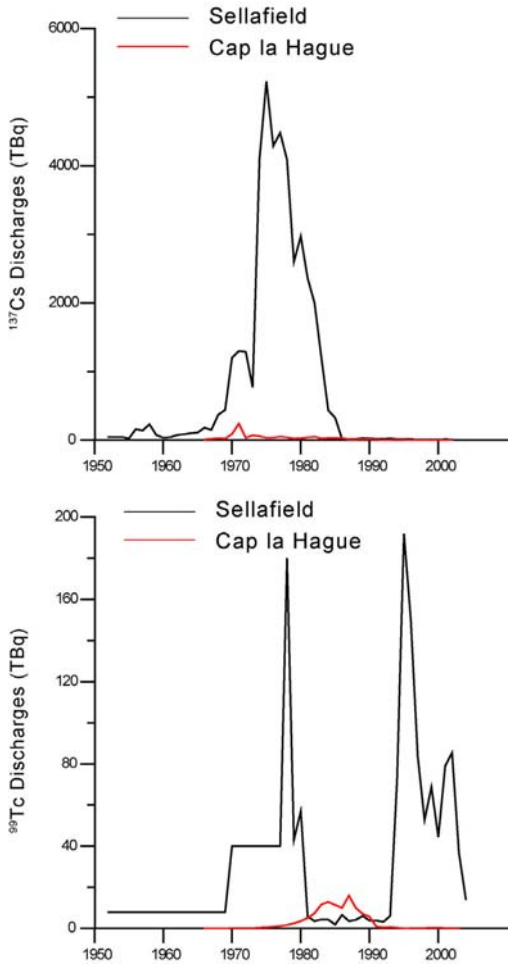

Figure 7. Discharges of ^{99}Tc and ^{137}Cs from Sellafield and Cap la Hague

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).

1.2.3 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).

1.2.4 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.

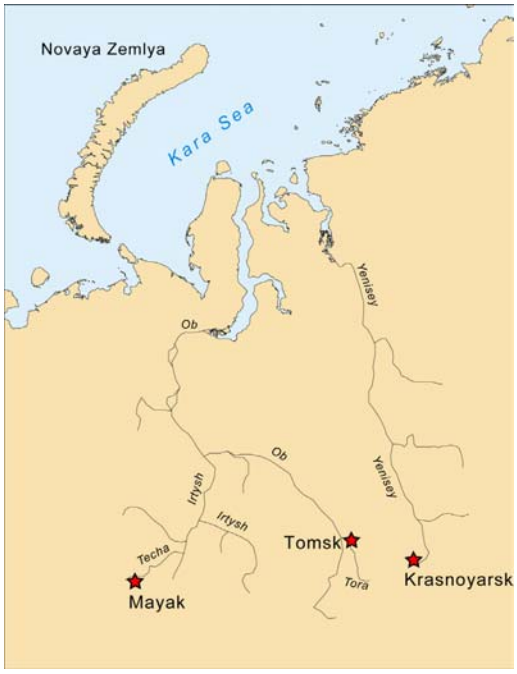

Figure 8. Location of Russian nuclear facilities within the Ob and Yenisey watersheds.

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 Kyshtym 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).

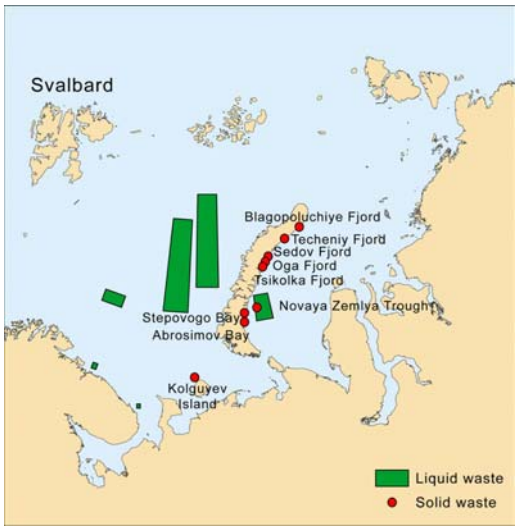

Figure 9. Sites in the Arctic used by the FSU for dumping radioactive waste.

Figure 10. Dumped containers in Stepovogo Bay, Novaya Zemlya (AMAP, 1998).

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.

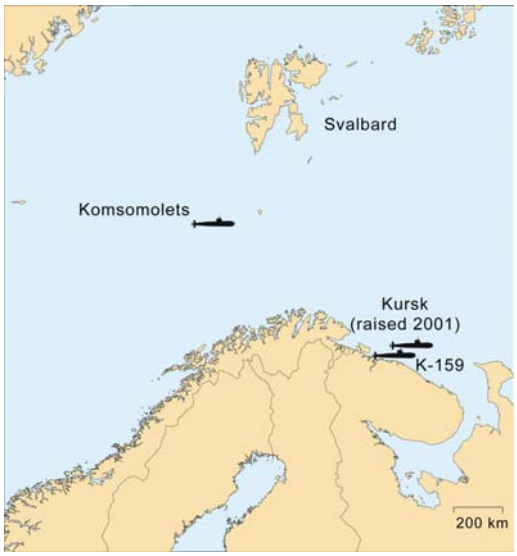

Figure 11. Location of submarine accidents in the Arctic region.

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.

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.

1.2.5 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.

1.2.6 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 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.

Figure 12. Coal mining wastes near Ny Ålesund.

1.3 The Mammals of Svalbard

There is a considerable difference in abundance and diversity of marine and terrestrial mammals in the Svalbard area, which is ultimately linked to the corresponding levels of primary productivity in the marine and terrestrial ecosystems. In the Svalbard marine environment, a combination of diverse water masses, shallow seas, ice-edge effects and nutrient rich upwellings give rise to areas of high primary production, which through diverse, but short food chains, support a range of resident and seasonally migrating marine mammals, including several phocid seals, walruses, whales and polar bears.

In the terrestrial environment, generally considered to be nutrient poor with lower rates of productivity, only the Svalbard reindeer (*Rangifer tarandus platyrhynchus*) and arctic fox (*Alopex lagopus*) are present with established populations. In the last century, attempts were made to introduce arctic hares (*Lepus arcticus*), mountain hares (*L. timidus*) and musk ox (*Ovibos moschatus*) to the archipelago, but no observations of these animals have been made in recent times. The accidental introduction of small rodents has occurred near some of the settlements, with a small population of voles (*Microtus rossiameridionalis*) observed in association with bird colonies in the Isfjorden area, but in general their distribution and numbers are considered to be range restricted and small (Henttonen *et al.*, 2001).

Studied Marine Mammals

1.3.1 Ringed Seal (Phoca hispida)

The ringed seal is the smallest of the seal species, with an adult mean length of about 130 cm and mean adult body mass between 50 and 90 kg (Lydersen and Gjertz, 1987). The most abundant and widely distributed seal species in the northern hemisphere (Reeves, 1998), ringed seals are also the most abundant seal on Svalbard,

Figure 13. Ringed seal. Photo ©Kit & Christian, NP.

with an estimated population size of more than a hundred thousand (Lydersen 1998). These seals occur in the archipelago all year round, but are most numerous during winter to early summer when there is ice in the fjords and bays. Ringed seals are opportunistic feeders, preying on a wide variety of pelagic, benthic and ice associated fauna (Gjertz and Lydersen, 1986; Weslawski *et al.*, 1994).

1.3.2 Bearded Seal (Erignathus barbatus)

The bearded seal is the largest of the northern phocid seals, with adult lengths of about 230 cm and weighing on average 270 to 275 kg (Andersen *et al.*, 1999), with maximum recordings well over 400 kg. Bearded seals are found in areas of relatively shallow water (<100m), avoiding areas of continuous fast-ice and are commonly found alongside leads in ice-covered areas or on drifting floes. The circumpolar population display some degree of seasonal movements mainly related to the distribution of sea-ice. On Svalbard, bearded seals are common all year around, throughout the archipelago, with a population probably in the thousands. These seals are predominantly benthic feeders, preying mainly on fish, molluscs and crustaceans (Hjelset *et al.*, 1999).

Figure 14. Bearded seal.

1.3.3 Hooded Seal (Cystophora cristata)

A pelagic, deep diving seal species, where females can reach 220 cm in length and weigh on average 200 kg, while adult males can reach 250 cm and 300 to 400 kg (Kovacs, 2000). Hooded seals can be found off-shore and in areas of broken drift-ice in the North Atlantic and are commonly found off the south-western coast of Spitsbergen in early and later summer and occasionally north of Spitsbergen and Nordaustlandet.

The global population is thought to be at least half a million animals. Hooded seal diet consists of a variety of deep-water fish species including Greenland halibut various redfish species in addition to squid and benthic invertebrates (Kovacs, 2000)

Figure 15. Hooded seal. Photo ©Kit & Christian, NP.

1.3.4 Polar Bear (Ursus maritimus)

The polar bear is regarded as a marine mammal due to its close association with sea ice. Adult males can have a body mass between 300 and 600 kg, while females are around half this size (Derocher and Wiig, 2002). Polar bears have a circumpolar distribution, with 20 populations currently recognised throughout the Arctic (Lunn *et al.*, 2002). The polar bears of Svalbard belong to the Barents Sea population that includes the area from East Greenland in the west to Franz Josef Land in the east, and is believed to have between 3000 and 6000 animals (Larsen, 1986). In the Svalbard archipelago polar bears are most frequently found in eastern and northern regions and display seasonal movement patterns that are closely related to the dynamic sea ice conditions in the Barents Sea, migrating as far south as Bjørnøya (Mauritzen, 2003). Polar bear diet consists mainly of ringed and bearded seals, although other marine mammals including harp seals, walruses and white whales, as well as scavenging can be important sources of food (Derocher *et al.*, 2002).

Figure 16. Polar bear. Photo ©M. Andersen, NP.

Studied Terrestrial Mammals

1.3.5 Svalbard Reindeer (Rangifer tarandus platyrhynchus)

Several sub-species of reindeer are found throughout the arctic region, with the Svalbard sub-species exclusive to the main islands of the archipelago. The current distribution is centred on Nordenskiold (~4000 animals) and Edgeøya and Barentsøya (~2500 animals) with smaller populations found in north western Spitsbergen, Nordaustlandet and in particular on Brøggerhalvøya (Øritsland, 1985). The diet of Svalbard reindeer consists of whatever vegetation is available, principally grasses in the short summer months and mosses during the winter (e.g. Staaland *et al*., 1983). The ability of these animals to survive Svalbard's harsh climate and limited food resources is due in part to its energy saving sedentary lifestyle, its well-developed ability to utilise the body's own reserves and excellent insulation (e.g. Nilssen *et al*., 1982; Orphin *et al*., 1985). However, Svalbard reindeer must still obtain a major part of their daily energy requirement during the long winter months through grazing (Tyler, 1987).

Figure 17. Svalbard Reindeer.

1.3.6 Arctic Fox (Alopex lagopus)

Found throughout the arctic region and over the entire Svalbard archipelago, arctic foxes are particularly abundant where food is plentiful, such as on the west coast of Spitsbergen where large seabird and waterfowl nesting sites occur and where significant numbers of reindeer occur on Spitsbergen, Edgeøya and Barentsøya. No estimates exist for the total population on the Svalbard archipelago, but a summer population density of 1-1.5 foxes per 10 km^2 has been calculated for the Sassendalen and Adventdalen area (Prestrud, 1992a). Food is available in excess during the summer but is very restricted during winter. In spring and summer diet consists predominantly of ringed seal pups, birds, eggs, young chicks, Svalbard rock ptarmigan (*Lagopus mutus hyperboreus*) and scavenging of reindeer and seal carcasses. During winter, arctic foxes prey on Svalbard rock ptarmigan but must rely on cached food, scavenged carcasses of birds, seals, reindeer or other foxes to survive (Prestrud, 1992b; Frafjord, 1993).

Figure 18. Arctic fox. Photo ©E. Fuglei, NP.

1.4 Vulnerability of Svalbard Marine and Terrestrial Mammals to Radioactive 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 constrain the degree of bioaccumulation of radionuclides within marine and terrestrial fauna and to investigate the possibility of any biomagnification of radionuclides through food chains and food webs. Through biomagnification, the mammals of Svalbard may be vulnerable to radionuclide contamination at even low ambient contamination levels, as they generally represent end members of their respective food chains.

Radionuclide 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

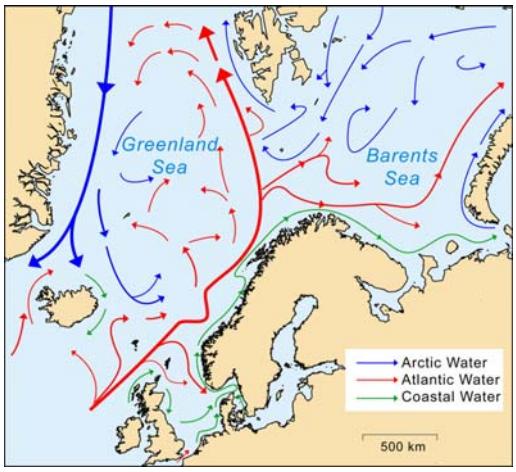

Figure 19. Oceanic circulation in the Northern Seas (adapted from Loeng (1998)).

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 (Kershaw *et al.*, 2003).
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

Figure 20. Sediment laden sea-ice off the west coast of Svalbard. Transport of radionuclides by sea-ice may have implications for the Arctic environment.

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 (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), while ^{137}Cs has been shown to biomagnify through marine food webs (Calmet *et al.*, 1992; Kasamatsu and Ishikawa, 1997; Watson *et al.*, 1999; Heldal *et al.*, 2003). The tendency for Arctic marine food chains to be short and dependent on benthic and sympagic systems, provides an efficient mechanism for the biomagnification of contaminants, while the longevity of top consumers in these food chains allows for the potential accumulation of contaminants over long periods of time. These observations may have important consequences for Arctic marine ecosystems in the event of significant levels of contamination.

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. 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's and 1960'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, due to the higher cation exchange capacities of humic material (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 and are typified by the food chain, lichen/moss – reindeer/caribou – top predator/man. 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, 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.

1.5 Previous Radiological Investigations of Svalbard Mammals

In comparison with abiotic matrices, there have been very few radiological studies of the marine and terrestrial mammals in the Svalbard region with available data generally limited to observations of the anthropogenic radionuclide ^{137}Cs. Marine mammals have received slightly more attention, due in part to the fact that populations of the seals and whales are found in areas along the transport pathways of radionuclide contamination from European Reprocessing Plants but additionally because of the importance of these animals in the diets of Arctic indigenous peoples.

In 1980, average activity concentrations of ^{137}Cs in the muscle of an unknown seal species and two polar bears caught in the Svalbard area were reported at 1.1 $\pm$0.1 and 6.0 $\pm$0.3 Bq/kg (d.w.) respectively, with the polar bear muscle samples showing an average ^{239,240}Pu activity concentration of 0.0044 Bq/kg (d.w.) (Holm *et al.*, 1983). More recently, ^{137}Cs activity concentrations in muscle of greenland, ringed and bearded seals caught off northern Svalbard in 1999, varied between 0.16 and 0.3 Bq/kg (w.w.), with lower levels observed in the liver (Carroll *et al.*, 2002). In minke whales (*Balaenoptera acutorostrata*) caught within the East Svalbard International Whaling Commission (IWC) management unit in 1998, the average ^{137}Cs activity concentration in muscle from fourteen animals was 0.298 $\pm$0.083 Bq/kg (w.w.) (Born *et al.*, 2002).

For the terrestrial mammals, previous studies are restricted to a report of ^{137}Cs activity concentrations of between 0.3 and 2.7 Bq/kg (w.w.) in muscle of Svalbard reindeer in 1980 (Kjos-Hansen and Rennesund, 1981) and in a later study in 1997, where ^{137}Cs activity concentrations were reported between 2.0 and 2.75 Bq/kg (d.w.) in antlers of Svalbard reindeer, with a value of 1.9 $\pm$0.3 Bq/kg (d.w.) from a single bone sample from an arctic fox (Negoita, 1999). In this latter study, ^{40}K activity concentrations were reported between 19.1 and 29.6 Bq/kg (d.w.) in the reindeer antlers, with 86.6 $\pm$18.5 Bq/kg (d.w.) detected in the arctic fox bone sample (Negoita, 1999).

2 Current levels of Radionuclides in Marine and Terrestrial Mammals of Svalbard

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

Marine Mammals

This section presents results on the levels of the anthropogenic radionuclides ^{90}Sr, ^{137}Cs, ^{238}Pu and ^{239,240}Pu and the natural radionuclides ^{40}K and ^{210}Po in marine mammals from the Svalbard area over the period 2000 to 2003.

These results are discussed in the light of previous radiometric investigations of marine mammals within and outside of the Svalbard marine environment, in the context of past and continuing sources of radionuclides to the Svalbard area.

2.1 ^{137}Cs in Marine Mammals

2.1.1 Ringed seals

Activity concentrations of ^{137}Cs in muscle of ringed seals from Spitsbergen in 2003 (Fig. 22) ranged from 0.40 ±0.03 to 0.61 ±0.06 Bq/kg (w.w.) (Table 1). This compares with previously reported ranges of ^{137}Cs muscle activity concentrations in ringed seals of 0.15 ±0.02 to 0.26 ±0.04 Bq/kg (w.w.) from NE Svalbard in 1999 and 1.6 to 2.8 Bq/kg (w.w.) from the Kara Sea in 1995 (Carroll *et al.*, 2002; Yoshitome *et al.*, 2003). Further afield, an average ^{137}Cs muscle activity concentration of 0.21 Bq/kg (w.w.) was reported for both eleven ringed seals from Alaska and Canada over the period 1995 to 1997 and for five ringed seals from Canada in 1999 (Cooper *et al.*, 2000; Yoshitome *et al.*, 2003). There was no correlation observed between ^{137}Cs activity concentrations in muscle and the weight of the individual animals ($P>0.05$).

The transfer of radionuclides in the marine environment can be quantified using concentration factors that are determined on the basis of the activity concentration ratio of a radionuclide between the organism of interest and the surrounding seawater (Equation 1).

Equation 1.

$$\text{Concentration Factor of radionuclide} = \frac{\textit{Concentration of radionuclide in biota}\ (\text{Bq/kg w.w.})}{\textit{Concentration of radionuclide in sea water}\ (\text{Bq/l})}$$

Concentration factors for ^{137}Cs in muscle of ringed seals from 2003 ranged from 200 and 305 (Table 1), using a ^{137}Cs sea water concentration of 2.0 Bq/m^3 (NRPA, unpublished data). This compares with a reported ^{137}Cs concentration factor range of 32 to 130 for ringed seals from

Figure 21. Concentration factors for ^{137}Cs in ringed seals in this study were within previously reported ranges for this species. Photo ©Kit & Christian, NP.

Figure 22. Location of ringed seals sampled in this study.

NE Svalbard in 1999 and 320 to 560 for ringed seals from the Kara Sea in 1995 (Carroll *et al.*, 2002; Yoshitome *et al.*, 2003).

Additionally, ^{137}Cs activity concentrations were determined in organs of three ringed seals from Spitsbergen in 2003 (Table 2). Activity concentrations of ^{137}Cs in all organs were lower than those in muscle from the same animal and in general followed the trend muscle>liver=kidney>brain. A similar trend has been reported for liver and kidneys of ringed seals from NE Svalbard in 1999 and from Alaska and Canada in 1995 to 1997 (Cooper *et al.*, 2000; Carroll *et al.*, 2002). Average concentration factors for ^{137}Cs in liver, kidneys and brain were 175, 179 and 102 respectively, using a ^{137}Cs sea water concentration of 2.0 Bq/m^3 (NRPA, unpublished data).

Year	Location	Sex	Age Class	^{137}Cs Bq/kg w.w.	^{137}Cs CF
2003	Billefjorden	M	Adult	0.48 ±0.06	240
		M	Adult	0.40 ±0.03	200
		M	Adult	0.57 ±0.06	283
		M	Adult	0.43 ±0.06	216
		M	Adult	0.40 ±0.03	201
		M	Adult	0.48 ±0.03	239
		M	Adult	0.60 ±0.06	301
		M	Adult	0.46 ±0.03	228
		M	Adult	0.41 ±0.03	203
		M	Adult	0.61 ±0.06	305
		M	Adult	0.45 ±0.03	224
		M	Adult	0.48 ±0.03	238
		M	Adult	0.54 ±0.03	268
		M	Sub-adult	0.57 ±0.03	286
		F	Adult	0.46 ±0.06	236

Table 1. Activity concentrations (Bq/kg w.w.) and concentration factors (CF) of ^{137}Cs in muscle of ringed seals from Billefjorden, Spitsbergen (2003). Age classes are; sub-adult (3-4 years) and adults (older than 4 years).

Tissue	^{137}Cs (Bq/kg w.w.)		
	Male Adult	Male Adult	Female Adult
Muscle	0.40 ±0.03	0.57 ±0.06	0.48 ±0.06
Liver	0.38 ±0.04	0.40 ±0.05	0.27 ±0.05
Kidney	0.34 ±0.05	0.50 ±0.07	0.23 ±0.04
Brain	0.27 ±0.07	0.13 ±0.05	<0.24

Table 2. Activity concentrations (Bq/kg w.w.) of ^{137}Cs in organs of ringed seals from Billefjorden, Spitsbergen (2003) compared to muscle activity concentrations. Adults are animals older than 4 years.

2.1.2 Bearded seals

Activity concentrations of ^{137}Cs in muscle of two bearded seals from West Spitsbergen (Fig. 24) in 2000 were 0.35 ±0.06 and 0.42 ±0.03 Bq/kg (w.w.), while ^{137}Cs muscle activity concentrations in the same species from the same location in 2002, ranged from 0.06 to 0.23 Bq/kg (w.w.) (Table 3). This compares with previously reported ^{137}Cs muscle activity concentration of 0.26 ±0.02 Bq/kg (w.w.) for a bearded seal from NE Svalbard in 1999 (Carroll *et al.*, 2002). Furthermore, an average ^{137}Cs muscle activity concentration of 0.23 Bq/kg (w.w.), assuming a d.w./w.w. ratio of 0.29, has been reported for four animals from Alaska and Canada over the period 1995 to 1997 (Cooper *et al.*, 2000).

Activity concentrations of ^{137}Cs in muscle of animals from 2000 and 2002 show a significant difference (P=0.006), although this result must be treated with caution due to the limited number of samples compared. Concentration factors for ^{137}Cs in muscle of the two bearded seals from 2000 were 153 and 181, using ^{137}Cs sea water concentration of 2.3 Bq/m^3 (Gäfvert *et al.*, 2003). This compares to a concentration factor range of 30 to 117 for bearded seals from 2002 (Table 3), using ^{137}Cs sea water concentration of 2.0 Bq/m^3 (NRPA, 2004).

Figure 23. Cesium-137 activity concentrations in two bearded seals from 2000 were significantly higher than those observed in animals sampled in 2002.

Figure 24. Location of bearded seals sampled in this study in 2000 and 2002.

These values are similar to a previously reported ^{137}Cs concentration factor range of 55 to 130 (based on a range of sea water concentrations) for a single bearded seal from NE Svalbard in 1999 (Carroll *et al.*, 2002).

Year	Location	Sex	Age Class	^{137}Cs Bq/kg w.w.	^{137}Cs CF
2000	Kongsfjorden	M	Adult	0.35 ±0.06	153
		M	Adult	0.42 ±0.03	181
2002	Kongsfjorden	M	Adult	0.15 ±0.03	76
		M	Adult	0.15 ±0.03	76
		M	Adult	0.19 ±0.03	96
		M	Adult	0.23±0.03	117
		M	Adult	0.06 ±0.03	30

Table 3. Activity concentrations (Bq/kg w.w.) and concentration factors (CF) of ^{137}Cs in muscle of bearded seals from Kongsfjorden, Spitsbergen (2000 and 2002). Adults are animals older than 4 years.

2.1.3 Hooded seals

Activity concentrations of ^{137}Cs in muscle of hooded seals from the North Greenland Sea and North West Svalbard in 2002 (Fig. 26) ranged from 0.21 ±0.03 to 0.42 ±0.03 Bq/kg (w.w.) or were below the limits of detection (Table 4). No previous assessment of ^{137}Cs activity concentrations or indeed any other radionuclide is available in the literature for hooded seals. Amongst the samples in this study with ^{137}Cs activity concentrations above the minimum detectable activity, no significant difference was observed between ^{137}Cs activity concentrations in muscle of male and female animals (P=0.099) and no correlation was observed between ^{137}Cs activity concentrations in muscle and the weight of the individual animals (P>0.05). Concentration factors for ^{137}Cs in muscle from hooded seals, in animals with detectable activity concentrations of ^{137}Cs, ranged from 83 to 167 (Table 4), using ^{137}Cs sea water concentrations of 2.5 Bq/m^3 and 2.1 Bq/m^3 (NRPA, 2004) for hooded seals from the North Greenland Sea and North West Svalbard respectively.

Figure 25. Cesium-137 activity concentrations in hooded seals were generally lower than for other seal species. Photo ©Kit & Christian, NP.

Figure 26. Location of hooded seals sampled in this study.

Year	Location	Sex	Age Class	^{137}Cs Bq/kg w.w.	^{137}Cs CF
2002	N Greenland Sea	M	Adult	<0.4	-
		M	Sub-adult	<1.1	-
		M	Sub-adult	0.36 ±0.06	145
		M	Juvenile	0.32 ±0.06	128
		M	Juvenile	0.42 ±0.03	167
		F	Adult	0.21 ±0.03	83
		F	Adult	0.33 ±0.06	131
		F	Adult	<0.2	-
2002	NW Svalbard	F	Adult	<0.3	-
		F	Sub-adult	0.27 ±0.06	127

Table 4. Activity concentrations (Bq/kg w.w.) and concentration factors (CF) of ^{137}Cs in muscle of hooded seals from the North Greenland Sea and North West Svalbard (2002). Age classes are; juvenile (1-2 years), sub-adult (3-4 years) and adults (older than 4 years).

2.1.4 Polar Bears

Activity concentrations of ^{137}Cs in muscle of polar bears from Svalbard over the period 2000 to 2003 (Fig. 28), ranged from 0.20 ±0.03 to 2.25 ±0.08 Bq/kg (w.w.) (Table 5). This range compares with a previously reported average ^{137}Cs activity concentration in muscle for two polar bears from East Svalbard in 1980 of 1.27 ±0.06 Bq/kg (w.w.) (Holm *et al.*, 1983; IAEA, 2004). More recently and from outside the Svalbard area, an average ^{137}Cs muscle activity concentration of 0.31 ±0.1 Bq/kg (w.w.) was reported for fourteen animals from Alaska and Canada over the period 1995 to 1997 (Cooper *et al.*, 2000). It is difficult to directly compare ^{137}Cs activity concentrations in muscle from animals in different years and locations, due to the lack of information on recent feeding behaviour for individual animals. Analysis of the stomach contents of the polar bears in this study suggests a range of feeding habits, from preying on seals and/or seabirds, scavenging on rubbish piles to fasting. However, information on the stomach contents of a polar bear may not be related to the current ^{137}Cs body burden within the animal. When considering all animals from all years, there was no significant difference in the ^{137}Cs activity concentrations in muscle of male and female animals (P=0.788). Additionally, there was no correlation between ^{137}Cs activity concentrations in muscle and the age or weight of the individual animals, nor the time of year in which the animals were killed (P>0.05).

A basic assumption in the calculation of concentration factors is that the organism is in equilibrium with ambient sea water radionuclide concentrations. With regard to polar bears, this assumption is difficult to maintain, due to possibility of marine and terrestrial feeding behaviours and episodic feeding intervals. However, for two of the polar bears killed in 2003, which are thought to have been feeding on seals at the time they were killed (i.e. displaying a marine feeding behaviour), ^{137}Cs concentration

Figure 27. The wide range in observed ^{137}Cs activity concentrations in muscle of polar bears probably reflects a range of feeding histories. Photo ©M. Andersen, NP.

factors for muscle of 223 and 237 can be determined using a sea water ^{137}Cs concentration of 2.0 Bq/m^3 (NRPA, unpublished data). These values are comparable to the IAEA (2004) recommended concentration factor for ^{137}Cs in polar bears of 1 x 10^2, based on data from the Svalbard region (Holm *et al.*, 1983).

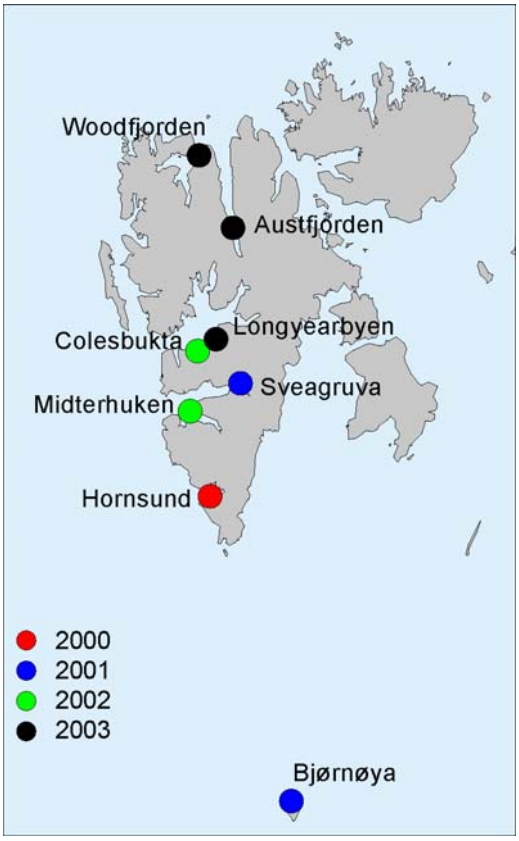

Figure 28. Location of polar bears sampled in this study from 2000 to 2003.

Year	Location	Sex	Age Class	^{137}Cs Bq/kg w.w.
2000	Hornsund	M	Juvenile	0.45 ±0.04
		F	Juvenile	0.44 ±0.05
2001	Svegruva	M	Adult	0.20 ±0.03
	Bjørnøya	F	Sub-adult	0.23 ±0.03
2002	Colesbukta	M	Adult	0.48 ±0.04
		F	Adult	2.25 ±0.09
		F	Juvenile	1.19 ±0.05
	Midterhuken	M	Juvenile	0.32 ±0.05
2003	Austfjorden	M	Adult	1.50 ±0.07
	Longyearbyen	M	Sub-adult	0.47 ±0.05
	Woodfjorden	M	Sub-adult	0.45 ±0.05

Table 5. Activity concentrations of ^{137}Cs (Bq/kg w.w.) in muscle of polar bears from Svalbard (2000 to 2003). Age classes are; juvenile (1-2 years), sub-adult (3-4 years) and adults (older than 4 years).

2.1.5 Inter-Species Comparison

Cs-137 activity concentrations for all marine mammals in this study can be considered to be low, with polar bears displaying a greater range of activity concentrations than seal species. The activity concentrations of ^{137}Cs in marine mammals are principally dependent on the concentrations within prey species and so differences in diet may account for some of the observed variation between species. Estimations of the dietary composition of the different seal species suggests that ringed seals tend to consume a greater percentage of fish in their diet, whereas hooded seals consume a high proportion of squid and bearded seals consume mainly benthic invertebrates (Pauly *et al.*, 1998). Concentration factors for ^{137}Cs recommended by the IAEA (2004) for fish (1×10^2) are one order of magnitude higher than for benthic invertebrates and cephalopods (1×10^1), which may explain the higher average ^{137}Cs activity concentration observed in ringed seals. Indeed, Yoshitome *et al.* (2003) reported that marine mammals feeding on fish showed higher concentrations factors for ^{137}Cs than those feeding on cephalopods. It is worth noting that the observed ^{137}Cs concentration factors for ringed seals in this study (196 to 299) are comparable to reported ranges for other predominantly piscivorous seal species (196 to 670) from the North, Irish and Baltic Seas, although actual body burdens of ^{137}Cs in seals from these areas are up to two orders of magnitude higher than those for ringed seals from Svalbard (Watson *et al.*, 1999; Holm and Leisveik, 2001). Concentration factors for all seal species in this study ranged from 3×10^1 to 3×10^2, which is lower than the recommended IAEA (2004) value for a generic seal species of 4×10^2. In addition to diet, differences in ^{137}Cs assimilation efficiencies between species, prey availability, feeding rates and migration patterns may all impact on the observed ^{137}Cs activity concentration within the muscle of a given marine mammal. The role of feeding rates may be particularly important in the bioaccumulation of ^{137}Cs in polar bears, which can display episodic feeding behaviour. This may account for the wider range of observed ^{137}Cs activity concentrations in polar bears in this study compared to any of the seal species (Fig. 29). In addition, it may be important to consider that ^{137}Cs activity concentrations in polar bears killed due to their proximity to man (i.e. 'problem' animals) may differ to those in animals living on the open sea-ice.

In seals, ^{137}Cs concentration factors, although often similar in magnitude, are typically higher than those for lower trophic levels, which suggests that ^{137}Cs is biomagnified through marine food chains to these consumers. In the case of polar bears, it is difficult to assess any degree of ^{137}Cs biomagnification through the marine food chain to these top predators due to the lack of recent feeding histories on animals prior to their death.

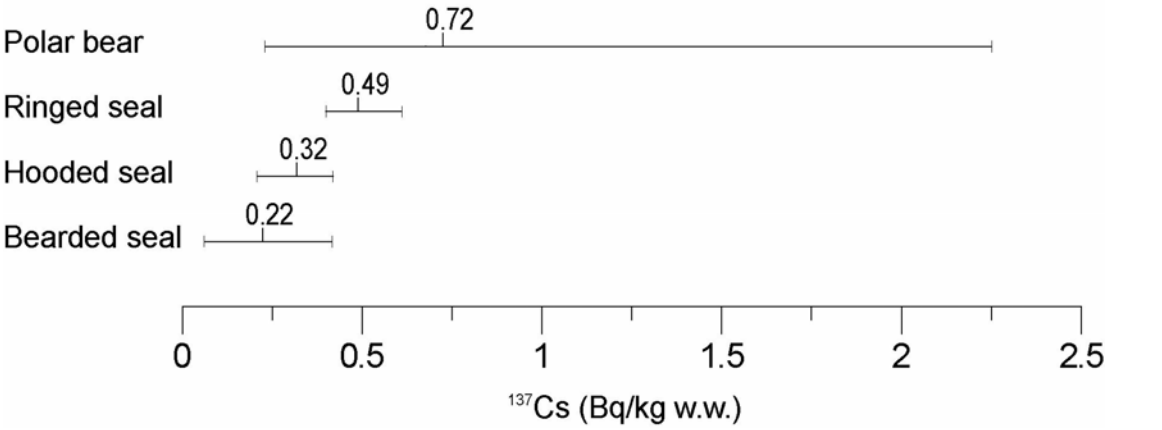

Figure 29. Comparison of range and average ^{137}Cs activity concentrations (Bq/kg w.w.) in marine mammals from the Svalbard area from 2000 to 2003.

Assuming that seals maintain a constant feeding rate, Watson *et al.* (1999) estimated the biological half life of ^{137}Cs to be 28 days for grey (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) from the North and Irish Seas. Using the same approach, Holm and Leisveik (2001) reported a lower estimated biological half life of 20 days for grey seals from the Baltic Sea. These biological half lifes for ^{137}Cs are short when compared to man, e.g. 110 days for adult males (Kendall *et al.*, 1987), which Watson *et al.*, (1999) suggested were due to a higher intake and internal turnover rate of potassium resulting from a marine diet, facilitating a faster turnover of ^{137}Cs.

Equation 2.

Assuming steady state, i.e. ^{137}Cs intake equals ^{137}Cs outtake:

Biological half life of ^{137}Cs ($T_{½}$)

$= \ln 2 \times F \times (W_{mm}/W_f) \times (Cs_{mm}/Cs_f)$

Where:

F = Fraction of the mammal (0.6) containing ^{137}Cs, i.e. muscle (Pentreath and Woodhead, 1988)

W_{mm} = Mass of the mammal

W_f = Mass of fish eaten each day. Assumed to be 5% of W_{mm}

Cs_{mm} = Concentration of ^{137}Cs in the lean tissues

Cs_f = Concentration of ^{137}Cs in fish.

Using the same approach as Watson (Equation 2) and concentration factors for fish species from Heldal *et al.* (2003), an average biological half life of 28 days can be estimated for the ringed seals from this study. However, using a more detailed diet as described by Pauly *et al.* (1998) and including other prey groups, the estimated average biological half life for ^{137}Cs in ringed seals increases to 40 days, illustrating the importance of dietary considerations in ^{137}Cs bioaccumulation.

2.2 ^{90}Sr in Marine Mammals

Activity concentrations of ^{90}Sr in the leg bones of three ringed seals from Spitsbergen in 2003 (Fig. 22) were all below the limits of detection (Table 6). Previously reported activity concentrations of ^{90}Sr in seal species are limited, with only a range of 0.10 to 0.18 Bq/kg (w.w.)

Year	Location	Sex	Age Class	^{90}Sr Bq/kg d.w.
2003	Billefjorden	M	Adult	<0.5
		M	Adult	<0.4
		F	Adult	<0.4

Table 6. Activity concentrations of ^{90}Sr (Bq/kg d.w.) in leg bones of ringed seals from Billefjorden, Spitsbergen (2003). Adults are animals older than 4 years.

reported for a variety of skeletal bones in Greenland seals from the White Sea area in 1996 (Rissanen *et al.*, 1999). At present there are no recommended values for concentration factors for ^{90}Sr in marine mammals. Current levels of ^{90}Sr in the Svalbard marine environment are low, with activity concentrations of ~1 Bq/m^3 reported for 2002 (NRPA, 2004).

2.3 ^{238}Pu and ^{239,240}Pu in Marine Mammals

Activity concentrations of ^{238}Pu and ^{239,240}Pu in muscle of ringed seals, bearded seals and polar bears from Svalbard were all below the limits of detection (Table 7). Reports of Pu isotope activity concentrations in marine mammals above the minimum detactable activity are limited. Holm *et al.* (1983) reported an average ^{239,240}Pu activity concentration of 0.0044 Bq/kg (d.w.) in muscle from two polar bears from Svalbard in 1980. Additionally, Watson *et al.* (1999) reported a range of 0.6 to 3.7 mBq/kg (w.w.) ^{239,240}Pu in liver from various marine mammals in UK waters. Similar findings to this study (i.e. concentrations below the limits of detection) have been previously reported for muscle and other tissues from a variety of seal species and polar bears (Anderson *et al.*, 1990; Rissannen *et al.*, 1999, 2000; Cooper *et al.*, 2000).

Current recomended concentration factors for Pu isotopes for seals (liver) and polar bears (muscle) are 8 x 10^0 and 7 x 10^1 respectively (IAEA, 2004).

	Year	Sex	Age Class	^{238}Pu mBq/kg w.w.	^{239,240}Pu mBq/kg w.w.
Ringed seal					
	2003	M	Adult	<2.8	<0.8
		M	Adult	<0.9	<1.8
		F	Adult	<0.5	<1.5
Bearded seal					
	2000	M	Adult	<0.7	<1.5
		M	Adult	<0.7	<1.6
Polar bear					
	2000	F	Adult	<0.9	<2.2
	2003	M	Adult	<0.6	<1.2

Table 7. Activity concentrations of ^{238}Pu and ^{239,240}Pu (mBq/kg w.w.) in muscle of marine mammals from Svalbard (2000 to 2003). Adults are animals older than 4 years.

2.4 ^{40}K in Marine Mammals

Activity concentrations of ^{40}K in muscle of all marine mammals from the Svalbard area were similar, with average activity concentrations ranging from 94 to 129 Bq/kg (w.w.) (Table 8). This range in values is similar to previously reported ranges in ^{40}K activity concentrations for seals (26 to 179 Bq/kg w.w.), dolphins (77 to 127 Bq/kg w.w.) and porpoises (54 to 126 Bq/kg w.w.) from different geographical areas (Anderson *et al.*, 1990, Calmet *et al.*, 1992; Berrow *et al.*, 1998, Watson *et al.*, 1999; Yoshitome *et al.*, 2003).

	n	Year	^{40}K (Bq/kg w.w.) Average ±SD	Range
Ringed seal	17	2003	106 ±18	91-172
Bearded seal	7	2000/2002	122 ±16	89-138
Hooded seal	10	2002	116 ±9	96-127
Polar bear	11	2000-2003	94 ±27	50-125

Table 8. Average and range of activity concentrations of ^{40}K (Bq/kg w.w.) in muscle of marine mammals from the Svalbard area (2000 to 2003).

Tissue	^{40}K (Bq/kg w.w.)		
	Male Adult	Male Adult	Female Adult
Muscle	102 ±3	108 ±3	99 ±4
Liver	128 ±4	175 ±6	168 ±5
Kidney	156 ±5	192 ±6	60 ±3
Brain	208 ±7	219 ±7	194 ±6

Table 9. Activity concentrations of ^{40}K (Bq/kg w.w.) in organs of three ringed seals from Billefjorden, Spitsbergen (2003) compared to muscle activity concentrations. Adults are animals older than 4 years.

Additionally, activity concentrations of ^{40}K were determined in organs of three ringed seals from Spitsbergen in 2003 (Fig. 22; Table 9). Activity concentrations of ^{40}K in all organs were typically higher than those in muscle from the same animal and in general followed the trend brain>kidney=liver>muscle, which is the reverse for that shown by activity concentrations of ^{137}Cs in these tissues.

2.5 ^{210}Po in Marine Mammals

Activity concentrations of ^{210}Po in muscle of ringed seals from Spitsbergen in 2003 ranged from 12 ±4 to 32 ±4 Bq/kg (w.w.) (Table 10). This range of values is similar to a previously reported ^{210}Po activity concentration range of 16 to 29 Bq/kg (w.w.) for ringed seals from Central West Greenland in 2001 (Dahlgaard *et al.*, 2004) and a range of 4 to 23 Bq/kg (w.w.), assuming a d.w./w.w. ratio of 0.29, for grey seals from the Baltic Sea in 1999 (Holm and Leisveik, 2001). In comparison with other marine organisms, Aarkrog *et al.* (1997) reported global average ^{210}Po activity concentrations of 2.4, 6 and 15 Bq/kg (w.w.) in fish, crustaceans and molluscs, respectively. Concentration factors for ^{210}Po in muscle of ringed seals ranged from 1.2 x 10^4 to 3.2 x 10^4, using an estimated seawater concentration of 1 Bq/m^3 (Aarkrog *et al.*, 1997).

Year	Location	Sex	Age Class	^{210}Po Bq/kg w.w.	^{210}Po CF
2003	Billefjorden	M	Adult	24 ±3	24000
		M	Adult	32 ±4	32000
		M	Adult	24 ±3	24000
		M	Adult	17 ±2	17000
		M	Adult	17 ±3	17000
		M	Adult	17 ±2	17000
		M	Adult	12 ±4	12000
		F	Adult	23 ±3	23000

Table 10. Activity concentrations (Bq/kg w.w.) and concentration factors of ^{210}Po in muscle of ringed seals from Spitsbergen in 2003. Adults are older than 4 years.

This range is a magnitude higher than the ^{210}Po concentration factor recommended for fish, but similar to those for other marine organisms (IAEA, 2004). Using (Equation 2) and a fish ^{210}Po concentration factor of 2000 (IAEA, 2004), an average biological half life of ^{210}Po in ringed seals can be estimated at 86 days. This estimate is 3 fold higher than that calculated for ^{137}Cs in the same species, using the same approach.

Terrestrial Mammals

This section presents results on the levels of the anthropogenic radionuclides ^{90}Sr, ^{137}Cs, ^{238}Pu and ^{239,241}Pu and the natural radionuclides ^{40}K and ^{210}Po in terrestrial mammals from the Svalbard area over the period 2000 to 2003.

These results are discussed in the light of previous radiometric investigations of terrestrial mammals from Svalbard and the Norwegian and European mainland, in the context of past and continuing sources of radionuclides to the Svalbard area.

2.6 ^{137}Cs in Terrestrial Mammals

2.6.1 Svalbard Reindeer

Activity concentrations of ^{137}Cs in muscle of Svalbard reindeer from Spitsbergen in 2003 (Fig. 31) ranged from 0.08 ±0.03 to 1.16 ±0.18 Bq/kg (w.w.), with one sample below the limit of detection (Table 11). This range is similar to a previously reported ^{137}Cs activity concentration range of 0.3 to 2.7 Bq/kg (w.w.) for muscle of Svalbard reindeer from the identical location (Colesdalen) on Spitsbergen in 1980 (Kjos-Hansen and Rennesund, 1980). In comparison, ^{137}Cs activity concentrations in muscle of Norwegian reindeer (*Rangifer tarandus tarandus*) from mainland Northern Norway (Finmark) in 2004 ranged from 56 to 177 Bq/kg (w.w.), up to 3 orders of magnitude higher than those for Svalbard reindeer (Eikelmann *et al.*, submitted). Furthermore, average ^{137}Cs activity concentrations in muscle of Norwegian reindeer from two areas in mainland Central and Southern Norway (Østre Namdal and Vågå) in 2000 to 2002 were even higher ranging from 1400 to 3000 Bq/kg (w.w.) (Skuterud *et al.*, in press a).

Figure 30. The importance of mosses in the diet of Svalbard reindeer may effect the assimilation of ^{137}Cs.

Figure 31. Location of Svalbard reindeer sampled in this study.

Year	Location	Sex	Age	^{137}Cs Bq/kg w.w.	^{137}Cs T_{ag} m^2/kg w.w.
2003	Colesdalen	M	1	0.20 ±0.05	2.0 x 10^{-4}
		M	1	0.37 ±0.06	3.7 x 10^{-4}
		M	<1	0.23 ±0.03	2.2 x 10^{-4}
		M	<1	0.15 ±0.03	1.5 x 10^{-4}
		F	9	0.90 ±0.06	9.0 x 10^{-4}
		F	6	0.08 ±0.01	8.1 x 10^{-5}
		F	5	0.31 ±0.06	3.1 x 10^{-4}
		F	5	0.10 ±0.02	1.0 x 10^{-4}
		F	4	0.31 ±0.02	3.1 x 10^{-4}
		F	4	0.23 ±0.02	2.3 x 10^{-4}
		F	4	0.09 ±0.04	8.9 x 10^{-5}
		F	3	1.16 ±0.18	1.2 x 10^{-3}
		F	3	0.17 ±0.05	1.7 x 10^{-4}
		F	3	0.12 ±0.01	1.2 x 10^{-4}
		F	3	0.09 ±0.01	8.7 x 10^{-5}
		F	3	<0.14	-
		F	<1	0.38 ±0.09	3.8 x 10^{-4}
		F	<1	0.18 ±0.03	1.8 x 10^{-4}
		F	<1	0.08 ±0.03	7.7 x 10^{-5}

Table 11. Activity concentrations (Bq/kg w.w.) and T_{ag} values of ^{137}Cs in muscle of Svalbard reindeer from Colesdalen, Spitsbergen (2003).

When considering all Svalbard reindeer samples in this study, no correlation was observed between ^{137}Cs activity concentrations in muscle and the age of the individual animals ($P>0.05$). However, amongst cow/calf reindeer pairs, it is interesting to note that ^{137}Cs activity concentrations in muscle of the calf were higher than those in the muscle of the cow (Fig. 32). This trend had been observed previously in Norwegian reindeer in summer and autumn (Pedersen *et al.*, 1993; Hove *et al.*, 1999) and was suggested by Hove *et al.* (1999) to occur through a combination of greater fodder intake by calves and a higher bioavailability of ^{137}Cs in milk as compared to the bioavailability of ^{137}Cs in fodder plants.

The transfer of radionuclides in the terrestrial environment can be quantified using aggregated transfer coefficients (T_{ag}) that are determined on the basis of the ratio between the activity concentration of a radionuclide in the organism of interest and the areal deposition of the radionuclide in the soil (Equation 3).

Equation 3.

$$T_{ag}\ (m^2/kg) = \frac{\text{Activity concentration in biota (Bq/kg)}}{\text{Radionuclide deposition in soil (Bq/m}^2\text{)}}$$

Using an estimated 2003 decay corrected ^{137}Cs deposition figure of 1 kBq/m^2 for soil in the area of Colesdalen, based on previously reported ^{137}Cs deposition for the area (Kjos-Hanssen and Tørresdal, 1981) and additional ^{137}Cs deposition from the Chernobyl accident (Pinglot *et al.*, 1994), ^{137}Cs T_{ag} values for Svalbard reindeer from 2003 range from 7.7 x 10^{-5} to 1.2 x 10^{-3} (w.w.) (Table 11). This range of values is similar to a range of 1.9 x 10^{-4} to 1.7 x 10^{-3} (w.w.) calculated for Svalbard reindeer from 1981, based on a ^{137}Cs soil deposition figure of 1.59 kBq/m^2 (Kjos-Hansen and Rennesund, 1980, Kjos-Hanssen and Tørresdal, 1981). In comparison, these T_{ag} values are a magnitude lower than a range of 4 x 10^{-3} to 1.2 x 10^{-2} (w.w.) reported for Norwegian reindeer from mainland Norway in 1998 (JRNEG, 2002). Care must be taken when comparing activity concentrations and T_{ag} values in reindeer, as both values can vary significantly throughout the year due to seasonal changes in diet (Åhman and Nylén, 1998). However, reindeer in this study and in those of Kjos-Hanssen and Rennesund (1980) and JRNEG (2002), were all sampled in either October or November. The difference in

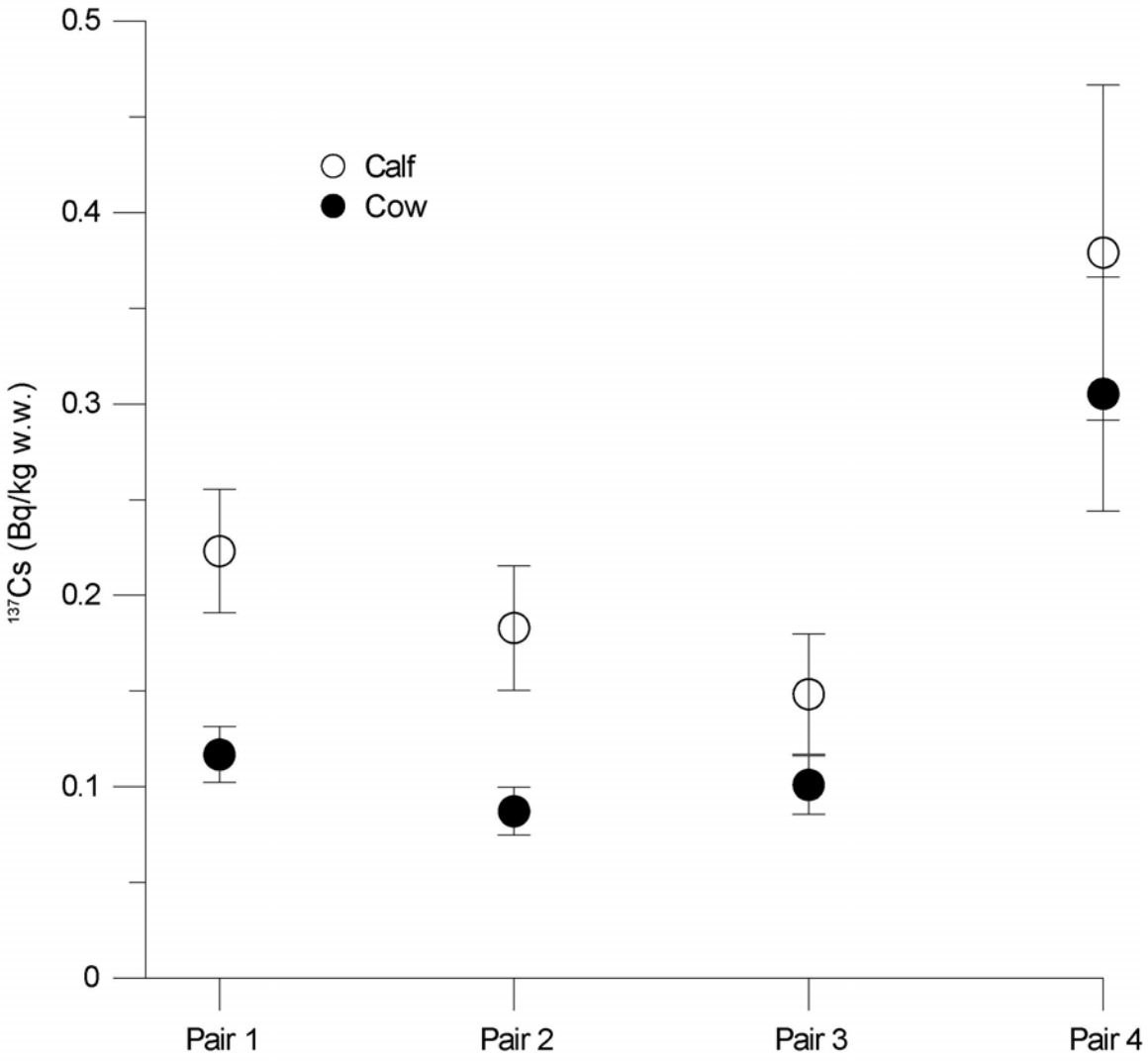

Figure 32. Activity concentrations and uncertainties of ^{137}Cs (Bq/kg w.w.) in muscle from related Svalbard reindeer cow (closed circles) and calf (open circles) pairs from Spitsbergen in 2003.

magnitude between ^{137}Cs activity concentrations observed in Svalbard reindeer and Norwegian reindeer is foremost a likely reflection of differences in ^{137}Cs deposition, but additionally, differences in diet, physiology and metabolism between the sub-species may have an impact (Staaland *et al.*, 1979, 1983; Nilssen *et al.*, 1984) . These latter differences may explain the observed differences in the magnitude of T_{ag} values between the sub-species.

Svalbard reindeer typically feed on grasses during summer and late autumn but additionally consume large quantities of mosses in both summer and winter (Staaland, 1984). Activity concentrations of ^{137}Cs in grasses and mosses from Kongsfjorden, north of Colesdalen, show ranges of 14 to 20 Bq/kg (d.w.) and 11 to 292 Bq/kg (d.w.) respectively (Dowdall *et al.*, 2005). However, uptake of ^{137}Cs by Svalbard reindeer may be lower than in Norwegian reindeer due to a combination of a higher mineral content in the diet of Svalbard reindeer (i.e. greater potassium intake) and the low digestibility of mosses (Staaland *et al.*, 1983). In addition, Svalbard

Tissue	^{137}Cs (Bq/kg w.w.)	
	Female Aged 4	Female Aged <1
Muscle	0.22 ±0.02	0.08 ±0.03
Liver	0.12 ±0.03	<0.12
Kidney	0.20 ±0.07	<0.43
Brain	0.21 ±0.07	<0.28

Table 12. Activity concentrations (Bq/kg w.w.) of ^{137}Cs in tissues of a Svalbard reindeer adult and calf (unrelated) from Colesdalen, Spitsbergen (2003) compared to muscle activity concentrations.

reindeer have been shown to have a larger digestive system and a lower metabolic rate than Norwegian reindeer which may affect ^{137}Cs assimilation rates (Staaland *et al.*, 1979; Nilssen *et al.*, 1984).

Additionally, ^{137}Cs activity concentrations were determined in organs of an unrelated Svalbard reindeer adult and calf from Spitsbergen in 2003 (Fig. 31; Table 12). In the reindeer adult, activity concentrations of ^{137}Cs in kidneys and brain were similar to that in muscle, whereas activity concentrations in liver were lower. In the reindeer calf, ^{137}Cs activity concentrations were all below the limits of detection. In comparison, ^{137}Cs activity concentrations in organs from Finnish reindeer have been reported as being lower than those in muscle, with the exception of kidneys, which showed higher activity concentrations than muscle (Rissanen *et al.*, 1990). Using the same ^{137}Cs deposition figure as for the calculation of muscle T_{ag} values, T_{ag} values for the liver, kidney and brain from the adult reindeer were 1.2 x 10^{-4}, 2.0 x 10^{-4} and 2.1 x 10^{-4} respectively.

2.6.2 Arctic Foxes

Activity concentrations of ^{137}Cs in the muscle of arctic foxes from Spitsbergen trapped between November 1 in 2001 and March 15 in 2002 (Fig. 34) ranged from 0.3 ±0.1 to 10.2 ±0.9 Bq/kg (w.w.), or were below the limit of detection (Table 13). In comparison, an average ^{137}Cs activity concentration of 7546 Bq/kg (w.w.) was reported in muscle of arctic foxes from mainland Norway (Dovrefjell) from 1989 to 1994 (Strand *et al.*, 1998). In the same study, average ^{137}Cs activity concentrations of 1368 and 54 Bq/kg (w.w.) were reported in muscle of arctic foxes from 1993 from the Kola Penninsula and Taimyr respectively (Strand *et al.*, 1998). Strand *et al.* (1998) ascribed the high activity concentrations of ^{137}Cs observed in arctic foxes from Dovrefjell to the practice of scavenging on dead reindeer that had accumulated high concentrations of ^{137}Cs following the Chernobyl Accident. Skogland *et al.* (1991) determined ^{137}Cs activity concentrations in reindeer from Dovrefjell to be in the range of 1000 to 3000 Bq/kg (w.w.), whereas in Svalbard, prey items of arctic foxes such as birds, Svalbard reindeer and ring seals generally have ^{137}Cs activity concentrations of the order of <1 Bq/kg (w.w.) (This study; Kjos-Hanssen and Rennesund, 1980; Holm *et al.*, 1983). When considering all arctic fox samples in this study, no correlation was observed between ^{137}Cs activity concentrations in muscle and the age of the individual animals ($P>0.05$).

Using an identical estimated ^{137}Cs deposition figure as for Svalbard reindeer (1 kBq/m^2), ^{137}Cs T_{ag} values for arctic foxes from Spitsbergen in 2001 and 2002 range from 3.4 x 10^{-4} to 1.0 x 10^{-2} (Table 13). Care must be taken with the use of these T_{ag} values for arctic foxes on Svalbard due to the possibility of a marine component in their diet. In comparison, ^{137}Cs T_{ag} values for arctic foxes from Dovrefjell in the study by Strand *et al.* (1998) can be estimated to range from 4.0 to 7.0 x 10^{-1}, based on a ^{137}Cs deposition range of 11 to 21 kBq/m^2 (Gaare, 1994). In the red fox (*Vulpes vulpes*), Gashak *et al.* (2003) reported a ^{137}Cs T_{ag} range of 1.3 x 10^{-3} to 2.2 x 10^{-2} in animals from the Chernobyl exclusion zone from 1988 to 2000. The difference in the magnitude between arctic fox T_{ag} values in Svalbard and mainland Norway is probably a combination of the relative importance of the marine component to the diet of arctic foxes on

Figure 33. The transfer of ^{137}Cs through the marine food chain to Arctic foxes on Svalbard must be considered, due to their predation on ring seals, seabirds and seabird eggs. Photo ©K. Blom.

Svalbard and the lower uptake of ^{137}Cs by Svalbard reindeer as compared to Norwegian reindeer relative to the amount of ^{137}Cs deposition (i.e. lower T_{ag} values).

Figure 34. Location of Arctic foxes sampled in this study when trapped between November 1 in 2001 and March 15 in 2002.

Year	Location	Sex	Age	^{137}Cs Bq/kg w.w.	^{137}Cs T_{ag} m^2/kg w.w.
2001-2002	Sassenfjorden	M	4	0.5 ±0.1	5.4×10^{-4}
		M	3	3.4 ±0.1	3.4×10^{-3}
		M	3	<0.7	-
		M	3	<0.2	-
		F	5	1.8 ±0.1	1.8×10^{-3}
	Eskerfossen	M	5	0.3 ±0.1	3.4×10^{-4}
	Kapp Wijk	M	6	1.2 ±0.1	1.2×10^{-3}
		M	3-5	10.2 ±0.9	1.0×10^{-2}
		M	>3	<0.7	-
		M	>2	<0.6	-
		M	2	<0.7	-
		M	1-2	2.0 ±0.2	2.0×10^{-3}
		M	1-2	<0.5	-
		F	>3	2.9 ±0.3	2.9×10^{-3}
		F	>2	0.7 ±0.1	7.4×10^{-4}
	Russerkeila	M	2	1.8 ±0.2	1.8×10^{-3}
		F	2	0.9 ±0.1	9.4×10^{-4}

Table 13. Activity concentrations of ^{137}Cs (Bq/kg w.w.) and ^{137}Cs T_{ag} values (m^2/kg w.w.) in muscle of Arctic foxes from various location on Spitsbergen (2001 to 2002).

2.6.3 Inter-Species Comparison

Activity concentrations of ^{137}Cs in arctic foxes were generally higher than in Svalbard reindeer and showed a wider range in values (Fig. 35). The narrow range of activity concentrations of ^{137}Cs in Svalbard reindeer probably reflects the similar temporal and dietary intake of ^{137}Cs by each individual animal, with the observed variations between adults and calves resulting through a combination of greater fodder intake by calves and the higher bioavailability of ^{137}Cs in milk. The arctic foxes in this study were trapped during the winter (November to March)

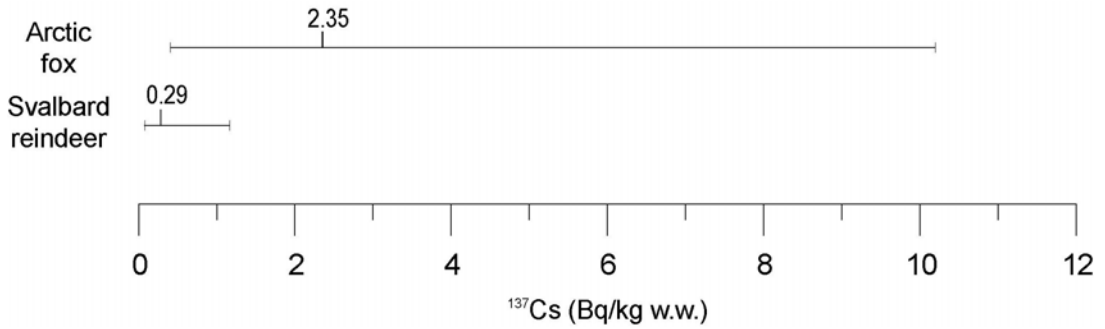

Figure 35. Comparison of range and average ^{137}Cs activity concentrations (Bq/kg w.w.) in arctic foxes (2002) and Svalbard reindeer from Spitsbergen.

when food is generally scarce and available only infrequently in the form of cached prey or opportunistic scavenging. The wide range of ^{137}Cs activity concentrations observed in arctic foxes is thus likely to reflect a range of different feeding histories (i.e. starving to recently fed) prior to the animals capture, in a similar fashion to the polar bears in this study. It has been estimated that arctic foxes can survive without food for up to 25 days, relying solely on its own energy reserves (Fuglei, 2000). Analysis of arctic foxes from Svalbard at the end of summer may yield a smaller range of activity concentrations, when prey, typically seabirds and seabird eggs, is more abundant.

The typically lower ^{137}Cs activity concentrations in Svalbard reindeer compared to those observed in vegetation from Svalbard (Dowdall *et al.*, 2005), would imply that ^{137}Cs is not biomagnified between the trophic levels of primary producer to primary consumer, in the terrestrial environment. Although arctic foxes are considered a terrestrial mammal, their diet in Svalbard features a marine component in the form of ring seals, seabirds and seabird eggs. Despite opportunistic scavenging on Svalbard reindeer and hunting of Svalbard rock ptarmigan, it is difficult to draw a direct comparison with ^{137}Cs activity concentrations observed in arctic foxes and those in Svalbard reindeer and Svalbard vegetation.

2.7 ^{90}Sr in Terrestrial Mammals

Activity concentrations of ^{90}Sr in the leg bones of an unrelated Svalbard reindeer adult and calf from Spitsbergen in 2003 (Fig. 31) were similar at 31 ±4 and 33 ±4 Bq/kg (d.w.) respectively. In comparison, Skuterud *et al.* (in press b) reported average ^{90}Sr activity concentrations in Norwegian reindeer bone from 2000 and 2001 of 840 and 558 Bq/kg (d.w.) from two areas in mainland Central and Southern Norway (Østre Namdal and Vågå), while Klevezal *et al.* (2001) reported an average ^{90}Sr activity concentration of 396 Bq/kg (d.w.) for reindeer bone from Northern Russia in 1994. Activity concentrations of ^{90}Sr in bone typically show no seasonal fluctuations and do not vary significantly between different bones (Staaland *et al.*, 1991; Hognestad and Lie, 1998). Therefore, the difference in magnitude between ^{90}Sr activity concentrations in Svalbard reindeer and Norwegian reindeer is probably a reflection of the latitudinal differences in ^{90}Sr deposition arising from the nuclear weapon test programme and the Chernobyl Accident. According to UNSCEAR (2000) ^{90}Sr deposition from atmospheric nuclear testing was approximately four fold higher in the 50 to 60° N lattitude band compared to 70 to 80° N, with contributions from the Chernobyl Accident comparable to fallout levels in the most contaminated areas in Norway (Bjørnstad *et al.*, 1990).

Year	Location	Sex	Age	^{90}Sr (Bq/kg d.w.)
2003	Colesdalen	F	4	31 ±4
		F	<1	33 ±4

Table 14. Activity concentrations of ^{90}Sr (Bq/kg d.w.) in leg bones of a Svalbard reindeer adult and calf (unrelated) from Colesdalen, Spitsbergen (2003).

2.8 ^{238}Pu and ^{239,240}Pu in Terrestrial Mammals

Activity concentrations of ^{238}Pu and ^{239,240}Pu in muscle of Svalbard reindeer and arctic foxes were all below the limits of detection (Table 15). Previous reports of Pu isotope activity concentrations in reindeer are limited to data from the 1960's and 1970's from mainland Scandanavia, with ^{239,240}Pu muscle activity concentrations ranging from 1.5 to 4.8 mBq/kg (w.w.) and liver activity concentrations up to 100 fold higher (Holm and Persson, 1975; Miettinen, 1975). Pu isotope activity concentrations in Svalbard reindeer might be predicted to be lower than those observed in mainland Scandanavia as the estimated fallout deposition for ^{239,240}Pu in the 80 to 70° N latitude band is roughly 4.5 fold lower than that estimated for the 70 to 60° N latitude band (Hardy *et al.*, 1973).

	Year	Sex	Age	^{238}Pu mBq/kg w.w.	^{239,240}Pu mBq/kg w.w.
Svalbard reindeer					
	2003	F	5	<0.8	<2.5
		F	<1	<0.5	<2.0
Arctic fox					
	2001-2002	M	3	<0.9	<3.2
	2002	M	3-5	<0.9	<0.8

Table 15. Activity concentrations of ^{238}Pu and ^{239,240}Pu (mBq/kg w.w.) in muscle of Svalbard reindeer and Arctic foxes from Spitsbergen (2001 to 2003).

There are no previous assesments of Pu isotopes available in the literature for arctic foxes. However, Beresford *et al.* (2004) estimated a Pu isotope T_{ag} value for arctic foxes based on allometric relationships of 2.2 x 10^{-6} (w.w.). Using an estimated ^{239,240}Pu deposition figure of 13 Bq/m^2 (Hardy *et al.*, 1973) and the above T_{ag} value, we can estimate that ^{239,240}Pu activity concentrations in arctic foxes in Svalbard might be of the order of 0.03 mBq/kg (w.w.).

2.9 ^{40}K in Terrestrial Mammals

Activity concentrations of ^{40}K in muscle of Svalbard reindeer and arctic foxes (Figs. 31 and 34) were similar, with average activity concentrations of 86 and 90 Bq/kg (w.w.) respectively. (Table 16). These values are similar to a range of ^{40}K activity concentrations observed in Norwegian reindeer of 79 to 148 Bq/kg (w.w.) (Eikelmann, pers. comm.; Skuterud, pers. comm.).

Additionally, activity concentrations of ^{40}K were determined in organs of an unrelated Svalbard reindeer adult and calf from Spitsbergen in 2003 (Fig. 31; Table 17). Activity concentrations of ^{40}K in organs from the adult Svalbard reindeer were similar to those in muscle with the exception of the brain, which were 2 fold higher. In the Svalbard reindeer calf, a similar trend was observed for the liver and brain as in the adult Svalbard reindeer, but activity concentrations of ^{40}K in the kidneys were 4 fold higher than those in muscle.

	n	Year	^{40}K (Bq/kg w.w.) Average ±SD	Range
Svalbard reindeer	19	2003	86 ±5	78-94
Arctic fox	17	2001-2002	90 ±14	75-113

Table 16. Average and range of activity concentrations of ^{40}K (Bq/kg w.w.) in muscle of Svalbard reindeer and Arctic foxes from Spitsbergen (2001 to 2003).

Tissue	^{40}K (Bq/kg w.w.)	
	Female Aged 4	Female Aged <1
Muscle	82 ±2	87 ±3
Liver	70 ±2	86 ±3
Kidney	82 ±3	375 ±11
Brain	189 ±6	209 ±7

Table 17. Activity concentrations of ^{40}K (Bq/kg w.w.) in organs of a Svalbard reindeer adult and calf (unrelated) from Spitsbergen (2003).

2.10 ^{210}Po in Terrestrial Mammals

Activity concentrations of ^{210}Po in the muscle of Svalbard reindeer from Spitsbergen in 2003 (Fig. 31) ranged from 2.1 ±0.4 to 7.3 ±0.9 Bq/kg (w.w.) (Table 18). This range is somewhat lower than reported ^{210}Po activity concentration ranges of 4.7 to 18.3 Bq/kg (w.w.) and 3.0 to 16.6 Bq/kg (w.w.) for Norwegian reindeer in 2000 and 2002 from two areas in mainland Norway (Skuterud *et al.* in press b). The uptake of ^{210}Po and it's grandparent ^{210}Pb by reindeer via the diet, is mainly as result of atmospheric deposition of ^{210}Pb onto vegetation surfaces, with lichens containing significantly higher levels of ^{210}Po and ^{210}Pb than higher plants (Holtzman, 1966). Atmospheric deposition of ^{210}Pb, occurring via the decay of the gaseous ^{222}Rn, is generally directly related to the landmass size, snow cover and precipitation (Hill, 1960; El-Daoushy, 1988). In the case of Svalbard, a relatively small landmass, covered mainly by snow and ice and receiving low amounts of precipitation, the deposition of ^{210}Pb should be lower than that received by mainland Norway, while the presence of permafrost may limit the exhalation of autochthonous ^{222}Rn.

When considering all Svalbard reindeer samples in this study, no correlation was observed between ^{210}Po activity concentrations in muscle and the age of the individual animals ($P>0.05$). However, amongst cow/calf reindeer pairs, it is interesting to note that ^{210}Po activity concentrations in muscle of the cow were higher than those in the muscle of the calf (Fig. 36), the opposite relationship to that seen for ^{137}Cs (Fig. 32).

Year	Location	Sex	Age	^{210}Po Bq/kg w.w.
2003	Colesdalen	M	1	4.0 ±0.6
		M	1	6.4 ±0.9
		M	<1	2.1 ±0.4
		M	<1	3.3 ±0.5
		F	9	3.2 ±0.5
		F	6	3.8 ±0.6
		F	4	5.6 ±0.7
		F	4	3.1 ±0.5
		F	4	6.4 ±0.7
		F	4	5.5 ±0.7
		F	4	5.0 ±0.6
		F	3	3.2 ±0.5
		F	3	2.4 ±0.4
		F	3	5.1 ±0.7
		F	3	5.5 ±1.0
		F	3	7.3 ±0.9
		F	<1	4.5 ±0.6
		F	<1	2.2 ±0.4
		F	<1	3.4 ±0.5

Table 18. Activity concentrations of ^{210}Po (Bq/kg w.w.) in muscle of Svalbard reindeer from Colesdalen, Spitsbergen (2003).

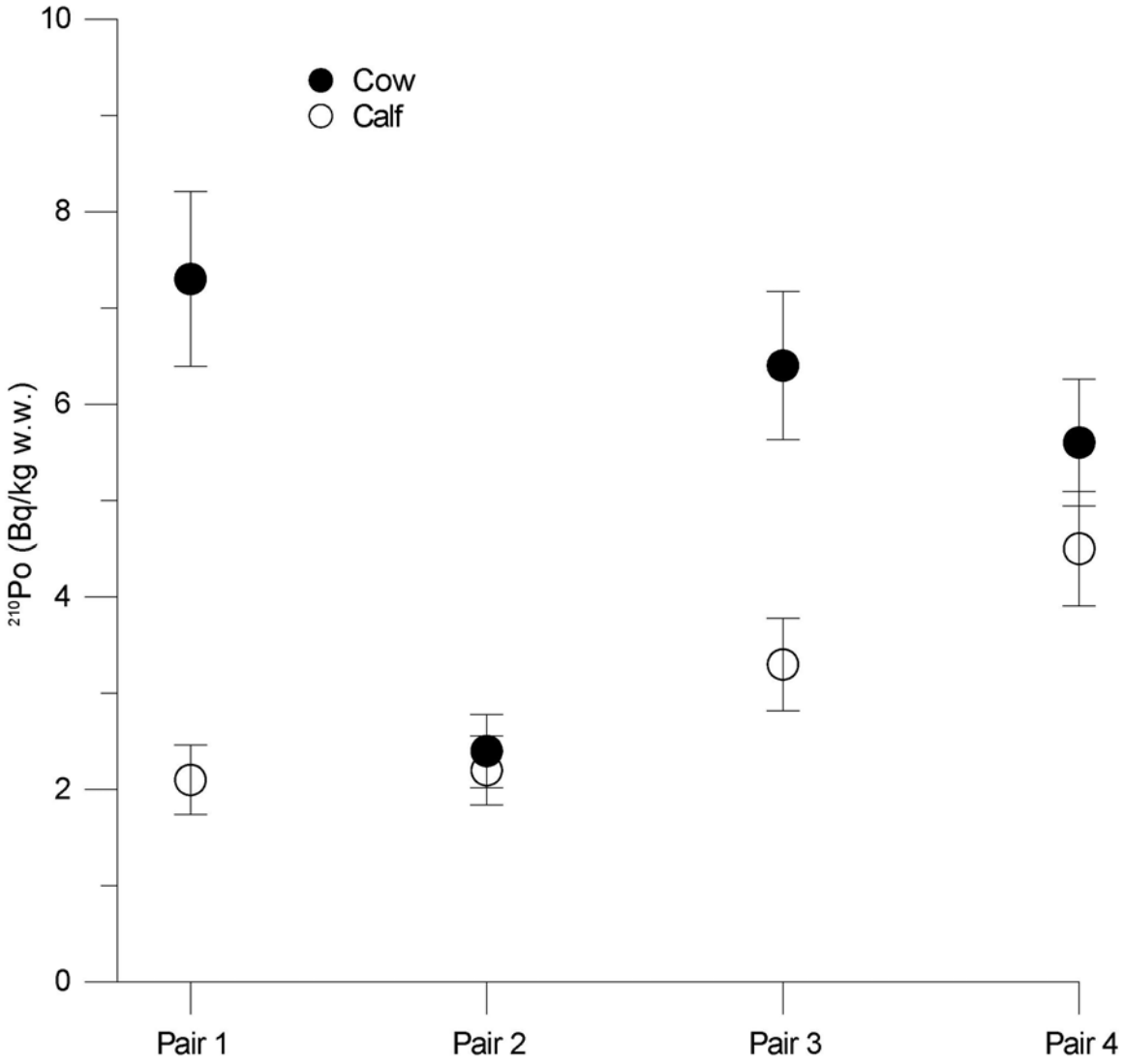

Figure 36. Activity concentrations and uncertainties of ^{210}Po (Bq/kg w.w.) in muscle from related Svalbard reindeer cow (closed circles) and calf (open circles) pairs from Spitsbergen in 2003.

3 Conclusions

Results of radiometric monitoring of Svalbard marine and terrestrial mammals indicates that levels of anthropogenic radioactive contamination in these fauna are generally low. Within marine mammals, activity concentrations of ^{137}Cs were generally higher in polar bears than seal species and amongst the seal species studied, ^{137}Cs activity concentrations were higher in ringed seals than hooded seals and bearded seals. In general, observed ^{137}Cs activity concentrations for specific species were similar to those reported previously for the same species in the wider Arctic region. The range of ^{137}Cs activity concentrations observed were wider in polar bears than seal species which probably reflects a wider range of feeding histories. Differences in ^{137}Cs activity concentrations between seal species is probably a reflection of differences in diet and prey ^{137}Cs concentrations. Based on comparisons with published and generic concentration factors, ^{137}Cs appears to biomagnify through marine food chains up to the trophic level represented by seal species. However, it is difficult to determine any further biomagnification of ^{137}Cs through the marine food chain to polar bears, representing the top predator, due to the lack of feeding histories on individual animals. Activity concentrations of ^{90}Sr in seal bones were low, reflecting the current low levels of this radionuclide in the marine environment around Svalbard. Activity concentrations of Pu isotopes in muscle of all marine mammals were all below the detection limit.

Activity concentrations of the natural radionuclide ^{40}K in all marine mammals were similar to previously reported ranges in seals, dolphins and porpoises from different geographical areas. Activity concentrations of the natural radionuclide ^{210}Po in ringed seals were similar to those reported previously for this species and other seal species. Within terrestrial mammals, activity concentrations of ^{137}Cs were generally higher and wider in range in arctic foxes than Svalbard reindeer, but lower in both species compared to reported ^{137}Cs activity concentrations in the same and related species from the Norwegian mainland. Compared with the Norwegian reindeer, uptake of ^{137}Cs by Svalbard Reindeer in relation to ^{137}Cs activity concentrations in plants appears to be low, with no obvious biomagnification of ^{137}Cs from primary producer to primary consumer. It is difficult to relate ^{137}Cs activity concentrations in arctic foxes in this study to those observed in Svalbard reindeer, due to the lack of feeding histories on individual animals. The wide range of ^{137}Cs activity concentrations observed in arctic foxes probably reflects the limited availability of food and opportunistic scavenging behaviour by Svalbard arctic foxes during the winter, when the animals were sampled.

Lower ^{90}Sr activity concentrations in the bones of Svalbard reindeer compared to Norwegian reindeer from mainland Norway is probably a reflection of the latitudinal differences in ^{90}Sr deposition arising from the nuclear weapon test programme and the Chernobyl Accident. Activity concentrations of Pu isotopes in muscle of Svalbard reindeer and arctic foxes were all below the detection limit.

Activity concentrations of the natural radionuclide ^{40}K in Svalbard reindeer were similar to those in arctic foxes from Svalbard and Norwegian reindeer from mainland Norway. Activity concentrations of the natural radionuclide ^{210}Po in Svalbard reindeer were lower than those reported in Norwegian reindeer from mainland Norway, which may reflect lower fluxes of ^{210}Po and ^{210}Pb into the Svalbard terrestrial environment and hence lower ^{210}Po concentrations in the diet of Svalbard reindeer.

Further study is required to clarify the situation regarding the biomagnification of radionuclides in the Arctic environment and in particular, to representative end members of both marine and terrestrial food webs. There is a need to further constrain the concentrations of radionuclides in

Arctic biota at each trophic level and to consider the importance of prey selection by different species occupying the same trophic level. Continued monitoring of marine and terrestrial mammals and other fauna in the Svalbard region is required to provide information on trends in radionuclide levels and the long-term consequences of existing and future sources of radioactive contaminants.

Acknowledgements

The authors wish to thank Rolf Longvatn from the University Centre on Svalbard (UNIS), the Governor of Svalbard, Tore Haug from the Norwegian College of Fishery Science, Bjørn Krafft from the Norwegian Polar Institute and the crew of the Jan Mayen Meteorological Station for contributing samples to this study.

Personal Communications

Eikelmann IM. Norwegian Radiation Protection Authority, Norway.

Skuterud L. Norwegian Radiation Protection Authority, Norway.

References

Aarkrog, 1993.

Aarkrog A. Radioactivity in polar regions: Main sources. In: Strand P, Holm E, eds. Environmental radioactivity in the Arctic and Antarctic: Proceedings of the International Conference on Environmental Radioactivity in Arctic and Antarctic, Kirkenes, 1993. Østerås: Norwegian Radiation Protection Authority, 1993: 15-34.

Aarkrog *et al*., 1987.

Aarkrog A, Boelskifte S, Duniec S, Hallstadius L, Holm E, Smith JN. Technetium-99 and Caesium-134 as long distance tracers in Arctic waters. *Estuarine Coastal and Shelf Science* 1987; 24: 637-647.

Aarkrog *et al*., 1997.

Aarkrog A, Baxter MS, Bettencourt AO, Bojanowski R, Bologa A, Charmasson S, *et al.* A comparison of doses from ^{137}Cs and ^{210}Po in marine food: a major international study. *Journal of Environmental Radioactivity* 1997; 34 (1): 69-90.

Academy of Science, 1991.

Academy of Science, 1991. Conclusion of the Commission on the estimation of the ecological situation in the Region of the production assoc. "MAYAK", organised by the direction of the Presidium of Academy Science. No. 1140-501// Russia. *Journal of Radiobiology* 1991; 31(3): 436-452.

AMAP, 1993.

AMAP, 1993. The monitoring programme for Arctic Monitoring and Assessment Programme. Oslo: Arctic Monitoring and Assessment Programme, AMAP, 1993.

AMAP, 1997.

AMAP, 1997. Arctic pollution issues: A state of the Arctic environment report. Oslo: Arctic Monitoring and Assessment Programme, AMAP, 1997.

Amundsen *et al*., 2001.

Amundsen I, Lind B, Reistad O, Gussgaard K, Iospje M, Sickel M. The Kursk accident. StålevernRapport 2001:5. Østerås: Norwegian Radiation Protection Authority, 2001.

Andersen *et al*., 1999.

Andersen M, Hjelset AM, Gjertz I, Lydersen C, Gulliksen B. Growth, age at sexual maturity and condition in bearded seals (*Erignathus barbatus*) from Svalbard, Norway. *Polar Biology* 1999; 21: 179-185.

Anderson *et al.* 1990.

Anderson SS, Livens FR, Singleton DL. Radionuclides in grey seals. *Marine Pollution Bulletin* 1990; 21:343-345.

Åhman and Nylén, 1998.

Åhman B, Nylén T. Radiocesium in reindeer in relation to ground deposition, levels in vegetation, season and diet. Progress report to the Swedish Radiation Protection Institute, Reindeer Husbandry Unit, Swedish University of Agricultural Sciences, Uppsala, 1998.

Barrie *et al.*, 1992.

Barrie LA, Gregor D, Hargrave B, Lake R, Muir D, Shearer R, *et al.* Arctic contaminants: Sources, occurrence and pathways. *The Science of the Total Environment* 1992; 122: 1-74.

Baskaran *et al.*, 1991.

Baskaran M, Kelley JJ, Naidu AS, Holleman DF. Environmental ^{137}Cs in subarctic and arctic Alaska following Chernobyl. *Arctic* 1991; 44: 346-350.

Baxter, 1993.

Baxter MS. Environmental radioactivity: a perspective on industrial contributions. *IAEA Bulletin* 1993; 35: 33–38.

Beck *et al.*, 1980.

Beck HL, Gogolak CV, Miller KM, Lowder WM. Perturbations on the natural radiation environment due to the utilisation of coal as an energy source. In: Gesell TF, Lowder WM (eds.). The natural radiation environment III : proceedings of an international symposium held at Houston, Texas 1978. USDOE CONF-780422. Springfield, Va. 1980 : 1521-1558.

Beresford *et al.*, 2004.

Beresford NA, Broadlet MR, Howard BJ, Barnett CL, White PJ. Estimating radionuclide transfer to wild species – data requirements and availability for terrestrial ecosystems. *Journal of Radiological Protection* 2004; 24: A1-A15.

Berrow *et al.*, 1998.

Berrow SD, Long SC, McGarry AT, Pollard D, Rogan E, Lockyer C. Radionuclides (^{137}Cs and ^{40}K) in harbour porpoises *Phocoena phocoena* from British and Irish coastal waters. *Marine Pollution Bulletin* 1998; 36: 569-576.

Bjurman *et al.*, 1990.

Bjurman B, Degeer LE, Vintersved I, Rudjord AL, Ugletveit F, Aaltonen H, *et al.* The detection of radioactive material from a venting underground nuclear explosion. *Journal of Environmental Radioactivity* 1990; 11: 1-14.

Bjørnstad *et al.*, 1990.

Bjørnstad HE, Lien HN, Salbu B. Bestemmelse av strontium-90 i jord og vegetasjon. In: Garmo TH, Gunnerød TB, eds. Radioaktivt nedfall fra Tsjenobyl-ulykken. Følger for norsk landbruk, naturmiljø og matforsyning, Ås, Norway.

Born *et al.* 2002.

Born EW, Dahlgaard H, Riget FF, Dietz R, Øien N, Haug T. Regional variation of caesium-137 in minke whales *Balaenoptera acutorostrata* from West Greenland, the Northeast Atlantic and the North Sea. *Polar Biology* 2002; 25: 907-913.

Brown *et al.*, 1999.

Brown JE, Kolstad AK, Brungot AL, Lind B, Rudjord AL, Strand P, *et al.* Levels of ^{99}Tc in seawater and biota samples from Norwegian coastal waters and adjacent seas. *Marine Pollution Bulletin* 1999; 38(7): 560-571.

Calmet *et al.* 1992.

Calmet D, Woodhead D, Andre JM. ^{210}Pb, ^{137}Cs and ^{40}K in three species of porpoises caught on the eastern tropical Pacific Ocean. *Journal of Environmental Radioactivity* 1992; 15: 153-169.

Carroll *et al.* 2002.

Carroll, J, Wolkers H, Andersen M, Rissanen K. Bioaccumulation of radiocaesium in Arctic seals. *Marine Pollution Bulletin* 2002; 44: 1366-1371.

CCMS/CDSM/NATO, 1995.

CCMS/CDSM/NATO, 1995. Cross-border environmental problems emanating from defence-related installations and activities. Vol.1: Radioactive contamination. Final report phase 1, 1993-1995. Committee on the Challenges of Modern Society. North Atlantic Treaty Organization. Report no. 204. Oslo(?): NATO, 1995.

CEC, 1990.

CEC, 1990. The radiological exposure of the population of the European Community from radioactivity in North European marine waters: Project 'Marina'. Radiation Protection 47. EUR 12483 EN. Luxembourg: Office for official Publications, 1990.

Chen *et al* 2003.

Chen Q, Aarkrog A, Nielsen S, Dahlgaard H, Hou X, Yixuan Y, *et al.* Procedures for determination of ^{238,239,240}Pu, ^{241}Am, ^{237}Np, ^{234,238}U, ^{228,230,232}Th, ^{210}Pb, ^{210}Po, ^{99}Tc and ^{90}Sr in environmental materials. NUK-202. Roskilde: Risø National Laboratory, 2003.

Cooper *et al*., 2000.

Cooper LW, Larsen IL, O'Hara TM, Dolvin S, Woshner V, Cota GF. Radionuclide contaminant burdens in arctic marine mammals harvested during subsistence hunting. *Arctic* 2000; 53(2): 174-182.

Dahlgaard, 1995.

Dahlgaard H. Transfer of European coastal pollution to the Arctic: Radioactive tracers. *Marine Pollution Bulletin* 1995; 31: 353-358.

Dahlgaard *et al*., 2004.

Dahlgaard H, Eriksson M, Nielsen SP, Joensen HP. Levels and trends of radioactive contaminants in the Greenland environment. *Science of the Total Environment* 2004; 331: 53-67.

Derocher *et al*., 2002.

Derocher AE, Wiig Ø, Andersen M. Diet composition of polar bears in Svalbard and the western Barents Sea. *Polar Biology* 2002; 25:448-452.

Derocher and Wiig, 2002.

Derocher AE, Wiig Ø. Postnatal growth in body length and mass of polar bears (*Ursus maritimus*) at Svalbard. *Journal of Zoology* 2002; 256:343-349.

Dethleff *et al*., 1998.

Dethleff D, Loewe P, Kleine E. The Laptev Sea flaw lead - detailed investigation on ice formation and export during 1991/1992 winter season. *Cold Regions Science and Technology* 1998; 27(3): 225-243.

Dowdall *et al*., 2005.

Dowdall M, Gwynn JP, Moran C, O'Dea J, Davids C, Lind B. Uptake of Radionuclides by Vegetation at a High Arctic Location. *Environmental Pollution*. 2005; 133: 327-332.

Eikelmann *et al*., submitted.

Eikelmann IMH, Selnæs ØG, Møller B, Skuterud L. Radiocesium in reindeer and lichens from different locations in sub-arctic Finnmark and Russia. In: The 6th International Conference on Environmental Radioactivity in the Arctic and Antarctic. 2–6 October 2005 Nice, France. Submitted.

El-Daoushy, 1988.

El-Daoushy F. A summary of the lead-210 cycle in nature and related applications in Scandinavia. *Environment International* 1988; 14: 305-319.

Eurola and Hakala, 1977.

Eurola S, Hakala UK. The bird cliff vegetation of Svalbard. *Aquilo Series Botanical* 1977; 15: 1-18.

Erickson, 1997.

Erickson MD. The procedures manual of the environmental measurement laboratory, Vol II, 28th edition. 1997. HASL 300. New York, U.S. Department of Energy.

Frafjord, 1993.

Frafjord K. Food habits of arctic foxes *Alopex lagopus* on the western coast of Svalbard. *Arctic* 1993; 46: 49-54.

Fuglei, 2000.

Fuglei E. Physiological adaptations of the arctic fox to high Arctic conditions. PhD Thesis, University of Oslo, Oslo, Norway.

Føyn and Sværen, 1997.

Føyn L, Sværen I. Distribution and sedimentation of radionuclides in the Barents Sea. *ICES Journal of Marine Science* 1997; 54(3): 333-340.

Gaare, 1994.

Gaare E. Monitoring of ^{137}Cs in the TOV-areas: Dividal, Børgefjell, Dovre/Rondane, Gutulia and Solhomfjell, the summer of 1993. NINA Oppdragsmelding 300, 1994.

Gaare and Staaland 1994.

Gaare E, Staaland H. Pathways of fallout radiocaesium via reindeer to man. In: Dahlgaard H, ed. Nordic radioecology: The transfer of radionuclides through Nordic ecosystems to man Amsterdam: Elsevier, 1994: 303-334.

Gaschak *et al.*, 2003.

Gaschak S, Chizhevsky I, Arkhipov A, Beresford NA, Barnett CL. The transfer of Cs-137 and Sr-90 to wild animals within the Chernobyl exclusion zone. In: Contributed papers to the International Conference on the Protection of the Environment from the Effects of Ionizing Radiation. Stockholm, 2003. IAEA-CN-109/100: 200-202.

Gjertz and Lydersen, 1986.

Gjertz I, Lydersen C. The ringed seal (*Phoca hispida*) spring diet in northwestern Spitsbergen, Svalbard. *Polar Research* 1986; 4: 53-56.

Godzik, 1991.

Godzik B. Heavy metals and macroelements in the tundra of Southern Spitsbergen: the effect of Little Auk *Alle alle* (L.) colonies. *Polar Research* 1991; 9(2): 121-131.

Grue and Jensen, 1976.

Grue H, Jensen B. Annual cementum structures in canine teeth in arctic foxes (*Alopex lagopus L.*) from Greenland and Denmark. *Danish Review of Game Biology* 1976; 10(3): 1-2.

Gäfvert *et al.*, 2003.

Gäfvert, T, Føyn L, Brungot AL, Kolstad AK, Lind B, Christensen C, *et al.* Radioactivity in the marine environment 2000 and 2001. Results from the Norwegian national monitoring programme (RAME). NRPA report 2003:8. Østerås: Norwegian Radiation Protection Authority, 2003.

Haagenrud, 1978.

Haagenrud H. Layers of secondary dentine in incisors as age criteria in moose (*Alces alces*). *Journal of Mammalogy* 1978; 59: 857-858.

Hansen-Bauer *et al.*, 1990.

Hansen-Bauer I, Solås MK, Steffensen EL. The climate of Spitsbergen. DNMI Rapport. Klima; 39/90. Oslo: Norwegian Meteorological Institute, 1990.

Hardy et al., 1973.

Hardy EP, Krey PW, Volchok HL. Global inventory and distribution of fallout plutonium. *Nature* 1972; 241: 444-445.

Headly, 1996.

Headly AD. Heavy metal concentrations in peat profiles from the high Arctic. *The Science of the Total Environment* 1996; 177: 105-111.

Heldal *et al.*, 2003.

Heldal HE, Føyn L, Varskog, P. Bioaccumulation of ^{137}Cs in pelagic food webs in the Norwegian and Barents Seas. *Journal of Environmental Radioactivity* 2003; 65: 177-185.

Hentonen *et al.*, 2002.

Henttonen H, Fuglei E, Gower C, Haukisalmi V, Ims RA, Niemimaa J et al. *Echinococcus multilocularis* on Svalbard: Introduction of an intermediate host has enabled the local life-cycle. *Parasitology* 2002; 123: 547-552.

Hill, 1960.

Hill CR. Lead-210 and polonium-210 in grass. *Nature* 1960; 187: 211-212.

Hjelle *et al.*, 1999.

Hjelle A, Piepjohn K, Saalmann K, Ohta Y, Thiedig F, Salvigsen O, *et al.* Geological map of Svalbard 1: 100,000, sheet A7G Kongsfjorden. Tromsø, Norwegian Polar Institute, 1999.

Hjelset *et al.*, 1999.

Hjelset AM, Andersen M, Gjertz I, Lydersen C, Gulliksen B. Feeding habits of bearded seals (*Erignathus barbatus*) from the Svalbard area, Norway. *Polar Biology* 1999; 21: 186-193.

Hognestad and Lie, 1998.

Hognestad IK, Lie GM. Opptak og omsetning av radioaktivt strontium hos rein (*Rangifer tarandus tarandus*). Gevir og knokler som indikator på ^{90}Sr-forurensning i reinens miljø. Ås, Norway, Inst. for biologi og naturforvaltning, Inst. for husdyrfag, Inst. for analytisk kjemi, Norges landbrukshøgskole. Hovedoppgave, 1998.

Holm and Persson, 1975.

Holm E, Persson RBR. Transfer of fallout plutonium in the food chain lichen-reindeer-man. In: Proceedings of the Symposium on Transuranium Nuclides in the Environment, San Francisco, USA, 1975: 435-440.

Holm and Leisvik, 2001.

Holm E, Leisvik M. Radiocaesium och polonium I säl från Östersjön, en preliminär studie. In: Ilus E, ed. Proceedings of the 8th Nordic Seminar on Radioecology, Rovaniemi, 2001. NKS-70. 80-83.

Holm *et al*., 1983.

Holm E, Persson BRR., Hallstadius L, Aarkrog A, Dahlgaard H. Radio-cesium and transuranium elements in the Greenland and Barents Seas. *Oceanologica Acta* 1983; 6(4): 457-462.

Holtzman, 1966.

Holtzman RB. Natural levels of lead-210, polonium-210 and radium-226 in humans and biota of the Arctic. *Nature* 1966; 210: 1094-1097.

Hove *et al*., 1999.

Hove K, Garmo T, Gaare E, Strand P. Husdyr og vilt. In: Harbitz O, Skuterud L, eds. Radioactiv forurensning – betydning for landbruk, miljø og befolkning. Landbruksforlaget AS, 1999: 103-130.

Hubbell, 1982.

Hubbell JH. Photon mass attenuation and energy-absorption coefficients from 1 keV to 20 MeV. *International Journal of Applied Radiation Isotopes* 1982; 33: 1269-90.

IAEA, 2004.

IAEA. International Atomic Energy Agency. Sediment distribution coefficients and concentration factors for biota in the marine environment. Technical Report Series No.422, Vienna, International Atomic Energy Agency, 2004.

Iosjpe *et al*., 2002.

Iosjpe M, Brown J, Strand P. Modified approach to modelling radiological consequences from releases into the marine environment. *Journal of Environmental Radioactivity* 2002; 60(1-2): 91-103.

Jensen and Nielsen, 1968.

Jensen B, Nielsen LB. Age determination in the red fox (*Vulpes vulpes L*.) from canine toothy sections. *Danish Review of Game Biology* 1968; 5(3): 1-15.

Johannessen and Henriksen, 1978.

Johannessen M, Henriksen A. Chemistry of snow meltwater: Changes in concentration during meltwater. *Water Resources Research* 1978; 14: 615-619.

JRNEG 2002.

JRNEG. Joint Russian-Norwegian Expert Group for investigation of radioactive contamination in the Northern areas. Long term consequences of potential contamination in the Northern areas: Northern Norway and Murmansk Oblast. Summary Report. NRPA report 2002:5. Østerås: Norwegian Radiation Protection Authority, 2002.

Karcher *et al*., 2003.

Karcher MJ, Gerland S, Harms IH, Iosjpe M, Heldal HE, Kershaw PJ, *et al*. The dispersion of ^{99}Tc in the Nordic Seas and the Arctic Ocean: a comparison of model results and observations. *Journal of Environmental Radioactivity* (in press 2003).

Kasamatsu and Ishikawa, 1997.

Kasamatsu F, Ishikawa, Y. Natural variation of radionuclide ^{137}Cs concentrations in marine organisms with special referenceto the effect of food habits and trophic level. *Marine Ecology Progress Series* 1997; 160: 109-120.

Kauranen and Miettinen, 1969.

Kauranen P, Miettinen JK. ^{210}Po and ^{210}Pb in the arctic food chain and the natural radiation exposure of Lapps. *Health Physics* 1969; 16: 287-295.

Kautsky, 1987.

Kautsky H. Investigations on the distribution of ^{137}Cs, ^{134}Cs and ^{90}Sr and the water mass transport times in the Northern North Atlantic and the North Sea. *Deutsche Hydrographische Zeitschrift* 1987; 40(2): 49-69.

Kendall *et al.* 1987.

Kendall GM, Kennedy, BW, Greenhalgh JR, Adams N, Fell TP. Committed doses to selected organs and committed effective doses from intakes of radionuclides. Didcot: NRPB-GS7, NRPB, 1987.

Kershaw *et al.*, 2003.

Kershaw PJ, Heldal HE, Mork KA, Rudjord AL. Variability in the supply, distribution and transport of the transient tracer ^{99}Tc in the NE Atlantic. *Journal of Marine Systems* (in press).

Kjos-Hanssen and Rennesund, 1981.

Kjos-Hanssen B, Rennesund JS. Cesium-137 i kjoett fra reinsdyr og ryper paa Svalbard 1980. *Nordisk Veterinaermedicin* 1981; 33: 371-373. (In Norwegian)

Kjos-Hanssen and Toerresdal, 1982.

Kjos-Hanssen B, Toerresdal O. Cesium-137 i jord og beiteplanter fra Spitsbergen og fastlands-Norge (Suldal) 1981. *Nordisk Veterinaermedicin* 1982; 34: 98-100.

Klevezal *et al.*, 2001.

Klevezal GA, Bakhur AE, Kalyakin VN, Krushinskaya NL. Dynamics of radionuclide concentrations in calcified tissues of reindeer in the western Russian Arctic. *Chemosphere* 2001; 42: 61-67.

Kolstad, 1995.

Kolstad AK. Expeditions to Komsomolets in 1993 and 1994. NRPA report 1995:7. Østerås: Norwegian Radiation Protection Authority, 1995. (In Norwegian)

Kovacs, 2000.

Kovacs KM. Hooded seal *Cystophora cristata*. In: Perrin WF, Würsig B, Thewissen JMG, eds. Encyclopedia of marine mammals. Academic Press, San Diego, 2000: 580-582.

Landa *et al.*, 1998.

Landa ER, Reimnitz E, Beals DM, Pochkowski JM, Winn WG, Rigor I. Transport of Cs-137 and Pu-239, Pu-240 with ice-rafted debris in the Arctic Ocean. *Arctic* 1998; 51(1): 27-39.

Larsen *et al.*, 1999.

Larsen E, Holo EN, Saltbones J, Stokke E. Kola consequence analyses: Evaluation of doses from a hypothetical accident at Kola Nuclear Power Plant. NRPA report 1999:10. Østerås: Norwegian Radiation Protection Authority, 1999. (In Norwegian)

Larsen, 1986.

Larsen T. Population biology of the polar bear (*Ursus maritimus*) in the Svalbard area. Norsk Polarinstitutt Skrifter, Nr. 184. 1986.

Loeng, 1991.

Loeng H. Features of the physical oceanographic conditions of the Barents Sea. *Polar Research* 1991; 10(1): 5-18.

Loeng, 1998.

Loeng H. Økosystemet i Barentshavet. Havklima. *Fisken og Havet* 1998; Special Issue 2, Havets Miljø: 13-17. (In Norwegian)

Lunn *et al*., 2002

Lunn NJ, Schliebe S, Born EW. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Nuuk, Greenland. IUCN, Gland, Switzerland and Cambridge, UK, 2002.

Lydersen, 1998

Lydersen C. Status and biology of ringed seals (*Phoca hispida*) in Svalbard. *NAMMCO Science Publication* 1998; 1: 46-62.

Lydersen and Gjertz, 1987.

Lydersen C, Gjertz I. Population parameters of ringed seals (*Phoca hispida* Schreber, 1775) in the Svalbard area. *Canadian Journal of Zoology* 1987; 65: 1021-1027.

Macdonald *et al*., 1993.

Macdonald, CR, Ewing LL, Stewart, REA, Wiewel AM, Harris DA. Evaluation of radionuclide levels and radiological dose in three populations of marine mammals in the Eastern Canadian Arctic. In: Society of Environmental Toxicology and Chemistry 14th Annual Meeting, Ecological Risk Assessment: Lessons Learned? 1993, Houston, 254.

Mauritzen, 2003.

Mauritzen M. Patterns and processes in female polar bear space-use. PhD-Thesis, University of Oslo, Norway, 2003.

McDonald *et al*., 2003.

McDonald RW, Harner T, Fyfe J, Loeng H, Weingartner T. AMAP assessment 2002: The influence of global change on contaminant pathways to, within, and from the Arctic. Oslo: Arctic Monitoring and Assessment Programme (AMAP), 2003.

Meese *et al*., 1997.

Meese DA, Reimnitz E, Tucker WB, Gow AJ, Bischof J, Darby D. Evidence for radionuclide transport by sea ice. *The Science of the Total Environment* 1997; 202(1-3): 267-278.

Miettinen, 1975.

Miettinen JK. Plutonium foodchains. In: Miller MW, Stannard JN, eds. Proceedings of the International Conference on Environmental Toxicity of Aquatic Radionuclides: Models and Mechanisms, New York, USA, 1975: 29-43.

Nakano, 1990.

Nakano Y. Quasi-steady problems in freezing soils: I. Analysis on the steady growth of an ice layer. *Cold Regions Science and Technology* 1990;17: 207-226.

Negoita, 1999.

Negoita TG. Radionuclide contamination in the Arctic areas. In: Strand P, Jølle T, eds. 4th International Conference on Environmental Radioactivity in the Arctic, Edinburgh, 1999. Extended abstracts. Østerås: Norwegian Radiation Protection Authority, 1999: 270-272.

NEFCO 1996

NEFCO 1996. The Nordic Environmental Finance Corporation. Proposals for environmentally sound investment projects in the Russian part of the Barents region. Prepared by the AMAP expert group. Helsinki 1996. www.nefco.org

Nies *et al.*, 1999.

Nies H, Harms IH, Karcher MJ, Dethleff D, Bahe C. Anthropogenic radioactivity in the Arctic Ocean. Review of the results from the joint German project. *The Science of the Total Environment* 1999; 237-238: 181-191.

Nilssen *et al.*, 1982.

Nilssen KJ, Sundsfjord JA, Blix AS. Energetics of the High Arctic (77-81°NL) Svalbard reindeer. *Federation Proceedings* 1982; 41: 1238.

Nilssen *et al.*, 1984.

Nilssen KJ, Johnsen HK, Rognmo A, Blix AS. Heart rate and energy expenditure in resting and running Svalbard and Norwegian reindeer. *American Journal of Physiology* 1984; 246: 963-967.

NRPA, 2004.

NRPA. Norwegian Radiation Protection Authority. Radioactivity in the Marine Environment 2002. Results from the Norwegian National Monitoring Programme (RAME). StrålevernRapport 2004:X. Østerås: Norwegian Radiation Protection Authority, 2004.

Nurnberg *et al.*, 1994.

Nurnberg D, Wollenburg I, Dethleff D, Eicken H, Kassens H, Letzig T, *et al.* Sediments in Arctic sea-ice – implications for entrainment, transport and release. *Marine Geology* 1994; 119(3-4): 185-214.

Oritsland, 1985.

Øritsland NA. Svalbardreinen og dens livsgrunnlag. Avslutningsrapport for MAB – Svalbardprosjektet 1975-1985. 1985: pp. 278.

Orpin *et al.*, 1982.

Orpin CG, Mathiesen SD, Greenwood Y, Blix AS. Seasonal changes in the ruminal microflora of the high arctic Svalbard reindeer (*Rangifer tarandus platyrhynchus*). *Applied and Environmental Microbiology* 1985; 50: 144-151.

Orvin, 1934.

Orvin AK. Geology of the Kings Bay region, Spitsbergen. Skrifter om Svalbard og Ishavet. No. 57. Oslo 1934.

Pauly *et al.*, 1998.

Pauly D, Trites AW, Capuli E, Christensen V. Diet composition and trophic levels of marine mammals. *ICES Journal of Marine Science* 1998; 55: 467-481.

Pedersen *et al.*, 1993

Pedersen Ø, Hove K, Staaland H. Seasonal variation and effective half life of ^{137}Cs in semi-domestic reindeer grazing mountain pasture contaminated by deposition from the Chernobyl accident. In: Strand P, Holm E, eds. International conference on environmental radioactivity in the Arctic and Antarctic, 1993: 303-304.

Pentreath *et al.*, 1982.

Pentreath RJ, Jefferies DF, Talbot JW, Lovett MB, Harvey BR. Transuranic cycling behaviour in marine environment. IAEA-TECDOC-265. Vienna: International Atomic Energy Agency, 1982: 121-128.

Pentreath and Woodhead, 1988.

Pentreath RJ, Woodhead DS. Towards the development of criteria for the protection of marine fauna in relation to the disposal of radioactive wastes into the sea. 2nd Edition, Vienna, IAEA, 1988: 213-243.

Pentreath, 1999.

Pentreath RJ. A system for radiological protection of the environment: some initial thought and ideas. *Journal of Radiological Protection* 1999; 19: 117-128.

Pinglot *et al.*, 1994.

Pinglot JF, Pourchet M, Lefauconnier B, Hagen JO, Vaikmae R, Punning JM, *et al.* Natural and artificial radioactivity in the Svalbard glaciers. *Journal of Environmental Radioactivity* 1994; 25: 161-176.

Prestrud, 1992a.

Prestrud P. Arctic foxes on Svalbard: population ecology and rabies. PhD Thesis 1992, Norwegian Polar Institute and the University of Oslo, Norway.

Prestrud, 1992b.

Prestrud P. Food habits and observations of the hunting behaviour of arctic foxes, *Alopex lagopus*, in Svalbard. *Canadian Field-Naturalist* 1992; 106: 225-236.

Putkonen, 1998.

Putkonen J. Soil thermal properties and heat transfer processes near Ny-Alesund, northwestern Spitsbergen, Svalbard. *Polar Research* 1998; 17(2): 165-179.

Reeves, 1998.

Reeves RR. Distribution, abundance and biology of ringed seals (*Phoca hispida*): an overview. *NAMMCO Science Publication* 1998; 1: 9-45.

Reimers and Nordby, 1968.

Reimers E, Nordby Ø. Relationships between age and tooth cementum layers in Norwegian reindeer. *Journal of Wildlife Management* 1968; 32: 957-961.

Rissanen *et al.*, 1990.

Rissanen K, Rahola T, Aro P. Distribution of cesium-137 in reindeer. *Rangifer* 1990; 2: 57-66.

Rissanen *et al.*, 1999.

Rissanen, K, Pempkowiak J, Ikaheimonen TK, Matishov DG, Matishov GG. ^{137}Cs, $^{239,240}Pu$, ^{90}Sr and selected metal concentrations in organs of Greenland seal pups in the White Sea area. In: Strand P, Jølle T, eds. 4th International Conference on Environmental Radioactivity in the Arctic, Edinburgh, 1999. Extended abstracts. Østerås: Norwegian Radiation Protection Authority, 1999: 186-189.

Rissanen et al., 2000.

Rissanen K, Ikauheimonen TK, Ylipieti J, Matishov DG, Matishov GG. Plutonium in algae, sediments and biota in the Barents, Pechora and Kara seas. In: Inaba J, Hisamatsu S, Ohtsuka Y, eds. Distribution and speciation of radionuclides in the environment. Rokkasho, Aomori (Japan): Institute for Environmental Sciences, 2000: 107-114.

Skogland *et al.*, 1991.

Skogland T, Strand O, Espelien IS. Den biologiske betydning av radiocesium I villrein. In: Gaare E, Jonsson B, Skogland T, eds. Sluttrapport fra NINA's radioøkologiprogram 1986-1990, 1991: 64-70.

Skuterud *et al.*, in press a.

Skuterud L, Gaare E, Eikelmann IM, Hove K, Steinnes E. Chernobyl radioactivity persists in reindeer. In press.

Skuterud *et al.*, in press b.

Skuterud L, Gwynn JP, Gaare E, Steinnes E, Hove K. ^{90}Sr, ^{210}Pb and ^{210}Po in lichen and reindeer in Norway. In press.

Staaland, 1984.

Staaland H. On the quality of Svalbard reindeer pasture in the summer and autumn. *Rangifer* 1984; 4: 16-23.

Staaland *et al.*, 1979.

Staaland H, Jacobsen E, White R. Comparison of the digestive tract in Svalbard and Norwegian reindeer. *Arctic Alpine Research* 1979; 11: 457-466.

Staaland *et al.*, 1983.

Staaland H, Brattbakk I, Ekern K, Kildemo K. Chemical composition of reindeer forage plants in Svalbard and Norway. *Holarctic Ecology* 1983; 6: 109-122.

Staaland *et al.*, 1991.

Staaland H, Bjørnstad H, Pedersen Ø, Hove K. Radiostrontium, radiocesium and stable mineral composition of bones of domestic reindeer from Vågå, Norway. *Rangifer* 1991; 11: 17-22.

Stokke, 1997.

Stokke E. Kola nuclear power plant. Consequence assessment for hypothetical severe accidents: Source term assessment. Report No. IFE/HR/F-97/1088. Kjeller: Institutt for energitekknikk / Institute for Energy Technology, 1997.

Strand *et al.*, 1998.

Strand O, Severinsen T, Espelien IS. Terrestrisk naturovervåkning – metaller og radioaktivitet i fjellrev. NINA Oppdragsmelding 560, 1998.

Strand *et al.*, 1993.

Strand P, Rudjord AL, Salbu B, Christensen G, Foyn L, Lind B, *et al.* Survey of artificial radionuclides in the Kara Sea 1993. In: Strand P, Holm E, eds. Environmental radioactivity in the Arctic and Antarctic: Proceedings of the International Conference on Environmental Radioactivity in Arctic and Antarctic, Kirkenes, 1993. Østerås: Norwegian Radiation Protection Authority, 1993: 53-65.

Strandberg 1997.

Strandberg M. Distribution of ^{137}Cs in a low Arctic ecosystem in West Greenland. *Arctic* 1997; 50: 216-223.

Taylor *et al.*, 1988.

Taylor HW, Svoboda J, Henry GHR, Wein RW. Post-Chernobyl ^{134}Cs and ^{137}Cs levels at some locations in northern Canada. *Arctic* 1988; 41: 293-296.

Thomas *et al.*, 1994.

Thomas PA, Sheard JW, Swanson S. Transfer of ^{210}Po and ^{210}Pb through the lichen-caribou-wolf food chain of Northern Canada. *Health Physics* 1994; 66: 666-677.

Tyler, 1987.

Tyler N. Estimating the daily dry matter intake of Svalbard reindeer in late winter. *Rangifer* 1987; 7: 29-32.

UNSCEAR 1982.

UNSCEAR 1982. Ionizing radiation: Sources and biological effects. United Nations Scientific Committee on the Effects of Atomic Radiation. New York: UNSCEAR, 1982.

UNSCEAR 2000.

UNSCEAR 2000. Sources and effects of ionizing radiation. Volume 1: Sources. United Nations Scientific Committee on the Effects of Atomic Radiation. New York. UNSCEAR, 2000.

Vakulovsky *et al.*, 1993.

Vakulovsky S, Nikitin A, Chumichev V. Radioactive contamination of the Russian Arctic seas from results of observations during the year 1960-1992 In: Strand P, Holm E, eds. Environmental radioactivity in the Arctic and Antarctic: Proceedings of the International Conference on Environmental Radioactivity in Arctic and Antarctic, Kirkenes, 1993. Østerås: Norwegian Radiation Protection Authority, 1993: 177-182.

Varskog *et al.* 1997

Varskog P, Bjerk TO, Ruud AB. pH-controlled EDTA titration as an alternative method for determination of the chemical yield of yttrium in ^{90}Sr-analyses. In: Jerome S, Inn K, eds. Conference on Rapid Radioactivity Measurments in Emergency and Routine Situations, 1997. Teddington, NPL 237-242.

Vinje and Kvambekk 1991.

Vinje T, Kvambekk AS. Barents Sea drift ice characteristics. *Polar Research* 1991; 10(1): 59-68.

Watson *et al.* 1999.

Watson WS, Sumner DJ, Baker JR, Kennedy S, Reid R, Robinson I. Radionuclides in seals and porpoises in the coastal waters around the UK. *Science of the Total Environment* 1999; 234: 1-13.

Weslawski *et al.*, 1994.

Weslawski JM, Ryg M, Smith TG, Øritsland NA. Diet of ringed seals (*Phoca hispida*) in a fjord of West Svalbard. *Arctic* 1994; 47: 109-114.

Wessel and Smith, 1998.

Wessel P, Smith WHF. New, improved version of the Generic Mapping Tools released. *EOS (Transactions AGU)* 1998; 79: 579.

Wright *et al.*, 1997.

Wright SM, Strand P, Sickel MAK, Howard BJ. Howard DC, Cooke AI. Spatial variation in the vulnerability of Norwegian arctic countries to radiocaesium deposition. *The Science of the Total Environment* 1997; 202: 173-184. 1997

Yoshitome *et al.*, 2003.

Yoshitome R, Kunito T, Ikemoto T, Tanabe S, Zenke H, Yamauchi M. *et al.* Global distribution of radionuclides (^{137}Cs and ^{40}K) in marine mammals. *Environmental Science and Technology* 2003; 37: 4597-4602.

APPENDIX I

This Appendix serves to detail sampling and analytical procedures used for the monitoring of the marine and terrestrial mammals as documented in this report.

Sampling – Marine mammals

Seal muscle and other tissue samples were obtained by the Norwegian Polar Institute from animals that had been collected in the course of other scientific studies (ringed and bearded seals) or through commercial hunting (hooded seals). Samples from polar bears were obtained through the Norwegian Polar Institute and the Governor of Svalbard from problem bears that had been shot on Svalbard. Age groups of individual animals were determined on the basis of social relationships, body size and sexual maturity. All tissue samples were stored frozen at -20 ºC until analysed.

Sampling – Terrestrial mammals

Svalbard reindeer muscle and tissue samples were provided by the University of Svalbard from animals that had been collected in the course of other scientific studies. All tissue samples were stored frozen at -20 ºC until analysed. Age of individual animals was determined by analysis of tooth cementum layers according to the method described by Reimers and Nordby (1968) and Haagenrud (1978). Arctic fox muscle samples were provided by the Norwegian Polar Institute from animals collected by commercial trappers. Foxes were caught using baited traps and the carcasses were skinned and stored frozen. Tissue samples were taken out and stored frozen at -20 ºC until analysis. Age of individual animals was determined by counting the annuli in the cementum of sectioned lower canine teeth according to the method described by Jensen and Nielsen (1968) and Grue and Jensen (1976).

APPENDIX II

Laboratory Preparation of Marine and Terrestrial Mammal Samples

All tissue samples were cleaned of excess blood, fat and remaining bone fragments and then dried at 105 ˚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 all soft tissue samples for the measurement of the gamma emitting isotopes ^{137}Cs and ^{40}K. ^{137}Cs was determined from its characteristic emission at 661 keV, ^{40}K via 1461 keV. 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 at 1332 keV was 1.9 keV and 40 %, respectively. 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 (AEA Technology) with attenuation coefficients taken from Hubbel (1982). 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.

^{90}Sr: Analysis

^{90}Sr determination was carried out by the Institute of Energy Technology (IFE, Norway) by chemical separation and analysis of the daughter product ^{90}Y. Bone samples were dried to constant weight at 105 ºC, and dry ashed at 700 ºC after addition of ^{85}Sr tracer and analysed using a modified HASL 300 procedure (Erickson, 1997; Varskog *et al.* 1997). Following chemical separation of Sr using fuming HNO_3 and precipitation of hydroxide and chromate, the samples were left for in-growth of ^{90}Y. Y was precipitated as oxalate, and collected by filtration, and the ^{90}Y activity measured by low level anticoincidence beta counters. Chemical yield was determined by ^{85}Sr tracer and titration of Y with EDTA.

^{238}Pu and $^{239,240}Pu$: Analysis

Concentrations of ^{238}Pu and $^{239,240}Pu$ were determined using alpha spectrometry. Samples consisted of 10 g of dried material. Recovery was determined by using ^{242}Pu as a yield monitor. Different radiochemical separation techniques were applied to separate plutonium 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 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.

^{210}Po: Analysis

^{210}Po was determined in muscle according to the method of Chen *et al.* (2003) on fresh (10g) and dried (2g) samples, using ^{209}Po as a yield tracer. Following wet ashing of a sample, ^{210}Po was spontaneously deposited onto silver disks from a dilute HCl solution at 90°C for three hours. Disks were then counted using ion-implanted silicon alpha detectors for 1-7 days. ^{210}Pb was not determined in any muscle sample as concentrations of ^{210}Pb in muscle from marine and terrestrial mammals are typically low or below the limits of detection (Kauranen and Miettinen, 1969; Macdonald *et al.*, 1993, Thomas *et al.*, 1994).

StrålevernRapport 2005:1
Virksomhetsplan 2005

StrålevernRapport 2005:2
Natural Radioactivity in Produced Water from the Norwegian Oil and Gas Industry in 2003

StrålevernRapport 2005:3
Kartlegging av historiske utslipp til Kjeller-området og vurdering av mulige helsekonsekvenser

StrålevernRapport 2005:4
Assessment of environmental, health and safety consequences of decomissioning radioisotope thermal generators in NW Russia

StrålevernRapport 2005:5
Environmental Impact Assessments in Arctic Environments
Protection of plants and animals

StrålevernRapport 2005:6
Anbefaling for opplæring av medisinske fysikere i stråleterapi i Norge

ISSN 0804-4910