The radiological environment of Svalbard

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This paper presents a detailed survey of the radiological environment of the Svalbard area carried out from 2000 to 2002, in both the marine and terrestrial environment. In the marine environment in 2001, ^{99}Tc activity concentrations in seawater were 0.13 to 0.36 Bq/m^3, 5 fold higher than those in 1994, reflecting the increase in ^{99}Tc discharges from Sellafield in the mid 1990s. Cs-137 activity concentrations in seawater were 2.23 to 2.43 Bq/m^3, ca. 10 fold lower than those in the 1980s, reflecting the reduction in discharge of this radionuclide. Pu-238, $^{239+240}$Pu and ^{241}Am activity concentrations in seawater were $<$0.3 to 0.7 mBq/m^3, 5.6 to 8.9 mBq/m^3 and 0.6 to 2.4 mBq/m^3 respectively, with activity ratios suggesting global fallout to be the dominant source. Tc-99 activity concentrations in brown algae were up to 18 fold higher than those in the 1980s with highest concentrations in *Fucus distichus* (25.7 to 58.7 Bq/kg d.w.). In the terrestrial environment, typical ^{137}Cs activity concentrations in soil were between $<$0.5 and 63 Bq/kg d.w. whilst activity concentrations of the natural radionuclides ^{238}U (17 to 72 Bq/kg d.w.), ^{226}Ra (21 to 70 Bq/kg d.w.), ^{232}Th (10 to 57 Bq/kg d.w.) and ^{40}K (115 to 818 Bq/kg d.w.) were similar to global averages. In terrestrial vegetation, ^{137}Cs activity concentrations varied from 29 to 292 Bq/kg d.w. in mosses, 30 to 140 Bq/kg d.w. in lichen and 19 to 109 Bq/kg d.w. in flowering plants. Elevated activity concentrations of ^{137}Cs, ^{238}U, ^{226}Ra, Pu isotopes and ^{241}Am were found in some matrices associated with seabird colonies.

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In comparison with the Norwegian mainland, Svalbard has received little attention with regard to the concentration and behaviour of radionuclides in marine and terrestrial environments. For the terrestrial environment, the majority of previous studies were conducted in the 1980s and were limited both in their spatial representativeness, the number and type of samples examined and the nuclides reported. For the marine environment, more data are available, both on a spatial and temporal scale. However, there is an overwhelming bias in the data for abiotic compartments (i.e. seawater and sediments), with few studies of radionuclide contamination in marine biota within the Svalbard region.

The amount of interest and attention focused on the region has recently increased, possibly because of the widely held perception of Svalbard as a pristine wilderness and because of the vulnerability of its ecosystems. Much of this attention has focused on the concentrations of pollutants in the region and its ecosystems, with radioactive contaminants being one of the main groups of pollutants under discussion (International Arctic Environmental Protection Strategy [IAEPS]; AMAP 1993). The current focus on

radioactivity within the Arctic regions in general is the result of 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, behaviour and effects of radioactive contaminants in High Arctic ecosystems.

Svalbard lies 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. Of the 543 atmospheric weapons tests conducted globally, 91 were carried out in the Arctic region by the former Soviet Union 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 30 PBq of ^{137}Cs from 87 of these tests alone. More recently, trends of radionuclides in the Svalbard marine environment have been dominated by the contributions of European reprocessing facilities. In particular, peak discharges of ^{137}Cs from Sellafield (UK) in the mid 1970s and later ^{99}Tc in the mid 1990s, following operation of the Enhanced Actinide Removal Plant (EARP), led to higher activity concentrations of these radionuclides in seawater and marine biota off the Norwegian and Svalbard coast (Hallstadius et al. 1982; Kershaw & Baxter 1995; Brown et al. 1999; Rudjord et al. 2001).

Historically, the emphasis of radiological protection has been directed towards health impacts on humans, but in recent times the need for developing a system for assessment of radiological impacts on both flora and fauna has come to the fore. This shift in emphasis has largely been due to the realization 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 focusing of attention towards site-specific processes influencing the behaviour and occurrence of radionuclides in constituent matrices therein. The productivity of the Arctic marine environment and the importance of that productivity to a number of industries necessitates public confidence with respect to levels of radioactive contamination in the Arctic.

This paper provides an overview of the current radiological state of the Svalbard area. The concentrations of a range of anthropogenic and natural radionuclides in both the terrestrial and marine biotic and abiotic environments of Svalbard in the period 2000 to 2002 are reported and discussed.

Sampling and analysis

Sampling methods

The majority of the marine and terrestrial samples discussed in this paper were collected during Norwegian Radiation Protection Authority (NRPA) field campaigns in 2000, 2001 and 2002 in the Kongsfjorden area and during RV *Polarstern* and RV *Lance* research cruises to the Fram Strait in 2000 and 2001 respectively. Investigations on the islands of Hopen and Bjørnøya were conducted during the KV *Senja* research cruise in 2000 (Fig. 1). Additional samples were taken at locations around Spitsbergen by the Norwegian Polar Institute during various field campaigns in 2001 and 2002. We also present data from the three seawater monitoring stations that were established in 2001 at the Ny-Ålesund settlement in Kongsfjorden and at the Norwegian Meteorological Institute's base stations on Bjørnøya and Hopen as part of NRPA's ongoing radionuclide marine monitoring programme Radioactivity in the Marine Environment (RAME), funded by the Norwegian Ministry of the Environment.

Seawater samples of 50 to 100 litres each for analysis of ^{99}Tc and 200 litres each for analysis of both ^{137}Cs and Pu isotopes were collected in polyethylene drums either from a boat or from the shore. Samples were analysed as soon as practicable after collection. In some cases, the samples were preconcentrated onboard vessels or in the field. Seaweed samples were collected from inter-

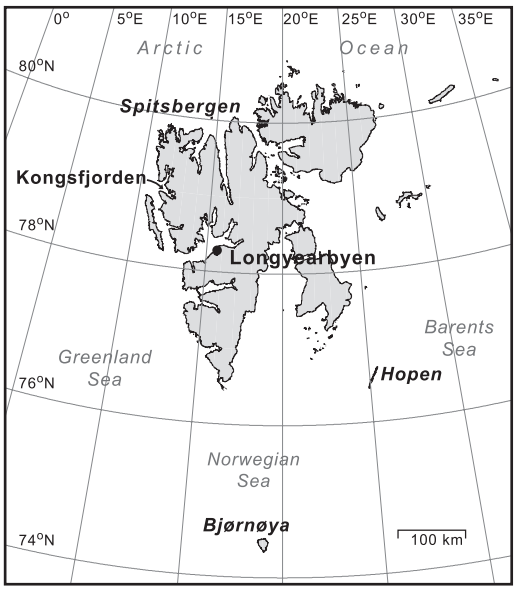


Fig. 1. Location of the principal sampling sites (Kongsfjorden, Hopen and Bjørnøya) in the Svalbard archipelago

tidal locations or by divers from deeper water. Terrestrial vegetation was sampled by cutting above-surface structures of healthy specimens. Both marine and terrestrial vegetation samples were frozen during transport to the laboratory. Soil samples were taken either by digging pits and removing samples from the walls or by coring. Samples were frozen during transport. All sampling was conducted in a manner that minimized impact on the environment of the area.

Sample preparation and analysis

Seawater samples were all filtered through 1 μm filters prior to analysis. Terrestrial vegetation samples were rinsed with distilled water; seaweed samples were rinsed in the water they were abstracted from. Solid samples were dried at a temperature of 105 °C to constant weight, homogenized using a stainless steel blender, sieved to 2 mm particle size and packed into standard counting geometries of volumes between 14 and 550 ml. Gamma emitting isotopes (^{40}K, ^{137}Cs, ^{238}U, ^{232}Th, ^{226}Ra) were analysed using a low background high resolution HPGe detector with 5 cm lead shielding. Nominal resolution and relative efficiency of the system were 1.9 keV at 1332 keV and 40%. Spectra were obtained for count times of 24 to 72 hours and analysed using the Genie 2000 software suite. The detector was calibrated using traceable standard solutions and all spectra were corrected for background and matrix interferences. ^{40}K and ^{137}Cs were quantified using their characteristic emissions at 1461 keV and 661 keV, respectively. U-238 was quantified via the emissions of the daughter ^{234}Th at 63 keV with which it is in secular equilibrium. The ^{238}U activity and the ^{238}U/^{235}U activity ratio of 21.4 were used to correct the 186 keV peak for its ^{235}U contribution, allowing ^{226}Ra quantification. Th-232 was assayed via the emissions of its granddaughter ^{228}Ac with which it may be assumed to be in secular equilibrium. For the measurement of ^{137}Cs in seawater, ^{137}Cs was preconcentrated by pumping the seawater through a sorbent filter impregnated with a copper hexacyanoferrate resin. The filters were subsequently air dried and ashed and the ash packed into standard geometries for gamma analysis.

Tc-99 preparation and analysis is described in detail in Dowdall et al. (2003). Briefly, ^{99}Tc was preconcentrated from seawater samples using ion exchange chromatography, isolated and purified using methods described in Chen et al. (2003) before being electroplated onto stainless steel planchettes for counting on a low background anti-coincidence shielded gas flow proportional counter. Seaweed samples (10 to 20 g) were digested using H_2SO_4 and HNO_3 prior to preconcentration. Chemical recovery was determined using ^{99m}Tc as a tracer, added at an early stage in the process and counted using a NaI well detector. All samples were blank corrected.

Pu and Am isotopes in seawater and soil were analysed according to the method described by Chen et al. (2003). Pu and Am isotopes in seawater samples were preconcentrated using a co-precipitation technique with ferric hydroxide, followed by further concentration, whilst soil samples (10 g) were ashed and digested before purification using ion exchange and solvent extraction. All samples were electroplated and counted on silicon semi-conductor detectors for one week. Chemical recoveries were determined by using ^{242}Pu and ^{243}Am as yield tracers.

All analyses were conducted according to the internal QA/QC procedures of NRPA laboratories which involve the use of blanks, standards, spikes and international reference materials. Splits were used to assess reproducibility and blank samples to assure freedom from contamination. Results presented for all terrestrial

matrices, marine sediment and marine vegetation are based on dry weight.

Uncertainties in all results were quadrature combinations of individual uncertainties in each factor of the analyses. These included uncertainties in calibrations, gravimetric and volumetric operations, statistical uncertainties in radiometric counting and nuclear data and a factor added to include any systematic uncertainty. Uncertainty budget analysis was conducted according to the methods outlined in Ellison et al. (2000). Uncertainties for averages are calculated by error propagation on individual measurements.

Results and discussion

Tc-99 in seawater

Activity concentrations of ^{99}Tc in seawater from the Svalbard area in 2001 ranged from 0.13 ± 0.03 Bq/m^3 to 0.35 ± 0.05 Bq/m^3 (Fig. 2), similar to those reported for the Svalbard area in 2000 (Gerland et al. 2002, Gerland et al. 2003; Kershaw et al. 2004), but 5 fold higher than 1994 pre-EARP values of 0.03 to 0.08 Bq/m^3 (Kershaw et al. 1999). In 2001, the highest concentrations of ^{99}Tc at the Svalbard monitoring stations were observed at Bjørnøya (0.35 ± 0.05 Bq/m^3) and Hopen (0.32 ± 0.04 Bq/m^3), whilst the highest concentration at Ny-Ålesund was 0.25 ± 0.03 Bq/m^3. In 2001, ^{99}Tc activity concentrations in the centre of the West Spitsbergen Current (WSC) immediately west of Spitsbergen were 0.24 ± 0.04 Bq/m^3, which was somewhat lower than those in 2000 of 0.39 ± 0.05 Bq/m^3 (Gerland et al. 2002, Gerland et al. 2003), but still 6 fold higher than 1994 pre-EARP values (Kershaw et al. 1999). The similarity between ^{99}Tc concentrations in the WSC and within Isfjorden and Kongsfjorden along the west coast of Spitsbergen, suggests a strong influence of the WSC on the exchange processes throughout these fjords, as noted in Kongsfjorden by Ingvaldsen et al. (2001) and Saloranta & Svendsen (2001).

The activity concentrations of ^{99}Tc in the Svalbard area are up to 15 fold lower than those measured at the NRPA monitoring station at Hillesøy, on mainland northern Norway, where the average activity concentration in 2001 was 1.25 Bq/m^3 (range 0.9 to 2.0 Bq/m^3) (Kolstad & Lind 2002). These values reflect the dominant oceanic surface circulation in the Northern seas and the concomitant transfer of EARP derived ^{99}Tc into the Norwegian Arctic. Lower ^{99}Tc activity concentrations are observed in the Svalbard area compared to Norwegian coastal waters for the same period because of the ingress of North Atlantic water into the Norwegian Coastal Current and the West Spitsbergen Current (WSC) with the resultant dilution of the EARP ^{99}Tc signal.

In 2001, sub-surface samples, taken in addition to surface samples at stations in Kongsfjorden and in the WSC, showed relatively uniform ^{99}Tc activity concentrations (Table 1). However, sub-surface samples taken in the WSC in 2000 at greater depths (465 to 1000 m) showed 0.17 and 0.04 Bq/m^3 of ^{99}Tc, 2 to 5 fold lower than at the surface (Gerland et al. 2002, Gerland et al. 2003). The uniform distribution of ^{99}Tc observed in the top 100 to 125 m of Kongsfjorden and the WSC is due to the conservative nature of ^{99}Tc in seawater and the apparent depth of the mixed layer at the time of sampling.

Cs-137 in seawater and sediments

In May 2001, the average ^{137}Cs activity concentration in seawater along the west coast of Svalbard was 2.33 ± 0.06 Bq/m^3 with a range of 2.26 to 2.43 Bq/m^3 (Fig. 3). These results are similar to two measurements taken in Kongsfjorden in 2000 of 2.32 ± 0.12 Bq/m^3 and 2.2 ± 0.1 Bq/m^3, but lower than previously reported values of 3 to 5 Bq/m^3 in 1994 (Kershaw et al. 1997). These

Table 1. Activity concentrations and associated uncertainties (1σ) of ^{99}Tc,^{137}Cs ^{238}Pu, $^{239+240}$Pu and ^{241}Am in surface and sub-surface seawater samples collected from Kongsfjorden and the West Spitsbergen Current (WSC) in May 2001.

Location	Depth (m)	Salinity (‰)	^{99}Tc (Bq/m^3)	^{137}Cs (Bq/m^3)	^{238}Pu (mBq/m^3)	$^{239+240}$Pu (mBq/m^3)	^{241}Am (mBq/m^3)
Kongsfjorden	0	34.4	0.25 ± 0.04	2.41 ± 0.14	0.7 ± 0.6	7.1 ± 1.7	0.7 ± 0.3
	125	34.6	0.29 ± 0.03	2.33 ± 0.11	< 1	6.1 ± 1.5	1.2 ± 0.4
WSC	0	34.5	0.24 ± 0.04	2.30 ± 0.11	0.5 ± 0.5	8.9 ± 1.4	2.4 ± 0.6
	100	34.1	0.22 ± 0.04	2.1 ± 0.1	1.5 ± 0.8	12 ± 2	2.4 ± 0.6

Fig. 2. Surface activity concentrations of ^{99}Tc (Bq/m^3) in seawater in the Svalbard area in 2001. Average values given for monitoring stations, with ranges in parentheses.

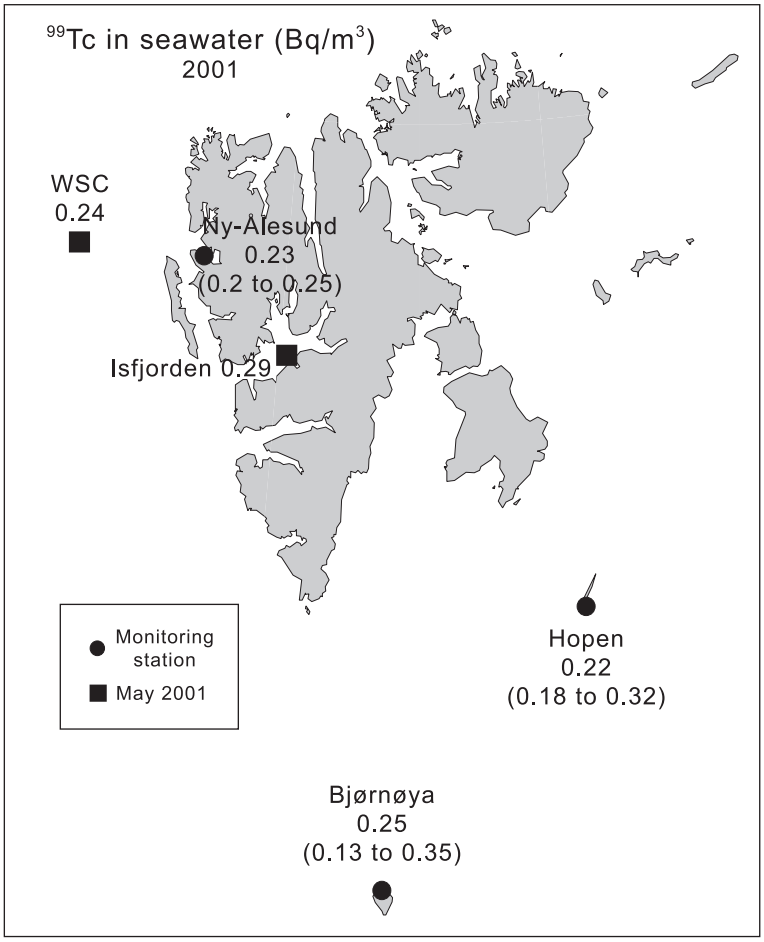
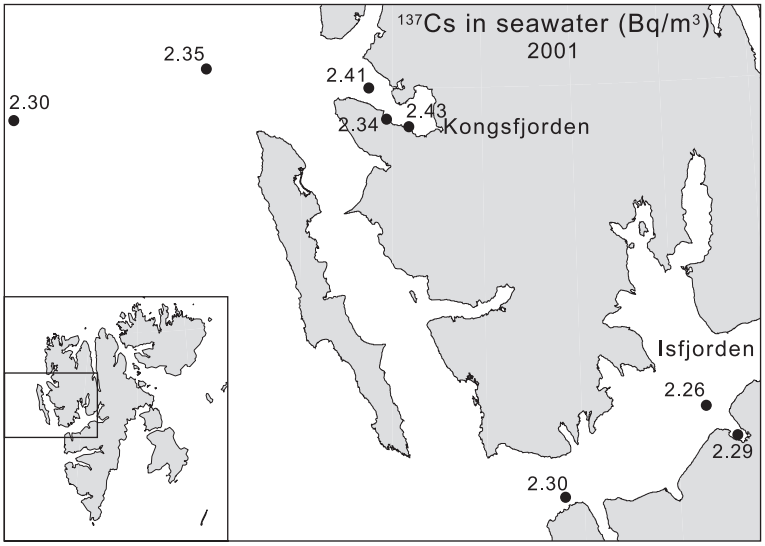


Fig. 3. Surface activity concentrations of ^{137}Cs in seawater (Bq/m^3) off western Svalbard in May 2001.



values reflect a continuing trend of ^{137}Cs in seawater in the Svalbard area from 20 to 30 Bq/m^3 (Hallstadius et al. 1982; Kershaw & Baxter 1995) in the early to mid 1980s. This decreasing trend is the result of decreased discharges of ^{137}Cs from Sellafield, from 4000 to 5000 TBq/a in the mid 1970s to less than 20 TBq/a since 1986 (Gray et al. 1995). The 2001 ^{137}Cs values are only slightly higher than the decay corrected (for 2001) global background value of 2.03 Bq/m^3 for North Atlantic seawater (Dahlgaard et al. 1995), which may be due to a combination of the transfer of Chernobyl affected water masses from the Baltic, Norwegian coastal areas and Barents Sea and continued low level discharges from the European reprocessing plants.

As was observed for ^{99}Tc, ^{137}Cs activity concentrations in surface and sub-surface seawater samples from the Svalbard western fjords and in the WSC were similar (Table 1), confirming the influence of the WSC on the western Svalbard environment and a deep well mixed layer at the time of sampling.

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

Pu-238, $^{239+240}$Pu and ^{241}Am in seawater

Activity concentrations of ^{238}Pu in 2001 in surface seawater off the west coast of Svalbard ranged from < 0.3 to 0.7 ± 0.6 mBq/m^3, whilst average activity concentrations of $^{239+240}$Pu and ^{241}Am were 7 ± 3 mBq/m^3 (range 5.6 to 8.9 mBq/m^3) and 1.1 ± 0.8 mBq/m^3 (range 0.6 to 2.4 mBq/m^3) respectively (Fig. 4). These values are similar to activity concentrations of ^{238}Pu (< 0.4 mBq/m^3), $^{239+240}$Pu (7.2 ± 1.1 mBq/m^3) and ^{241}Am (0.8 ± 0.7 mBq/m^3) in a seawater sample collected in Kongsfjorden in 2000. For $^{239+240}$Pu, these values are in good agreement with the range of previously reported values for the Svalbard area and Barents Sea of 6.5 to 9.9 mBq/m^3 (Kershaw et al. 1999; Rudjord et al. 2001). Furthermore, the average ^{238}Pu/$^{239+240}$Pu activity ratio of 0.087 ± 0.047 in surface seawater in 2001 is comparable to the range of 0.07 to 0.08 observed by Hallstadius et al. (1986) west of Spitsbergen. Despite the large uncertainties, the order of magnitude of these ratios would suggest that global fallout is the dominant source. However, we cannot exclude the possibility of non-fallout contributions from the European reprocessing plants including the remobilization of Pu isotopes from Irish sea sediments as suggested by Hallstadius et al. (1986).

The activity concentration of ^{241}Am in the WSC (2.4 ± 0.6 mBq/m^3) in 2001 is comparable to the reported average in the northern seas of 2.1 ± 0.9 mBq/m^3, which was ascribed to the radioactive decay of global fallout ^{241}Pu and not to plutonium produced at Sellafield (Hallstadius et al. 1986). Am-241 activity concentrations appear to decrease on approach to the Svalbard coast (Fig. 4); this is possibly due to a greater flux of ^{241}Am to the particulate phase nearer Svalbard through association with terrestrially derived suspended particulate matter. The ^{241}Am/$^{239+240}$Pu activity ratios in surface seawater ranging from 0.09 to 0.27 in 2001 are similar to an average activity ratio of 0.13 ± 0.04 previously reported for the northern seas (Holm et al. 1983). Activity concentrations of $^{239+240}$Pu and ^{238}Pu/$^{239+240}$Pu activity ratios were similar in sub-surface and surface seawater samples in Kongsfjorden, whereas activity concentrations of ^{241}Am and ^{241}Am/$^{239+240}$Pu activity ratios were higher in the sub-surface sample. The reverse was true for the WSC, where higher $^{239+240}$Pu and ^{238}Pu/$^{239+240}$Pu activity ratios were observed in the sub-surface sample, as has been observed previously in the Barents Sea (Rudjord et al. 2001), whilst activity concentrations of ^{241}Am and ^{241}Am/$^{239+240}$Pu activity ratios were similar in both sub-surface and surface seawater samples.

Tc-99 and ^{137}Cs in marine algae

Results from marine algae (brown algae only), collected from Kongsfjorden in 2000, 2001 and 2002 and from Bjørnøya and Hopen in 2001, are summarized in Table 2. Data for *Fucus distichus* represent bulked samples, whilst data for *Laminaria* spp. represent individual specimens.

For each year and location, ^{99}Tc activity concentrations in *F. distichus* (25.7 to 58.7 Bq/kg) were generally higher than those in *Laminaria* spp. (5.8 to 28.9 Bq/kg), as has been previously

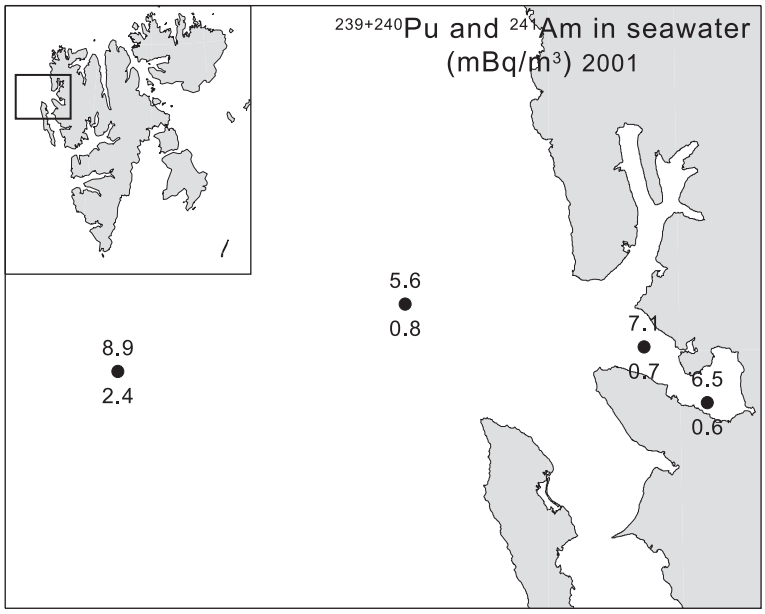


Fig. 4. Surface activity concentrations of $^{239+240}$Pu (above) and ^{241}Am (below) in seawater off western Svalbard in May 2001 (both mBq/m^3).

reported for samples from coastal areas around mainland Norway (Kolstad & Lind 2002). In addition, ^{99}Tc activity concentrations in marine algae from Hopen and Bjørnøya were generally higher compared to those observed in Kongsfjorden for the same year. These differences are likely to be the result of the higher ^{99}Tc activity concentrations in the marine environment around Bjørnøya and Hopen compared to Kongsfjorden. Similarly, ^{99}Tc activity concentrations in *F. distichus* from the Svalbard area were 7 to 8 fold lower than reported average values in *F. vesiculosus* (which were 318 ± 36 Bq/kg (d.w.) in 2000 and 321 ± 68 Bq/kg (d.w.) in 2001) from a coastal site on the northern Norwegian mainland (Kolstad & Lind 2002). This is consistent with the observed gradient in ^{99}Tc activity concentrations in seawater between the Svalbard area (lower ^{99}Tc) and mainland coastal sites in Norway (higher ^{99}Tc). However, ^{99}Tc activity concentrations in *F. distichus* and *Laminaria* spp. in 2000 to 2002 were up to 7 and 18 fold higher respectively than average values in *Fucus* spp. and *Laminaria* spp. from the Svalbard area in 1980 and 1981 (Holm et al. 1984).

Average ^{99}Tc concentration factors (CFs) in seaweed samples (defined as Bq/kg dry weight in biota per Bq/l in seawater) from Kongsfjorden, Bjørnøya and Hopen were calculated using single or, where possible, average concentrations of ^{99}Tc in seawater. Since the inferred CFs represent snapshot and not equilibrium values, care must be

Table 2. Activity concentrations and associated uncertainties (1σ) (Bq/kg d.w.) and concentration factors (CF) of ^{99}Tc in marine algae from Kongsfjorden (K), Hopen (H) and Bjørnøya (B) collected in 2000, 2001 and 2002. n = 1 unless otherwise indicated. Average values given where (a) n = 5, (b), n = 2 and (c) n = 3.

	2000		2001		2002	
Species	^{99}Tc (Bq/kg)	CF	^{99}Tc (Bq/kg)	CF	^{99}Tc (Bq/kg)	CF
Fucus distichus (K)	34.3 ± 3.3	137000	42.9 ± 1.9[a]	156000[a]	28.6 ± 2.1[b]	168000[b]
Fucus distichus (H)	-	-	58.7 ± 5.8	280000	-	-
Fucus distichus (B)	-	-	26.6 ± 2.6	140000	-	-
Laminaria spp. (K)	6.8 ± 0.5[c]	32000[c]	5.8 ± 0.4[b]	16000[b]	28.9 ± 2.8	120000
Laminaria sp. (H)	-	-	15.1 ± 1.5	72000	-	-
Laminaria sp. (B)	-	-	25.1 ± 4.6	132000	-	-

taken in their use. However, the range in ^{99}Tc CFs in *F. distichus* from the Svalbard area during 2000 to 2001, of 1.4×10^5 to 2.8×10^5, is in good agreement with the previously reported range of values of 1.5×10^5 to 2.6×10^5 for *F. vesiculosus* between 1998 and 2001 from a coastal site on the northern Norwegian mainland (Kolstad & Lind 2002). Likewise, ^{99}Tc CFs for *Laminaria* spp. from the Svalbard area in the sampling period are, with the exception of two samples, of the same order of magnitude as a 2001 value of 1.4×10^4 for a *Laminaria* sp. from Lista in southern Norway (Kolstad & Lind 2002). Cs-137 activity concentrations in all species of marine algae sampled from Kongsfjorden in 2001 and 2002 were below the analytical minimum detectable activity, with the exception of one sample of *F. distichus* collected in 2001 (1.3 ± 0.3 Bq/kg) and one sample of *Laminaria* sp. collected in 2002 (0.7 ± 0.1 Bq/kg). Cs-137 activity concentrations in all samples of marine algae collected from Hopen and Bjørnøya in 2001 that were above the minimal detectable activity were in the range 0.4 to 1.2 Bq/kg. Svalbard marine algae ^{137}Cs values from 2000 to 2002 are comparable to those of ^{137}Cs in *F. vesiculosus* collected from mainland northern Norway between 1997 and 1999 of 0.2 to 0.8 Bq/kg d.w. (Rudjord et al. 2001). The similarity between ^{137}Cs activity concentrations in marine algae in the Svalbard area and mainland northern Norway, as opposed to the 7 to 8 fold difference in ^{99}Tc values, is a reflection of the similar ^{137}Cs seawater concentrations and lower CFs for ^{137}Cs in brown algae compared to CFs for ^{99}Tc.

Anthropogenic and natural radionuclides in soil

The average ^{137}Cs activity concentration in surface soils (top 3 cm) in the Svalbard area in 2001 was 21 ± 1 Bq/kg. On Spitsbergen, the average ^{137}Cs activity concentration in surface soils was 14 ± 1 Bq/kg (range 0.9 to 39 Bq/kg) inside the Kongsfjorden area and 34 ± 1 Bq/kg (range < 0.5 to 63 Bq/kg) in other areas (Tables 3 and 4). Cs-137 activity concentrations in soil samples from Hopen and Bjørnøya were in the range of 5.4 to 42 Bq/kg. These ^{137}Cs activity concentrations are in good agreement with previously reported values for the Svalbard area of < 1.5 to 35.8 Bq/kg (Negoita 1997, 1999).

A series of soil cores taken in the Kongsfjorden area in 2001 (Table 4) showed that ^{137}Cs decreased rapidly with depth, with ca. 80 % of ^{137}Cs residing

Table 3. Activity concentrations and associated uncertainties (1σ) of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K in the top 3 cm of soil from locations on Spitsbergen outside of Kongsfjorden and Bjørnøya and in the top 5 to 10 cm from Hopen in 2001. n = 1 unless otherwise indicated. Average values given where (a) n = 3 and (b) n = 2.

Location	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Northern Spitsbergen					
Liefdefjorden[a]	31 ± 1	36 ± 3	37 ± 3	47 ± 1	776 ± 29
Woodfjorden[a]	51 ± 2	43 ± 3	38 ± 3	41 ± 1	690 ± 33
Wijdefjorden[b]	35 ± 2	39 ± 4	47 ± 6	55 ± 1	773 ± 28
Southern Spitsbergen					
Longyearbyen	< 0.7	34 ± 8	39 ± 10	39 ± 1	706 ± 23
Semmeldalen	3 ± 1	47 ± 5	70 ± 8	46 ± 2	724 ± 31
Bjørnøya	42 ± 1	17 ± 4	30 ± 5	14 ± 1	170 ± 20
Hopen (0 - 5 cm)	5 ± 1	36 ± 6	38 ± 5	43 ± 1	665 ± 21
Hopen (0 - 10 cm)	7 ± 3	43 ± 4	47 ± 6	43 ± 2	646 ± 49

Table 4. Average activity concentrations and associated uncertainties (1σ) of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K (Bq/kg d.w.) and ^{226}Ra/^{238}U ratios in soil cores collected within Kongsfjorden in 2001.

Depth (cm)	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)	^{226}Ra/^{238}U
0 - 3 (n = 10)	14 ± 1	33 ± 2	41 ± 2	26 ± 1	338 ± 6	1.40
3 - 6 (n = 10)	2.0 ± 0.1	38 ± 2	40 ± 2	27 ± 1	342 ± 7	1.07
6 - 9 (n = 9)	1.0 ± 0.3	42 ± 2	44 ± 2	31 ± 1	387 ± 7	1.04
9 - 12 (n = 5)	1.0 ± 0.4	40 ± 3	42 ± 4	29 ± 1	350 ± 12	1.03

within the top 3 cm of soil. The lack of any discernable downward migration is consistent with previous assessments of the vertical distribution of ^{137}Cs in Arctic tundra (e.g. Taylor et al. 1988; Baskaran et al. 1991; Strandberg 1997). A ^{137}Cs soil/vegetation deposition value of 1.6 kBq/m^2, calculated from a high resolution soil core (12 cm deep with a surface area of 357 cm^2) in Kongsfjorden in 2002, is in good agreement with reported decay corrected (2001) estimated deposition values of 0.4 to 2.2 kBq/m^2 (AMAP 1997; UNSCEAR 2000). As this ^{137}Cs deposition value is similar in magnitude to the decay corrected (2001) pre-Chernobyl Svalbard soil deposition values of 0.9 to 1.4 kBq/m^2 measured by Kjos-Hanssen & Tørresdal (1982) and Hallstadius et al. (1982), this suggests that atmospheric weapon test fallout is the dominant source of ^{137}Cs in this area. For comparison, the 1998 ^{137}Cs deposition values in soil in northern Norway (Troms and Finnmark) were 0.086 to 6.2 kBq/m^2 (JRNEG 2002). The higher values measured in northern Norway indicate the greater contribution of fallout ^{137}Cs from the Chernobyl accident at these lower latitudes.

The range of activity concentrations of the natural radionuclides ^{238}U (17 to 72 Bq/kg d.w.), ^{226}Ra (21 to 70 Bq/kg d.w.), ^{232}Th (10 to 57 Bq/kg d.w.) and ^{40}K (115 to 818 Bq/kg d.w.) found in soil samples taken from across the Svalbard area were in broad agreement with average Norwegian and global concentrations (UNSCEAR 2000) and displayed little variation with respect to sample location. Despite available information on the geology of the area (Hjelle 1993), the lack of information on permafrost depth at individual sampling sites prevents interpretation of how natural radionuclide activity concentrations in the soil reflect those in the underlying lithology. Unlike ^{137}Cs, the four natural radionuclides, ^{238}U, ^{226}Ra, ^{232}Th and ^{40}K, showed little variation in activity concentration with soil depth. This may indicate that the processes that normally govern vertical redistribution of these radionuclides have either been diminished or retarded by the influence of the climate of the area, the presence of permafrost below the active layer and the relatively low level of biological activity in the soils for much of the year. This hypothesis is supported by the constancy of the ^{226}Ra/^{238}U ratio with depth, the average ratio being 1.4 at the surface of the soil column and approaching 1.0 at depths of 9 to 12 cm.

The primary processes responsible for disruption of the ^{226}Ra/^{238}U secular equilibrium in surface soils are leaching, dissolution and precipitation via the action of percolating ground or surface water (Ivanovich 1994). The low ^{226}Ra/^{238}U ratios exhibited by the Svalbard soils and the lack of variation with depth appear to indicate the absence or retardation of processes that are responsible for segregation of decay chain radionuclides in more temperate climates.

Cs-137 in vegetation

Samples of mosses, lichens and flowering plants were collected from around Kongsfjorden in 2001 and 2002 and from Bjørnøya and Hopen in 2001 (Table 5). Due to limitations of sample sizes, the reported activity concentrations for all terrestrial plants represent bulked samples. The average ^{137}Cs activity concentrations in Svalbard mosses, lichen and flowering plants were 124 ± 8 Bq/kg, 82 ± 2 Bq/kg and 54 ± 1 Bq/kg respectively. The highest activity concentrations observed in each of the three vegetation types were 292 ± 70 Bq/kg in *Racomitrium ericoides* (moss), 140 ± 4 Bq/kg in *Cetraria nivalis* (lichen) and 109 ± 3 Bq/kg in *Cassiope tetragona* (flowering plant).

The 2001 and 2002 activity concentrations of

Table 5. Activity concentrations and associated uncertainties (1σ) of ^{137}Cs (Bq/kg d.w.) in bulked samples of mosses, lichens and flowering plants from Kongsfjorden (K), Hopen (H) and Bjørnøya (B) collected in 2001 and 2002. n = 1 unless otherwise indicated. Average values given where (a) n=2 and (b) n=6.

Species	^{137}Cs (Bq/kg)
Moss	
Racomitrium ericoides (K)	292 ± 70
Ditrichum flexicaule (K)	216 ± 36
Bryum sp. (K)[a]	145 ± 22
Amphidium lapponum (K)	29 ± 1
Sanionia uncinata (K)	117 ± 4
Hygrohypnum polare (H)[a]	63 ± 3
Aulocomnium turgidum (H)	245 ± 21
Sanionia uncinata (B)[a]	61 ± 2
Lichen	
Cetraria nivalis (K)[a]	108 ± 2
Cladonia mitis (B)	30 ± 1
Flowering plants	
Silene acaulis (K)[a]	33 ± 1
Cassiope tetragona (K)	109 ± 3
Carex nardina (K)	64 ± 2
Dryas octopetala (K)[b]	61 ± 4

^{137}Cs in vegetation are in reasonable agreement—allowing for decay correction and uncertainties over plant ages—with the limited number of previously reported ^{137}Cs activity concentrations in terrestrial plants on Svalbard from the early 1980s (Kjos-Hanssen & Tørresdal 1982; Holm et al. 1983; Aarkrog et al. 1984). ^{137}Cs activity concentrations were generally higher in mosses and lichens than in flowering plants, reflecting their ability to collect ^{137}Cs due to their large surface area and nutrient uptake characteristics. Considered together, ^{137}Cs activity concentrations in terrestrial flora on Svalbard are similar to those recently reported in a range of terrestrial flora from Finnmark, northern Norway, but up to 2 fold lower than those reported for flora collected further south in Troms, northern Norway (JRNEG 2002).

Areas of elevated radioactivity

Several areas of localized enrichment of radionuclides within soil were identified in the Kongsfjorden area (Table 6). These areas were primarily associated with the presence of large colonies of seabirds, for example at Krykkefjellet close to the Ny-Ålesund settlement and at sites on the Brøgger peninsula, where both ^{137}Cs and ^{238}U activity concentrations were elevated in well drained soils compared to soils remote from seabird colonies within the Svalbard area. At the Krykkefjellet seabird colony, mounds of aggregated faecal and nesting material (AFNM) have developed directly below the seabird colony and the leaching of nutrients from the AFNM mounds has led to the formation of a thin peat soil at the base of these mounds. Cs-137 activity concentrations in the AFNM mounds were up to 3 fold higher than those observed in well drained soils at this site, whilst activity concentrations of ^{238}U and ^{226}Ra were 3 to 4 fold lower. Activity concentrations of these radionuclides appear to increase with depth within the mounds of AFNM, especially for ^{137}Cs. This suggests that either the input of these radionuclides has reduced over time or that the radionuclides are being leached from surface layers by percolating water. In the peat soil at the base of the AFNM mounds, ^{137}Cs and ^{238}U activity concentrations were up to 2 fold higher than those observed in the AFNM and up to 20 and 8 fold higher respectively than average values for typical Kongsfjorden soils. Cs-137 and ^{238}U activity concentrations in the underlying mineral soil were up to 30 and 9 fold lower than those

Table 6. Activity concentrations and associated uncertainties (1σ) of ^{137}Cs, ^{238}U, ^{226}Ra, ^{232}Th, ^{40}K, ^{238}Pu, $^{239+240}$Pu and ^{241}Am (Bq/kg d.w.) and activity ratios in terrestrial matrices associated with seabird colonies on the Brøgger peninsula and at Krykkefjellet, Kongsfjorden, compared to a sample of Kongsfjorden soil from a site remote from any seabird colony. AFNM: aggregated faecal and nesting material. Average values given where n > 1. Table continues on opposite page.

	^{137}Cs (Bq/kg)	^{238}U (Bq/kg)	^{226}Ra (Bq/kg)	^{232}Th (Bq/kg)	^{40}K (Bq/kg)
Kongsfjorden – well drained soil (n = 1)					
0 - 3 cm	6 ± 2	26 ± 7	38 ± 8	31 ± 1	542 ± 17
Brøgger – well drained soil (n = 2)					
0 - 10 cm	48 ± 1	59 ± 4	41 ± 6	7 ± 1	116 ± 9
0 - 5 cm	101 ± 3	47 ± 7	18 ± 7	7 ± 1	55 ± 8
Krykkefjellet – well drained soil (n = 3)					
0 - 5 cm	76 ± 1	161 ± 5	173 ± 9	24 ± 1	380 ± 15
Krykkefjellet – AFNM (n = 1 except (a) where n = 8)					
0 - 10 cm	103 ± 2[a]	72 ± 3[a]	64 ± 3[a]	20 ± 1[a]	392 ± 8[a]
0 - 15 cm	85 ± 3	65 ± 8	64 ± 10	19 ± 1	394 ± 29
15 - 30 cm	69 ± 2	56 ± 7	61 ± 8	18 ± 1	406 ± 13
30 - 45 cm	80 ± 2	73 ± 4	55 ± 7	18 ± 1	451 ± 14
45 - 60 cm	148 ± 4	69 ± 5	94 ± 8	21 ± 1	534 ± 40
Krykkefjellet – peat soil (n = 6)					
	207 ± 2	102 ± 5	51 ± 4	31 ± 1	516 ± 13
Krykkefjellet – mineral soil underneath peat soil (n = 6)					
	42 ± 1	34 ± 2	27 ± 3	29 ± 1	640 ± 9

in the peat soil, demonstrating the strong association of these radionuclides with the upper peat layers. The distribution of the radionuclides in the different soil types below the seabird colony suggests that elevated activity concentrations in the peat soil originate from the leaching of radionuclides from the AFNM mounds. This is supported by the disruption of the $^{226}Ra/^{238}U$ secular equilibrium within the peat soil, with an average $^{226}Ra/^{238}U$ ratio of 0.64, compared to a ratio of ca. 1 for the underlying mineral layer.

Activity concentrations of ^{238}Pu, $^{239+240}Pu$ and ^{241}Am in AFNM mounds below the Krykkefjellet seabird colony were up to 33 fold higher than those in a typical Kongsfjorden soil, whilst those in peat soil at the base of the AFNM mounds were up to 73 fold higher. Despite these elevated values, the average $^{238}Pu/^{239+240}Pu$ and $^{241}Am/^{239+240}Pu$ activity ratios for these samples are of a similar magnitude to the global fallout ratios for Svalbard of 0.025 and 0.37 respectively (Hardy et al. 1973; Holm et al. 1983). In addition, $^{239+240}Pu/^{137}Cs$ activity ratios in both AFNM and peat soil below the Krykkefjellet colony are similar to a $^{239+240}Pu/^{137}Cs$ activity ratio determined for a typical Kongsfjorden soil and to an estimated global fallout $^{239+240}Pu/^{137}Cs$ activity ratio of 0.03 between 70 and 80°N (Hardy et al. 1973; UNSCEAR 2000).

The elevated concentrations of radionuclides at the base of seabird colonies on Svalbard suggest that there is either a greater flux of radionuclides into these particular terrestrial environments and/or that radionuclides are retained to a greater degree at these sites compared to other terrestrial environments away from seabird colonies. The formation of nutrient enriched soils through guano deposition at the base of seabird colonies in Arctic areas has been observed both on Spitsbergen and on Novaya Zemlya (Goryachkin et al. 1998), whilst the ability of such nutrient enriched soils to adsorb and enrich various radionuclides has been well documented (Owen 1991; Dowdall & O'Dea 1999; Schleich et al. 2000). A possible source of additional radionuclides may arise from the seabirds themselves, as Headley (1996) and Godzik (1991) previously ascribed elevated levels of heavy metals in soils near Svalbard seabird colonies to their transfer from the marine environment via the consumption of marine organisms by seabirds and subsequent deposition of guano on the terrestrial environment. There is some evidence that the levels of radionuclides, such as ^{210}Po and ^{210}Pb, in decapod shrimp and other pelagic organisms from the Atlantic Ocean can be in the order of kBq/kg (Cherry & Heyraud 1981) and it is therefore feasible that the elevated activity concentrations of radionuclides near seabird colonies were introduced in a similar fashion.

In contrast to the AFNM and peat soil, average ^{137}Cs activity concentrations in mosses and flowering plants growing beneath the Krykke-

^{238}Pu (Bq/kg)	$^{239+240}Pu$ (Bq/kg)	^{241}Am (Bq/kg)	$^{238}Pu/^{239+240}Pu$	$^{241}Am/^{239+240}Pu$	$^{239+240}Pu/^{137}Cs$
0.01±0.01	0.20±0.03	0.05±0.02	0.056±0.035	0.27±0.09	0.035±0.013
-	-	-	-	-	-
-	-	-	-	-	-
-	-	-	-	-	-
0.13±0.02	3.6±0.3	1.6±0.1	0.036±0.006	0.44±0.05	0.028±0.002
-	-	-	-	-	-
-	-	-	-	-	-
-	-	-	-	-	-
0.14±0.02	4.1±0.3	1.8±0.2	0.036±0.006	0.43±0.05	0.027±0.002
0.25±0.02	6.9±0.2	2.8±0.1	0.038±0.003	0.42±0.02	0.033±0.001
-	-	-	-	-	-

fjellet seabird colony (Table 7) were 4 fold and 2 fold respectively lower than average values in mosses and flowering plants taken from other areas in Svalbard. It is possible that this observation is an artefact because of the different species compared, but it could also be the result of increased plant growth and associated grazing pressures, which are thought to be higher at nutrient enriched seabird colony sites (Eurola & Hakala 1977). Alternatively, the unique nature of the substrates at the bird colonies, with respect to nutrient content, may have some impact on how this nuclide is taken up by vegetation.

Conclusions

Results of radiometric monitoring on the Svalbard archipelago indicate that anthropogenic radioactive contamination is generally low, as per the wider Arctic region. Although activity concentrations of ^{99}Tc in Svalbard seawater have increased since the early 1990s, in association with increased discharges from the Enhanced Actinide Removal Plant at Sellafield (UK), they are an order of magnitude lower than activity concentrations reported from coastal waters surrounding the Norwegian mainland. A similar trend is observed in the ^{99}Tc activity concentrations in seaweed from Svalbard and the Norwegian mainland, but contemporary values in seaweed from Svalbard are higher than those in the early 1980s. Pu isotope activity ratios in seawater suggest global fallout as the dominant source, while declining ^{137}Cs activity concentrations in seawater are probably maintained above global fallout values by the oceanic import of European coastal run-off containing ^{137}Cs from Chernobyl.

The terrestrial environment is primarily contaminated with Pu isotopes and ^{137}Cs from the historical atmospheric nuclear weapon testing programmes and to a lesser extent, ^{137}Cs from the Chernobyl accident in 1986. Typically, ^{137}Cs activity concentrations are higher in certain Arctic flora (e.g. mosses and lichens) than in the surrounding soils. Of particular interest is the occurrence of localized enrichments of radionuclides in areas associated with seabird colonies. Further study is required to assess the implications, if any, such enrichment processes may have on local ecosystems at present and in the event of possible future contamination.

Continued monitoring of both the marine and terrestrial environment in the Svalbard region is required for a number of reasons. Only through regular monitoring can information be provided on trends in radionuclide levels and the long-term consequences of existing and future radioactive contamination of the Svalbard region. Furthermore, impact assessments of anthropogenic contaminants are largely achieved by determining the dose delivered by the contaminant. This assessment can only be considered meaningful when conducted in the context of the natural background dose, upon which the contaminant dose is superimposed. Information and data on background doses is somewhat sparse for High Arctic environments. This situation requires greater attention, if effective radiological protection is to be provided following any significant rise in radioactive contamination.

Table 7. Activity concentrations and associated uncertainties (1σ) in bulked samples of mosses and flowering plants from the Krykkefjellet seabird colony, Kongsfjorden, collected in 2001 and 2002 (Bq/kg d.w.). n = 1 unless otherwise indicated. Average values given where (a) n = 20, (b) n = 2 and (c) n = 3.

Species	^{137}Cs (Bq/kg)
Moss	
Bryum sp.	11 ± 3
Dicranoweisia sp.	16 ± 1
Sanionia uncinata[a]	40 ± 1
Flowering plants	
Cochlearia groenlandica	59 ± 2
Saxifraga cernua[b]	13 ± 1
Deschampsia alpina[c]	17 ± 1

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