StrålevernRapport ▪ 2007:10

INSTITUTE OF MARINE RESEARCH

Radioactivity in the Marine Environment 2005

Results from the Norwegian Marine Monitoring Programme (RAME)

Statens strålevern
Norwegian Radiation Protection Authority

Reference:
NRPA. Radioactivity in the Marine Environment 2005. Results from the Norwegian National Monitoring Programme (RAME). StrålevernRapport 2007:10. Østerås: Norwegian Radiation Protection Authority, 2007.

Authors:
Gäfvert T[1], Sværen I[2], Gwynn J[1], Brungot A L[1], Kolstad A K[1], Lind B[1], Alvestad P[2], Heldal H E[2], Strålberg E[3], Christensen G C[3], Drefvelin J[1], Dowdall M[1], Rudjord A L[1]
[1]Norwegian Radiation Protection Authority, [2]Institute of Marine Research, [3]Institute of Energy Technology

Key words:
Radioactivity, marine environment, RAME, monitoring, Norway

Abstract:
This report presents results of monitoring of radioactivity in seawater and biota collected along the Norwegian coast, the Barents Sea, the North Sea and the Skagerrak in 2005. An overview of discharges from Norwegian sources and data concerning the long-range transport of radionuclides from European nuclear facilities is included.

Referanse:
Statens strålevern. Radioaktivitet i det marine miljø. Resultater fra det nasjonale overvåkningsprogrammet (RAME) 2005. StrålevernRapport 2007:10. Østerås: Statens strålevern, 2007. Språk: engelsk.

Forfattere:
Gäfvert T[1], Sværen I[2], Gwynn J[1], Brungot A L[1], Kolstad A K[1], Lind B[1], Alvestad P[2], Heldal H E[2], Strålberg E[3], Christensen G C[3], Drefvelin J[1], Dowdall M[1], Rudjord A L[1]
[1]Statens strålevern, [2]Havforskningsinstituttet, [3]Institutt for energiteknikk

Emneord:
Radioaktivitet, marin miljø, RAME, overvåking, Norge

Resymé:
Rapporten inneholder resultater fra overvåkningen av radioaktivitet i sjøvann og biota i 2005 langs norskekysten og i Barentshavet, Nordsjøen og Skagerrak. En oversikt over utslipp fra norske kilder og utslippsdata fra europeiske nukleære anlegg som er relevante for langtransport av radioaktivitet til norske havområder er inkludert i rapporten.

Head of project: Anne Liv Rudjord
Approved:

Per Strand

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

---

42 pages
Published 2007-12-01
Printed number 200 (07-12)
Cover design: Lobo media, Oslo
Printed by Lobo media

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 2007:10

Radioactivity in the Marine Environment 2005
Results from the Norwegian Marine Monitoring Programme (RAME)

Statens strålevern

Norwegian Radiation
Protection Authority
Østerås, 2007

Contents

1 Introduction

The issue of present and potential radioactive contamination in the marine environment has received considerable attention in Norway in recent years. In the late 1980s several accidents and incidents involving nuclear-powered submarines, demonstrated that the risk of the release of radionuclides into the Barents Sea should be considered more carefully. In particular, it became evident that better documentation concerning the radioactivity levels in fish and other seafood was important for the seafood export industries. Furthermore, in the early 1990s, information concerning the dumping of nuclear waste emerged through bilateral environmental cooperation between Norway and Russia. In the years that followed, concern grew regarding the safety of military and civil nuclear installations in the northwest of Russia. This concern was associated not only with possible reactor accidents, but also with the prolonged or sudden release of radionuclides from radioactive waste facilities.

In addition to the potential threats outlined above, radionuclides originating from nuclear weapons fallout, the Chernobyl accident and waste discharged from European reprocessing facilities have been detected in the Norwegian marine environment. In 1994 and 1995, the discharge of ^{99}Tc from the reprocessing facility at Sellafield in the UK increased sharply, and although this discharge has been reduced, it has continued at a high level up to 2003. The discharge of ^{99}Tc was substantially reduced from 2004, but it will take 3-4 years before this is observable in Norwegian waters. There has been much public concern about the consequences of such kinds of release, as the radionuclides discharged to the Irish Sea are transported by ocean currents via the North Sea into the Norwegian coastal current and to the Barents Sea. In response to this concern, programmes for the monitoring of radioactivity in the marine environment have been established. Due to the economic importance of the fishing industry and its vulnerability to contamination, as well as any rumours of radioactive contamination, one of the main objectives of these programmes is to document levels and trends of radionuclides in the Norwegian marine environment.

Other industrial activities, such as mining and oil production, may change the distribution of naturally occurring radionuclides in the environment. The discharge of radium from water produced by oil installations is one area that has recently received special attention.

In Norway there are currently two monitoring programmes concerned with radioactivity in the marine environment, both coordinated by the Norwegian Radiation Protection Authority (NRPA). One is funded by the Ministry of the Environment and focuses on monitoring of radioactivity in the marine environment both in coastal areas and in the open seas, the other by the Ministry of Fisheries which focuses on monitoring of radioactivity in commercially important fish species.

The marine monitoring programme includes the compilation of discharge data from Norwegian sources, in addition to the collection of discharge data relevant for the long-range transport of radionuclides from various sources. Liquid discharge data for 2005 from nuclear installations and recent trends in such discharges are summarised in Chapter 2.

During 2005, samples for monitoring radioactivity in the marine environment were collected in the Barents Sea, the North Sea and at permanent coastal stations along the Norwegian coastline. Results from the analysis of these samples are presented in Chapters 4 and 5. In Chapter 6, a summary of the findings and the conclusions is given. In the Appendix, technical information regarding sample preparation techniques and analytical methods employed in the laboratories are presented.

2 Sources of radionuclides in the marine environment

This chapter provides updated information and an overview of radionuclide discharges and other sources that are relevant to the Norwegian marine environment. The collection and updating of this information is an integral part of the marine monitoring programme RAME (Radioactivity in the Marine Environment), the main purpose of which is to present recent trends in radionuclide discharges and other sources of radioactivity in the marine environment. Information on discharges from Norwegian sources and on long-range transport of radionuclides from various distant sources is included. Anthropogenic radionuclides are discharged from the nuclear facilities of the Institute for Energy Technology, IFE Kjeller and IFE Halden. Discharges from these facilities are authorised by the NRPA. Unsealed radioactive materials used in medicine and science will primarily be released to the marine environment via sewage treatment plants. The utilisation of such unsealed sources is regulated through guidelines issued by the NRPA.

In recent years, increased attention has been devoted to releases containing elevated levels of naturally occurring radionuclides. Such releases are the result of offshore oil production and, probably, leaching of mine tailings.

The discharge of radionuclides from Norwegian sources is only detectable in the local environment, near the discharge point. The long-range transport of radionuclides originating from fallout from atmospheric nuclear weapons tests (conducted mainly in the 1950s and 1960s), the Chernobyl accident in 1986 and from reprocessing of nuclear fuel are still the main contributors to the general levels of anthropogenic radionuclides found in Norwegian waters.

2.1 Discharges of anthropogenic radionuclides from Norwegian sources

2.1.1 IFE Kjeller and Amersham Health

IFE Kjeller is located about 20 km east of Oslo. The facilities include a heavy-water cooled and moderated research reactor with a thermal effect of 2 MW, called JEEP II, a metallurgical laboratory, production facilities for medical radioactive isotopes, and a radioactive waste treatment plant for low-level and intermediate-level waste (LLW and ILW). The liquid LLW originates from these facilities. Liquid effluent is discharged through a designated pipeline to the River Nitelva about 100 km from the sea. The river empties into Lake Øyeren where the water is mixed with the water from the Glomma River. The River Glomma empties into the Oslofjord at the city of Fredrikstad.

A plant manufacturing radiopharmaceutical products, managed by the private company GE Healthcare (former Amersham Health), operates in close collaboration with IFE Kjeller. Authorisation for IFE Kjeller discharges also includes the discharge from this production facility.

Table 2.1. *Liquid discharges (MBq) from IFE Kjeller (IFE 2006a; IFE, 2005a; IFE, 2004a; IFE, 2003a; IFE, 2002a), 2001-2005 (including Amersham Health).*

Nuclide	*2001*	*2002*	*2003*	*2004*	*2005*
^{241}Am	0.35	0.7	0.005	0.003	0.03
^{124}Sb	-	<0.06	<0.2	<0.1	<0.1
^{125}Sb	-	<0.11	2.7	0.075	<0.7
^{144}Ce	-	1.6	3.3	1.1	0.054
^{134}Cs	2.5	2.3	1.2	0.35	0.08
^{137}Cs	25.7	25	22	7.9	0.51
^{3}H	$1.5\cdot10^{6}$	$2.4\cdot10^{6}$	$2.8\cdot10^{6}$	$2.9\cdot10^{5}$	$4.1\cdot10^{5}$
^{59}Fe	-	<0.15	<0.2	<0.3	<0.2
^{125}I	310	350	540	327	50
^{131}I	107	6.0	15	27	39
^{58}Co	-	<0.10	0.26	<0.1	<0.1
^{60}Co	74	60	55	24	6.7
^{51}Cr	-	<0.5	<0.6	<2	<1.3
^{54}Mn	-	<0.14	<0.2	<0.2	<0.2
^{95}Nb	-	0.17	0.42	<0.06	<0.1
^{238}Pu	$3\ 10^{-4}$	0.011	0.002	0.01	0.007
$^{239+240}$Pu	0.04	0.04	0.034	0.15	0.15
^{103}Ru	-	<0.07	0.14	0.10	<0.3
^{106}Ru	-	<0.9	2.0	<4	<2.2
^{90}Sr	1.23	0.55	0.33	2.6	0.34
^{110m}Ag	-	1.2	2.1	0.089	1.2
^{65}Zn	3.8	<0.4	0.27	0.1	8.4
^{95}Zr	-	0.13	0.14	<0.2	<0.2

The discharge limit authorised by the NRPA is based on the annual dose to any member of a critical group of the population along the River Nitelva, and shall not exceed 1 μSv. Each year, IFE Kjeller reports discharge data, the results of their environmental monitoring programme and calculations of effective doses resulting from discharges, to the NRPA. The reported discharges for the period 2001-2005 are summarised in Table 2.1.

Table 2.2. *Effective doses (μSv) to any member of the critical group from river water exposure pathways as reported by IFE Kjeller (IFE 2006a; IFE, 2005a; IFE 2004a; IFE, 2003a; IFE 2002a).*

2001	*2002*	*2003*	*2004*	*2005*
0.21	0.18	0.17	0.07	0.02

Effective doses have been calculated by IFE for an individual in the hypothetical critical group exposed through the annual consumption of 20 kg of fish from the river and 100 hours per year presence on the riverbanks. In 2005, the calculated effective dose to this critical group was 0.02 μSv, corresponding to 2 % of the annual dose limit of 1 μSv. The effective dose to the critical group varies, as can be seen in Table 2.2, corresponding to between 2 and 21 % of the dose limit, with an average of 13 % for the period 2001-2005.

2.1.2 IFE Halden

The Halden Boiling Water Reactor (HBWR) is located in the town of Halden, in the southeast of Norway, close to the Swedish border. The HBWR is heavy-water-cooled and moderated with a thermal effect of 20 MW. The reactor was commissioned in 1959.

IFE Halden is authorised to discharge a limited amount of radioactivity from the research reactor to the River Tista (which empties into the Iddefjord). The discharge limit for liquid waste to the marine environment is based on a dose limit of 1 μSv per year to a hypothetical critical group, and the actual annual discharge is typically 1.5-10 % of the authorised limit. Liquid discharges in the period 2001-2005, and the corresponding doses to the critical group are presented in Tables 2.3 and 2.4.

Table 2.3. *Liquid discharge (MBq) from IFE Halden, 2001 - 2005 (IFE 2006b; IFE, 2005b; IFE, 2004b; IFE, 2003b; IFE 2002b).*

Nuclide	*2001*	*2002*	*2003*	*2004*	*2005*
^{124}Sb	-	0.53	-	-	0.003
^{125}Sb	130	18	0.06	0.002	0.037
^{141}Ce	6.4	0.82	0.29	0.68	0.61
^{144}Ce	14	5.7	3.2	16	8.8
^{134}Cs	2	18	8.8	8.0	18
^{137}Cs	58	110	130	70	130
^{3}H	$2.4\cdot10^{5}$	$1.1\cdot10^{6}$	$2.7\cdot10^{5}$	$5.4\cdot10^{5}$	$5.3\cdot10^{5}$
^{59}Fe	-	0.32	-	0.75	0.88
^{131}I	0.04	0.9	3.6	0.68	0.42
^{109}Cd	0.46	0.33	0.016	-	-
^{58}Co	49	5.1	3.3	8.7	41
^{60}Co	440	88	62	68	99
^{51}Cr	290	150	130	210	180
^{54}Mn	7	0.5	0.18	0.73	2.5
^{56}Mn	-	-	-	2.8	-
^{95}Nb	40	12	8.5	11	12
^{103}Ru	2.8	0.59	0.17	0.45	0.42
^{106}Ru	2	0.18	-	-	-
^{90}Sr	-	-	3.3	2.1	4.9
^{110m}Ag	0.5	0.005	0.17	0.001	0.94
^{95}Zr	16	5.4	3.3	4.9	5.3

The calculation of the effective dose to the critical group is based on the following assumptions:

- An annual consumption of 30 kg of fish from the Iddefjord,
- 200 hours per year exposure on the shore of the fjord,
- 50 hours per year bathing in the fjord, and
- 1000 hours per year boating on the fjord.

Table 2.4. *Effective doses (μSv) to the critical group from marine exposure pathways as reported by IFE Halden, 2000 - 2004 (IFE 2006b; IFE, 2005b; IFE, 2004b; IFE, 2003b; IFE 2002b).*

2001	*2002*	*2003*	*2004*	*2005*
0.09	0.02	0.014	0.016	0.025

2.2 Unsealed radioactive substances in the medical sector

Unsealed radioactive substances are used in hospitals, research laboratories and various industrial activities. According to regulations from 1981, laboratories etc. handling unsealed radioactive substances must be authorised by the NRPA. With this authorisation, laboratories and hospitals are allowed to discharge activity into the sewage system according to predefined limits. On January 1, 2004, new regulations came into force, describing requirements for handling and discharge of unsealed radioactive sources. At the moment all discharges require authorisation.

Unsealed radioactive substances used in medicine dominate the anthropogenic radioactive discharges to the sewage system. Radioactive solutions are given to patients for diagnostic and therapeutic purposes, and enter the sewage systems mainly by the excretion of urine and faeces. The two most used radionuclides in nuclear medicine for diagnostic and therapeutic purposes are ^{99m}Tc (which decays to ^{99}Tc with a half-life of 6 h) and ^{131}I ($T_{1/2}$ 8.04 d). The discharge of these radionuclides has been estimated according instructions published by OSPAR. The percentage of the administered doses that is discharged as liquid waste is:

^{131}I Ablation therapy:
100% of the administered dose.

^{131}I Thyrotoxicosis treatment :
for in-patients, 50% of the administered dose;
for out-patients, 30% of the administered dose;
for patients who cannot be classified as in-patients or out-patients, 50% of the administered dose.

^{99m}Tc/^{99}Tc
100% of the amount of ^{99m}Tc administered has decayed to ^{99}Tc and that it is all discharged.

Under these assumptions the total discharged activity of ^{131}I and ^{99}Tc in 2005 from the medical sector was 1390 GBq and 0.08 MBq, respectively.

2.3 Discharge of produced water containing ^{226}Ra and ^{228}Ra

Large volumes of produced water, containing dissolved ^{226}Ra and ^{228}Ra, are discharged into the sea during oil exploitation. Produced water may be formation water trapped in the reservoirs for millions of years or a mixture of formation water and seawater, when seawater has been injected in order to maintain pressure in the reservoirs during oil and gas production. Although the levels of ^{226}Ra and ^{228}Ra are not technologically enhanced, the natural concentrations of these isotopes in the saline formation water are generally much higher (about 3 orders of magnitude) than the background concentration in seawater (a few mBq per litre).

The concentration of radium in the produced water may change over the lifetime of the well. Injection of large volumes of seawater, to maintain the pressure within the well, may lead to dilution of the radium in the produced water, but at the cost of greater water-to-oil ratios later in the production process.

In 2005 147 10^6 m^3 of produced water was discharged to the marine environment from the Norwegian oil and gas industry, while 32 10^6 m^3 was reinjected in the reservoirs. The total activity of ^{226}Ra, ^{228}Ra and ^{210}Pb discharged was about 390 GBq, 380 GBq and 50 GBq, respectively (OLF, 2006). There is a somewhat higher uncertainty in the discharge figures for ^{228}Ra and ^{210}Pb, since some operators only reported the discharge of ^{226}Ra. Earlier investigations of radium in produced water from the Norwegian continental shelf have shown that the activity concentration of ^{228}Ra is approximately equal to the activity concentration of ^{226}Ra (NRPA, 2005a). For some fields where ^{210}Pb was not reported it was assumed that the activity concentration in the produced water was 0.4 Bq l^{-1} (which was the average for all fields that reported ^{210}Pb activity concentrations in produced water). In Fig. 2.1 the activity of ^{226}Ra discharged from Norwegian oil fields are presented. Slightly below 50 % of the total discharged acitivity of ^{226}Ra was discharged from the Troll B and C platforms.

Photo: NRPA

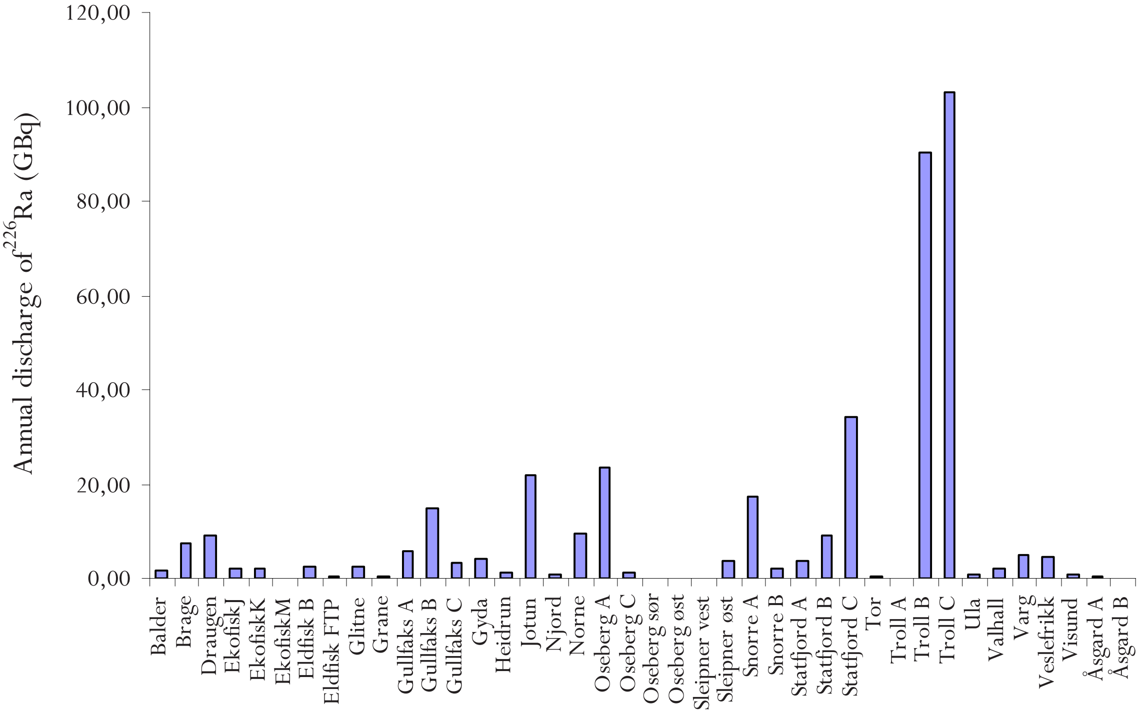

Figure 2.1. *Estimated discharged activity of* ^{226}*Ra from Norwegian oil and gas fields in 2005 (OLF, 2006).*

2.4 Long-range transport of radionuclides

There are several real and potential sources of radioactive contamination of the northeast Atlantic and the Arctic marine environment, as shown in Figure 2.2.

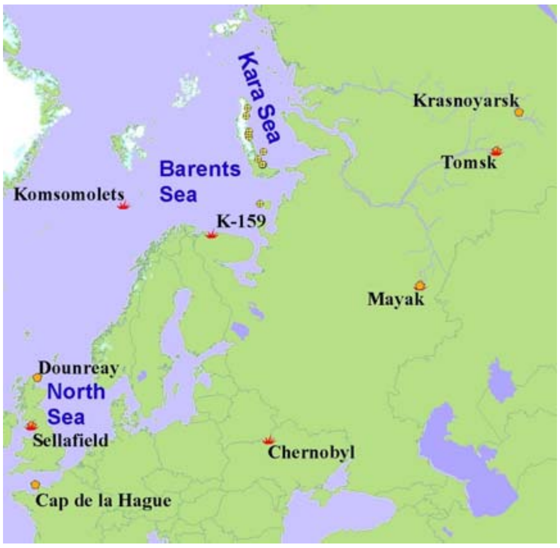

Figure 2.2. *Sources of radionuclides in the northern marine environment: Chernobyl in the Ukraine, the reprocessing plants at Sellafield, Cap de la Hague and Dounreay, the dumping sites for nuclear waste in the Kara Sea, the sites of the sunken submarines Komsomolets and K-159 and Russian nuclear installations (Mayak, Tomsk and Krasnoyarsk) releasing radionuclides to the Russian rivers Ob and Yenisey.*

The primary sources are fallout from atmospheric nuclear weapons testing during the 1950s and 1960s, discharged radionuclides from reprocessing plants and fallout from the Chernobyl accident.

Radioactive contaminants discharged into northern European marine waters can reach Norwegian waters by ocean current transportation. The main ocean currents of the area are presented in Figure 2.3. Some elements, such as Pu and Am, are particle-reactive and will to a large extent sink and end up in the sediments. From the sediments the radionuclides can later be remobilised and transported away by local currents. Remobilised plutonium and ^{137}Cs from Irish Sea sediments contaminated by previous Sellafield discharges are one of the main sources of these elements in Norwegian marine waters. Other elements, such as ^{99}Tc and ^{90}Sr, are not particle-reactive and will follow currents and can be transported large distances away from the discharge point.

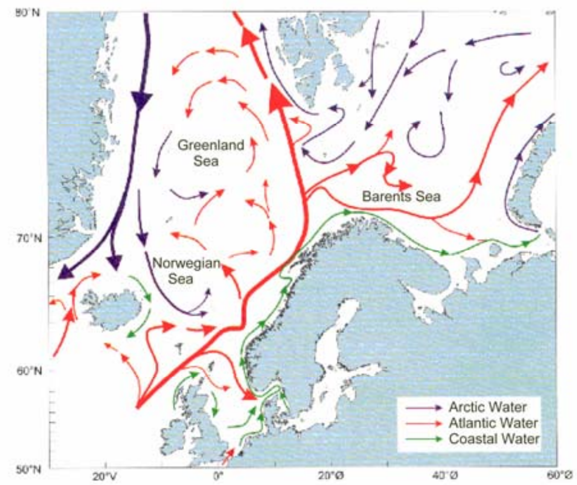

Figure 2.3. *Overview of the main current system in the North Sea, Norwegian Sea, Greenland Sea and Barents Sea (Aure* et al., 1998*).*

2.4.1 Discharge of radionuclides from European nuclear installations

Spent nuclear fuel can be managed in different ways. In a few countries, for instance in France and the UK, spent nuclear fuel is reprocessed. The major part of the waste from these facilities, such as fission products, is stored pending final management and disposal, but a small part is discharged as liquid waste to the marine environment. Discharge limits are set by the relevant national authority.

Sellafield (formerly Windscale) is located on the west coast of Cumbria in the UK. Further information about this reprocessing plant can be found in Amundsen *et al*., (2003).

The other reprocessing plant in Europe affecting Norwegian marine waters is located at Cap de la Hague in France. Except for tritium (^{3}H), the discharge of liquid radioactive waste has generally been lower here than from Sellafield.

Springfields is located on the west coast of the United Kingdom, and liquid waste is discharged to the Ribble estuary. This facility is mainly involved in the manufacture of fuel elements for nuclear reactors and the production of uranium hexafluoride. The Dounreay facilities on the northern coast of Scotland were established in 1955, and have mainly been used in the development of technology for fast breeder reactors. Liquid discharge of radioactive waste from Cap de la Hague, Sellafield, Springfields and Dounreay in 2005 is presented in Table 2.6.

Table 2.6. *Liquid discharge of radionuclides (TBq) from Sellafield, Cap de la Hauge, Springfields and Dounreay to the marine environment in 2005 (OSPAR, 2007 and Environment Agency* et al, *2006).*

Nuclide	Cap de la Hague	Sellafield	Springfields	Dounreay
^{3}H	$1.35\ 10^{4}$	$1.57\ 10^{3}$	-	0.099
Total-α	$2.15\ 10^{-2}$	0.25	0.25	$7.39\ 10^{-4}$
Total-β	11.5	42.9	103	$5.91\ 10^{-3}$
^{14}C	8.27	5.26	-	-
^{60}Co	0.23	0.66	-	-
^{90}Sr	0.50	12.7	-	0.102
^{99}Tc	0.06	6.7	0.06	-
^{106}Ru	5.84	1.85		-
^{129}I	1.40	0.30	-	-
^{134}Cs	$6.1\ 10^{-2}$	0.16	-	-
^{137}Cs	0.71	5.86	-	0.021
^{144}Ce	$7.7\ 10^{-4}$	0.54		-
$^{239+240}Pu$	$1.08\ 10^{-3}$	0.20	-	-
^{241}Pu	0.11	5.5	-	-
^{241}Am	$2.48\ 10^{-3}$	0.034	-	-
^{237}Np	$1.23\ 10^{-4}$	0.05	0.0018	-
^{230}Th	-	-	0.085	-
^{232}Th	-	-	$3.30\ 10^{-4}$	-
U-α	-	-	0.046	-
U	53.4 kg	369 kg	-	-

In addition to the direct discharges from re-processing, the remobilisation of ^{137}Cs and plutonium from contaminated sediments in the Irish Sea acts as a secondary source of radio-nuclides in the marine environment. Substantial discharges of ^{238}Pu, $^{239+240}$Pu, ^{241}Am and ^{137}Cs and other radionuclides in the 1970s and early 1980s resulted in widespread contamination of Irish Sea sediments. It has been shown that these radionuclides can be redissolved and transported out of the Irish Sea. An annual loss of 1.2 TBq $^{239+240}$Pu and 86 TBq ^{137} Cs from the Irish Sea has been estimated by Cook *et al.*, (1997), while Leonard *et al.*, (1999) estimated annual losses of plutonium based on remobilisation from surface sediments to be 0.6 TBq. The half-lives of plutonium and americium in Irish Sea sediments were estimated to be 58 years and ~1000 years, respectively. This implies that remobilisation of radionuclides from Irish Sea sediments is a significant source of anthropogenic radio-nuclides in the North Sea and Norwegian coastal areas, and will remain so for a long time.

2.4.2 Global fallout from nuclear weapons testing

From a global point of view, atmospheric nuclear weapons tests, conducted between 1945 and 1980, are the largest source of radioactive contamination in the environment. The most intensive test period was between 1952 and 1962, during which the USSR and USA were responsible for most of the tests. The last atmospheric nuclear detonation was conducted in China in 1980. In total, it has been estimated that 948 PBq ^{137}Cs, 622 PBq ^{90}Sr, 6.52 PBq ^{239}Pu and 4.35 PBq ^{240}Pu have been released to the environment (UNSCEAR, 2000a), the major part in the northern hemisphere, where most of the tests took place.

Fallout from atmospheric nuclear weapons tests can still be found in the marine environment. About 60 % of the released activity was deposited in the world's oceans. The distribution of the fallout today depends on parameters such as, ocean currents, diffusion and sedimentation rates. Earlier expeditions in the Atlantic have shown that the concentration of ^{137}Cs in surface water is about a few Bq m^{-3} (about twice as much as ^{90}Sr), between 45° S and 45° N, with a slight increase to the north (Bourlat *et al.,* 1996). The activity concentration of $^{239+240}$Pu shows the same trend, but the activity concentrations are about 1000 times lower. Reasons for this are that less plutonium was released, but also that plutonium is more particle reactive and has a higher sedimentation rate, which leads to a shorter residence time in the water.

2.4.3 Outflow of Chernobyl-contaminated water from the Baltic Sea

The worst nuclear accident in history occurred on April 26, 1986 in Chernobyl, Ukraine. In the initial explosion and the following fire, about 3-4 % of the nuclide inventory in the core was released to the atmosphere. The major part consisted of volatile radionuclides such as inert gases, iodine isotopes and ^{134}Cs and ^{137}Cs. Many of the released nuclides had short half-lives and can not be observed in the environment today. The radionuclide that will affect the environment in a long-term perspective is ^{137}Cs, with a physical half-life of 30 years. In total about 85 PBq of ^{137}Cs was released to the environment (UNSCEAR, 2000b). The Baltic Sea was the most affected marine area, where about 5 PBq was deposited (Evans, 1991). Shortly after the accident the activity concentration of ^{137}Cs was very high in part of the Baltic Sea. Due to low water exchange and a constant supply from rivers it will take a relatively long time for the levels to decrease. The average activity concentration of ^{137}Cs in the southern Baltic Sea surface water in 2005 was reported to 43 Bq m^{-3} (Zalewska and Lipska, 2006). Due to the water exchange through the Danish Straits it has been estimated that about 40 TBq flows from the Baltic Sea into the Kattegat annually (Dahlgaard, 2002). The ^{137}Cs will subsequently be transported by the Baltic current which continues as the Norwegian coastal current. This is one of the major sources of ^{137}Cs in the Norwegian marine environment, together with remobilised ^{137}Cs from the seabed of the Irish Sea.

2.5 Other present and potential sources of radioactivity in the northern marine environment

In addition to discharges from the western European reprocessing plants, northern marine waters have also received an input of various radionuclides from Russian reprocessing plants situated on the tributaries of the Russian rivers Ob and Yenisey (see Figure 2.2). These are the Mayak Production Association in the southern Urals and the Siberian Chemical Combine (Tomsk-7) north of the town of Tomsk, and the Krasnoyarsk Mining and Chemical Combine (KMCC). Uncertainty exists regarding the magnitude of radionuclide contributions from these sources, although estimates have been made. For example, between 1958 and 1993 routine discharges from the KMCC are estimated to have led to an input of 30-100 TBq of ^{137}Cs into the Kara Sea; a level which is of a similar order of magnitude to inputs from Sellafield-derived ^{137}Cs to the area (Vakulovsky *et al.*, 1995). However, the fact that the prevailing surface currents tend to transport contamination away from the Kara Sea in predominantly easterly and northerly directions, suggests that the resultant inputs to the Barents Sea has been relatively low. Nonetheless, there is a potential for significant radionuclide contamination following large accidental releases of radioactivity from these nuclear complexes.

Several other sources exist which also represent potential sources of radionuclides in the Barents and Norwegian Seas. Among these are radioactive waste containers dumped in the Barents and Kara Seas by the Former Soviet Union (FSU) and the sunken submarines the Komsomolets and the K-159 in the Norwegian Sea and the Barents Sea. The dumping areas and the locations of the sunken submarines are given in Figure 2.2. Several investigations have been conducted to detect possible leakage from these sources.

As part of the bilateral environmental cooperation with the Russian Federation, a joint Russian-Norwegian expert group has been investigating the condition of the contained dumped objects and has analysed sediment and water samples at these sites.

According to Strand *et al.* (1998), elevated levels of radionuclides in sediments collected in the vicinity of the dumped objects demonstrated that leakage had occurred, but no significant inputs of radionuclides to the open sea from this source have been detected. Following an accident on April 7, 1989, the Russian nuclear submarine the Komsomolets, lies at a depth of 1700 metres, 180 km southwest of Bear Island in the Norwegian Sea. The submarine contains a nuclear reactor and two torpedoes with nuclear warheads. Kolstad (1995) reported low activity concentrations of ^{134}Cs in seawater samples collected near the submarine, showing that leakage of radiocaesium isotopes may have occurred. Samples of water and sediments collected around the submarine by the Institute of Marine Research and analysed with respect to ^{137}Cs are presented in Figure 2.4.

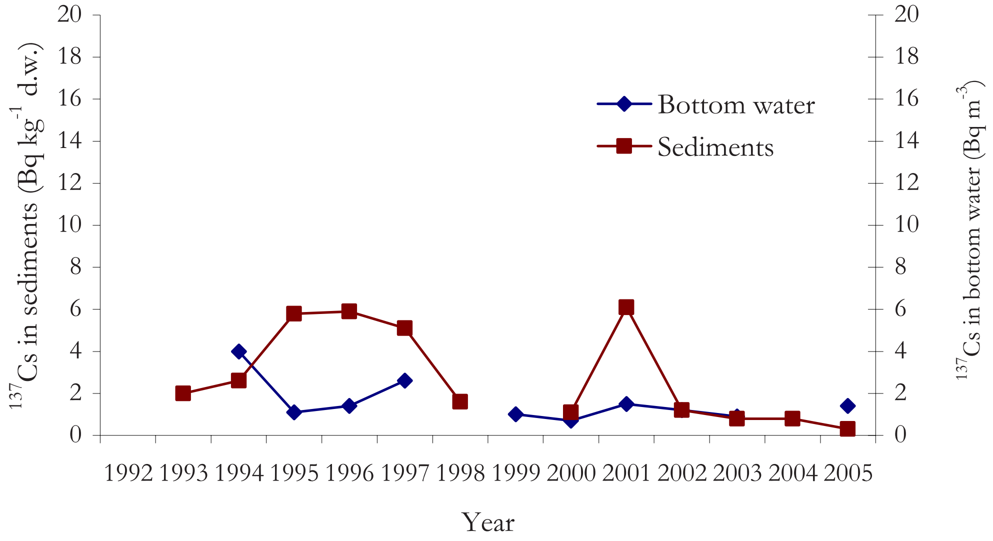

Figure 2.4. *^{137}Cs in samples from the position of the sunken nuclear submarine "Komsomolets".*

3 Collection of samples

In 2005 samples were collected in the Barents Sea, the North Sea, in selected fjords and at permanent coastal stations. A geographic overview of the sampling area covered by the marine monitoring programme, with the fixed coastal stations marked, is shown in Figure 3.1.

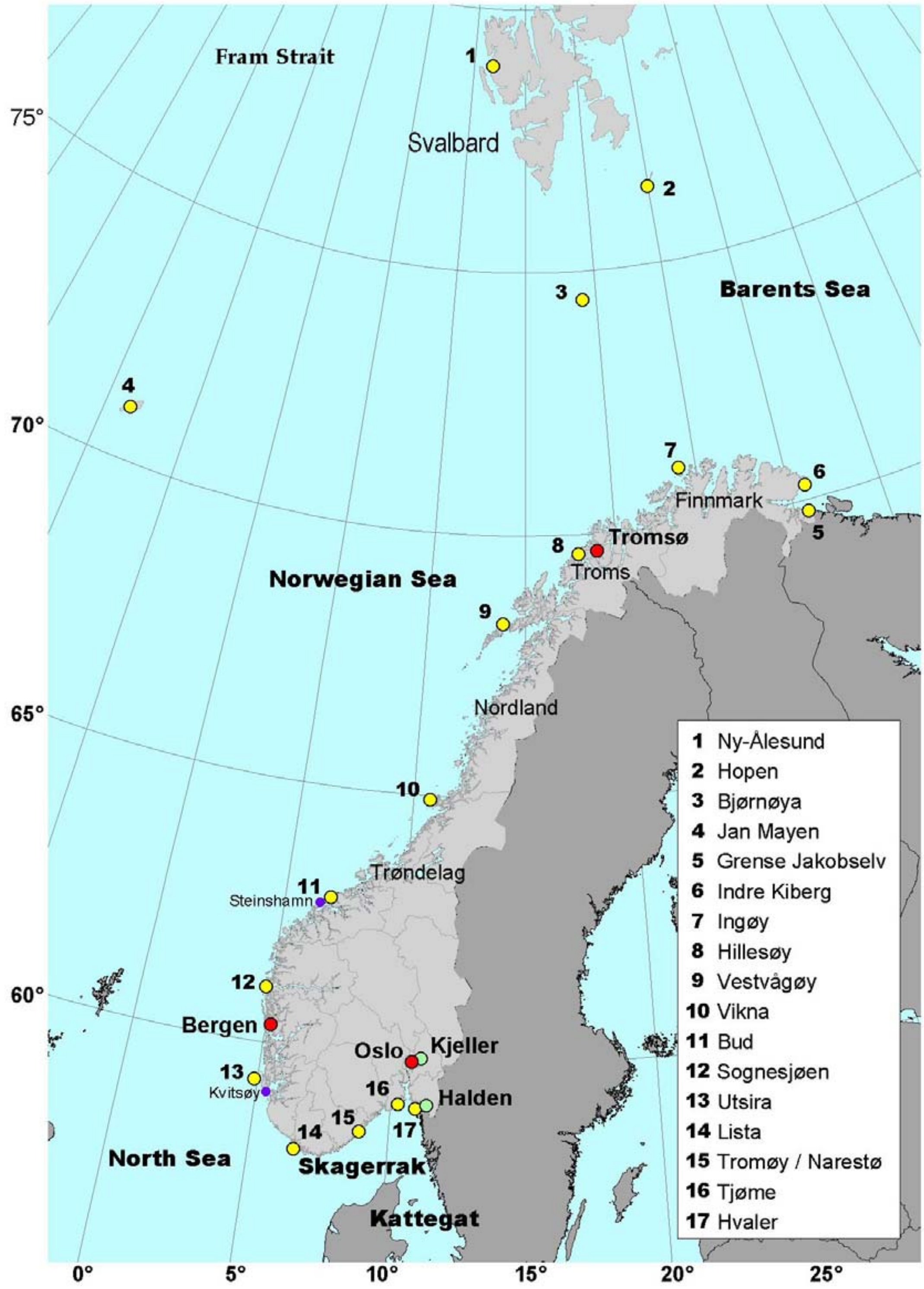

Figure 3.1. *Geographic overview of the sampling area covered by the marine monitoring programmes. (Fixed coastal sampling stations are indicated by yellow dots).*

Sampling and analysis were carried out by the Norwegian Radiation Protection Authority (NRPA), the Institute of Marine Research (IMR) and the Institute for Energy Technology (IFE).

In 2005 samples from the Barents Sea, were collected in the period Aug 15 – Sept 30 by the IMR and NRPA from the vessels R/V "G.O. Sars" and R/V "Johan Hjort" During the expeditions samples of surface water were collected which were later analysed for ^{228}Ra, ^{226}Ra, ^{99}Tc, ^{137}Cs, ^{90}Sr, ^{241}Am and plutonium isotopes. Sediment was also sampled and later analysed for ^{137}Cs. Samples were also collected in the southern part of the North Sea in the period Nov 26 – Dec 18 by the IMR from R/V "G.O. Sars".

Fig. 3.2. *R/V G O Sars in Bergen (Photo: NRPA).*

3.1 Seawater

For ^{99}Tc, ^{90}Sr and ^{226}Ra 50-100 litres of unfiltered water was collected at each sampling station and later transported to various laboratories for analysis.

For the analysis of ^{137}Cs, $Cu_2[Fe(CN)_6]$-impregnated cotton filters are used as sorbents (Roos *et al.,* 1994). The system consists of a prefilter (1 micron) and two $Cu_2[Fe(CN)_6]$-impregnated cotton filters connected in series (Fig 3.3). Assuming identical collection efficiency for the $Cu_2[Fe(CN)_6]$-impregnated cotton filters, the amount of caesium passing through the prefilter can be calculated by considering the two filters as the first two terms in a geometrical series, which has the sum:

$$A(^{137}Cs) = A_1/(1-A_2/A_1)$$

where A_1 and A_2 are the ^{137}Cs activities on the first and second impregnated filters, respectively.

Figure 3.3. *Filter system used for analysis of ^{137}Cs in seawater.(Photo: NRPA).*

Seawater samples of 200 litres (unfiltered) were collected for the determination of the activity concentrations of $^{239+240}$Pu and ^{241}Am. To collect deep water samples a CTD/Rosette multi bottle sampler was used (Fig 3.4). Plutonium-242 and ^{243}Am were used as chemical yield determinants. The precipitation of plutonium and americium from the water was performed *in situ* according to the analytical procedure described by Chen *et al.* (1991).

Fig. 3.4. *CTD/Rosette multi-bottle sampler.*

3.2 Sediment

Sediment samples were retrieved from the seabed using a Smøgen box corer. Profiles were collected from the box by slowly inserting PVC tubes into the sediment sample. Surface sediment samples were obtained from the upper 2 cm layer of the sediment remaining in the box-corer.

3.3 Seaweed

The NRPA collected seaweed (*Fucus vesiculosus*) samples from eight stations along the Norwegian coastline. At Hillesøy in northern Norway, seaweed is collected every month and at the other locations once per year. In addition, IFE performs monthly or annual seaweed sampling at eleven locations along the coastline, from the Russian border in the north to the Swedish border in the south.

3.4 Fish, crustaceans and molluscs

The IMR collected fish and shrimp samples from commercial fishing locations in the Barents Sea, The Norwegian Sea, the North Sea with Skagerrak and the Norwegian coastal areas. Samples of cod were prepared twice a year by combining muscle samples of 50 grams from 25 individual fish. At the request of the NRPA, Labora A/S in Salten was sampling different species of fish and crustaceans from the northern part of Norway.

Lobsters (*Homarus gammarus*) from two sites in Norway have been analysed for ^{99}Tc and ^{137}Cs. From one region (Værlandet) samples were delivered by IMR. From the other site (Sandøya, Arendal), a local fisherman delivered the lobsters to the NRPA immediately after they had been caught.

In addition some few samples of mussels (*Mytilus edulis*), kamtchatca crab (*Paralithodes camtschaticus*), shrimps (*Pandalus borealis)* and sea gooseberry *(Pleurobrachia Pileus)* were collected and analysed for ^{137}Cs.

4 Radioactivity in seawater and sediments

4.1 Technetium-99 in seawater

Tc-99 has a physical half-life of $2.13 \cdot 10^5$ years, and originates from the decay of ^{99}Mo, which is formed either as a fission product from ^{235}U or ^{239}Pu or by neutron activation of ^{98}Mo. The main source of ^{99}Tc in Norwegian waters is liquid discharge from the reprocessing plant at Sellafield. In oxygenated seawater ^{99}Tc is present as the highly soluble pertechnetate ion (TcO_4^-). Due to its conservative behaviour in seawater, TcO_4^- has the ability to be transported by ocean currents without being significantly affected by sedimentation processes. From the Irish Sea, ^{99}Tc is transported by ocean currents to the North Sea and via the Norwegian Coastal Current up to the Barents Sea. The transit time (the time between a specific discharge and the maximum activity concentration from that discharge reaching the sampling location) for ^{99}Tc to reach this area (Hillesøy) from the Irish Sea has been estimated to 3-4 years (Brown *et al.*, 2002; Dahlgaard, 1995).

During 2005, seawater samples were collected in the Barents Sea and at coastal station in the Skagerrak and analysed for ^{99}Tc. The results are presented in Figure 4.1 and Table 4.1. In the Barents Sea the activity concentration of ^{99}Tc ranged from 0.1 to 0.7 Bq m^{-3}. The activity concentration of ^{99}Tc in the Skagerrak ranged from 0.7 to 1.5 Bq m^{-3}.

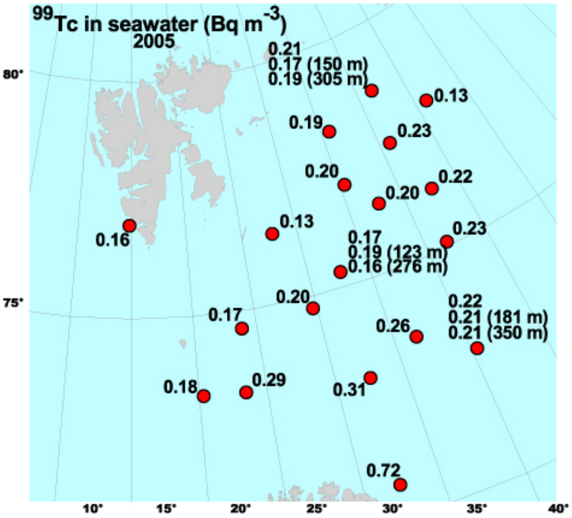

Figure 4.1. *Activity concentration (Bq m^{-3}) of ^{99}Tc in seawater samples collected in the Barents Sea in 2005.*

The activity concentrations found in the Barents Sea are similar or lower than those observed in 2002 (NRPA, 2004). Monthly samples have also been collected at Hillesøy (Fig. 4.2). Average activity concentration in seawater at Hillesøy in 2005 was 0.88 ± 0.10 Bq m^{-3}, which is similar to what was observed in 2003 and 2004.

During 2005, sea water samples were collected from coastal sites around the Svalbard archipelago. Tc-99 activity concentrations from Svalbard coastal sites (including monitoring stations on Hopen and Bjørnøya) ranged from 0.10 to 0.20 Bq m^{-3}. Comparison of these values with earlier observations (Gwynn et al., 2004) reveal that sea water activity concentrations for 2005 are generally lower than those reported for same region in 2000 and 2001, but comparable to values from 2004 (NRPA, 2006). Current activity concentrations of ^{99}Tc in sea water in the Svalbard region remain above 1994 pre-EARP values of 0.04 Bq m^{-3} (Kershaw et al., 1999).

Table 4.1. *Activity concentration (Bq m^{-3}) of ^{99}Tc in seawater samples collected from coastal stations in 2005.*

Location	Date	^{99}Tc (Bq m^{-3})
Tjøme	May	0.7
	September	1.2
Narestø	May	0.95
	September	1.1
Lista	May	1.2
	September	1.5
Gr. Jakobselv	May	0.5
	September	0.95
Hopen	January	0.18
	Mars	0.20
	October	0.18
Bjørnøya	Mars	0.23
	June	0.15
Jan Mayen	Mars	0.08
	September	0.07
	December	0.16
Kongsfjorden (outer fjord)	July	0.20
Kongsfjorden (inner fjord)	July	0.12
Ny Ålesund	July	0.10

The activity concentrations of ^{99}Tc found at the coastal stations at Tjøme, Narestø and Lista in the southern part of Norway are similar or lower than those observed in 2003.

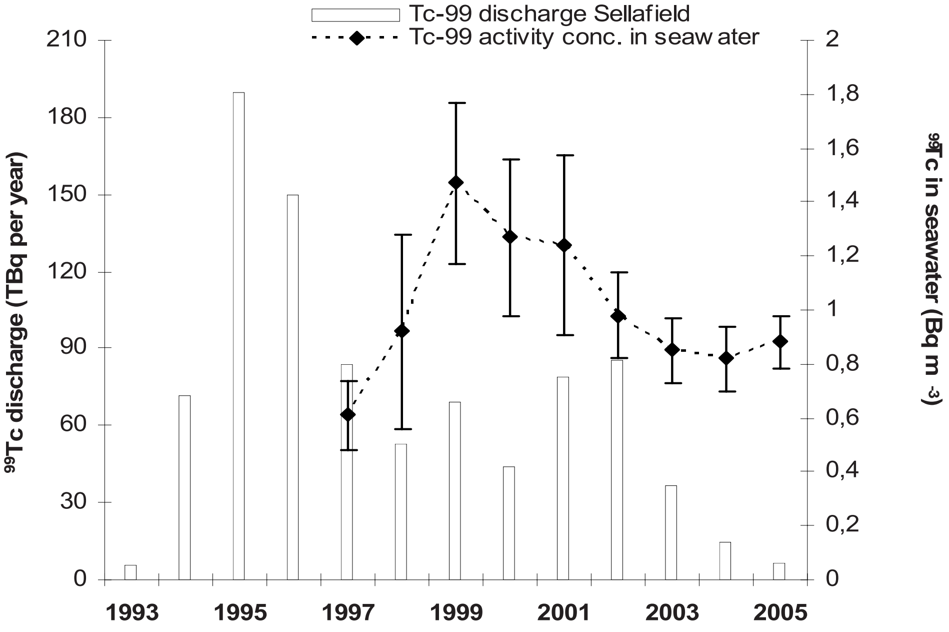

Figure. 4.2. *Annual average activity concentration (Bq m^{-3}) of ^{99}Tc in seawater at Hillesøy and annual discharge of ^{99}Tc from Sellafield.*

4.2 Strontium-90 in seawater

Strontium-90 is a fission product with a physical half-life of 29 years. Similar to ^{99}Tc, ^{90}Sr is a conservatively behaving element in the marine environment.

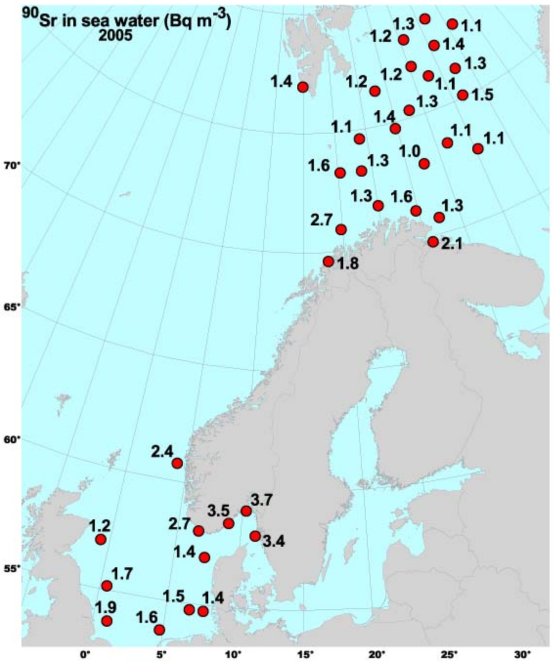

Figure 4.3. Activity concentration (Bq m^{-3}) of ^{90}Sr in surface water samples collected in the Barents Sea, the North Sea and the Skagerrak in 2005.

Seawater has been sampled in the Barents Sea, the North Sea and the Skagerrak and analysed for ^{90}Sr. The main sources of ^{90}Sr in these waters are discharge of liquid waste from reprocessing plants (mainly Sellafield), fallout from atmospheric nuclear weapons tests conducted mainly in the 1950s and 1960s and outflow of water from the Baltic Sea. The annual discharge of ^{90}Sr from Sellafield between 1998 and 2005 has been in the range of 13 to 31 TBq. The results for 2005 are presented in Figure 4.3. Typical activity concentration in Baltic Sea seawater have been reported to be in the range from 8 to 12 Bq m^{-3} (Vartti *et al*., 2006).

In the North Sea and in the Skagerrak, the activity concentration ranged from 1.2 Bq m^{-3} to 3.7 Bq m^{-3}. The highest concentrations were found off the southern coast of Norway in the Skagerrak. In the Barents Sea the levels of ^{90}Sr ranged from 1.0 to 2.7 Bq m^{-3}, which is similar to the levels observed in 2002 (NRPA, 2004).

4.3 Caesium-137 in seawater and sediment

Caesium-137 is a fission product with a half-life of 30 years. The main sources of ^{137}Cs in the North Sea and the Skagerrak are fallout from atmospheric nuclear weapons tests in the 1950s and 60s, outflowing water from the Baltic Sea and ^{137}Cs remobilised from Irish Sea sediments. Runoff from land, from the areas with the highest Chernobyl fallout can also contribute locally in coastal water. Like ^{99}Tc and ^{90}Sr, ^{137}Cs is also a conservatively behaving radionuclide in seawater.

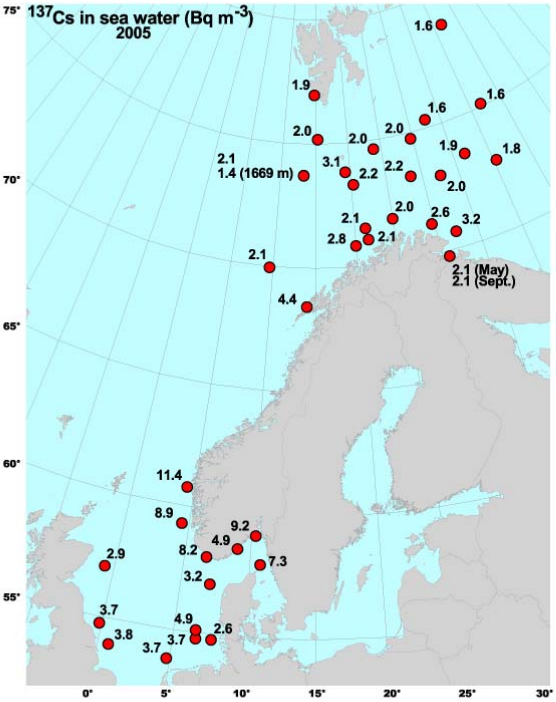

Figure 4.4. Activity concentration (Bq m^{-3}) of ^{137}Cs in seawater samples collected in the North Sea, the Skagerrak and the Barents Sea in 2005.

Observed levels of ^{137}Cs in the North Sea, the Skagerrak and the Barents Sea are shown in Figure 4.4.

The activity concentration in surface water ranged from 2.6 to 9.2 Bq m^{-3}, in the North Sea/Skagerrak area. The highest concentrations 9.2 and 11.4 Bq m^{-1} were observed in the Skagerrak and the Norwegian coastal current which is mainly explained by outflowing Baltic Sea water. Typical activity concentration in Baltic seawater is between 30 and 50 Bq m^{-3} (Vartti *et al.*, 2006).

Caesium-137 has also been analysed in surface sediments (upper 2 cm layer) from the Baltic Sea, the northern North Sea and selected fjords. The results are presented in Fig. 4.5 and Fig. 4.6 and range from 0.3 to 415 Bq kg^{-1} (d.w.). The highest activity concentration was found in Sognefjorden. The reason for the elevated levels of ^{137}Cs can be explained by direct fallout from the Chernobyl accident in 1986 and runoff of ^{137}Cs from nearby areas affected by the Chernobyl accident.

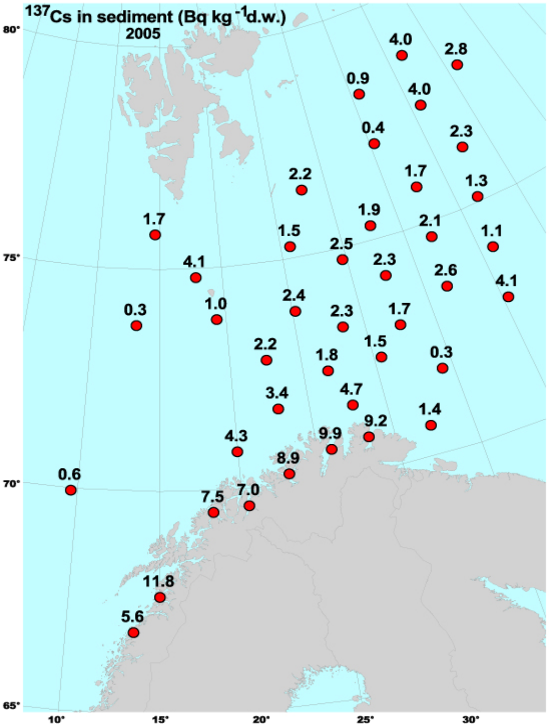

Figure 4.5. *Activity concentration (Bq kg^{-1} d.w.) in surface sediment from the Barents Sea in 2005.*

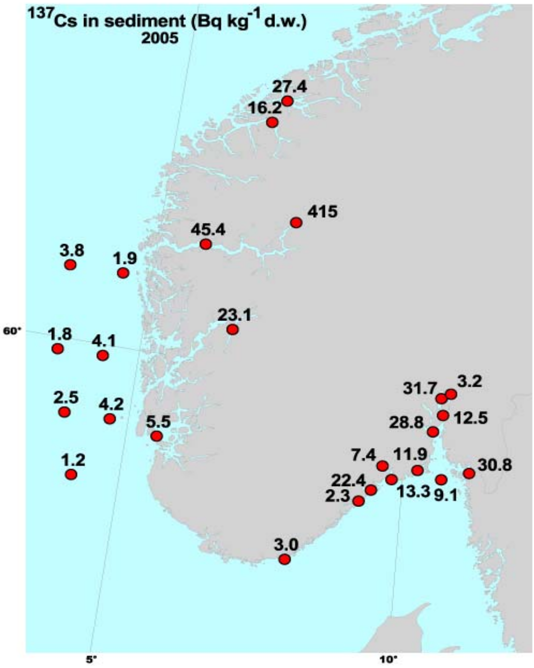

Figure 4.6. *Activity concentration (Bq kg^{-1} d.w.) of ^{137}Cs in surface sediment from the North Sea and selected fjords in 2005.*

4.4 Plutonium-239+240 in seawater

Plutonium-239 ($T_{1/2}$ = 24 110 y) and ^{240}Pu ($T_{1/2}$ = 6 563 y) belong to the transuranium elements and are mainly produced by neutron capture (with subsequent beta decay) by ^{238}U and ^{239}Pu, respectively. The behavior of plutonium in the marine environment is complex due to its different possible oxidation states. Under oxidizing conditions Pu is generally found in the more soluble +V and +IV state, while in reducing environments the more particle-reactive +III and +IV states dominate. The main source of $^{239+240}$Pu in northern Norwegian marine waters is global fallout from atmospheric nuclear weapons tests in the 1950s and 1960s. Another possible source is remobilised plutonium from Irish Sea sediments.

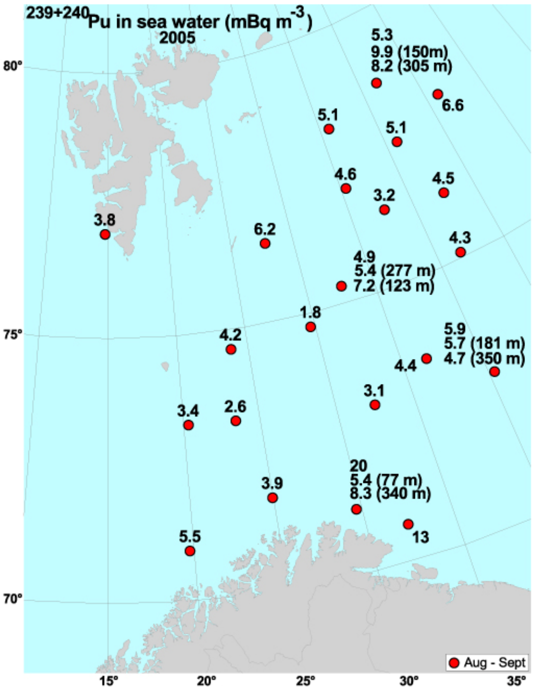

Figure 4.7. *Activity concentration (mBq m^{-3}) of $^{239+240}$Pu in surface water samples collected in the Barents Sea and along the coast in 2005.*

Observed levels in samples collected in 2005 are presented in Fig. 4.7 and Fig. 4.8 and range from 2.6 to 29 mBq m^{-3}. The highest level was observed in water collected off the coast of Scotland, showing that Sellafield is a source to plutonium in the North Sea, where part of the $^{239+240}$Pu comes from remobilised plutonium from contaminated Irish Sea sediments. The levels of $^{239+240}$Pu are generally lower than those observed in 2001 (Gäfvert *et al.*, 2003).

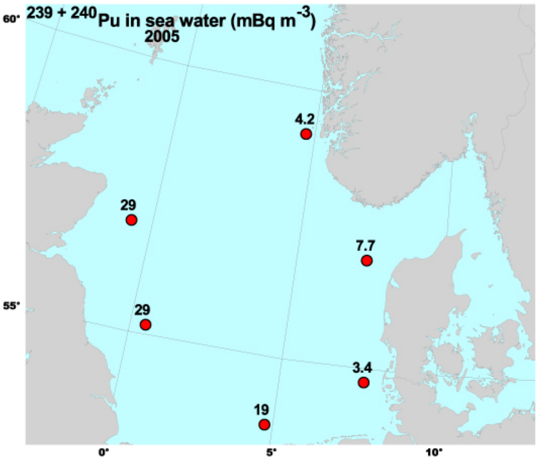

Figure 4.8. *Activity concentration (mBq m^{-3}) of $^{239+240}$Pu in surface water samples collected in the North Sea in 2005.*

4.5 Americium-241 in seawater

Americium-241 belongs to the transuranium elements and has a physical half-life of 432 years. It is produced, for example, in nuclear reactors by successive neutron capture by ^{239}Pu. Americium-241 is finally formed by beta-decay of ^{241}Pu ($T_{1/2}$ = 14.35 y). Main sources of ^{241}Am in the environment are fallout of ^{241}Pu from nuclear weapon tests in the 1950s and '60s and the discharge of ^{241}Am and ^{241}Pu from reprocessing plants.

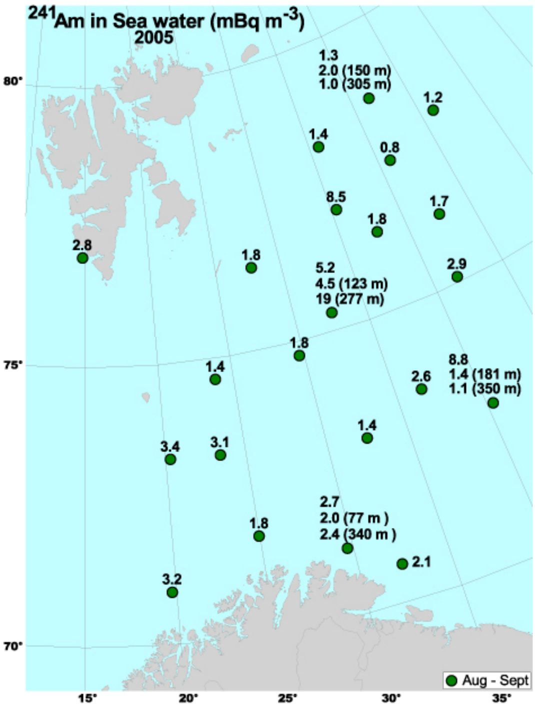

Figure 4.9. *Activity concentration (mBq m^{-3}) of ^{241}Am in seawater from the Barents Sea in 2005.*

Measured activity concentrations of ^{241}Am in the Barents Sea in 2005 are presented in Figure 4.9 and range from 0.8 to 19 mBq m^{-3}. The ^{241}Am found in the water column today, can be due to both the present discharge from Sellafield of ^{241}Am and earlier discharges of ^{241}Pu (which decays to ^{241}Am).

4.6 Radium-226+228 in seawater

Radium-226 is a naturally occurring radionuclide with a physical half-life of 1 600 years. As a member of the natural decay chain starting with ^{238}U ($t_{1/2}$ = 4.47·10^{9} years), it is continuously produced by the decay of ^{230}Th, and can be found in different concentrations in the environment. In the marine environment ^{226}Ra is naturally supplied from both the sediments (the mother nuclide ^{230}Th is particle reactive and is mainly found attached to sediment particles) and by river water to the oceans. Typical activity concentration of ^{226}Ra in Atlantic surface water has been reported to be about 1.3 Bq m^{-3} (IAEA, 1990). Radium-228 ($T_{1/2}$ = 5.75 y) is the daughter of ^{232}Th and is mainly produced in marine sediments. The highest concentration is found in coastal water where the mixing with oceanic water is limited. An anthropogenic source of ^{226}Ra and ^{228}Ra in the North Sea is produced water from oil reservoirs. Due to the relatively high solubility of radium, produced water can contain more than a thousand times the activity concentration normally found in seawater.

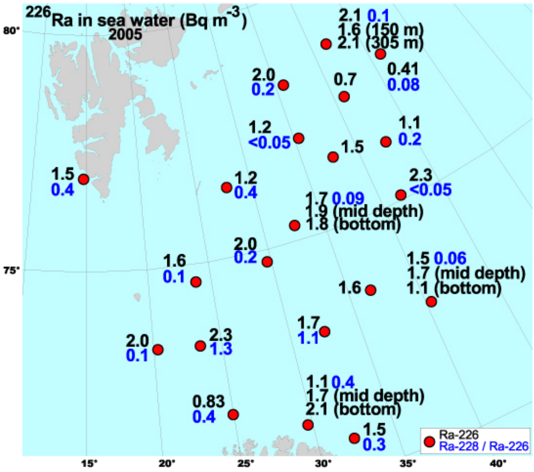

Figure 4.10. *Activity concentration (Bq m^{-3}) of ^{226}Ra and ^{228}Ra/^{226}Ra ratio in seawater from the Barents Sea in 2005.*

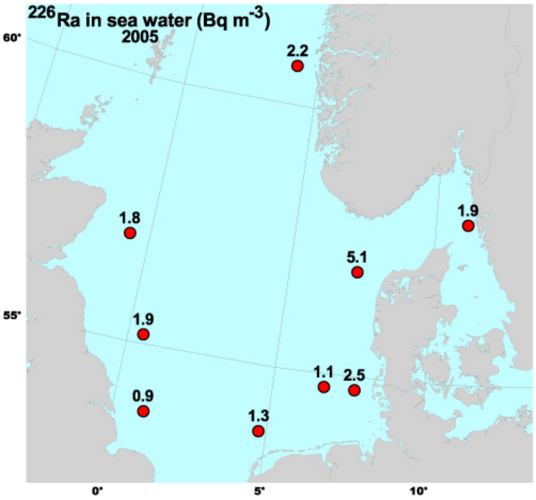

Figure 4.11. *Activity concentration (Bq m^{-3}) of ^{226}Ra seawater from the North Sea in 2005.*

The activity concentration of ^{226}Ra observed in the Barents Sea, the North Sea and the Skagerrak in 2005 were in the range of 0.4 - 5.1 Bq m^{-3} (Fig. 4.10 and 4.11). The ^{228}Ra/^{226}Ra ratios were generally below 1, and the lowest ratios were found in arctic water.

5 Radioactivity in biota

5.1 Technetium-99 in seaweed

Figure 5.1. *Brown seaweed (*Fucus vesiculosus*) (Photo:NRPA).*

Seaweed is an excellent bioindicator for ^{99}Tc in he marine environment. It has a very high ability to concentrate ^{99}Tc from seawater and is easy accessible in most coastal areas. During 2005, seaweed (*Fucus vesiculosus,* Fig 5.1) was collected at the permanent coastal sampling stations along the Norwegian coastline and analysed for ^{99}Tc. At two locations, Hillesøy and Utsira, sampling was performed monthly. At the remaining sites sampling was conducted in August or September. The results of the analyses are presented in Figure 5.2 and range from 73 to 265 Bq kg^{-1} (d.w.), where the highest activity concentration (348 Bq kg^{-1} d.w.) was found in one of the monthly samples collected at Utsira. Compared with the results from 1999-2001 (Rudjord *et al.*, 2001; Gäfvert *et al.*, 2003), the levels of ^{99}Tc have decreased at most sampling sites. The trend can also be seen in Figures 5.3 and 5.4 which show the annual average activity concentration of ^{99}Tc in *Fucus vesiculosus* at Utsira and Hillesøy, together with the annual discharge of ^{99}Tc from Sellafield. The maximum levels in Figures 5.3 and 5.4 are the response to the increased discharge from Sellafield in the mid 1990s. From Figures 5.3 and 5.4, one can see that the peak in ^{99}Tc levels in *F. vesiculosus* occurred after about 3-4 years at Utsira and about 4-5 years at Hillesøy further down-stream. The increasing ^{99}Tc levels in seaweed from Utsira and Hillesøy since 2003 can be explained by the relatively high discharge of ^{99}Tc from Sellafield in 2001 and 2002.

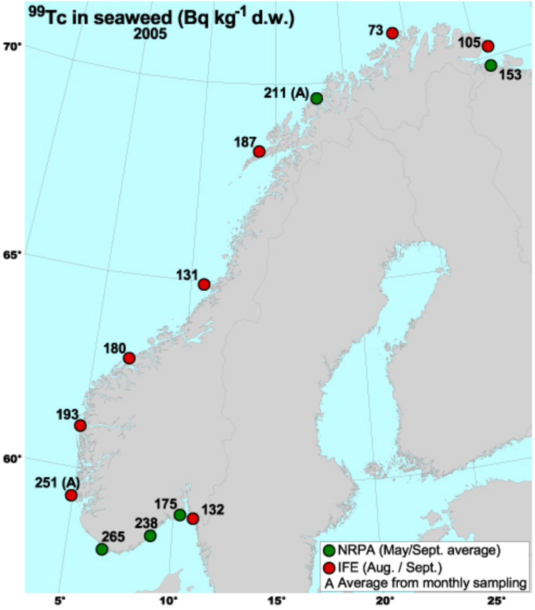

Figure 5.2. *Levels of ^{99}Tc in* Fucus vesiculosus *sampled along the Norwegian coastline in 2005.*

At Hillesøy both seawater and *F. vesiculosus* have been analysed monthly with respect to ^{99}Tc since 1997. In Figure 5.5 one can see that the activity concentration in the seaweed responded rapidly with the increased ^{99}Tc levels in the seawater up to mid 2001. From mid 2001 and onwards, a decreasing trend is observed for the ^{99}Tc level in the water, while the activity concentration in the seaweed has decreased at a slightly slower rate, and shows relatively large fluctuations. Laboratory experiments have shown that the elimination of Tc from macro-algae generally has an initial rapid component (biological $T_{1/2}$ of 1-3 days) followed by a slower component with a biological $T_{1/2}$ of 20-200 days (Beasley and Lorz, 1986; Topocuoglo and Fowler, 1984).

Tc-99 was also analysed in samples of *Ascophyllum nodosum* collected at Utsira in the period 1997 - 2005 (Fig. 5.6). *Ascophyllum nodosum* is known to have a relatively high concentration factor for ^{99}Tc. Holm and Rioseco (1986) have shown that the concentration factor for ^{99}Tc in *Ascophyllum nodosum* are about twice as high as for Fucus vesiculosus. The highest level found at Utsira, 614 Bq kg^{-1} (d.w.), was observed in 1999. Since then the levels have decreased and in 2005 the levels was found to be 325 Bq kg^{-1} (d.w.).

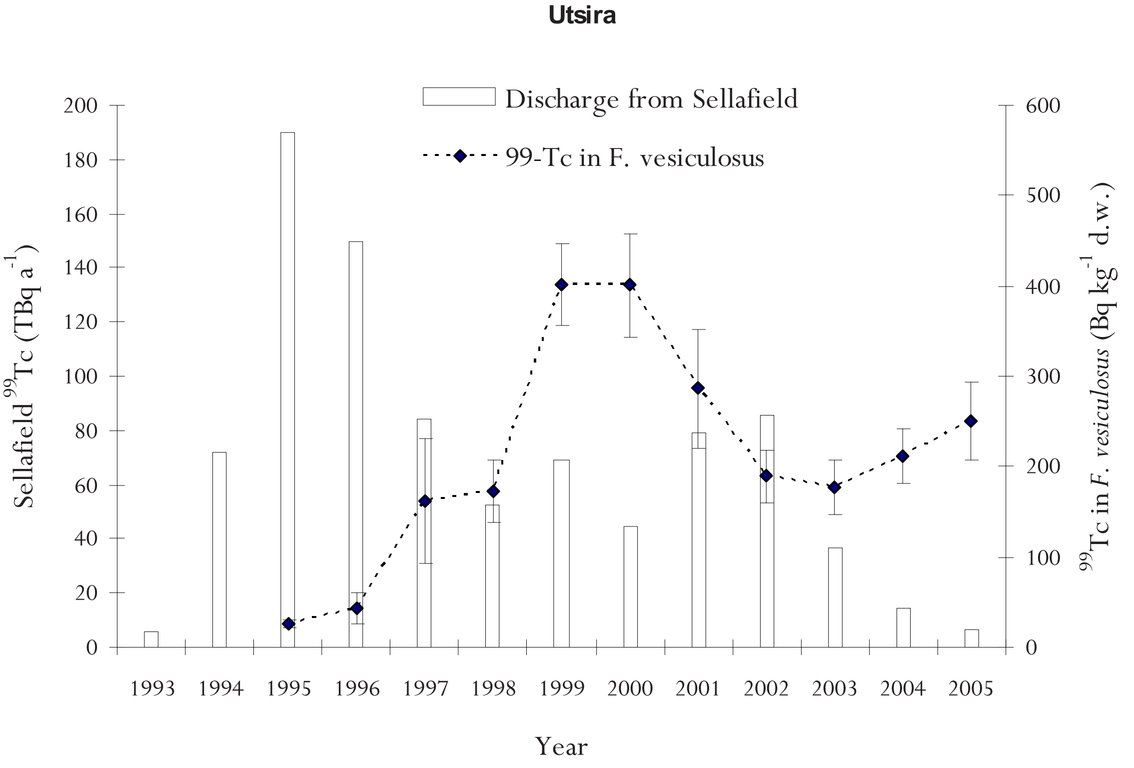

Figure 5.3. *Annual liquid discharge of ^{99}Tc from Sellafield (primary axis) and annual average (with 95 % confidence limits) ^{99}Tc activity concentration in brown algae* (Fucus vesiculosus*) sampled at Utsira (data provided by IFE) in the period 1995-2005 (secondary axis).*

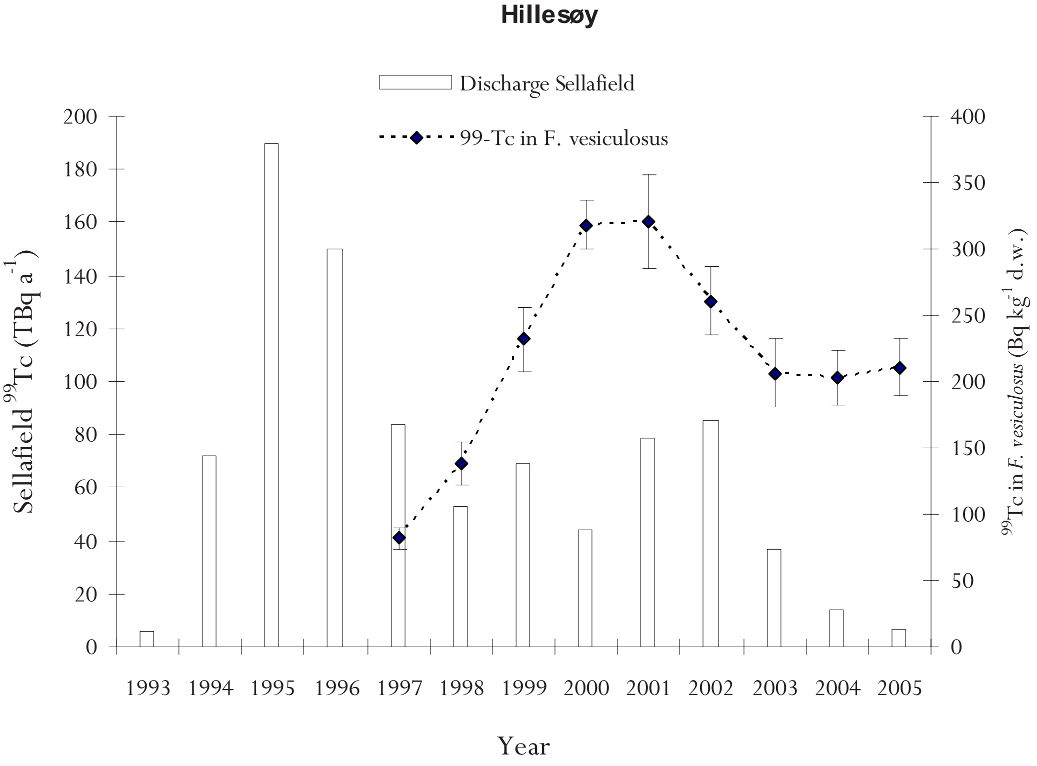

Figure 5.4. *Annual liquid discharge of ^{99}Tc from Sellafield (primary axis) and annual average (with 95 % confidence limits) ^{99}Tc activity concentration in brown algae (*Fucus vesiculosus*) sampled at Hillesøy in the period 1997-2005 (secondary axis).*

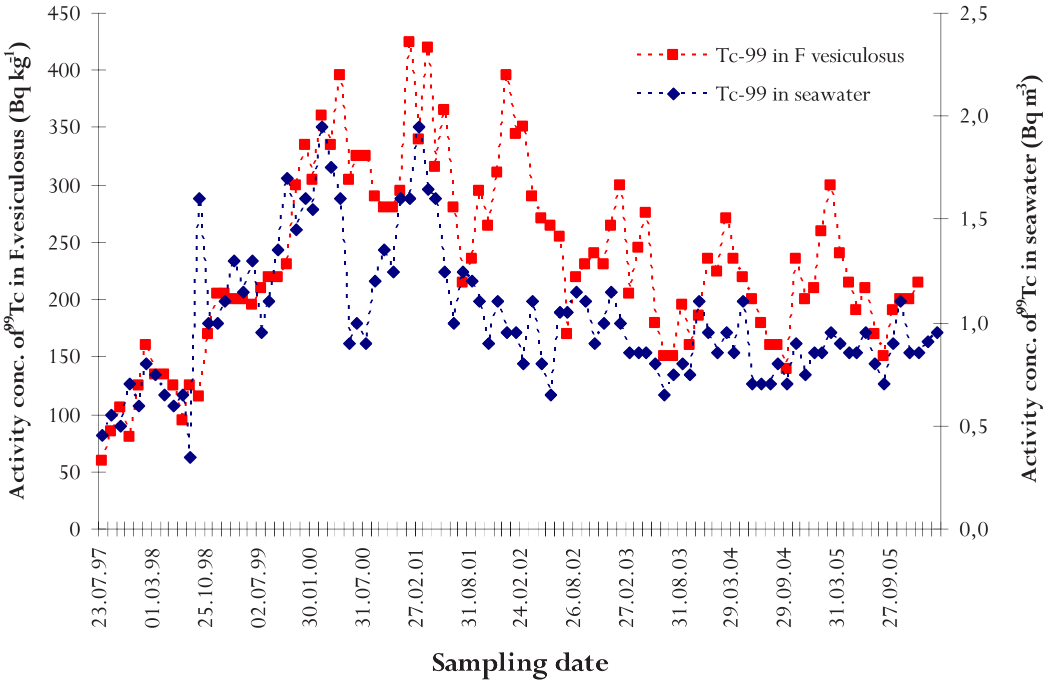

Figure 5.5. *Activity concentration of ^{99}Tc in seawater and seaweed sampled at Hillesøy in the period 1997 to 2005.*

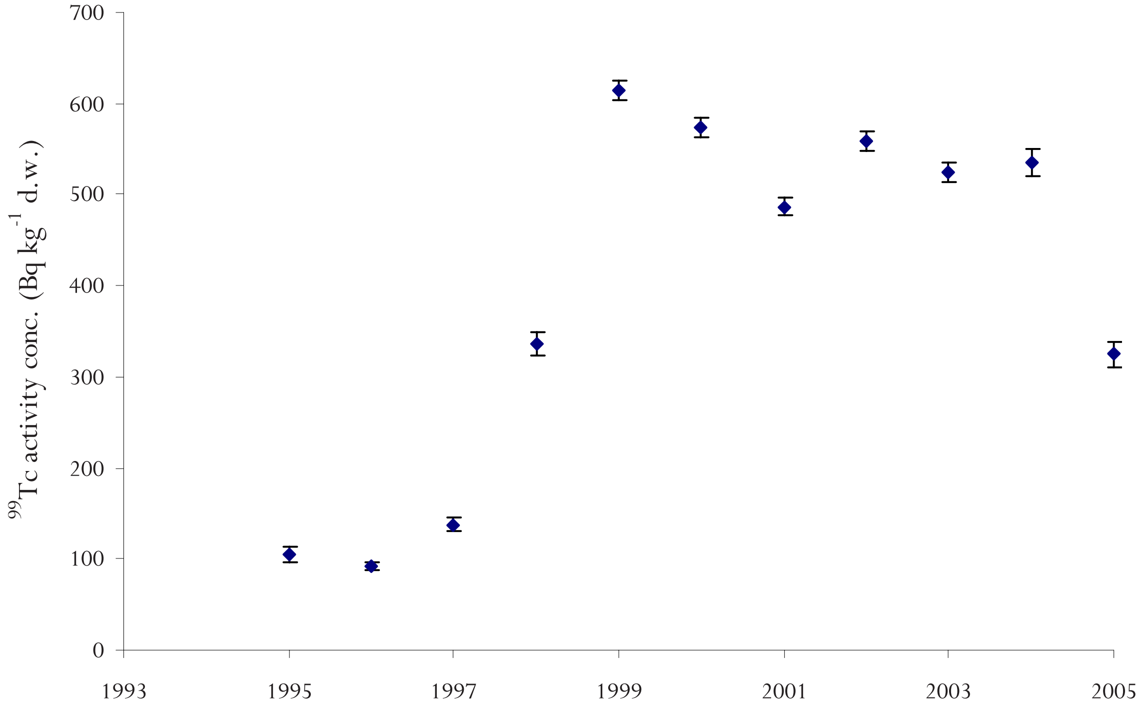

Figure 5.6. *Activity concentration (Bq kg^{-1}d.w.) of ^{99}Tc in* Ascophyllum nodosum *sampled at Utsira (data provided by IFE).*

5.2 Caesium-137 in seaweed

5.2.1 Caesium-137 in seaweed

Fucus vesiculosus has also been widely used as a bioindicator for ^{137}Cs. The accumulation of ^{137}Cs in brown algae is, however, not as pronounced as for ^{99}Tc. The uptake of ^{137}Cs also depends on the salinity of the surrounding sea water, with higher uptake at lower salinities (Carlsson and Erlandsson, 1991).

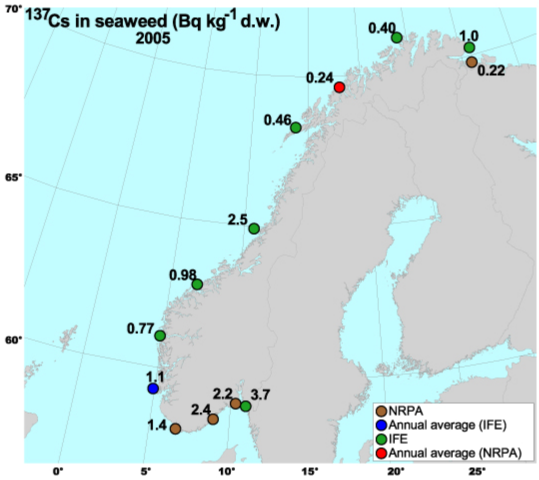

Figure 5.7. *Levels of ^{137}Cs (Bq kg^{-1} d.w.) in* Fucus vesiculosus *sampled along the Norwegian coastline in 2005.*

In 2005, samples of *Fucus vesiculosus* from the permanent coastal stations (see Figure 3.1) were analysed with respect to ^{137}Cs. The results are presented in Figure 5.7, and range from 0.22 Bq kg^{-1} (d.w.) in the north to 3.7 Bq kg^{-1} (d.w.) close to the Swedish border. The reason for the higher activity concentration in *Fucus vesiculosus* in the southern part of Norway is a combination of two effects. First, the salinity in the water is lower and secondly, that the activity concentration of ^{137}Cs in the seawater is higher due to outflowing Baltic Sea water contaminated by the Chernobyl accident. In Figure 5.8 the activity concentration of ^{137}Cs in the seaweed is plotted together with the salinity of the water for 11 of the permanent coastal stations. The peak in the ^{137}Cs levels in seaweed collected at Vikna can be explained by run-off of ^{137}Cs from land, since this area was affected by fallout from the Chernobyl accident in 1986.

Comparing those results with the levels found at the coastal sampling stations in 2001, 2002, 2003 and 2004 (Gäfvert *et al.*, 2003; NRPA, 2004, NRPA 2005b and NRPA 2006), one can see that the activity concentration of ^{137}Cs in *F. vesiculosus* has been relatively stable during recent years. In the southern part of Norway the levels have been in the range 1.2 to 6.0 Bq kg^{-1} (d.w.), while in the northern part all results have been below 1 Bq kg^{-1} (d.w).

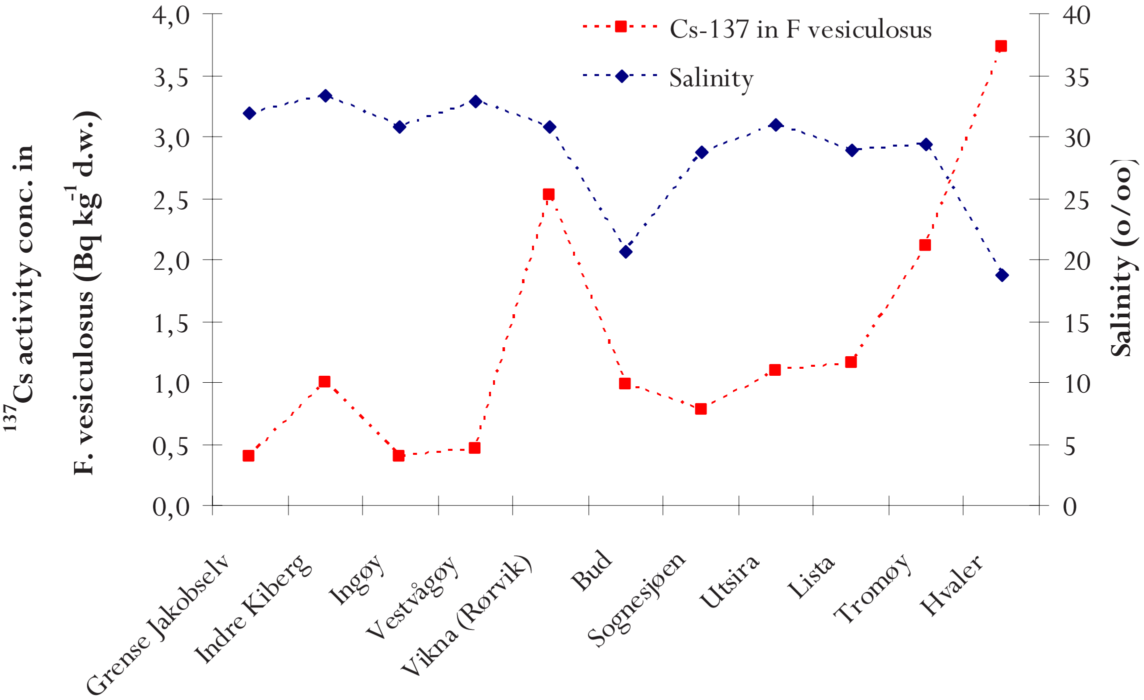

Figure 5.8. *Activity concentration (Bq kg^{-1} d.w.) in* Fucus vesiculosus *vesus salinity in seawater sampled along the Norwegian coastline in 2005 (data provided by IFE).*

5.3 Technetium-99 and caesium-137 in crustaceans and molluscs

In 2005, samples of lobster (*Homarus gammarus*) were collected at Sandøya and in Sogn og Fjordane (see Figure 5.9) and analysed for ^{99}Tc. Lobsters have previously been identified as a marine organism with a high ability to accumulate ^{99}Tc from seawater (Swift, 1985; Smith *et al.,* 1998; Smith *et al.,* 2001). The observed levels are presented in Table 5.1 and range from 2.0 to 42.0 Bq kg^{-1} w.w. The levels in Sandøya are lower than those found in 2003 and 2004 at the same site, and for Værlandet approximately the same as in 2003 and 2004. The results show, however, a relatively large variation, even for samples collected in the same area. It is also observed a clear difference between female and male lobsters where female lobsters have a significantly higher uptake of ^{99}Tc.

Figure 5.9. *Geographic overview of the sampling areas.*

Table 5.1. *Average activity concentration and range of ^{99}Tc in lobster (*Homarus gammarus*) tail muscle in 2005.*

Location	Gender (F/M)	n	^{99}Tc Average (range) (Bq kg^{-1} w.w.)
Sandøya	F	5	10.3 (5.1 - 15.4)
	M	5	2.8 (2.0 - 4.4)
Værlandet	F	3	20.3 (6.3 - 42.0)
	M	2	4.2 (3.6 - 4.8)

A number of lobster samples were also analysed for ^{137}Cs. The levels were considerably lower than for ^{99}Tc and ranged from < 0.1 to 0.45 Bq kg^{-1} (w.w.).

Mussels (*Mytilus edulis*), kamtchatca crab (*Paralithodes camtschaticus*), shrimps (*Pandalus borealis)* and sea gooseberry *(Pleurobrachia Pileus)* have been collected off the northern coast of Norway and analysed for ^{137}Cs. The results are presented in Table 5.2.

Table 5.2. *Activity concentration of ^{137}Cs in Blue mussels, shrimps, kamtchatca crab and Sea Gooseberry, in 2005.*

Location/Species	^{137}Cs (Bq kg^{-1} w.w.)
Nordland/Blue mussels *(Mytilus edulis)*	< 0.1
Nordland/Shrimps (*Pandalus borealis*)	< 0.1
Finnmark/Kamtchatca crab (*Paralithodes camtschaticus*)	< 0.1
The North Sea/ Sea Gooseberry *(Pleurobrachia Pileus)*	< 0.1

5.4 Caesium-137 in fish

Commercially important fish species, including farmed salmon, as well as other fish species have been collected from Norwegian marine waters and analysed with respect to ^{137}Cs.

The activity concentrations of ^{137}Cs in fish caught in the Barents Sea, the Norwegian Sea, the North Sea and in Skagerrak are presented in Table 5.3. The levels of ^{137}Cs in all sampled fish species from these areas range from 0.1 to 0.55 Bq kg^{-1} (w.w.).

The activity concentrations of ^{137}Cs in various fish species collected along the Norewgian coastline from Finnmark in the north to Sogn og Fjordane in the west are given in Table 5.4. The results from this area range from 0.20 to 0.55 Bq kg^{-1} (w.w.).

Generally, the levels of ^{137}Cs in Norwegian marine waters can be considered low. In the Norwegian marine environment the highest concentrations of ^{137}Cs are found in coastal water. These results could be explained as a result of Chernobyl run-off of ^{137}Cs from land and outflow of Chernobyl fallout from the Baltic Sea to the Norwegian Coastal Current.

In Figure 5.10, typical activity concentrations of ^{137}Cs found in cod in northern European waters are presented. In the Baltic Sea, the levels of ^{137}Cs in fish are higher due to higher levels of ^{137}Cs in the water, but also due to the lower salinity of the water, resulting in a higher uptake in fish. In the Irish Sea, the levels are influenced by past and present discharges of ^{137}Cs from the reprocessing plant at Sellafield.

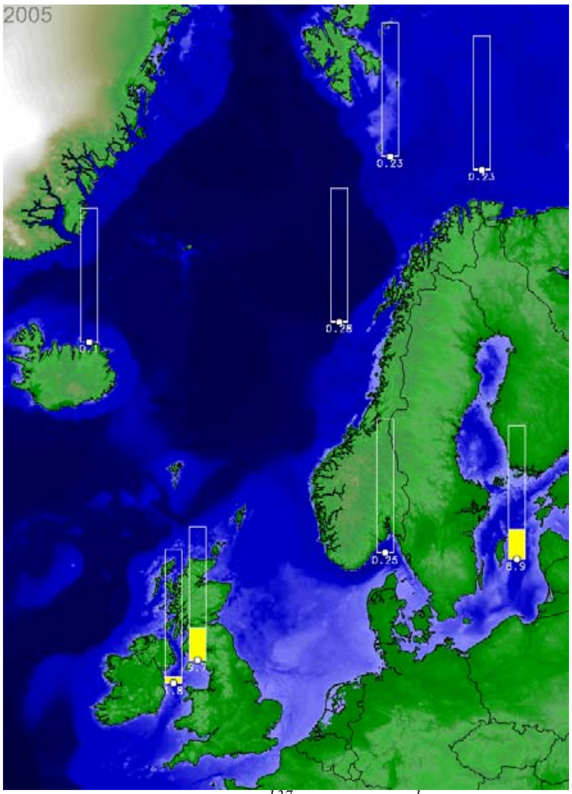

Figure 5.10. *Levels of ^{137}Cs (Bq kg^{-1}w.w.) in cod from different marine areas (data from EA, EHS, FSA & SEPA (2005 and this report).*

The activity concentration of ^{137}Cs in cod from the Barents Sea have been analysed annually since 1992 (Fig. 5.11). All obtained results have been below 1 Bq kg^{-1} w.w. (and most results below 0.5 Bq kg^{-1} w.w), and the levels are showing a slightly decreasing trend in the period 1992 - 2005.

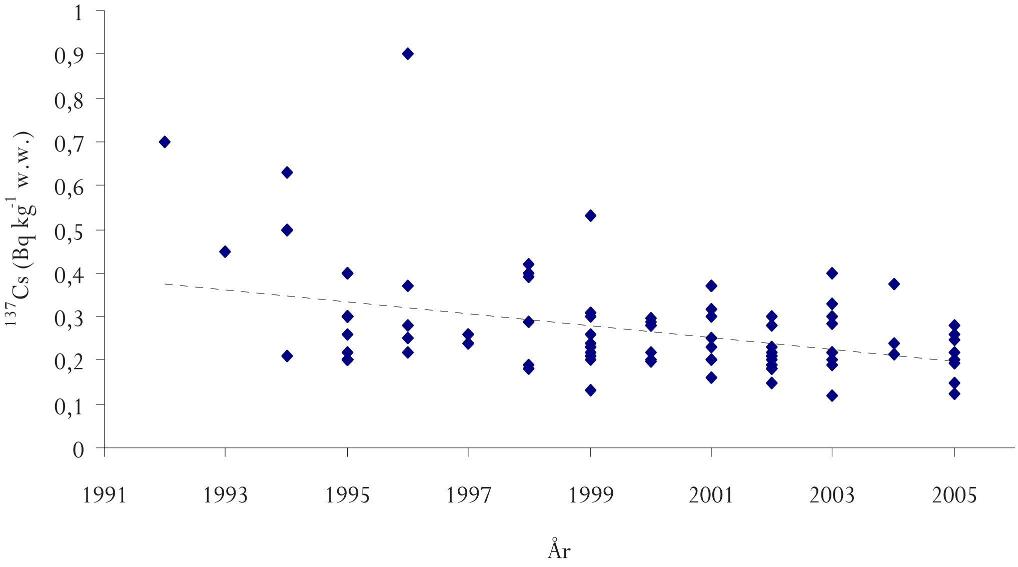

Figure 5.11. *Activity concentration (Bq kg^{-1} w.w.) of ^{137}Cs in cod (Gadus morhua L.) the Barents Sea in the period 1992 - 2005.*

Table5.3. *Levels of ^{137}Cs activity concentrations (Bq kg^{-1} w. w.) for various fish species collected from the Barents Sea, Norwegian Sea, the North Sea and in the Skagerrak, in 2005*

Species	**Location**	**No. of samples (total no. of fish)**	**^{137}Cs in muscle tissue (Bq kg^{-1} w.w.)**
Cod *(Gadus morhua L.)*	Barents Sea	7 (175)	0.15 ± 0.01 to 0.28± 0.10
Saithe *(Pollachius virens)*		2 (50)	0.30 ± 0.09 and 0.31 ± 0.10
Haddock *(Melanogrammus Aeglefinus)*		7 (175)	0.10 ± 0.09 and 0.22 ± 0.10
Long Rough Dab *(Hippoglossoides Platessoides)*		6 (150)	0.10 ± 0.01 to 0.20± 0.10
Golden Redfish *(Sebastes Marinus)*		2 (50)	0.20 ± 0.01 and 0.30± 0.10
Polar Cod *(Boreogadus Saida)*		3 (75)	< 0.3
Greenland Halibut *(Reinhardtius Hippoglossoides)*		4 (100)	0.10 ± 0.01 to 0.30± 0.10
Cod *(Gadus morhua L.)*	Norwegian Sea	1 (10)	0.23 ± 0.10
Saithe *(Pollachius virens)*		1 (25)	0.22 ± 0.10
Haddock *(Melanogrammus Aeglefinus)*		1 (25)	0.21 ± 0.10
Cod *(Gadus morhua L.)*	The North Sea	1 (25)	0.29 ± 0.12
Saithe *(Pollachius virens)*		2 (50)	0.32 ± 0.10 and 0.35 ± 0.13
Atlantic herring *(Clupea harengus)*		2(50)	< 0.8
Haddock *(Melanogrammus Aeglefinus.)*		2 (50)	0.12 ± 0.09 and 0.17 ± 0.09
Mackerel *(Scomber scombrus)*		2 (50)	0.13 ± 0.12 and 0.16 ± 0.10
Long Rough Dab *(Hippoglossoides Platessoides)*		1 (25)	0.10 ± 0.06
European plaice *(Pleuronectes Platessa)*		1 (25)	0.23 ± 0.11
Dab *(Limanda Limanda)*		1 (25)	0.21 ± 0.10
Sprat *(Sprattus Sprattus)*		2 (0,5 kg)	0.21 ± 0.10 and 0.23 ± 0.10
Blue Whiting *(Micromesistius Poutassou)*		1 (25)	0.22 ± 0.10
Norway pout *(Trisopterus Esmarkii)*		1	0.15 ± 0.10
Horse mackerel *(Trachurus Trachurus)*		2	0.29 ± 0.06 and 0.41 ± 0.18
Haddock *(Melanogrammus Aeglefinus.)*	Skagerrak	1(25)	0.55 ± 0.07

Table 5.4. *Levels of ^{137}Cs activity concentrations (Bq kg^{-1} w. w.) for various fish species collected off the coast of Finnmark in the north to Sogn og Fjordane in the western part of Norway, in 2005.*

Species	**Location**	**No. of samples (total no. of fish)**	**^{137}Cs in muscle tissue (Bq kg^{-1} w.w.)**
Cod *(Gadus morhua L.)*	Coastal waters of Finnmark	6 (150)	0.20 ± 0.06 to 0.31 ± 0.15
Saithe *(Pollachius virens)*		1 (25)	0.25 ± 0.07
Cod *(Gadus morhua L.)*	Coastal waters of Nordland and Trøndelag	4(100)	0.25 ± 0.10 to 0.55 ± 0.10
Saithe *(Pollachius virens)*		1 (25)	0.53 ± 0.05
Haddock *(Melanogrammus Aeglefinus.)*		1 (25)	0.48 ± 0.05
Salmon (*Salmo salar*)		1 (25)	0.31 ± 0.05
Atlantic herring *(Clupea harengus)*	Coastal waters of the Western part of Norway	3 (75)	0.22 ± 0.15 to 0.54 ± 0.06
Haddock *(Melanogrammus Aeglefinus.)*		1 (25)	0.33 ± 0.10
Long Rough Dab *(Hippoglossoides Platessoides)*		1 (25)	0.29 ± 0.05
Sprat *(Sprattus Sprattus)*		2 (0,5 kg)	0.40 ± 0.08 and 0.49 ± 0.09
Norway pout *(Trisopterus Esmarkii)*		1 (25)	0.26 ± 0.10

5.5 Caesium-137, polonium-210 and lead-210 in seabirds

Caesium-137 was determined in muscle in a range of seabirds from Svalbard archipelago. Activity concentrations of ^{137}Cs were either low or, as in the majority of cases, below detection limits. Where ^{137}Cs activity concentrations were above detection limits (n=6), observed values ranged from 0.08 ±0.02 to 0.18 ±0.05 Bq kg^{-1} (w.w.)

Polonium-210 is known to concentrate in marine organisms to a higher extent than other naturally occurring alpha emitters, particularly in certain organs and is typically the greatest contributor to dose under natural conditions. However, little is known concerning the trophic transfer and resulting activity concentrations of ^{210}Po and ^{210}Pb in seabirds, an important consumer group in marine food webs.

Polonium-210 and ^{210}Pb were determined in different tissues in a range of seabirds from Svalbard archipelago. Activity concentrations of ^{210}Po in muscle showed some variation between species, with mean activity concentrations ranging from 1.10 ±0.48 Bq kg^{-1} (w.w.) in glaucous gulls (*Larus hyperboreus*) to 13.67 ±7.24 Bq kg^{-1} (w.w.) in little auks (*Alle alle*). The observed variation in ^{210}Po activity concentrations probably reflects the differences in diet between the species. For example, little auks feed predominantly on copepods which are known to accumulate high levels of ^{210}Po (Carvalho, 1988), whereas glaucous gulls typically predate on other seabirds. Activity concentrations of ^{210}Pb in muscle were negligible.

Activity concentrations of ^{210}Po in pooled seabird kidney samples were between 7 and 21 fold higher than mean muscle values and generally showed the same trend across species as for activity concentrations in muscle (Table 5.5). Activity concentrations of ^{210}Pb in pooled seabird kidneys were generally below 1 Bq kg^{-1} (w.w.) with the exception of the pooled sample from northern fulmars (*Fulmarus glacialis*) which showed an activity concentration of 5.75 ±0.20 Bq kg^{-1} (w.w.). Polonium-210/^{210}Pb activity ratios in kidneys varied considerably with minimum and maximum values of 8 and 508 respectively. Data on ^{210}Po and ^{210}Pb in liver was only available for two seabird species, with activity concentrations of ^{210}Po 3 fold lower in liver than in kidneys from the same species and with activity concentrations of ^{210}Pb below 0.2 Bq kg^{-1} (w.w.). Observed liver ^{210}Po/^{210}Pb activity ratios were of the order of 230 and 444.

Table 5.5. *Activity concentrations of ^{210}Po and ^{210}Pb in different tissues of seabird species*

		^{210}Po (Bq kg^{-1} w.w.)			^{210}Pb (Bq kg^{-1} w.w.)	
Species	n	Muscle^{1}	Kidney^{2}	Liver^{2}	Kidney^{2}	Liver^{2}
Kittiwake (*Rissa tridactyla*)	5	3.89 ±1.48	31.9 ±0.3	-	0.42 ±0.02	-
Brunnich's Guillemot (*Uria lomvia*)	6	11.6 ±5.1	131.4 ±1.0	42.9 ±0.3	0.26 ±0.02	0.19 ±0.01
Glaucous gull (*Larus hyperboreus*)	2	1.10 ±0.48	22.9 ±0.3	7.18 ±0.1	0.39 ±0.02	0.016 ±0.002
Northern fulmar (*Fulmarus glacialis*)	3	6.48 ±3.32	47.0 ±0.6	-	5.75 ±0.20	-
Little auk (*Alle alle*)	5	13.7 ±9.9	94.0 ±0.7	-	0.67 ±0.04	-

1 - Mean value of individual samples; 2 - Pooled sample.

6 Summary and conclusions

In 2005 samples of seawater, sediment, and seaweed were collected in the Barents Sea, the North Sea, the Skagerrak and at a number of coastal stations. Data on radioactivity levels and trends in these areas together with data on liquid discharges from nuclear installations and recent trends in such discharges are summarised in this report.

6.1 Sources

The liquid discharges from the nuclear facilities of IFE at Kjeller and in Halden in 2005 are within authorised limits according to the estimates made by the operator. The effective dose to a member of the critical group from liquid radioactive discharge at Kjeller was estimated to be 0.02 µSv in 2004, which correspond to 2 % of the dose limit. The effective dose to the critical group from IFE Halden liquid radioactive discharge was estimated to be 0.025 µSv in 2004, corresponding to 2.5 % of the dose limit. The discharge of anthropogenic radionuclides from Norwegian sources is only detectable in the environment close to each discharge point and has no significant impact on the large-scale distribution of these radionuclides in the marine environment.

Produced water from offshore oil production may contain enhanced levels of naturally occurring radium isotopes. The reported total discharge of ^{226}Ra in produced water from Norwegian offshore installations in 2005 was 390 GBq, where about 50 % was discharged from 2 platforms, Troll B and Troll C.

The long-range transports of radionuclides originating from nuclear weapons fallout, the Chernobyl accident and from reprocessing of spent nuclear fuel are still the main contributors to anthropogenic radionuclides found in Norwegian waters. In addition to direct discharges from reprocessing, water from the Baltic Sea and remobilisation of ^{137}Cs and plutonium from contaminated Irish Sea sediments act as secondary sources of radionuclides to the Norwegian marine environment.

6.2 Radioactivity in seawater and sediment

In 2005, samples of seawater and sediment were collected in the North Sea and the Skagerrak and analysed with respect to ^{137}Cs, ^{99}Tc, ^{90}Sr, ^{238}Pu, $^{239+240}$Pu, ^{241}Am, ^{226}Ra and ^{228}Ra. A general trend seen in most samples is that the levels of radioactivity are similar, or slightly lower than have been observed in recent years.

6.2.1 Caesium-137 in seawater and sediment

Observed levels of ^{137}Cs in surface water in the Barents Sea, the North Sea, the Skagerrak and along the Norwegian coast in 2005 ranged from 1.4 to 11.4 Bq m^{-3}. The highest activity concentrations were found in the Skagerrak and in the Norwegian Coastal Current. The observed levels in Skagerrak are lower in 2005 than in 2004. The relatively high activity concentrations in the Skagerrak in 2004 were most likely due to a large outflow of seawater from the Baltic Sea. The observed activity concentration of ^{137}Cs in the Barents Sea and the North Sea are similar to the levels found in 2003 and 2004. The activity concentration of ^{137}Cs in sediment ranged from 0.4 to 415 Bq kg^{-1} (d.w.). The highest levels were found in Sognefjorden and are the result from fallout and run-off from the Chernobyl accident in 1986.

6.2.2 Technetium-99 in seawater

Samples of seawater were collected in the Barents Sea and at a number of coastal stations in the Skagerrak, Barents Sea and the Norwegian Sea. Levels of ^{99}Tc ranged from 0.13 to 0.72 Bq m^{-3} in the Barents Sea and from 0.7 to 1.5 Bq m^{-3} in seawater at coastal stations in the Skagerrak. Average activity concentration from monthly sampling at Hillesøy was 0.88 ± 0.1, which is lower than the peak values observed in 1999/2000, but similar to the levels found in 2003 and 2004. Due to the reduced discharge from Sellafield in 2003, the levels are expected to decrease in 2006 and 2007.

6.2.3 Strontium-90 in seawater

In the North Sea and in the Skagerrak, the activity concentration in seawater ranged from 1.2 Bq m^{-3} to 3.7 Bq m^{-3} in 2005. The highest concentrations were found off the southern coast of Norway in the Skagerrak. The levels of ^{90}Sr in the North Sea and the Skagerrak are similar to those observed in 2001 and 2004. In the Barents Sea the levels of ^{90}Sr ranged from 1.0 to 2.7 Bq m^{-3}, which is similar to the levels observed in 2002.

6.2.4 Plutonium-239+240 and americium-241 in seawater

Observed levels of $^{239+240}$Pu in the North Sea in 2005 ranged from 3.4 to 29 mBq m^{-3}, where the highest level was found off the coast of Scotland. The observed levels of $^{239+240}$Pu in the North Sea in 2005 were lower than those found in the same area in 2001.

In the Barents Sea the activity concentration of $^{239+240}$Pu in seawater ranged from 1.8 to 20 mBq m^{-3}. Except for somewhat higher activity concentrations in two samples collected off the northern coast of Norway, the levels in the Barents Sea in 2005 are similar or lower than in 2002. The reason for the slightly elevated concentrations is not easy to explain due to the low number of samples.

The activity concentration of ^{241}Am in seawater from the Barents Sea ranged from 1.0 to 19 mBq m^{-3}. Except for two samples with slightly elevated levels of ^{241}Am, the activity concentration of ^{241}Am in the Barents Sea are similar to what was observed in 2002. Also for ^{241}Am, the reason for the slightly elevated concentrations is not easy to explain due to the low number of samples.

6.2.5 Radium-226 and radium-228 in seawater

Radium-226 and ^{228}Ra were analysed in seawater samples collected in the Barents Sea and the North Sea and in the Skagerrak. The activity concentration of ^{226}Ra in the samples ranged from 0.4 to 5.1 Bq m^{-3}, where the highest levels were found in coastal water from the North Sea and the Skagerrak.

6.3 Radioactivity in biota

6.3.1 Technetium-99 in seaweed

Samples of *Fucus vesiculosus* sampled at the permanent coastal stations showed activity concentrations in the range 73 to 265 Bq kg^{-1} (d.w.). For most stations the levels were similar in 2005 compared to observed levels in the period 2002 - 2004. The levels in 2005 are however significantly lower than the peak values observed in 1999/2000. Monthly sampling at Hillesøy and Utsira all indicate that the peak of ^{99}Tc from the Sellafield discharge in 1995 and 1996 has passed these sites. The activity concentration of ^{99}Tc in seaweed from Utsira showed a slight increase in 2005 compared to 2003 and 2004, which can be explained by the relatively high Sellafield discharge of ^{99}Tc in 2001 and 2002.

6.3.2 Caesium-137 in seaweed

The activity concentration of ^{137}Cs in *F. vesiculosus* sampled at the coastal stations was in the range 0.21 to 3.7 Bq kg^{-1} (d.w.), where the highest levels were found in the southern part of Norway, where the impact from outflowing Baltic Sea water is highest. Apart from a higher activity concentration of ^{137}Cs in the water, lower salinity also leads to a higher uptake of ^{137}Cs in the seaweed. A comparison with results from 2000 and onwards shows that the levels have been relatively stable in recent years.

6.3.3 Technetium-99 and caesium-137 in crustaceans and molluscs

Samples of lobster (*Hommarus gammarus*), mussels (*Mytilus edilus*) and other marine species have been sampled at different locations and analysed for ^{137}Cs and ^{99}Tc. The activity concentrations of ^{137}Cs in all analysed samples were low, < 0.4 Bq kg^{-1} (w.w.). Technetium-99 was analysed in lobsters where the average level in female lobsters was 38 Bq kg^{-1} (w.w.) and 4 Bq kg^{-1} in male lobsters.

6.3.4 Caesium-137 in fish

Different species of fish, commercially important and others, have been sampled in the Barents Sea, Norwegian Sea, the North Sea and the Skagerrak. The activity concentrations of ^{137}Cs in fish are generally low. All reported results in 2005 were below 1 Bq kg^{-1} (w.w.).

References

Amundsen et al., 2003
Amundsen I, Brekken A, Liland A. Utslipp av radioaktive stoffer fra Sellafield-anleggene. En gjennomgang av britiske myndigheters regulering av utslippstillatelser. StrålevernRapport 2003:2. Østerås: Statens strålevern, 2003 (In Norwegian, English summary). http://www.nrpa.no/dokumentarkiv/Stralevernrapport2_2003.pdf (06.01.06)

Aure et al., 1998
Aure J, ed. Havets miljø 1998. Fisken og havet, særnr. 2-1998. Bergen: Havforskningsinstituttet, 1998. (In Norwegian)

Beasley and Lorz, 1986
Beasley TM, Lorz HV. A review of the biological and geochemical behaviour of technetium in the marine environment. Journal of Environmental Radioactivity 1986; 3: 1-22.

Bourlat et al., 1996
Bourlat Y, Milliès-Lacroix J-C, Le Petit G, Bourguignon J. ^{90}Sr, ^{137}Cs and $^{239+240}$Pu in world ocean water samples collected from 1992 to 1994. In: Guéguéniat P, Germain P, Métivier H, eds. Radionuclides in the oceans. Les Ulis: Institute de Protection et de Surete Nucleaire / Les editions de physique, 1996: 75-93.

Brown et al., 2002
Brown J, Iospje M, Kolstad AK., Lind B, Rudjord AL, Strand P. Temporal trends for ^{99}Tc in Norwegian coastal environments and spatial distribution in the Barents Sea. Journal of Environmental Radioactivity 2002; 60: 49-60.

Carlsson L. and Erlandsson B., 1991
Carlsson L, Erlandsson B. Effects of salinity on the uptake of radionuclides by *Fucus vesiculosus* L. Journal of Environmental Radioactivity 1991; 13: 309-322.

Carvalho, 1988
Carvalho FP. 210Po in marine organisms: A wide range of natural radiation dose domains. Radiation Protection Dosimetry 1988; 24: 113-117.

Chen et al., 1991
Chen Q et al. Determination of plutonium in environmental samples by controlled valence in anion exchange. RISØ-M-2856. Roskilde: Risø National Laboratory, 1991.

Chen et al., 2001
Chen Q, Aakrog A, Nielsen SP, Dahlgaard H, Lind B, Kolstad AK, Yu, Y. Procedures for determination of ^{239,240}Pu, ^{241}Am, ^{237}Np, ^{234,238}U, ^{228,230,232}Th, ^{99}Tc, ^{210}Pb and ^{210}Po in environmental materials. Risø-R-1263. Roskilde: Risø National Laboratory, 2001.

Cook et al., 1997
Cook GT, Mac Kenzie AB, Mc Donald P and Jones SR . Remobilization of Sellafield-derived radionuclides and transport from the North-East Irish Sea. Journal of Environmental Radioactivity 1997; 35: 227-241.

Dahlgaard, 1995
Dahlgaard. (1995). Transfer of European Coastal Pollution to the Arctic: Radioactive tracers. Marine Pollution Bulletin, Vol. 31, Nos 1-3, pp 3-7.

Dahlgaard, 2002
Dahlgaard H. Baltic ^{137}Cs outflow through the Danish Straits indicates remobilisation. In: Strand P, Børretzen P, Jølle T, eds. International conference on radioactivity in the environment, Monaco 2002. Extended abstracts. Österås: Norwegian Radiation Protection Authority, 2002: 516-520.

Environment Agency *et al*., 2006
Environment Agency, Environment and Heritage Service, Food Standards Agency & Scottish Environment Protection Agency, 2006. Radioactivity in food and the environment 2005. RIFE-11. Preston 2006.

Evans, 1991
Evans S. Impacts of the Chernobyl fallout in the Baltic Sea ecosystem. In: Moberg L, ed. The Chernobyl fallout in Sweden. Stockholm: Swedish Radiation Protection Institute, 1991: 109-127.

Gwynn et al., 2004
Gwynn JP, Dowdall M, Lind B. The radiological environment of Svalbard. StrålevernRapport 2004:2. Østerås: Norwegian Radiation Protection Authority, 2004.

Gäfvert et al. 2003
Gäfvert T, Føyn L, Brungot AL, Kolstad AK, Lind B, Christensen GC, Strålberg E, Drefvelin J, Rudjord AL.. Radioactivity in the marine environment 2000 and 2001. Results from the Norwegian National Montoring Programme (RAME). StrålevernRapport 2003:8. Østerås: Norwegian Radiation Protection Authority, 2003. http://www.nrpa.no/dokumentarkiv/StralevernRapport8_2003.pdf (06.01.06)

Holm and Rioseco, 1986
Holm and Rioseco, 1986, Technetium-99 in Algae from Temperate and Arctic Waters of the North Atlantic. In: Technetium in the Environment (Ed. G. Desmet and C. Myttenaere), 1986, p. 53, Elsevier Applied Science Publ., Proc. of a Seminar on the Behaviour of Technetium in the Environment, Cadarache, France, 23-26 October 1984.

IAEA, 1990
IAEA (International Atomic Energy Agency). The Environmental Behaviour of Radium. Vol. 1. Technical Reports Series No. 310. ISBN 92-0-125090-9. Vienna, 1990

IFE, 2002a
IFE. Måling av radioaktive stoffer i utslipp til luft, i avløpsvann og i prøveobjekter fra IFE Kjellers omegn og Nitelva. Rapport for 2001. IFE Rapport nr: IFE/I-2002-09. Kjeller: Institutt for energiteknikk, IFE, 2002.

IFE, 2003a
IFE. Kontroll av radioaktive stoffer i utslipp til luft og vann og i prøveobjekter fra IFE Kjellers omegn, Nitelva og Himdalen. Rapport for 2002. IFE Rapport nr: IFE/I-2003/016. Kjeller: Institutt for energiteknikk, IFE, 2003.

IFE, 2004a
IFE. Kontroll av radionuklider i utslipp til luft og vann og i prøveobjekter fra IFE Kjellers omegn, Nitelva og Himdalen. Rapport for 2003. IFE Rapport nr: IFE/I-2004/012. Kjeller: Institutt for energiteknikk, IFE, 2004.

IFE, 2005a
IFE. Kontroll av radionuklider i utslipp til luft og vann og i prøveobjekter fra IFE Kjellers omegn, Nitelva og Himdalen. Rapport for 2004. IFE Rapport nr: IFE/I-2005/009. Kjeller: Institutt for energiteknikk, IFE, 2005.

IFE, 2006a
IFE. Måling av radioaktive stoffer i utslipp til luft, i avløpsvann og i prøveobjekter fra IFE Kjellers omegn og Nitelva. Rapport for 2005. IFE Rapport nr: IFE/I-2006/012. Kjeller: Institutt for energiteknikk, IFE, 2006.

IFE, 2002b
IFE. Utslipp fra Haldenreaktoren i 2001. Sv-rapport 511. Halden: Institutt for energiteknikk, IFE, 2002.

IFE, 2003b
IFE. Utslipp fra Haldenreaktoren i 2002. Sv-rapport 541. Halden: Institutt for energiteknikk, IFE, 2003.

IFE, 2004b
IFE . Utslipp fra Haldenreaktoren i 2003. Sv-rapport 586. Halden: Institutt for energiteknikk, IFE, 2004.

IFE, 2005b
IFE. Utslipp fra Haldenreaktoren I 2004. Sv-rapport 629. Halden: Institutt for energiteknikk, IFE, 2005.

IFE, 2006b
IFE. Utslipp fra Haldenreaktoren i 2005. Sv-rapport 666. Halden: Institutt for energiteknikk, IFE, 2006.

Kershaw et al., 1999.
Kershaw PJ, McCubbin D, Leonard KS. Continuing contamination of North Atlantic and Arctic waters by Sellafield radionuclides. The Science of the Total Environment 237-238: 119-132.

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

Leonard et al., 1999
Leaonard KS, McCubbin D, Blowers P, Taylor BR. Dissolved plutonium and americium in surface waters of the Irish Sea, 1973-1996. Journal of Environmental Radioactivity 1999; 44:129-158.

NRPA, 2004.
NRPA. Radioactivity in the marine environment 2002. Results from the Norwegian National Monitoring Programme (RAME). StrålevernRapport 2004:10. Østerås: Norwegian Radiation Protection Authority, 2004. http://www.nrpa.no/dokumentarkiv/StralevernRapport10_2004.pdf (06.01.06)

NRPA, 2005a
NRPA. Natural radioactivity in produced water from the Norwegian oil and gas industry in 2003. StrålevernRapport 2005:2. Østerås: Norwegian Radiation Protection Authority, 2005. http://www.nrpa.no/dokumentarkiv/StralevernRapport2_2005.pdf

NRPA, 2005b
NRPA. Radioactivity in the Marine Environment 2003. Results from the Norwegian National Monitoring Programme (RAME). StrålevernRapport 2005:20. Østerås: Norwegian Radiation Protection Authority, 2005.

NRPA, 2006.
NRPA. Radioactivity in the Marine Environment 2004. Results from the Norwegian National Monitoring Programme (RAME). StrålevernRapport 2006:14. Østerås: Norwegian Radiation Protection Authority, 2006.

OLF, 2006
OLF (Oljeindustriens Landsforening, Stavanger). Miljørapport 2005. Utslippsdata fra de enkelte innterningene på norsk sokkel, se nettsiden til Miljørapport 2005: http://www.olf.no/publikasjoner/miljorapporter/?32711 (18.09.07)

OSPAR, 2007
OSPAR Commission. Liquid discharges from nuclear installations in 2005, including exceptional discharges from decommissioning and management of legacy radioactive wastes. Radioactive substances series, Publication number 340/2007. London: OSPAR, 2007.

Roos et al., 1994
Roos P, Holm E, Persson RBR. Comparison of AMP precipitate method and impregnated $Cu_2[Fe(CN)_6]$ filters for the determination of radiocesium concentrations in natural waters. Nuclear Instruments and Methods in Physics Research A 1994; 339: 282-286.

Rudjord et al., 2001
Rudjord AL, Føyn L, Brungot AL, Kolstad AK, Helldal HE, Brown J, Iosjpe M, Christensen G. Radioactivity in the marine envioronment 1999. StrålevernRapport 2001:9. Østerås: Norwegian Radiation Protection Authority, 2001. http://www.nrpa.no/dokumentarkiv/StraalevernRapport9_2001.pdf (06.01.06)

Smith, 1987
Smith AL. Radioactive scale formation, Offshore Technology Conference, Journal of Petroleum Technology, June 1987.

Smith et al., 1998
Smith DL, Knowles JF, Winpenny K. The accumulation,retention and distribution of ^{95m}Tc in crab (*Cancer pagurus L*) and lobster (*Homarus gamarus L*). A comparative study. Journal of Environmental Radioactivity 1998; 40: 113-135.

Smith et al., 2001
Smith V, Fegan M, Pollard D, Long S, Hayden E, Ryan P. Technetium-99 in the Irish marine environment. Journal of Environmental Radioactivity 2001; 56: 269-284.

Strand, 1998
Strand P, Balonov M, Aarkrog A, Bewers MJ, Howard B, Salo A, Tsaturov YS. Radioactivity. In: AMAP assessment report: Arctic pollution issues, Chapter 8. Oslo: Arctic Monitoring and Assessment Programme, AMAP 1998: 525-620. Also available as offprint from AMAP secretariat, Oslo.

Swift, 1985
Swift D. J. The accumulation of ^{95m}Tc from sea water by juvenile lobsters (*Homarus gammarus*). Journal of Environmental Radioactivity 1985; 2: 229-243.

Topcuoglu and Fowler, 1984
Topcuoglu S, Fowler SW. Factors affecting the biokinetics of technetium (^{95m}Tc) in marine macroalgae. Marine Environmental Research 1984; 12: 25-43.

UNSCEAR (2000a)

UNSCEAR. United Nations Scientific Committee on the Effects on Atomic Radiation. Sources and effects of ionizing radiation: United Nations report to the general assembly, with scientific annexes. Volume I: Sources. New York: United Nations, 2000.

UNSCEAR (2000b)
UNSCEAR. United Nations Scientific Committee on the Effects on Atomic Radiation, Sources and effects of ionizing radiation: United Nations report to the general assembly, with scientific annexes. Volume II: Effects. New York: United Nations, 2000.

Vakulovsky et al., 1995
Vakulovsky SM, Kryshsev II, Nikitin AI. Radioactive contamination of the Yenisei River. Journal of Environmental Radioactivity 1995; 29: 225-236.

Varskog et al., 1997
Varskog P, Bjerk TO, Ruud AB. Ph-control EDTA titration as an alternative method for the determination of the chemical yield of yttrium in ^{90}Sr analysis. In: Rapid radioactivity measure-ments in emergency and routine situations. Teddington: National Physical Laboratory, NPL, 1997: 237-241.

Vartti et al., 2006
Vartti V-P, Illus E, Mattila J, Klemola S, Ikäheimonen TK. Monitoring of radionuclides in the Baltic Sea in 2005. Helsinki Commission. Project Group for Monitoring of Radioactive Substances in the Baltic Sea (MORS), Eleventh meeting, Monaco 2006. HELCOM MORS-PRO 11/2006, Document 3/3. Helsinki: Radiation and Nuclear Safety Authority, STUK, 2006. http://sea.helcom.fi:15037/dps/docs/documents/Monitoring%20and%20Assessment%20Group%20(MONAS)/MORS%20PRO%2011,%202006/33.pdf#search=%22Monitoring%20of%20radionuclides%20in%20the%20Baltic%20Sea%20in%202005%22 (18.09.06)

Zalewska and Lipska, 2006
Zalewska and Lipska. ^{137}Cs and ^{90}Sr distribution in the Southern Baltic Sea in 2004 and 2005. Helsinki Commission. Project Group for Monitoring of Radioactive Substances in the Baltic Sea (MORS), Eleventh meeting, Monaco 2006. HELCOM MORS-PRO 11/2006, Document 3/2.

Appendix

Analytical methods

During recent years, laboratories at the Norwegian Radiation Protection Authority, and the Institute of Marine Research have both been accredited according to the requirements of NS-EN ISO/IEC 17025. Laboratories and organisations that are not named here have not been accredited according to the requirements of NS-EN ISO/IEC 17025. The scope and conditions governing the accreditation of the institutes mentioned above are quite different, but they have all been accredited for gamma spectrometric measurements. For example, the NRPA is accredited for gamma spectrometric measurements in the energy interval 100-1800 keV (except ^{226}Ra), while the other organisations are accredited for gamma spectrometric measurements of ^{137}Cs. The analytical techniques employed at each institution are described below. The results in this report are not claimed to be accredited results as not all results are accredited.

The Norwegian Radiation Protection Authority (NRPA)

NRPA analyses alpha, beta and gamma emitters, using the procedures described in this Appendix.

Determination of ^{99}Tc activity

To determine the activity concentration of ^{99}Tc in seawater, samples of 50 litres were filtered through a 1-micron polypropylene cartridge to remove suspended particulate matter. Seaweed and other biota samples were dried, milled and homogenized. A 10-20 g dried sample was transferred to a specially designed bottle and carbonised and then dissolved by adding concentrated H_2SO_4 followed by HNO_3. ^{99m}Tc was added to all samples for chemical recovery determination.

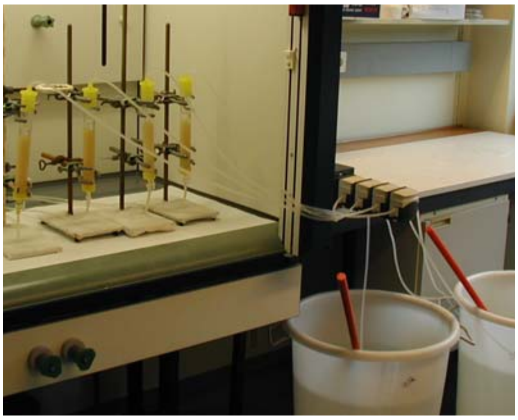

Figure A.1. *Fifty litres of seawater is passed through an ion-exchange column.*

The analytical procedure used is a modified version of that developed by Risø National Laboratory, Denmark (Chen *et al*., 2001). The procedure is outlined briefly below. Technetium is initially separated from the matrix by ion-exchange chromatography using a BIO-RAD AG1-X4 (100-200 mesh) column (Figure A.1) and separation techniques such as precipitation and solvent extraction are then applied before the technetium is electro-deposited onto stainless steel discs (Figure A.2). The chemical yields were determined by gamma counting of the ^{99m}Tc tracer in a NaI well-type detector. Typically, the radio-chemical yields varied between 70 % and 85 %. After one week, the ^{99}Tc activity was measured using a low-background anti-coincidence beta counter (Model Risø GM-25-5).

Figure A.2. *Dried sample material in different containers ready for gamma measurements. To the right, ^{99}Tc preparations ready for counting.*

The limits of detection for 10 g seaweed and 50 l seawater have been calculated to be approximately 0.5 Bq kg^{-1} (d.w.) and 0.10 Bq m^{-3}, respectively. The limit of detection may vary slightly owing to variations in chemical yield, counting efficiency and the mass of the sample. The total uncertainty in ^{99}Tc analysis is normally around 10 %.

Detection of gamma emitters

At NRPA analyses of gamma-emitting nuclides are performed with HPGe detectors. All the detectors are situated in a low-background laboratory to ensure low detection limits. The HPGe detectors have relative efficiencies in the range of 23 % to 40 %. The resolution of the detectors, the full width at half maximum (FWHM) at 1332 keV, was less than 1.9 keV. Three of the detectors cover the energy interval 50-2000 keV, and two cover the interval 20-2000 keV.

Caesium-absorbing filters (sometimes also the prefilter) from the filtering system for seawater samples were dried separately at 105°C and ashed at 450°C before the activity was determined with an HPGe detector. The counting time varied from 1 to 4 days.

Samples of seaweed were dried at 105°C and homogenized, and placed in containers prior to gamma counting. Sediment samples were freeze-dried and placed in containers prior to gamma counting. Samples were counted for a minimum of 2 days. At NRPA, the detection limits for ^{137}Cs and ^{134}Cs in the sediment samples ranged from approximately 0.3-2 Bq kg^{-1} (d.w.) Uncertainties in the analysis include uncertainties from counting statistics, calibration and sample preparation and are normally in the range 5% to 20%.

The Institute of Marine Research (IMR)

Detection of gamma emitters

Sediment samples were transported deep-frozen to IMR, where they were subsequently ground, freeze-dried, homogenized and transferred to polyethylene containers of appropriate size prior to analysis. The gamma counting for ^{137}Cs detection was performed in IMR's low-background laboratory using a HPGe detector with 30 % relative efficiency and an HPGe detector with 60 % relative efficiency. Both detectors have electric cryostat cooling systems, and 10 cm lead shielding.

Institute for Energy Technology (IFE)

Determination of ^{99}Tc

Samples of dried and homogenized *Fucus vesiculosus* were analysed with regard to ^{99}Tc content. The pretreatment was performed at Kjeller and the analysis at the Department of Radiation Physics, at Lund University. The following analytical procedure was used at the Department of Radiation Physics at Lund University, Sweden. Technetium was extracted into tri-butyl-phosphate (TBP) from sulphuric acid-hydrogen fluoride solution. Technetium was then back-extracted from the organic phase with a sodium hydroxide solution from which the technetium was electrodeposited onto stainless steel discs, and ^{99m}Tc was used as a radiochemical yield determinant. After decay of the yield determinant, ^{99}Tc was measured with an anti-coincidence-shielded GM counter (Holm *et al.*, 1984).

Determination of ^{90}Sr

For ^{90}Sr, the standard method using fuming nitric acid was used. The recovery of ^{90}Sr in the analytical process was monitored by adding ^{85}Sr as a yield determinant, and the recovery of the daughter nuclide ^{90}Y was determined by titration with EDTA (Varskog *et al.*, 1997). Finally, ^{90}Y was measured with an anti-coincidence-shielded GM counter.

Detection of gamma emitters

At IFE analyses of gamma-emitting nuclides are performed with low-background HPGe detectors. Samples of seaweed were dried at 105°C and then homogenized, and placed in suitable containers prior to gamma counting.

Determination of $^{239+240}$Pu

Samples of dried and homogenized *Fucus vesiculosus* were ashed and treated with *aqua regia* before separation of plutonium isotopes. ^{242}Pu was used as a tracer for radiochemical yield determination. The separation process involved solvent extraction with 10 % TIOA/Xylene followed by anion exchange on AG1-X4 columns. The plutonium fraction were then electrodeposited on a stainless steel disc and counted using alpha spectrometry.

Risø National Laboratory

Determination of ^{226}Ra and ^{228}Ra in seawater

Radium was coprecipitated with MnO_2 from 10 l of seawater, after ^{133}Ba had been added as a yield determinant. The MnO_2 precipitate was then dissolved in hydrochloric acid and hydrogen peroxide, and after the addition of sulphuric acid, K_2SO_4 and $Pb(NO_3)_2$, radium was coprecipitated with $PbSO_4$. The precipitate was dissolved in EDTA at pH = 10, and transferred to a liquid scintillation vial. After measuring the radiochemical yield with gamma spectrometry, a liquid scintillation cocktail was added (OptiFluor O). The sample was then stored for about four weeks after which the activity of ^{226}Ra was measured through ^{214}Po with a low-background, liquid scintillation counter (Quantulus). The laboratory work was carried out at Risø National Laboratory. The ^{228}Ra/^{226}Ra ratio was determined by pumping several hundred litres of seawater through a MnO_2-impregnated polyethylene filter. The filter was later ashed and ^{226}Ra and ^{228}Ra were analysed by gamma spectrometry.

Department of Radiation Physics, Lund University

Determination of $^{239+240}$Pu and ^{241}Am in seawater

The concentrations of ^{238}Pu, $^{239+240}$Pu and ^{241}Am were measured in samples of 200 litres of seawater. ^{242}Pu and ^{243}Am were added for chemical yield deter-mination. Precipitation from the water samples was achieved using the analytical procedure described by Chen *et al.*, (1991). Different radiochemical separation techniques were applied to separate plutonium and americium from other nuclides using solvent extraction with 10 % TIOA/xylene solution and ion-exchange chromatography with a BIO-RAD AG1-X4 (100-200 mesh) column. Purified americium and plutonium fractions were electrodeposited on stainless steel discs and the activity measured in semiconductor silicon detectors.

Statens strålevern
Norwegian Radiation Protection Authority

StrålevernRapport 2007:1
Virksomhetsplan 2007

StrålevernRapport 2007:2
Representative doser i Helse Øst. Representative doser for røntgendiagnostikk rapportert fra virksomheter i Helse Øst høsten 2006

StrålevernRapport 2007:3
Radioecological consequences of a potential accident during transport of radioactive materials along the Norwegian coastline

StrålevernRapport 2007:4
Measuring radon levels at high exposures with alpha-track detectors. Calibration and analysis

StrålevernRapport 2007:5
Upgrading the Regulatory Framework of the Russian Federation for the Safe Decommissioning and Disposal of Radioisotope Thermoelectric Generators

StrålevernRapport 2007:6
Stråledose til screena kvinner iMammografiprogrammet i 2005 og 2006

StrålevernRapport 2007:7
Implementation of the Obligations of the Convention on Nuclear Safety in Norway

StrålevernRapport 2007:8
Årsrapport fra persondosimetritjenesten ved Statens strålevern 2006

StrålevernRapport 2007:9
Implementation of the Obligations of the Convention on Nuclear Safety in Norway. The fourth Norwegian Report in Accordance with Article 5 of the Convention

Lobo Media 43521

ISSN 0804-4910