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# Contamination and Radiation Exposure

## Evaluation and Measures in the Nordic Countries after the Chernobyl Accident

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**Sammanfattning/Abstract:**

This is a summary of the major consequences in the Nordic countries of the Chernobyl accident during a time period of 5 - 10 years after the accident 1986. There is information given and discussions on the emergency system at that time, the deposition in the Nordic countries, the dispersion in the environment, resulting external and internal doses, countermeasures and their costs and effects. More than 10 % of total release of cesium-137 was deposited over the Nordic countries, deposition density up to about 200 kBqm$^{-2}$ and the estimated collective effective doses over 50 years about 24 000 manSv.

This article has been presented in a BfS/SSK-Seminar "Ten Years After Chernobyl, a Summation" Munich, 6 - 7 March, 1996.

Detta är en sammanfattning av konsekvenserna i de Nordiska länderna av Tjernobylolyckan. Beredskap, nedfall, beläggning, spridning i miljön, resulterande doser, motåtgärder beskrivs. Beläggning upp till ungefär 200 kBqm$^{-2}$, cirka 24 000 manSv över 50 år beräknas från nedfallet som motsvarade mer än 10 % av totala utsläppet.

Denna artikel har presenterats på ett BfS/SSK seminarium "Tio år efter Tjernobyl, en sammanfattning" i München, 6 - 7 mars, 1996.

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# Contamination and Radiation Exposure. Evaluation and Measures in the Nordic Countries after the Chernobyl Accident

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Summary

In the night 25/26 April 1986 it all started. The radioactive releases from Chernobyl reached the Nordic countries late 27 April. At noon the following day all the world knew that an accident had occurred - two and a half days after it started.

In April 1986 there was, generally speaking, quite a high level of preparedness for nuclear accidents in the Nordic countries but still, as it was shown, quite insufficient for accidents abroad with such extensive countrywide consequences as the Chernobyl accident proved to have.

The total deposition over the Nordic countries was about 10 PBq Cs-137 and the major part was found in wet deposition areas. As a total of about 80-90 PBq Cs-137 is assumed to have been released from Chernobyl, more than 10% was deposited over the Nordic countries with the average Cs-137 concentration of 1.2 $kBqm^{-2}$ in Denmark, 11 in Finland, 5.1 in Norway and 10 in Sweden.

The deposition from the Chernobyl accident added to the present fallout from the atmospheric tests in early 60's. The two depositions were of the same order of magnitude e.g in Sweden 4.25 PBq of Cs-137 from Chernobyl and 1.25 PBq from the atmospheric tests. However the fallout during the 60's was rather evenly distributed with an average of about 3 $kBqm^{-2}$ while the Chernobyl deposition was extremely variable from almost zero up to about 200 $kBqm^{-2}$.

The deposition of I-131 was also of interest from the radiation protection point of view. However because of the relatively short half-life of I-131 (8 days) there was only a problem the first weeks when care had to be taken in allowing the cows to go outdoors to avoid contamination of milk.

The deposition of Sr -90 and its dose consequences for people were minor in comparison with Cs 134+137, about two order of magnitude less.

The deposition of Pu 239+240 was on average less than 1% of the earlier deposition from nuclear test fallout (40 $Bqm^{-2}$). The radiological impact of Pu is insignificant. There were also some observations of so called hot particles in the air over the Nordic countries. They were not considered to imply any significant radiological risk.

The Cs-activity on the ground has decreased by physical decay and by wash-out/run-off processes followed by outflow in rivers into surrounding seas. However, the outflow to the Baltic Sea waters decreased rapidly and the total outflow from land during the first three years corresponds only to about 2% of the total Chernobyl fallout over Finland and Sweden as compared to 7% diminution caused by physical decay. Therefore it could be concluded that much of the Cs-137 deposited on ground still remains in the soil and in stream sediments in catchment areas of lakes. This is true also for the lake sediments. The major part of cesium is in the upper 5 cm.

Most of the radiocesium entering the lakes was rapidly transferred to the bottom by sedimentation. The total amount of Cs-137 in most lakes has not changed significantly more than by physical decay since 1986, and the sediments contain from 80 to almost 100% of the Cs-137 in lakes. The fallout from the Chernobyl accident caused heavy contamination of thousands of lakes in the Nordic countries. In Sweden alone about 14000 lakes were contaminated so seriously that the fish had activity concentrations exeeding the level applied for fish not allowed to be sold on the open market (1500 Bq/kg of Cs-137). There are still several thousands lakes of that kind and the problems with lakes and fish are predicted to exist even into the 21st century.

The contamination of terrestrial vegetation in the first period after the Chernobyl accident was primarily by direct contamination, particularly in the southern part of the Nordic countries. In the northernmost part there was still some snow cover at the time of the accident. Gradually the contamination was caused by root uptake of activity deposited on ground. An assumed transfer factor of 0.1 $m^2kg^{-1}$ the first year and 0.01 $m^2kg^{-1}$ following years does not seem to underestimate the uptake in grass in most cases.

For edible parts of crops the transfer factors lie between 0.001 and 0.01 $m^2kg^{-1}$ during 1986 and between 0.0001 and 0.001 $m^2kg^{-1}$ 1987-1990. Because of these low values contamination of agricultural products has not been a problem.

That is also true for milk (cow). The mean Cs-137 activity in dairy milk in 1986 and 1987 varied from 0.6 to 20 $Bql^{-1}$ in the Nordic countries. The aggregated transfer factor ($Bql^{-1}$ milk per $Bqm^{-2}$ ground deposition) has decreased for all countries. The transfer of Cs-137 from vegetation to cows' milk did not change significantly during 1986-1992 and ranged from 0.005 to 0.03 $dl^{-1}$ ($Bql^{-1}$ milk per Bq ingested daily by the cow) with a general mean of about 0.01 $dl^{-1}$. The effective ecological half-life for Cs-137 in milk was 1-2 years for all Nordic countries. The maximum activity concentration in milk was reached 8-9 days after the cows had been taken out for grazing.

The average transfer factors for sheep ($Bqkg^{-1}$ meat fresh weight per $kBqm^{-2}$ soil) for the years 1990-1993 are for Denmark 0.6, Finland 0.8, Iceland 15, Norway 39 and Sweden 47 (one place only). The ecological half-life for Cs-137 in sheep has been estimated to be about 13 years in undisturbed areas. Contamination of sheep has been a problem particularly in Norway.

Forest and part of alpine areas are important as pasture for grazing animals, for hunting, for berries, mushrooms etc. After 2-4 years after the fallout the transfer factor varied between 0.005 and 0.1 $m^2kg^{-1}$ dry weight of wild plants. In more acid forests, the transfer factor could be 10 times higher and in fungus fruit bodies further 10 times higher. As the ground contamination in many areas was 50 $kBqm^{-2}$ and more, it is easily understood that forest berries and other edible wild plants can be heavily contaminated. This is particularly a problem with mushrooms (transfer factor around 1-10 $m^2kg^{-1}$). The ecological half-life of Cs-137 in forest plants may be 5-10 years or more.

Contaminated plants also mean contamination of the animals in the forest. Moose and deer are most significant from the radiation dose point of view. As there are several hundred thousands of hunters in the Nordic countries there is potentially a large critical group and therefore the activity of Cs-137 in moose and deer has been given high attention. Still 10 years after the Chernobyl accident, moose and deer constitute a problem in some parts of the Nordic countries. Until more is known, it is suggested that the effective ecological half-life would be the same as the physical half-life of Cs-137 (30 years).

Reindeer are of particular interest because reindeer meat is an important part of the diet of the Laplanders and because reindeer meat can be heavily contaminated because of the reindeer's consumption of contaminated lichen. The contamination of reindeer was a great problem in Norway and Sweden, and activity levels up to 150kBq/kg were found. In Sweden in 1988 more than a third of the slaughtered animals had an activity concentration of more than 1000 $Bqkg^{-1}$ . In Finland the levels were generally less than 1000 $Bqkg^{-1}$. The estimated effective ecological half-life of Cs-137 in reindeer is 3-4 years tending to become longer.

The radiation dose during the first weeks was dominated by the dose caused by inhalation and the external dose from the cloud and from radionuclides deposited on the ground. In most parts of the Nordic countries the estimated effective dose caused by external radiation during the first year after the accident was less than 0.5 mSv. In some areas it was up to about ten times that for a few hundred people. The collective doses over 50 years are estimated to be for Denmark 700, Finland 6000, Norway 5000 and Sweden 5000 manSv. That corresponds to a total individual dose on average for the Nordic countries of about 1 mSv which is less than 1% of the natural radiation over that period of time.

The internal doses are mainly caused by ingestion of Cs-134+Cs-137 with food. The levels of internal contamination depends on diet, composition of food, place of food production, countermeasures to reduce activity concentration and time after 1986. The corresponding collective effective internal doses caused by contaminated food over 50 years would be about 150 manSv for Denmark, 3500 for Finland, 2500 for Norway and 1000 for Sweden making about 7000 manSv in total.That means that the internal dose in average over 50 years for the Nordic countries is about 0.3 mSv.

It was early concluded that measures like evacuation, sheltering and intake of stable iodine were not justified. But other measures were taken e.g giving information and advice to general public and specific groups (farmers etc), making measurements in the environment, on food and on man, issuing regulations, restrictions and advice and improving the emergency preparedness systems. Decisions on countermeasures in the Nordic countries were in the area of agriculture, reindeer, sheep, game and fresh water fish. Examples are changing farming habits, using cesium binders for animals, moving reindeer to uncontaminated areas, feeding sheep with uncontaminated fodder etc.

The costs of countermeasures, compensation, equipment, information etc because of the Chernobyl accident have up to now been 150-250 million US$ for the Nordic countries together. The objectives of these actions have generally been reached. The measures taken were sometimes cost-effective i.e the cost per saved manSv was reasonable as compared with a Nordic agreed value of 100 000 $ per manSv. But sometimes not, particularly not for more expensive countermeasures.

The situation today is that although the radiation levels have decreased a factor 2 or more, sometimes much more in Sweden and Norway, there are still areas and food that need special attention. The areas are those most heavily contaminated and the food is reindeer, sheep, moose, roe deer, fish and mushrooms, particularly from these areas. There are still several thousand lakes with fish containing more radiocesium than allowed for fish in the open market.

The emergency preparedness has been improved in all countries and there is now a much better knowledge of possible consequences and preparedness for taking decisions on countermeasures, making measurements and giving advice. There is also a much improved system for early warning and for exchange of data and information.

## The first day

In the morning of 28 April 1986 workers arriving to the Forsmark nuclear power plant in Sweden were stopped at the entrance gamma/beta monitor as being contaminated. The first reaction of the radiation protection staff and management at Forsmark was that there had been an accidental release somewhere at the plant and preparation was made for evacuation of those people not absolutely necessary for radiation protection and safe operation of the reactors.

The Swedish Radiation Protection Institute, SSI, was informed by the responsible person at the Forsmark plant soon after their observations in the morning of 28 April at about 10 o´clock am. It was first understood as "a good exercise of the emergency organization of the plant". However more information was continously reported to SSI from activity measurements in other parts of Sweden and it was soon concluded that the activity was not caused by any release from Forsmark. Information of activity measurements in Finland and Norway, air measurements from ground, weather information and dispersion trajectories indicated that the releases originated from some reactor in Ukraine. After contacts with Moscow on governmental level it was announced from Moscow later that day that there was an accident in Chernobyl Unit 4, had occurred about two and a half days earlier. From that time and for several months thereafter the life was not the same for anyone in the Nordic countries and particularly not for people at the Radiation Protection and Nuclear Safety Authorities and for any other person involved and engaged in and worried about the consequences of the Chernobyl accident.

## The existing and missing parts of the emergency system at that time

Already from 1960's there was a preparedness in the Nordic countries of nuclear bombtest fallout in terms of gamma- and air monitoring stations, routine measurements on environmental samples like milk, reindeer meat etc and action levels were defined for food consumption, evacuation etc.

Because Finland and Sweden have nuclear power programs (4 operating power reactors in Finland and 12 in Sweden) they also have emergency preparedness plans particularly adjusted for domestic accidents. In Denmark and Norway there are research reactors in Risö and Halden-Kjeller respectively and some emergency preparedness is justified for these reactors but the major sources of concern in these countries as a basis for their emergency preparedness plans were the reactors in their neighbour countries. In Iceland the preparedness was quite limited because of the long distance to any reactor. In conclusion, in April 1986 there was generally speaking quite a high level of preparedness for nuclear accidents in the Nordic countries but still, as it was shown, quite insufficient for accidents abroad with such extensive countrywide consequences as the Chernobyl accident proved to have.

The parts of the emergency system that worked well were the gamma monitoring stations at ground level, even if there was an uncomplete data transfer system, the airmonitoring at ground level, for instance that of the 7 stations of the National Defence Research Establishment (FOA) in Sweden, the air measurements made at high altitude from airplanes of the Airforce, the airborne gamma measurements at low altitude from airplanes (e.g that of the Swedish Geological Company (SGAB)), the ground deposition measurements made by experts of the Nordic radiation protection authorities, the university institutes, the nuclear power plants, the nuclear reactor research institutes and others. There was also an emergency preparedness in the level of knowledge in all related areas because of research in and experiences of the earlier fallout situations during the 60's and follow up studies in the 70's. Contact points were also preorganized and meetings between several domestic organisations and authorities with responsibilities in an emergency situation could

easily be arranged. Good contacts also existed between the Nordic countries and several meetings were organized between responsible authorities in radiation protection, nuclear safety, food control, agriculture etc.

But there were also lacking parts in organization, experience, knowledge etc and because of that, rapid development and new decisions had to be made in many areas. In Sweden the SSI gamma stations worked well and gave very good information about the general geographical distribution and levels of the deposition expressed in gamma doserates but the routines for immediate alarm to SSI were insufficient. Figure 1 shows the measured gamma levels on the island Öland which was one of the first Swedish areas reached by the radioactive plume (P E Kjelle, 1991). Already in the evening of 27 April the activity reached Sweden, but SSI was not notified and by that all emergency activities were delayed more than 12 hours. In Finland there was a strike among civil servants so there was nobody to look at the information given by the measurement network on the 27 April and the information reached the authorities not until the morning of the 28 April.

Even if the mobile measurement equipments on land were working well after some delay they were too few and the reporting routines were not sufficiently trained. Many samples on vegetation and food were taken and had to be measured at the laboratories for Sr-90 and plutonium and the limited capacities were quite evident in the beginning. There was a great lack of values on intervention levels for various situations, and many decisions had to be taken in this respect. The intervention levels from 1960's were more or less forgotten or not found to be applicable.

However, the greatest surprise was experienced in the area of information. The emergency plans contained organization and personnel for information but it appeared to be much too small. In Sweden for instance, during the first days more than 1000 telephone calls were received per day and it was necessary to ask all available people from the whole of Sweden, that were reasonably competent in radiation protection and radiobiology in particular, to join the information center at SSI and answer questions. A number of publications such as brochures etc had to be produced and distributed. This was the situation also in other Nordic countries.

All these and other insufficiences have later on been remedied.

## The deposition in the Nordic countries

The Chernobyl accident releases started the night of April 25-26 and continued for ten days. Airborne radioactive material reached the Nordic countries on April 27 and the cloud was transported northward during the following days. On May 2 the releases from the Chernobyl reactor increased again for about four days and also part of this release affected the Nordic countries. The releases and their dispersions over Europe is discussed in detail in the UNSCEAR 1988 report. The dispersions during the first week is shown in Fig 2 (STUK, 1991). Fig 3 shows the ground deposition in $kBqm^{-2}$ of Cs-137 in Denmark, Finland, Norway and Sweden as the result of all releases from the Chernobyl accident. The ground deposition in Iceland and Greenland was very low (H Dahlgaard et al 1994).

The deposition pattern in the Nordic countries is the result of the variation of amount and composition of releases from the Chernobyl accident, the wind direction, the fractionation effects during the atmospheric transport and the weather conditions, particularly the precipitation in affected areas (K Edvarson, 1991, H Arvela et al, 1990, M Suomela et al, 1991).

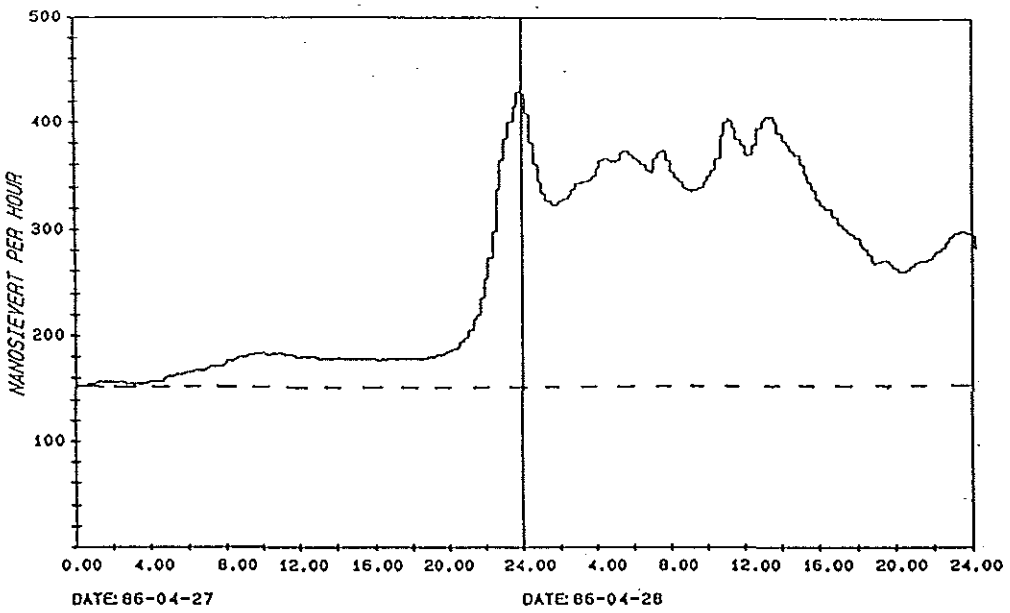


Fig 1 48-hour gamma levels for station: Ölands s:a udde. (P E Kjelle, 1991)

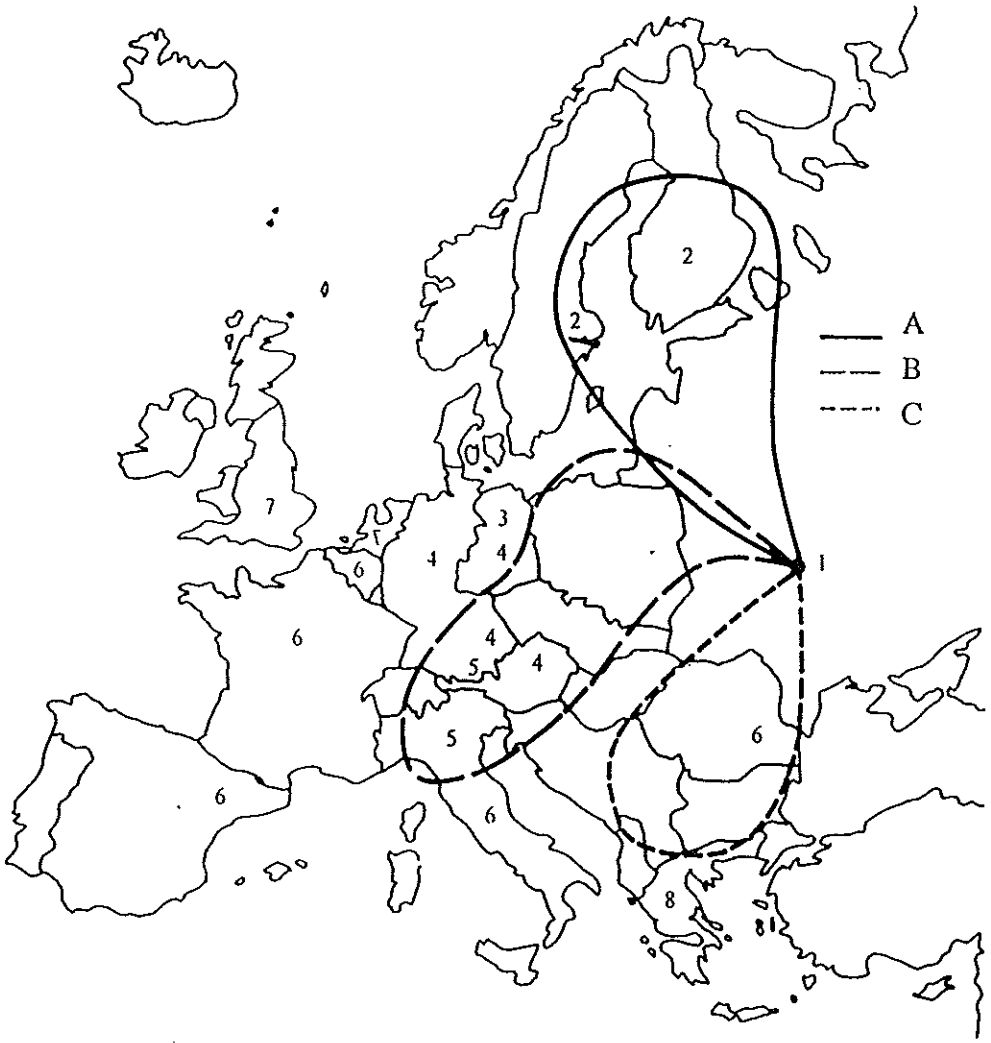


Fig 2 The distribution of air releases from the Chernobyl accident during the first week. Letters A, B and C refer to the distribution in Europe of releases from Chernobyl April 26, April 27-28 and April 29-30. The numbers in the figure are the times of arrival at respective points counted as number of days after April 26. So number 2 means arrival April 27. (STUK 1991)

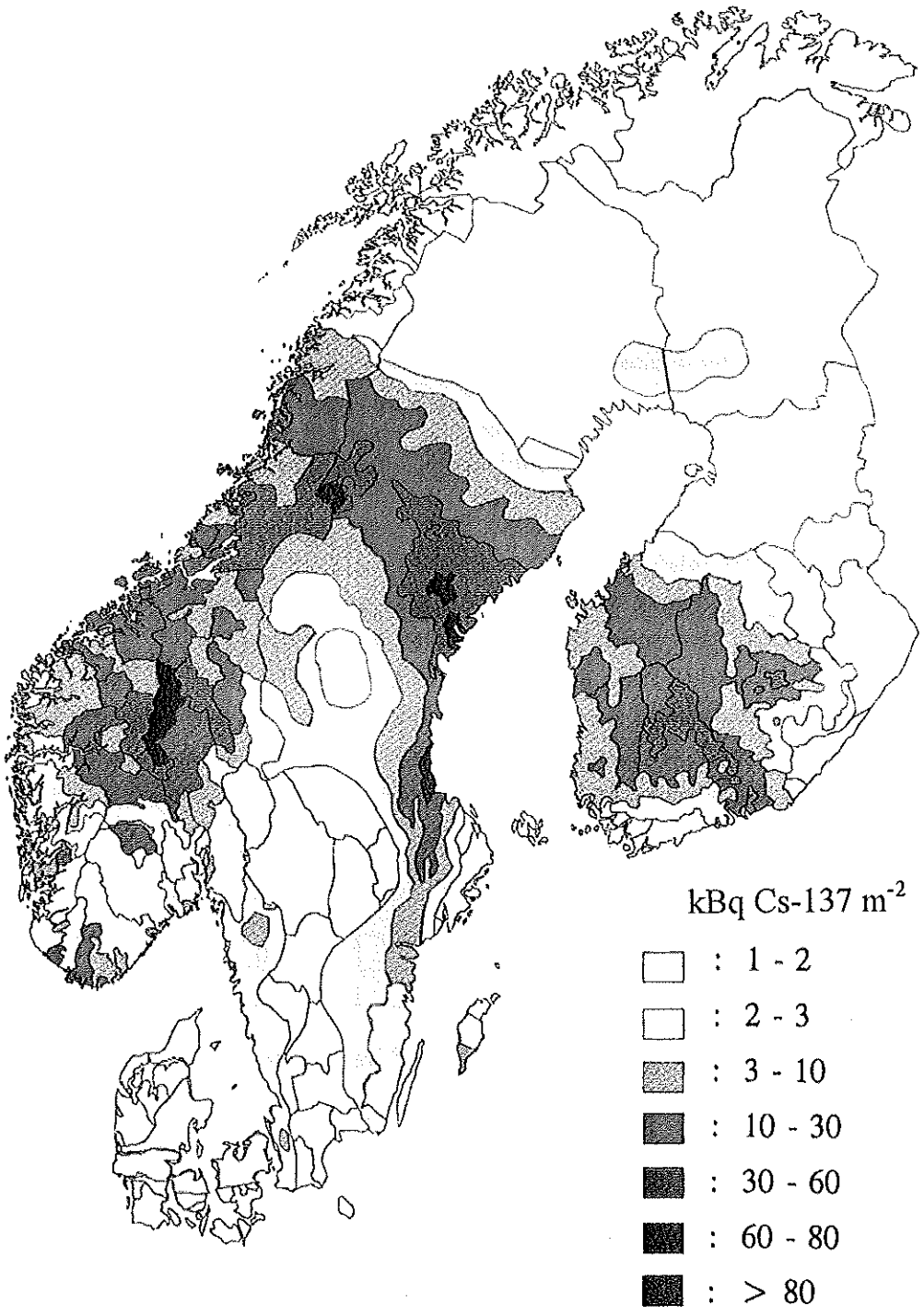


**Fig 3** Ground deposition of Cs-137 , kBq $m^2$, in Denmark, Finland, Norway and Sweden resulting from the Chernobyl accident. (H Dahlgaard et al, 1994)

The total deposition over the Nordic countries was about 10 PBq Cs-137 and the major part was found in wet deposition areas. As about 80 PBq Cs-137 is assumed to have been released (Gudiksen et al, 1989), (or about 60 PBq Cs-134 and about 90 PBq Cs-137 and 2000 PBq I-131 in later estimate corresponding to 20-40% of the total radiocesium and 50-60% of the radioiodine in the reactor core (information från IAEA Chernobyl conference 1996)), more than 10% was deposited over the Nordic countries with the average Cs-137 concentration of 1.2 $kBqm^{-2}$ in Denmark (even distribution), 11 in Finland (surface weighted), 5.1 in Norway (population weighted) and 10 in Sweden (surface weighted). As seen from Fig 3 the distribution of the ground deposition of Cs-137 was very uneven mainly depending on the direction of the movement of the plume and the wash-out deposition by rainfall. There was also great geographical variation of the nuclide ratios because of fractionation effects caused by i.a varying meteorological conditions during the transport and because different source of release (the first and second major releases from Chernobyl) had different nuclide relative compositions and qualities (L Devell et al, 1986). This is illustrated in Table 1 (from L Devell, 1991).

**Table 1.** Activity ratios relative to Ce-141.

| Nuclide | In core | Hot particle | Air samples April 28 | May 8 |
|---|---|---|---|---|
| Ce-141 | 1 | 1 | 1 | 1 |
| Zr-95 | 0.9 | 0.9 | 1 | 1 |
| Mo-99 | 1.3 | 1.0 | 5 | 90 |
| Ru-103 | 0.9 | 0.9 | 1.3 | 13 |
| I-131 | 0.56 | 0.6 | 200a) | 625b) |
| Te-132 | 0.6 | 0.9 | 7 | 580 |
| Cs-137 | 0.05 | 0.06 | 7 | 18 |
| Ba-140 | 0.9 | 0.9 | 2.6 | 20 |
| Np-239 | 6.4 | 18 | 12 | - |

Figures corrected for decay to the time of the accident.
a) Total iodine. Particulate 40.
b) Total iodine. Particulate 250.

More than 20 gammaradionuclides were identified in the air during the initial period and maximum concentrations at ground level of any radionuclide was about 10 $Bqm^{-3}$. Volatile fission products like I-131, I-133, Cs-134 and Cs-137 were dominant but also less volatile radionuclides were present like Zr-95, Ce-141, Ce-144 and Np-239.

The deposition from the Chernobyl accident added to the present fallout from the atmospheric tests in early 60's. The two depositions were of the same order of magnitude e.g in Sweden 4.25 PBq of Cs-137 from Chernobyl and 1.25 PBq from the atmospheric tests (De Geer et al 1978). However the fallout during the 60's was rather evenly distributed with an average of about 3 $kBqm^{-2}$ while the Chernobyl deposition was extremely variable from almost zero up to about 200 $kBqm^{-2}$. The Chernobyl Cs-137 deposition was simultaneously followed by Cs-134 deposition, half-life two years, and the ratio Cs-137/Cs-134 was 1.7. Because all Cs-134 from the earlier fallout has decayed (initially also very little as compared with Cs-137), the spatial distribution of Chernobyl fallout was estimated from measurements on Cs-134.

The measurements (in Sweden) on deposition were made by airborne or *in situ* gamma-measurements. The results were assessed as average surface equivalent deposition density. The actual deposition determined by gammaspectrometric measurements on soil samples was in average 1.6 times higher than that based on in situ gamma-measurements (first year) (K Edvarson, 1991). This correction factor for penetration in soil increased continously to 2.4 in 1989 (L Moberg et al 1996). The correction factor for estimating equivalent dose rates from aerial measurements increased from 1.15 to 1.25 only.

The deposition of I-131 was also of interest from the radiation protection point of view. The integrated air concentration of I-131 was even larger (about 25 times (UNSCEAR 1988)) than that of radiocesium. However because of the relatively short half-life of I-131 (8 days) there was only a problem the first weeks as regards contamination of grass-cow-milk. The deposition was 10-150 $kBqm^{-2}$ and care had to be taken in allowing the cows to go outdoors. The deposition of Sr -90 and its dose consequences for people were minor as compared with Cs 134+137, about two order of magnitude less. The Sr/Cs ratio of the deposition also had great geographical variations. In Finland this ratio varied from 0.002 to 0.12 at different deposition sampling stations (H Aaltonen et al, 1990).

Also the Pu/Cs ratio of the deposition had great geographical variations. The first emission starting 26 April had a higher Pu/Cs ratio than that of the second major emission started 2 May. The deposition in southern Sweden and Denmark was mainly caused by the second emission and had a ratio of only about $10^{-6}$ while the first emission affecting middle and north of Sweden and Norway and middle and south of Finland had a ratio of about $10^{-4}$. During the fallout in the 60's this ratio was about $10^{-2}$. The deposition of Pu 239+240 was in average less than 1% of earlier deposition from nuclear test fallout (40 $Bqm^{-2}$) but at a few places reached up to the same order of magnitude (E Holm, 1991).

There were also some observations of so called hot particles in the air over the Nordic countries. They were quite rare and were not considered to imply any significant radiological risk even though the activity of some of them was of the order of 10 kBq mainly of the radionuclides Ru-103 and Ru-106.

## Dispersion in the environment. Radiological aspects.

Some time after the accident the major interest was directed to Cs-137+134 because of their dominating amount and half life. The activity of the Cs-isotopes in various parts of the environment has continously changed by natural as well as man-made causes. The man-made causes are either specific countermeasures or normal uses of the environment. The natural causes are those which normally correspond to the radioecological behavior of radionuclides in the environment including the human being. The two phenomena are often closely connected and sometimes completely integrated.

### *Air*

The concentration of radionuclides in Nordic air after the Chernobyl accident decreased irregularly after the accident depending on time, current emissions, wind and other weather conditions. Fig 4 shows the air concentration of iodine-131 in the middle of Sweden (L Devell 1991), Fig 5 the air concentration of cesium-137 in Nurmijärvi (in the southern half of Finland) (L Blomqvist et al, 1987) and Fig 6 the air concentration of Pu 239+240 at some various places in Denmark, Finland,

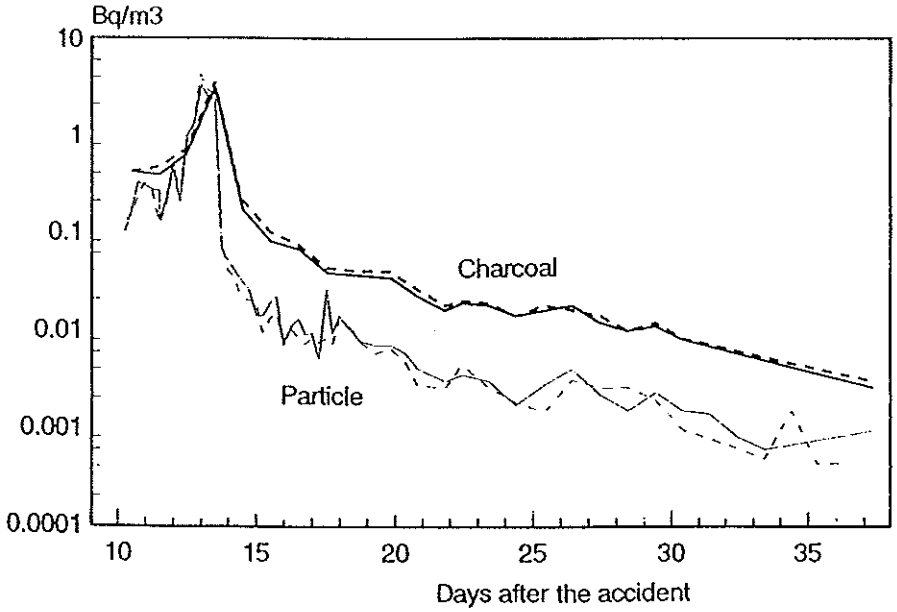


Fig 4 Iodine-131 in air sampled by charcoal or particle filters. (L Devell, 1991)

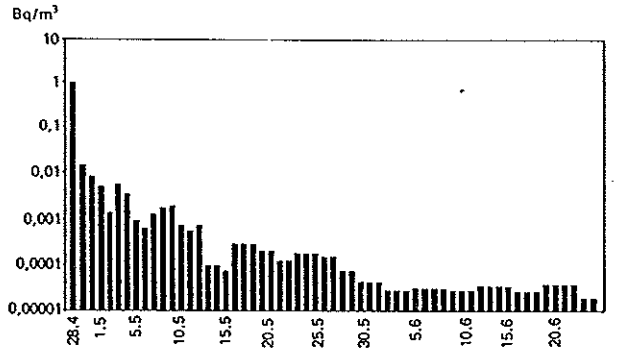


Fig 5 Cesium 137 concentration in Finland. (L Blomqvist et al, 1987)

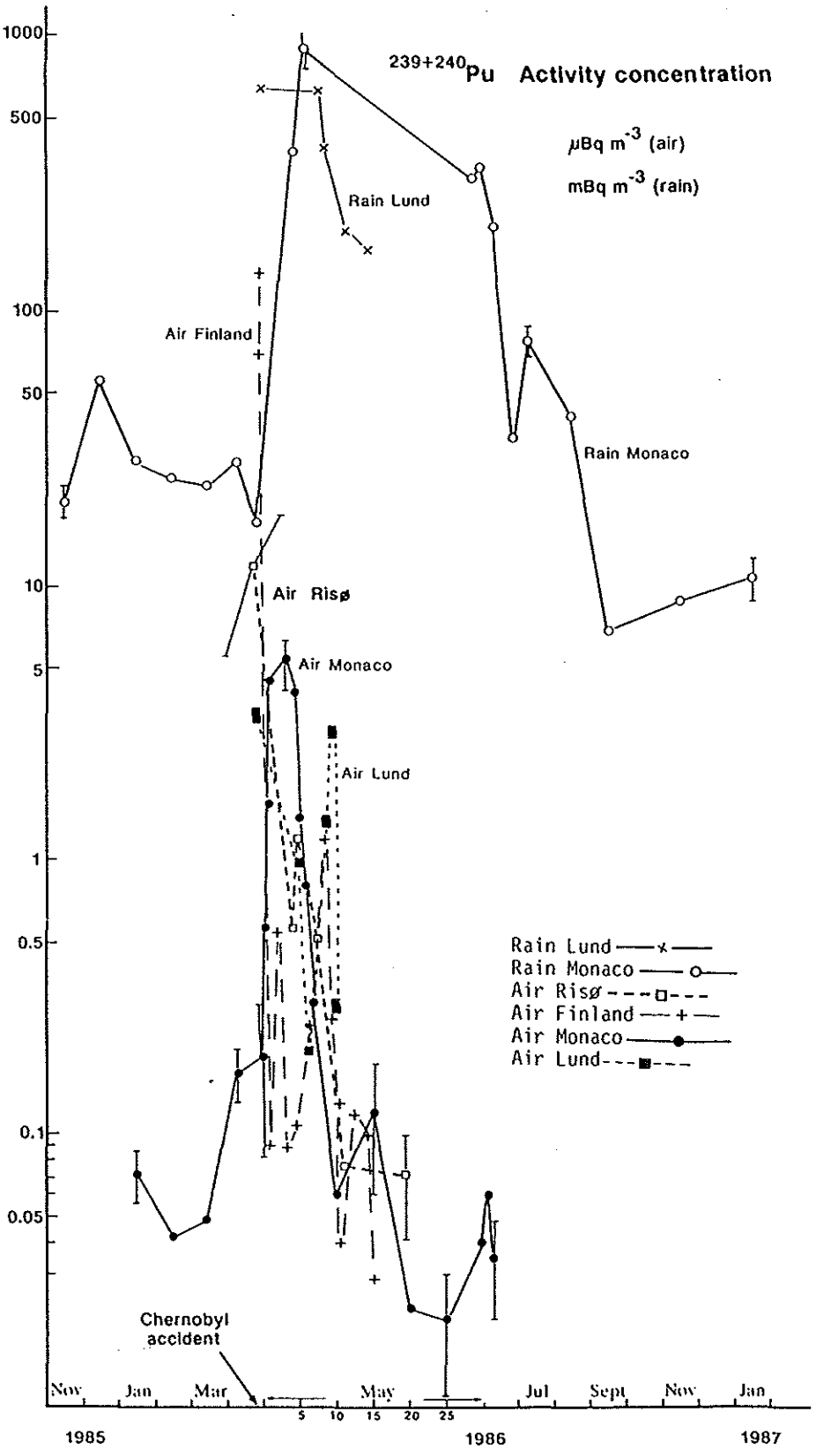


Fig 6 The Pu-239, 240 activity concentration in air and rain at Risø (Denmark), southern Sweden (Lund), Finland (Nurmijärvi) and southern Europe (Monaco) following the Chernobyl accident. (E Holm, 1991)

Sweden and southern Europe (Monaco) (E Holm, 1991). The air concentration several weeks after the accident and thereafter is influenced by resuspension, which is higher in areas with dry deposition than in those with wet and varies by season with a peak in spring and a minimum in fall and winter (J Vintersved et al, 1991). The concentrations are very low, though.

*Ground and transport to water*

The radionuclide of major interest today is Cs-137 even though there is some radioecological interest of other radionuclides like Sr-90. The Cs-activity on ground has decreased by physical decay and by wash-out/run-off processes. The actual amount left on ground today can be approximately assessed from studies of gammaradiation from the ground and by studies of out-flow in rivers into the Norwegian Sea, the Atlantic, the North Sea and the Baltic Sea including the core areas the Bothnian Sea and the Gulf of Finland.

Measurements of water from Swedish and Finnish rivers to the Gulf of Bothnia in 1986, 1987 and 1989 gave an input of 181, 81 and 24 TBq of Cs-137 respectively, see Fig 7 (S Evans, 1991). Finnish river data are found in references (R Saxén and H Aaltonen, 1987, R Saxén, 1990 and R Saxén and U Koskelainen, 1992). This can be compared with the fallout over Gulf of Bothnia, 1.9 PBq. The total inventary of Cs-137 from the Chernobyl accident in the water column of the Baltic Sea was 5.2 PBq (H Dahlgaard et al, 1988). As concluded, the out-flow to the Baltic Sea waters decreased rapidly and the total out-flow from land during the first three years corresponds only to about 2% of the total Chernobyl fallout over Finland and Sweden as compared to 7% diminution caused by physical decay.

Some part of the ground deposition that has actually migrated from soil into water has been trapped in the sink of lakes, water reservoirs and rivers mainly in their bottom sediment layers. The transport from soil to the water ecosystem to lakes etc decreased quite fast the first years with an approximate half-life of 1.5 - 3 years (B Sundblad et al, 1991). This can be compared with the estimated half-life of 0.5 - 1 year the first three years after the accident of the amount transported by rivers to the sea. The rapid decrease in amount of transported materials may be a result of the effect of the sink of lakes. On the other hand studies made on lakes in Norway, Sweden and Finland during several years, show that the output from a lake can be higher than input from the catchment area depending on leakage from the sediments of the lake. During 5 years of observation the loss in one of the lakes was 30% of the initial deposition while the corresponding loss in the catchment area was only 1-8% (depending on the size of the area). The general conclusion is, however, that much of the Cs-137 deposited on ground still remains in the soils and the stream sediments in catchment areas of lakes (H E Bjørnstad et al, 1994). This is also true for the lake sediments. There are also evidence of very low downward migration of Cs-137 in soil since 1987 and more than 90% is normally found in the upper 5-10 cm of soil (B Sundblad et al, 1991) with the major part in the upper 5 cm (D H Oughton and B Salby, 1994).Therefore, the major part of the reduction of activity on ground should be caused by physical decay. This is supported by airborne gammameasurements. However, it cannot be unambiguously proved by the observations of gammaradiation from ground made since 1986 (P E Kjelle, 1996).

*The Baltic Sea system*

The change of the Cs activity of the Baltic Sea system by transport with water currents has been studied by indirect methods using the algae Fucus vesiculosus as a bioindicator (L Carlsson and P Snoeijs, 1994). The results are illustrated by Fig 8 (from that publication). That the decrease of

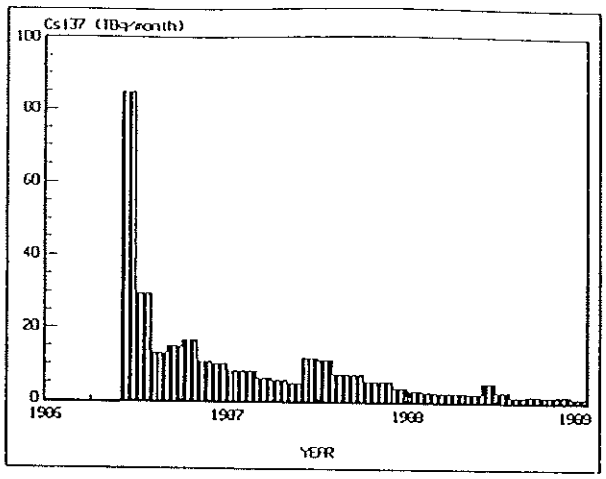


Fig 7 Riverine inflow of Cs-137 to the Gulf of Bothnia 1986-1988. (S Evans, 1991)

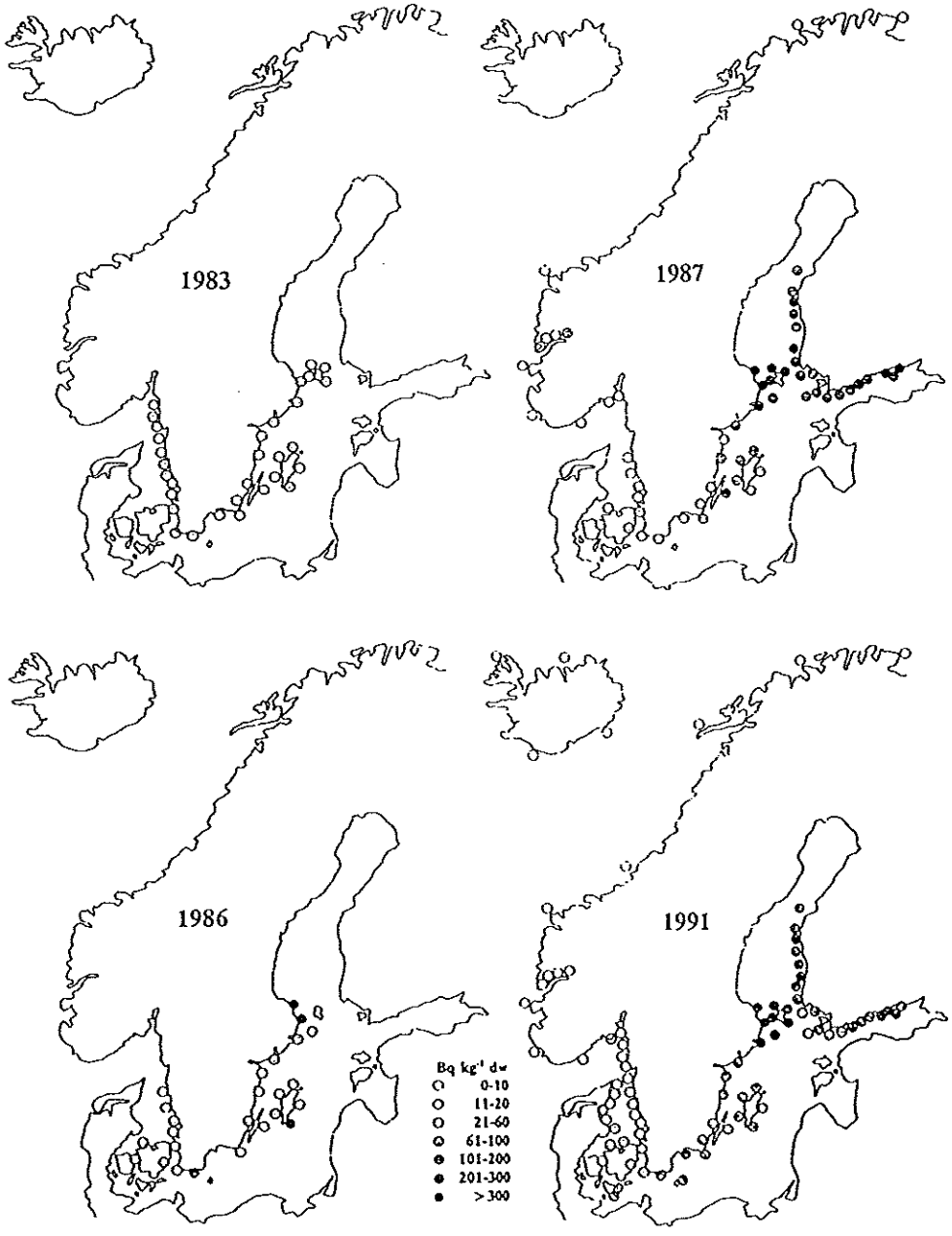


Fig 8 Activity concentrations of $^{137}Cs$ (Bq $kg^{-1}$ dw) in Fucus vesiculosus in Nordic coastal waters in 1983, 1987 and 1991. (L Carlsson and P Snoeijs, 1994)

activity along the Gulf of Finland since 1987 is larger than in the Gulf of Bothnia is assumed to depend on larger water turnover in the Gulf of Finland than in the Gulf of Bothnia and on larger addition of Cs-137 to the latter from heavily contaminated drainage areas of the rivers in Finland entering the Gulf of Bothnia. The increased activity along the southern Swedish coast indicates an addition of Cs-137 caused by Cs-137 in the out-flow from the Baltic Sea. As can be seen the activity along the Norwegian coast is low, mostly below 5 $Bqkg^{-1}$ dry weight. Outside Iceland the activity is very low below 0.5 $Bqkg^{-1}$ dry weight mainly from other sources than the Chernobyl accident.

*Lakes*

The turnover of radiocesium in the ecosystem of lakes is very complicated and variable (A Broberg, 1994). The role of sedimentation is important for the bioavailability of radiocesium in the lake ecosystem. Generally speaking, most of the radiocesium entering the lakes was rapidly and strongly bound to particles and transferred to bottom by sedimentation.

However, many factors influenced the rate and way of sedimentation 1986 and years thereafter. In-flow of materials to a lake and production of organic materials increased the sedimentation, lakes with large areas of shallow waters had an enhanced sedimentation process, redistribution and transportation of radiocesium occurred to deeper parts of a lake etc. The deposited radiocesium on the bottom of a lake was covered by sediments of lower activity concentrations or mixed to a more homogeneous distribution, all factors influencing the bioavailability. The exchange of radiocesium between sediments and water varied also in different parts of a lake depending on varying local water flows. The total amount of Cs-137 in most lakes has not changed significantly more than physical decay since 1986, and the sediments contain from 80 to almost 100% of the Cs-137 in lakes (M Meili, 1994).

*Transfer to fish*

The contamination of fish was predominantly caused by uptake of radiocesium in contaminated food. The intake from water was negligable (T Andersson and M Meili, 1994). The turnover rate in a given fish depends on fish size and temperature of the water. Both intake and excretion is higher in the summer than in the winter and the turnover in a fish may be higher in smaller fish than in larger because of faster metabolism.

The maximum transfer to fish occurred normally in the increasing order small perch - trout and char - larger perch - pike. The times to reach the peak values of Cs-137 vary. In some studies, times about 1 year for perch and 1-2 years for pike were reported (B Sundblad, 1991, R Saxén and U Koskelainen, 1992). Other studies and modelling work indicate that peak values occur later, for pike 5-10 years after the accident. The ecological half-life for cesium in pike has been assessed to 10-15 years and about 2 years for perch (L Håkansson, 1991). However, the ecological half-life of Cs-137 in fish is not constant but increases with time (R Saxén, 1994). Seasonal variations of the content of Cs-137 in fish have been found with maximum in wintertime because of low metabolism rate at lower temperatures. There are also great variations in relative uptake in fish between lakes (more than one order of magnitude) that are not easily understood. The results reflect the complex relations between a number of influencing factors on the balance between input from drainage areas, sedimentation, resuspension, out-flow and the chemical and physical qualities of the lake water and its content of various materials.

The fallout from the Chernobyl accident caused heavy contamination of thousands of lakes in the Nordic countries. In Sweden alone about 14000 lakes were contaminated so seriously that the fish had activity concentrations exceeding the level applied for fish not allowed to be sold at the open market (1500 Bq/kg of Cs-137). There are still several thousands lakes of that kind and considering the large variations between lakes and fish species as reported above, the problems with lakes and fish are predicted to exist even into the 21st century (L Håkansson, 1991).

*Transfer to agricultural products*

The contamination of terrestrial vegetation in the first period after the fallout caused by the Chernobyl accident was primarly by direct contamination. This was true particularly in the southern part of the Nordic countries. In the northernmost part there was still some snow cover and the vegetation had not started. Gradually the contamination was caused by root uptake of activity deposited on ground.

There are great differences in the agricultural ecosystems in the Nordic countries. In the north there are grassland and mountains and animal husbandry is dominating while in the southern part, land is used for cultivation of grain, crops, sugar beet, legumes, potatoes together with animal husbandry. Because there was also a largely varying degree of land contamination the agriculture contamination became complex and unpredictable.

The transfer factor for agricultural products in terms of $Bqkg^{-1}$ per $Bqm^{-2}$ ($m^2kg^{-1}$) depends on the qualities of soil, kind of product, soil preparation, the way of harvest, season etc. The change of transfer factors for grass and barley grown in different soils are shown in Table 2 (from data by K Rosen, 1991).

**Table 2.** Transfer factor ($m^2kg^{-1}$) of Cs-137 from soil to vegetation.

| Product | Soil | 1987 | 1988 | 1989 | 1990 |
|---|---|---|---|---|---|
| Hay grass | Peat soil | 0.028 - 0.034 | 0.010 - 0.042 | 0.005 - 0.015 | |
| Hay grass | Organic soil | 0.033 - 0.077 | 0.007 - 0.016 | 0.013 | |
| Hay grass | Sandy soil | 0.0045-0.0085 | 0.0021-0.0031 | 0.0025 | |
| Pasture grass | Loamy soil | 0.023 - 0.050 | 0.022 - 0.032 | 0.008 - 0.017 | |
| Pasture grass | Sandy soil | 0.10 - 0.12 | 0.08 - 0.10 | 0.08 - 0.09 | |
| Barley grain | Peat soil | 0.005 | 0.0018 | 0.0008 | 0.0005 |
| Barley straw | Peat soil | 0.012 | 0.0035 | 0.0012 | 0.001 |
| Barley grain | Sandy soil | 0.00035 | 0.0007 | 0.00025 | |
| Barley straw | Sandy soil | 0.001 | 0.001 | 0.0004 | |

As seen from the Table 2 the transfer is decreasing these years and variations occur depending on product and soil. The high value for uptake to pasture grass in sandy soil depends on the low clay and potassium content of the soil. As a rule of thumb an assumed transfer factor of 0.1 $m^2kg^{-1}$ the first year and 0.01 $m^2kg^{-1}$ following years does not seem to underestimate the uptake in grass in most cases. In barley it is lower. Table 3 is a summary of transfer factors (aggregated transfer factors) for various crops in the Nordic countries (except Iceland) for the years 1986 - 1990. The first value is for 1986 and the value within parentheses is the average value for 1987 - 1990 (from data by M Strandberg, 1994).

Table 3. Transfer factor ($m^2kg^{-1}$) for 1986 and (within parentheses) average of values for 1987-1990 in the Nordic countries in various crops. All crops are in fresh weight (except for barley in Sweden). All values should be multiplied with $10^{-3}$.

| Country/Crop | Barley | Potatoe | Cabbage | Carrot | Pea |
|---|---|---|---|---|---|
| Denmark | 1.3 (0.60) | 0.17 (0.092) | 0.18 (0.050) | 0.083 (0.052) | 0.16 (0.012) |
| Finland | 0.26 (0.0063) | 0.19 (0.066) | 0.17 (0.036) | 0.10 (0.027) | 0.15 |
| Norway | 0.81 | 0.56 | | | |
| Sweden | 7.0 (2.1) | 2.0 (1.4) | 1.3 | 0.7 (0.88) | 1 |

The relatively high values for Sweden seem to depend on the specific soil (peat soil and silt loam soil) but the conclusion is that the uptake of fallout activity from Chernobyl accident to vegetation from soil decreases rapidly after the first year 1986 as the contamination changes from direct fallout to uptake from soil through roots. The values during 1987 - 1990 lie within ±30% of the average with exception of the Swedish value for barley. In 1987 it is underestimated by a factor of 2 and for 1988-1990 overestimated by the same factor (this is also true for potatoes for 1988-1990). The estimated effective half-lives vary between the countries and the crops from a few years to about 10 years. A general conclusion is that the uptake through roots is low in the Nordic agriculture ecosystem. Most of the transfer factors for edible parts of crops lie between 0.001 and 0.01 $m^2kg^{-1}$ during 1986 and between 0.0001 and 0.001 $m^2kg^{-1}$ 1987-1990.

The transfer factor for Sr-90 in soil and vegetation was studied in Norway the years 1989-1992 (D H Oughton and B Salbu, 1994). No clear tendency of a decrease (or increase) can be seen and the transfer factors varied between 0.02 and 0.3 $m^2kg^{-1}$ (mostly "old" Sr-90).

*Transfer to animals and animal products*

*Milk*

Measurements were made on dairy milk or dry milk and on milk from individual farms in all Nordic countries. The mean Cs-137 activity in dairy milk in 1986 and 1987 varied from 0.6 to 20 $Bql^{-1}$ in the Nordic countries. The change of activity in milk during 1986 to 1992 is shown in Fig 9 (from H S Hensen and J Andersson, 1994). The aggregated transfer factor ($Bql^{-1}$ milk per $Bqm^{-2}$ ground deposition) decreased for all countries as seen in Fig 10. The transfer of Cs-137 from vegetation to cows' milk did not change significantly during 1986-1992 and ranged from 0.005 to 0.03 $dl^{-1}$ ($Bql^{-1}$ milk per Bq ingested daily by the cow) with a general mean of about 0.01 $dl^{-1}$. The effective ecological half-life for Cs-137 in milk was 1-2 years for all Nordic countries. The maximum activity concentration in milk was reached 8-9 days after the cows had been taken out for grazing.

Studies on I-131 in milk indicate about the same transfer factor (grass to milk) as for Cs-137 (E Wallström et al, 1991). The effective ecological half-life was 4-5 days and the maximum activity concentration was reached 3-5 days after the cows had been taken out for grazing.

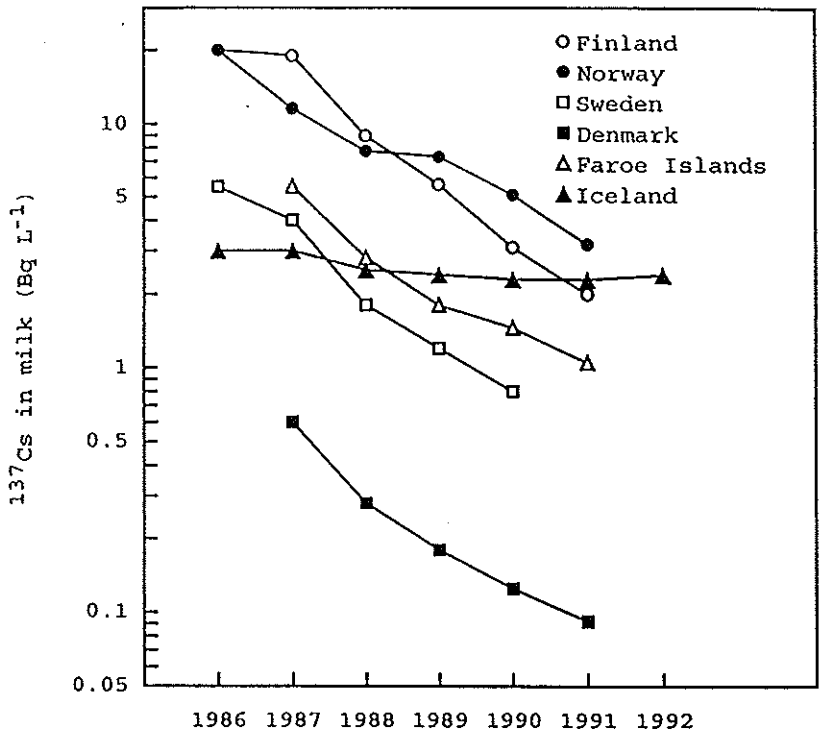


Fig 9 Mean Cs-137 activity concentration in cows' milk from dairies or dry-milk factories in the Nordic countries from 1986 to 1992. Values refer to Chernobyl Cs-137, except in Iceland where global Cs-137 only was included.
(H S Hensen and J Andersson, 1994)

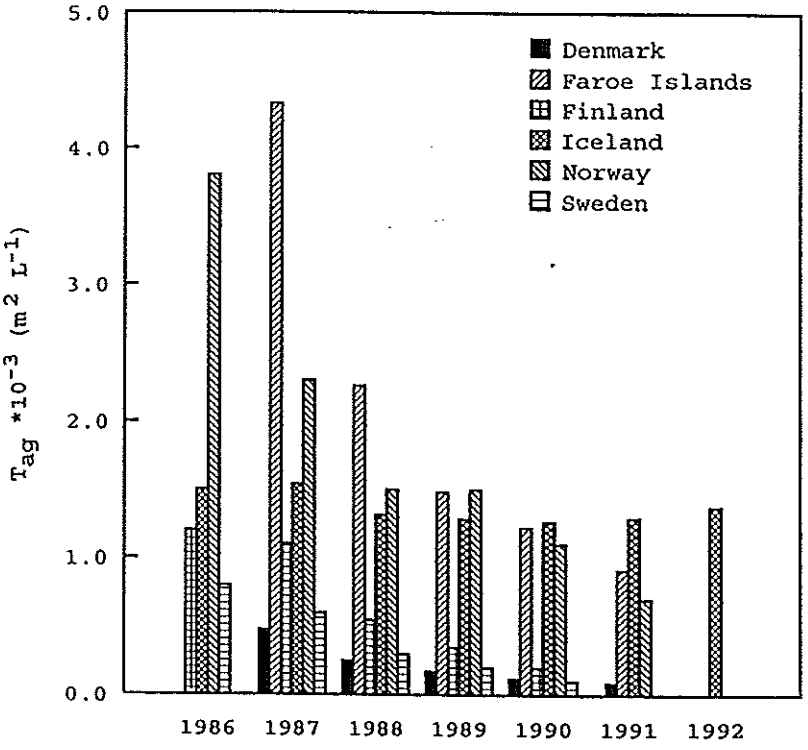


Fig 10 Mean aggregated transfer coefficients ($T_{ag}$) of Chernobyl Cs-137 to cows' milk in the Nordic countries (global Cs-137 in Iceland) from 1986 to 1992. Estimations based om mean ground deposition, where needed corrected according to the areal distribution of milk production and mean Cs-137 content in dairy milk.
(H S Hensen and J Andersson, 1994)

*Sheep*

The lamb production in the Nordic countries is most important in Norway and Iceland and least important in Denmark and Finland. The annual per caput consumption of mutton in Denmark, Finland and Sweden is 0.6 kg and in Norway it is 9 times more and in Iceland 40 times more. The production occurs on uncultivated pastures and on natural or semi-natural environments. The downward migration is normally very low which leads to a longlasting plant Cs-137 availability. However the absolute availability can vary depending on soil characteristics. That is reflected in the transfer factor soil-grass in areas where lambs are grazing. The average aggregated transfer factors ($Bqkg^{-1}$ meat fresh weight per $kBqm^{-2}$ soil) for the years 1990-1993 are for Denmark 0.6 ±100%, Finland 0.8 ±70%, Iceland 15 ±5%, Norway 39 ±10% and Sweden 47 ±25% (one place only) (K Hove et al, 1994).

These values have been relatively constant over the years 1990-1993 except for Denmark and Finland where there has been a decrease, a factor 2-5. Herbage to lamb transfer factors ($Bqkg^{-1}$ meat per $Bqkg^{-1}$ grass) have been fairly equal between the countries with values between 0.25-0.70. The ecological half-life for Cs-137 in sheep has been estimated to be about 13 years in undisturbed areas (P Strand, 1994).

*Forest and alpine ecosystems*

Forest and parts of alpine areas are important as pasture for grazing animals, for hunting, for berries, mushrooms etc. and of course for wood production. Some of the northern parts of the Nordic countries was covered by snow at the time of fallout from the Chernobyl accident and during melting substantial translocations could occur from higher level areas to lower level areas. That led sometimes locally to a more inhomogenous ground contamination than given by the deposition distribution (R A Olsen, 1994).

The contamination occured as in other areas by direct contamination and by root uptake and after one year the dominating part in fresh plants was by root uptake. After 2-4 years after the fallout in 1986 more than 90% of the Cs-137 activity was more or less stabilized in the upper 4cm and only about 15% was bioavailable. The bioavailability was only little affected by time. The aggregated transfer factor decreased rapidly the first 3 years to a more stable condition and varied between 0.005 and 0.1 $m^2kg^{-1}$ dry weight. In more acid forests the transfer factor could be 10 times higher, 0.08-0.3 $m^2kg^{-1}$ and in fungus fruit bodies further 10 times higher, 3-11 $m^2kg^{-1}$ . The last transfer factor is correlated with the ability to accumalete stable cesium. In edible plants the transfer factors are found to be 0.04-0.09 $m^2kg^{-1}$ for bilberry, 0.03-0.12 $m^2kg^{-1}$ for lingonberry and 0.13 $m^2kg^{-1}$ for cloudberry (K J Johansson, 1994 and K J Johansson et al,1991). As the ground contamination in many areas was 50 $kBqm^{-2}$ and more it is easily understood that forest berries and other edible plants can be heavily contaminated. This is particularly a problem with mushrooms (transfer factor around 1-10 $m^2kg^{-1}$ ).

It is too early to determine the ecological half-life of Cs-137 in forest plants but it may be 5-10 years or more. Example of change of activity concentration with time in some forest plants are given in Fig 11 (from R Bergman et al, 1991).

Contaminated plants also mean contamination of the animals in the forest. The most important animals as game meat in the Nordic countries are the moose, roe deer and hare. Birds like mallard, pheasant and grouse are also important as game meat in Denmark and Norway. The birds have usually quite a low transfer factor 0.01 $m^2kg^{-1}$ and lower (K J Johansson, 1994) and for hare it

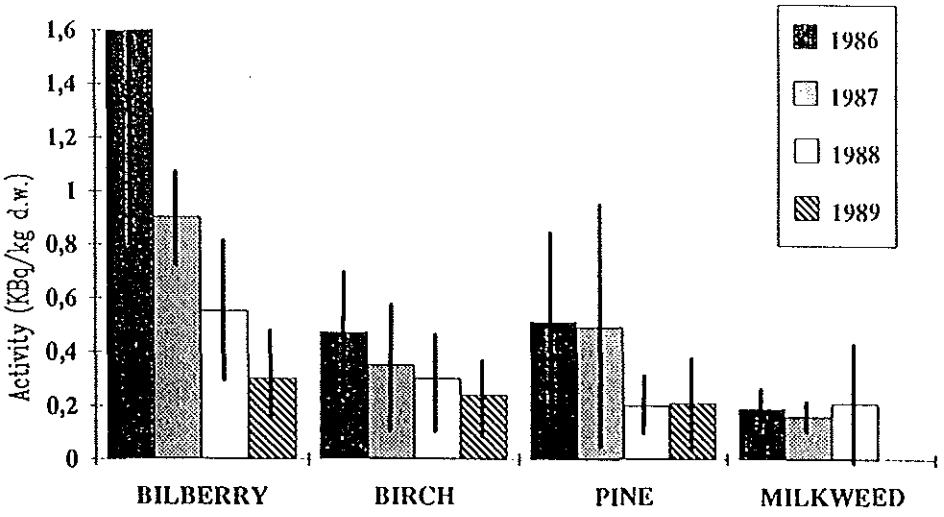


Fig 11 The concentration of caesium 137 during 1986-1989 in "key"-plants based on pooled data from samples in July (bilberry twigs, birch twigs and milkweed) and October (bilberry twigs, birch twigs and pine). (R Bergman et al, 1991)

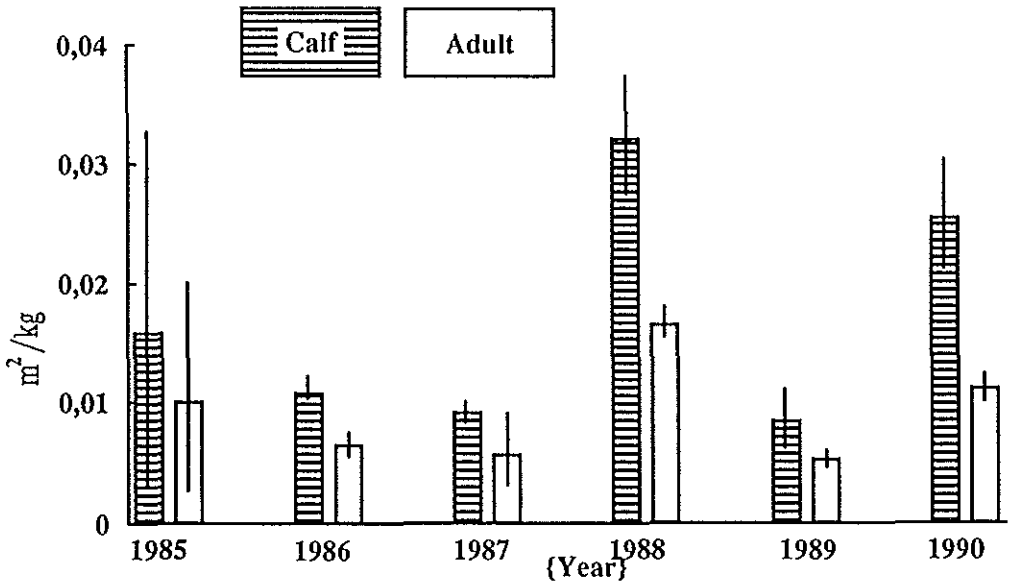


Fig 12 The quotient ($m^2$/kg) of Cs-137 concentration in moose muscle to ground deposition in Sweden during the hunting season in 1985-1990. (R Bergman et al, 1991)

might be around 0.03 $m^2kg^{-1}$. For moose transfer factors between 0.01 and 0.03 $m^2kg^{-1}$ have been found and 0.03 to 0.14 $m^2kg^{-1}$ for roe deer.

Considering the amount of meat consumed, moose and deer are most significant from the radiation dose point of view. As there are several hundred thousands of hunters in the Nordic countries (only in Sweden there are about 300.000) there is potentially a large critical group and therefore the activity of Cs-137 in moose and deer has been given high attention.

There are changes of activity concentration in these animals depending on the access to various plants with variable activity concentration. An important contributor is mushroom which normally has a high transfer factor. Fig 12 shows the variation of transfer factor for moose during 1985-1990 which reflects the variability of available Cs-137 in the forest plants. Still 10 years after the Chernobyl accident moose and deer constitute a problem in some parts of the Nordic countries. It is suggested until more is known that the effective ecological half-life would be the same as the physical half-life of Cs-137.

A plant of special interest in the alpine areas is the lichen. The ability of lichen to retain fallout activity is well known from studies after the fallout of the nuclear weapon tests in the 50's and 60's. Its large retaining ability is due to the large surface area per unit mass, slow growth and long lifetime. The ecological half-life of Cs-137 is around 10 years. Most of the activity is in the upper 3 cm of the lichen carpet (P Roos, C Samuelsson, S Mattsson, 1991).

Reindeer accumulate fallout nuclides in their bodies by eating lichen, particularly in the winter (1-4 kg per day). Because reindeer meat is an important part of the diet of the Laplanders it has a special radiological significance. In Sweden some 70.000 reindeer were located in the most contaminated areas up to 80 $kBqm^{-2}$ and more locally (O Eriksson et al, 1991). The contamination of reindeer was a great problem particularly in Norway and Sweden, and activity levels up to 150kBq/kg were found (P Strand, 1994). In Sweden 1988 more than a third of the slaughtered animals had activity concentration more than 1000 $Bqkg^{-1}$ (SSI-88).

The cesium intake shows large seasonal variations depending on the feeding habits of reindeer. The lichen constitute the major diet (70-80%). The seasonal variation of the Cs-137 intake by the diet of the reindeer has been predicted as is shown in Fig 13 (from E Gaare and H Staaland, 1994). That means that there are similar seasonal variations of the ceasium activity in meat with a maximum in the early spring and a minimum in the autumn, see Fig 14 with measurements up to 1992. The biological half-life of cesium in reindeer is only 1-3 weeks (E Gaare and H Staaland, 1994) and therefore the levels of meat activity follow the variations of activity intake with relatively short delay. With constant intake there is an equilibrium activity concentration in reindeer after 1-2 months. The ecological half-life of Cs-137 in reindeer in the first 5-8 years has been found to be 3-4 years but it tends to increase (B and G Åhman 1994). In comparison, the corresponding half-life for the nuclear weapon test fallout from the 60's is around 10 years.

The aggregated transfer factor is found to be 0.4-0.8 $Bqkg^{-1}$ per $Bqm^{-2}$ during wintertime and somewhat lower in the summer (0.1-0.2). Assuming the contamination of ground was 50 $kBqm^{-2}$ the expected levels would be 20-40 $kBqkg^{-1}$. Cs-137 concentration in meat is similar to or higher than the activity concentration of the lichen the reindeer eat. Activity concentration up to 50 $kBqkg^{-1}$ or more has been found wherefore meat concentrations of the same order of magnitude can be expected.

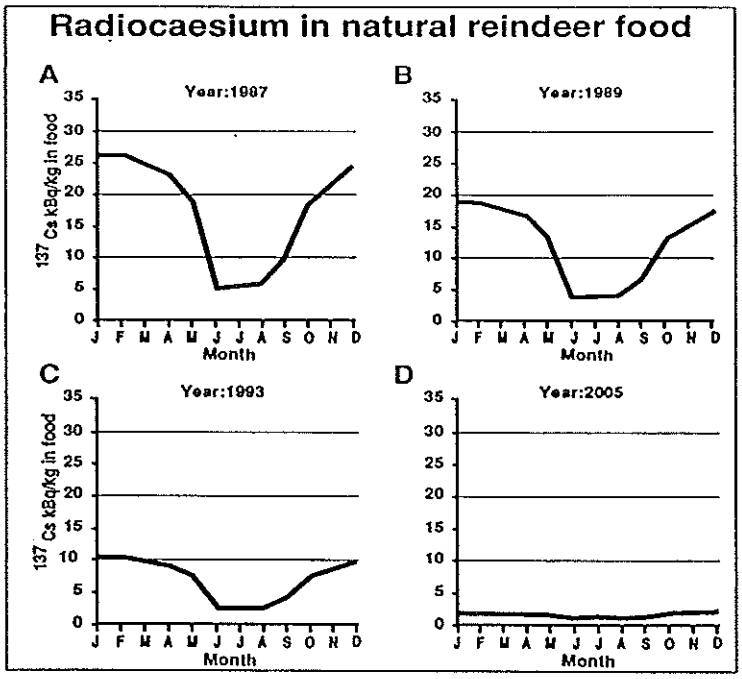


Fig 13 Predicted seasonal variations in Cs-137 concentrations in reindeer food. Further explanation in text. (E Gaare and H Staaland, 1994)

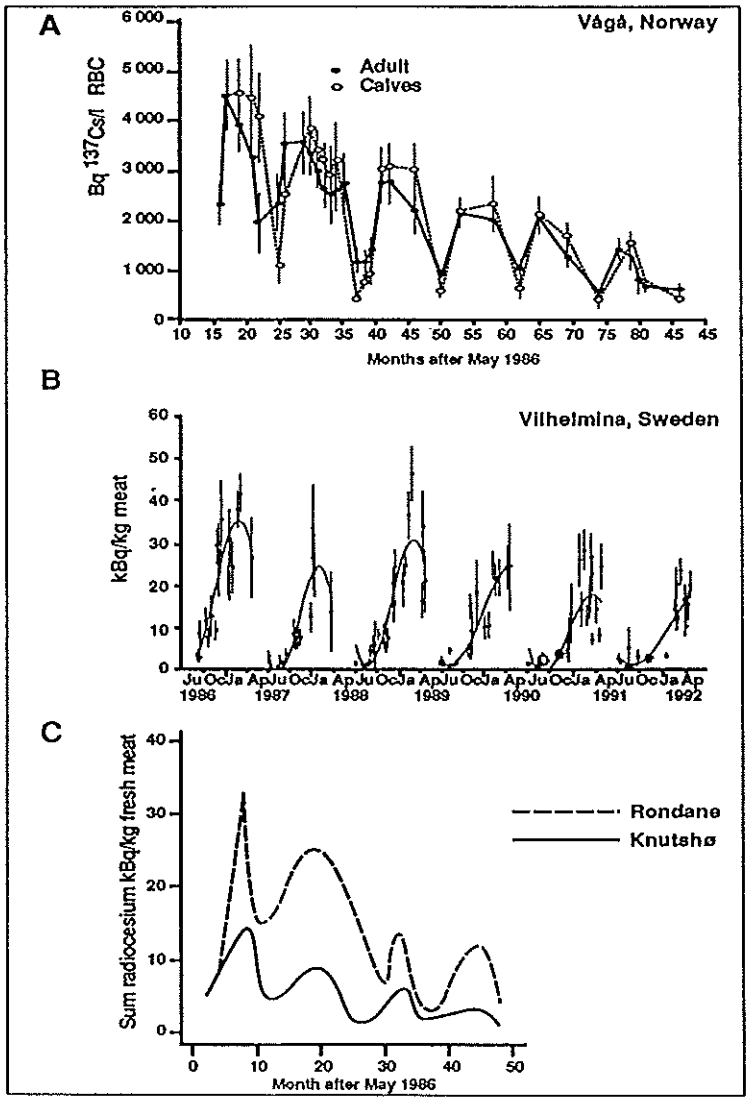


Fig 14 A: Seasonal variation in red blood cell (RBC) Cs-137 activity in domestic reindeer from Vågå, Norway. In meat from reindeer B: slaughtered in Vilhelmina, Sweden and C: wild reindeer, Norway. (E Gaare and H Staaland, 1994)

*Other Contaminations*

Besides those parts of the ecosystem described above there were other parts of that system of minor importance but that nevertheless influenced the exposure of man and the environment. Examples are other animals on ground than those mentioned, birds, fishes in the sea etc. Drinking water was very little affected if taken from wells. During the first days, when the radionuclides were still in the air, large airfilters in industries etc could be so contaminated that some precaution was justified. Contamination of sludge in sewage treatment plants was another problem. This sludge is used as fertilizer in many farms and in May 1986 the activity concentration at some places was up to 100 $kBqkg^{-1}$ (20% dry substance). This was in areas with a ground deposition of 50-70 $kBqm^{-2}$. In spring 1987 the activity levels had decreased factor 100-1000 (SSI-88).

Contamination of agricultural soil could cause problems from resuspension of deposited cesium during ploughing etc. However, measurements during 1986 and 1987 did not indicate any significant problem. The air concentration of Cs-137 was found to be a few $\mu Bqm^{-3}$ (SSI-88).

Another example was the contamination of peat. Peat is used as fertilizer and for cumbustion for production of hot water. In the first case there is a risk of contamination of food and in the second case there may be problems with airborn activity and activity in the ash. Therefore intervention levels were given (see below).

## Resulting external and internal doses

*External doses*

The radiation dose during the first weeks was dominated by the dose caused by inhalation (in Sweden resulting in about 150 manSv, see SSI-88), the external dose from the cloud and the external radiation from radionuclides deposited on the ground. The radionuclide composition of the fallout from the Chernobyl accident varied between different parts of the Nordic countries depending on time of deposition and kind of deposition (dry or wet). In wet deposition the cesium isotopes were relatively more significant in terms of resulting external dose rate than other radionuclides. Generally speaking the first days after 28 April the dominating radionuclides were I-131, Te-132 and Ba-140. Zr-95 and Nb-95 also gave significant contribution to external dose rates the first days in some areas. However, after 1-2 months the dominating radionuclides in most areas were Cs-134 and Cs-137 (in that order from the point of view of dose rates). An example of how various radionuclides contributed to the effective dose rate as a function of time is shown in Fig 15. This is from southern part of the wet deposition area in Sweden (K Edvarson, 1991).

The resulting external dose rate in air decreases in time due to physical decay and penetration of radionuclides into the ground. However after a few years the penetration has stabilized in a layer of 3-5 cm depth. The effective dose rate is influenced by snow cover and shielding in houses. The population weighted shielding factor caused by snow is about 0.9 in the Nordic countries and the one caused by buildings taking account of an assumed 15 percent out-door occupancy is about 0.3 (R Finck, 1991).

The estimated effective dose caused by external radiation during the first year after the accident was in most parts of the Nordic countries below 0.5 mSv. In some areas it was up to about ten times that for a few hundred people. The first year average effective dose and the collective doses for the first year and 50 years are shown in Table 4.

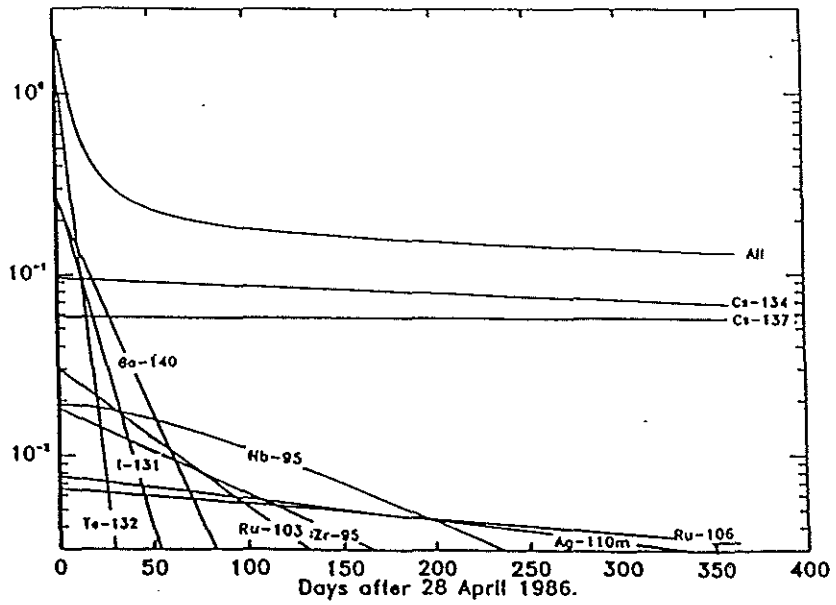


**Fig 15** Effective dose equivalent rate in microsievert per hour (Uppsala county). (K Edvarson, 1991)

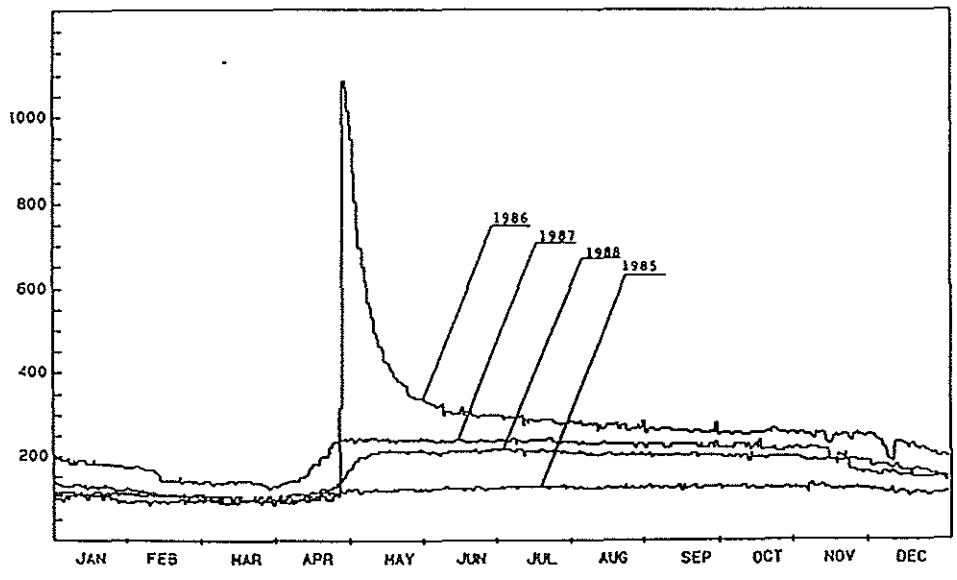


**Fig 16** Annual change of gamma levels (nSv $^{h-1}$) for station: Umeå. (P E Kjelle, 1991)

Table 4. First year effective dose and collective doses in the Nordic countries caused by external radiation.

| Country | First year mean effective dose, mSv | Collective dose 1st year, manSv | Collective dose 50 years, manSv |
|---|---|---|---|
| Denmark | [0.04][1)] | [200][1)] | 700 [6)] |
| Finland | 0.15[2)] | 750[1)] | 5500[7)] |
| Norway | 0.08[3)] | 300[1)] | 5000[1)] |
| Sweden | 0.08[4)] | 600[5)] | 5000[5)] |

1) Approximate assessment from data in references below
2) STUK 1991.
3) T Strand, P Strand and J Baarli, 1987.
4) SSI-88.
5) K Edvarson, 1991.
6) Risø 1986-1992.
7) M Suomela et al 1996

The decrease of external dose rate in air since 1987-88 is at least according to physical decay of Cs-134 and Cs-137. Since the two isotopes contributed quite similarly to the effective dose in the first years, the external dose rate is expected to have decreased about a factor 3 up to 1996. Fig 16 shows the annual change from 1985 to 1988 of the external dose rate in one of the heavily contaminated areas in the northern part of Sweden (P E Kjelle, 1991).

*Internal doses*

The internal doses are mainly caused by ingestion of Cs-134 + Cs-137 with food. The levels of internal contamination depends on diet, composition, place of food production, countermeasures to reduce the activity concentration and time after 1986. The countermeasures and their effects are discussed below.

Differences between the Nordic countries in soil qualities, agriculture methods, diet composition etc are reflected in different intake with food per unit deposition and can be expressed as different radioecological sensitivities for the Chernobyl derived radiocesium contamination of the ground. The radioecological sensitivity for Cs-137 in diet is defined as the infinite time-integrated concentration of cesium in diet per unit deposition (A Aarkrog, 1979).

Table 5 shows some estimated values of the radioecological sensitivity after Chernobyl. It is to be observed that for another fallout situation the values may be different (H Dahlgaard et al, 1994).

Table 5. Radioecological sensitivities in total diet for Chernobyl Cs-137 in the Nordic countries.

| Country | Sensitivity $Bqakg^{-1}/kBqm^{-2}$ |
|---|---|
| Denmark | 4,4 |
| Finland | 13 |
| Iceland | - |
| Norway | 33 |
| Sweden | 20 |

The range given in Table 5 is somewhat higher than that given by UNSCEAR 1988, 1 to 9 Bqakg$^{-1}$ per kBqm$^{-2}$ with a geometric mean of 2.6 Bqakg$^{-1}$ per kBqm$^{-2}$ of Cs-137.

The resulting dose caused by intake of radionuclides with food is the end result of a combination of independent influencing factors like activity concentration in food, dietary composition, age and sex. Interrelated factors are biological half-life, chemical composition and physical qualities. The transfer factor from total diet to body burden is defined as the quotient of time integrated concentration and dietary concentration. The value is estimated by multiplying total food consumption kga$^{-1}$ by 143 Bqd per Bq (mean residence time in body) and divide with 365 da$^{-1}$ and 70 kg (body weight). The value for the Nordic countries is about 3.0 Bqakg$^{-1}$ (body) per Bqakg$^{-1}$ (diet) (UNSCEAR 1988).

An intake of 1 Bq Cs-137 corresponds to 5.6 x 10$^{-3}$ Bqakg$^{-1}$ in the body (1 Bq x 143d/360 da$^{-1}$ x 70kg). The effective dose caused by time integrated concentration in the body is for Cs-137 equal to 2.5 µSv per Bqakg$^{-1}$. Therefore an intake of 1Bq Cs-137 leads to 13 nSv (adults). The corresponding value for Cs-134 is 19 nSv. That means that an effective dose of 1 mSv corresponds to an intake of 75000 Bq Cs-137 or 50000 Bq Cs-134.

Using the UNSCEAR values for the transfer factor for deposition to diet 2.6 Bqakg$^{-1}$ for Cs-137 and the other transfer factors given above the total transfer factor can be calculated. The total transfer factors in units of µSv per kBqm$^{-2}$ are given in Table 6 (from UNSCEAR 88) for the northern countries. The contribution from external radiation is also included.

**Table 6.** Total transfer factor for effective dose based on Cs-137 deposition, µSv per kBqm$^{-2}$.

| **Pathway/ radionuclides** | µSv per kBqm$^{-2}$ (Cs-137) | |
|---|---|---|
| | **First year** | **Total** |
| External gamma | | |
| Cs-137 | 2,2 | 73 |
| Cs-134 | 2,5 | 7 |
| Other | 5,6 | 6 |
| Subtotal | 10 | 86 |
| | | |
| Ingestion | | |
| Cs-137 | 15 | 35 |
| Cs-134 | 11 | 23 |
| I-131 | 1 | 1 |
| Subtotal | 27 | 59 |
| **Total (rounded)** | **40** | **150** |

These are the general principles and values for calculating doses from data on ground deposition and dietary intakes. Another method is to measure the whole body content of cesium in people and make the assessments from that. The two results are not often identical which illustrates the differences between international average values and the actual regional and local values of those parameters that are significant for the resulting dose (see below).

In the Nordic countries there are groups of people and food that are of special interest from the radiation protection point of view. These are hunters and fishermen, reindeer keeper, berry pickers and other people eating much wild produced food. Special food are reindeer meat, mutton, moose and deer meat, freshwater fish, mushrooms. Other food like milk (with the exception of goatmilk), milkproducts, pork and beef, grain products, vegetables, potatoes and fruit have not been a problem in the Nordic countries after the Chernobyl accident. The mother's milk has not been a problem either, 1-5 $Bql^{-1}$ was measured in Sweden in most contaminated areas and exceptionally up to 10 $Bql^{-1}$ when the mother had eaten contaminated fish or reindeer (SSI-88). It has been estimated that mother's milk concentration of cesium is about 15% of the concentration in the woman's body.

An example of a Laplander family diet of special interest is given in Table 7 (W Becker).

**Table 7.** Consumption of some food by Laplander families.

| Food | Kg per person and year |
|---|---|
| Reindeer meat | 50-150 |
| Moose meat | 5-15 |
| Fresh water fish | 10-40 |
| Berries | 10-20 |

Because these food products in most contaminated areas might have activity concentrations much above 1000 $Bqkg^{-1}$ the corresponding doses would be up to 10 $mSva^{-1}$ or more. If the Cs-137 activity concentration is 1500 $Bqkg^{-1}$ (the limit in Sweden for this kind of food) and the consumption is 200 kg of this food per year, the resulting effective dose caused only by Cs-137 would be 4 $mSva^{-1}$, disregarding the leak-out effect of cooking.

These are the extremes. Because there are limits for activity concentration in food to be sold and because of additional precautions of farmers and food production industries the actual activity levels were and are much below the limits. A reliable way to assess the average intake by food is to measure on "food baskets" which are baskets with annual average composition of daily food consumption in various parts of a country. More than hundred kinds of food compose the basket and is measured and the daily average intake is calculated. In 1986 the average intake in Sweden was estimated to be about 1900 Bq of Cs-137 (7 months), in 1987 3000 Bq (SSI-88) and in 1994 274 Bq per year. For 1994 the intake corresponds to an effective dose of 3.6 μSv (H Möre et al, 1995). In most contaminated parts of Sweden the average annual intake in 1994 was 815 Bq of Cs-137 and the corresponding dose about 10 μSv.

How well do these results agree with results that are assessed otherwise? Questions relate to the effective half-life of cesium in the body, the diet composition, the ecological half-life, the total transfer factor and others.

*i) The effective half-life:*
A dose conversion factor of $1.3 \times 10^{-8}$ $SvBq^{-1}$ of Cs-137 has been used (ICRP 89) in the calculation of the dose (3.6 μSv). ICRP uses a half-life of 104 days but whole body measurements give a shorter time, 81 days (R Falk et al, 1991). Similar values have been reported also in Finland (E Häsänen and T Rahola, 1971, M Suomela, 1968). That means that the dose might be overestimated by about 20%. On the other hand, an intake of 274 Bq per year (Sweden) corresponds to a

body burden at equilibrium of 1.3 $Bqkg^{-1}$ using a half-life of 81 days. Whole body measurements 1994 gave a body burden of 2.0 $Bqkg^{-1}$ which means that 1.3 $Bqkg^{-1}$ might be an underestimation of body burden of about 50%.

*ii) The diet composition:*
It might be that the foodbasket composition does not correctly reflect the true food composition after all. Wild produce food (moose, roe deer, reindeer, fresh water fish, mushroom, berries) from contaminated areas might play a greater role in average consumption then assumed. It is proposed that as much as 30, 70 and 80% of total intake of Cs-137 might come from wild produce food in Norway, Finland and Sweden respectively (H Dahlgaard et al, 1994). These discrepancies are still to be examined.

*iii) Ecological half-life:*
There are many different values given in literature on ecological half-life. They vary between different soil, plants and location from a few years to 10-30 years, which influence the calculated individual and collective dosecommitment.

In conclusion, there are so many uncertainties in the assumptions necessary for calculation of resulting doses caused by intake of radiocesium that the most reliable way is to use whole body measurements and reasonable dose conversion factors (Sv per $Bqkg^{-1}$ body burden). Fig 17 is a summary of doseestimates for the years 1986-1990 based on whole body measurements (M Suomela and T Rahola, 1994, reassessed 1996). Results of whole body measurements in Sweden are given in Fig 18 (L Moberg et al 1996).

The corresponding collective effective doses over 50 years would be about 150 manSv for Denmark, 3500 for Finland, 1500-2500 for Norway and 1100 manSv for Sweden making about 7000 manSv in total. UNSCEAR 1988 has estimated the total collective effective dose caused by internal radiation to be about 10000 manSv which probably is an overestimation.

## Countermeasures

The fallout from the Chernobyl accident affected the Nordic countries very differently. Some parts were affected only marginally others quite heavily and in combination with specific environmental conditions and living habits resulted in significant contamination of vegetation, animals and man. One example is the ecological chain lichen-reindeer-man, which in Sweden and Norway could lead to dose commitments caused by Cs-137 in the diet of the order of a hundred mSv unless precautionary actions were taken.

There were several reasons to effectuate countermeasures to avoid high individual doses (Laplanders, fish consumers, hunters etc), to avoid doses to large population groups, if that was justified, and to avoid unnecessary doses, if that was easily done and if they caused societal/ psychological concern. It was early concluded that measures like evacuation, sheltering and intake of stable iodine were not justified. But other measures were taken e.g giving information and advice to general public and specific groups (farmers etc), making measurements in the environment, on food and on man, issuing regulations, restrictions and advice and improving the emergency preparedness system.

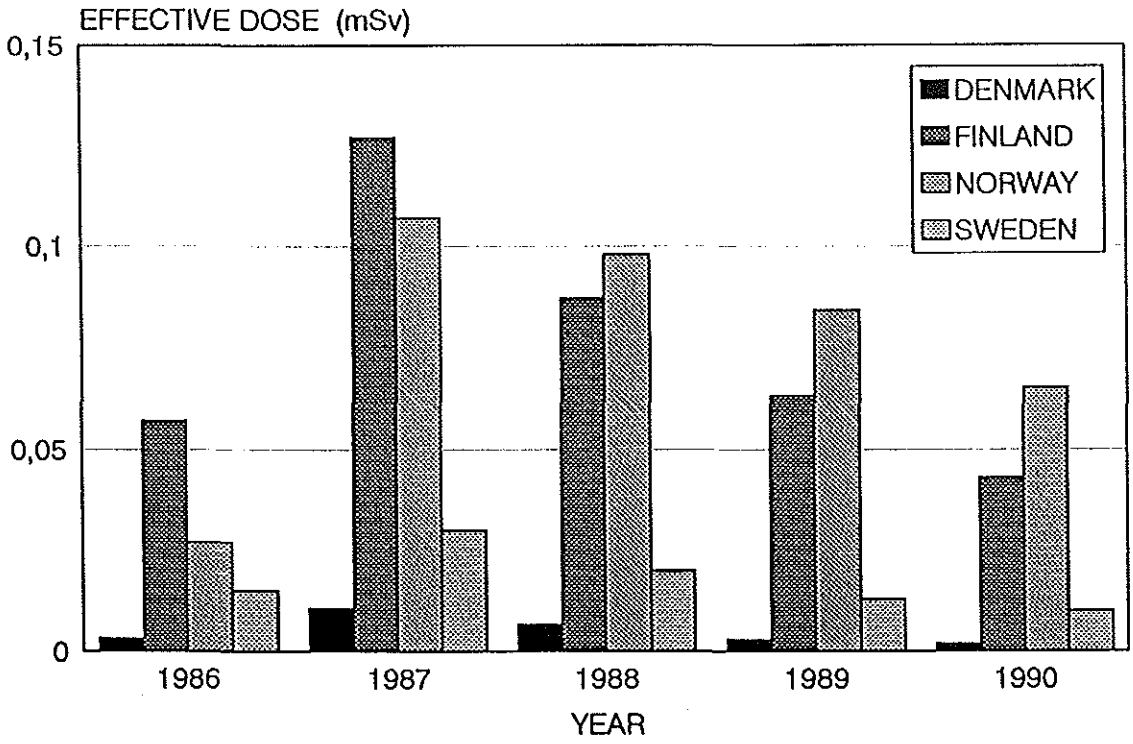


Fig 17 Effective internal doses from Cs-134 and Cs-137 delivered to the Danish, Finnish and Swedish populations and to the Sel group (Norway) in 1986-1990.
(M Suomela and T Rahola, 1994, reassessed 1996)

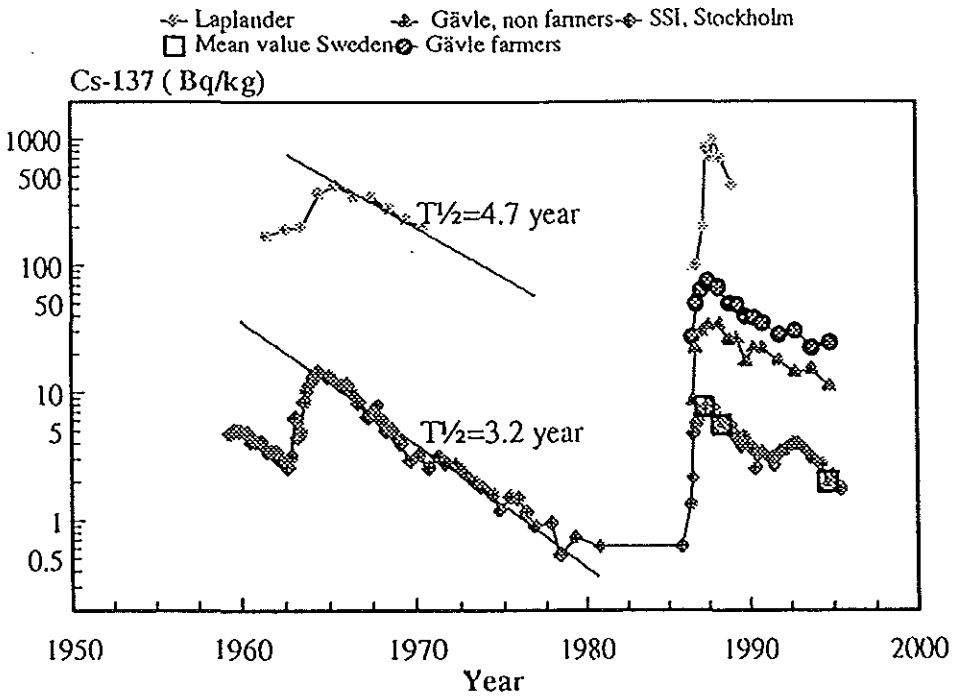


Fig 18 Measured body burden of Cs-137 (Bq/kg body weight) in a number of groups in Sweden between 1959 and 1995. During the period 1965 to 1975, the yearly intake of Cs-137 decreased with half a time of 3 - 5 years. The squares show the measured average body burden of the whole population. (L Moberg et al, 1996)

*Intervention levels*

In spite of the fact that the responsible authorities in the Nordic countries met early and repeatedly to inform each other and to harmonize the intervention levels and countermeasures, the efforts were not altogether successful in the sense that they reached the same values. Special national considerations had to be taken and this was mutually understood and accepted by all Nordic countries. The following summarizes the interventions and intervention levels (for prohibiting sale and import of food) in the affected Nordic countries, after the Chernobyl accident.

*Denmark:* The CEC values were used i.e 370 $Bqkg^{-1}$ (of Cs 134+137) for milk and dairy products and 600 $Bqkg^{-1}$ for other food (NEA 1989).

*Finland*: Max 5 mSv (effective dose) and max 50 mSv to a single organ first year and not more than 1 mSv $a^{-1}$ as an average over 50 years. The chosen intervention levels in terms of activity concentrations were 2000 $Bql^{-1}$ of I-131 in milk and drinking water, 1000 $Bqkg^{-1}$ of Cs 137 in milk, meat and others. Reindeer was no major Chernobyl related problem in Finland. No intervention levels for reindeer.

*Norway:* Intake via food max 400 kBq of radiocesium the first year and 50 kBq $a^{-1}$ following years. Breastfeeding and pregnant women max 50 kBq $a^{-1}$ all years.

1000 $Bqkg^{-1}$ of I-131 in all food. 300 $Bqkg^{-1}$ radioceasium in milk, dairy food, infant food (May-June 1986), changed to 370 $Bqkg^{-1}$ in June 1986. 600 $Bqkg^{-1}$ in other food, changed in November 1986 to exclude reindeer for which the intervention level of 6000 $Bqkg^{-1}$ was chosen (3000 $Bqkg^{-1}$ 1994). The reasons for that were socio-logical/psychological/cultural/economical. Unless changes had been made more than 85% of the reindeer had had to be discarded and there was an impending risk that the Laplanders and their reindeer breeding should cease to exist as a culture. In July 1987 6000 $Bqkg^{-1}$ should also apply to freshwater fish and game. (P Strand et al 990).

*Sweden:* First month 5 $kBqkg^{-1}$ I-131 and 10 $kBqkg^{-1}$ radiocesium for imported food and 2 $kBqkg^{-1}$ I-131 and 1 $kBqkg^{-1}$ radiocesium for domestic food. The idea behind that was that consumption of imported food was much less than domestic and therefore a higher value could be used. The values 1 and 10 were rounded off values and it was assumed that in practice the intake of cesium by food should be much less than corresponding to these figures (which was later on proved to be the case).

The values were soon changed to one value for all food 300 $Bqkg^{-1}$ Cs-137 but 2 $kBqkg^{-1}$ I-131 in milk and dairy products. The intention was that the average dose over 50 years should, even for most exposed group of individuals, be less than 1 mSv $a^{-1}$ and for a single year not more than 5 mSv. 300 $Bqkg^{-1}$ Cs 137 could in worst case give more than 1 mSv $a^{-1}$. Assuming that 1 mSv effective dose is caused by an intake of 75 000 Bq Cs-137 *or* 50 000 Cs-134, that Cs-137/Cs-134 in food was 1.7 and that about 10% of the dose by ingestion the first year is caused by other radionuclides in food, a level of 300 $Bqkg^{-1}$ of Cs-137 (corresponding to 500 Bq per day) should cause a total dose of about 5 mSv $a^{-1}$. However, after a few years 300 $Bqkg^{-1}$ Cs-137 would only give rise to about 2,5 mSv $a^{-1}$.

In order to avoid serious contamination of milk in spring 1986, cattle were not to be put on pasture outdoors if the deposition density was greater than 10 kBq $m^{-2}$ I-131 and 1 kBq $m^{-2}$ Cs 134 +137 in grass (or 3 kBq$m^{-2}$ radiocesium as determined by airborne gamma spectrometry and measurements in situ). Cesium $Bql^{-1}$ milk/cesium $Bqm^{-2}$ grass = 1:3 i.e 300 $Bql^{-1}$ milk corresponds to about 1000 $Bqm^{-2}$ grass. 25 June 1986 all pastures in Sweden were cleared.

In June 1987 300 Bq $kg^{-1}$ Cs-137 was split into 300 $Bqkg^{-1}$ for staple food and 1500 $Bqkg^{-1}$ for special food like reindeer, game, berries, freshwater fish, mushroom etc. This change of intervention level caused the percentage of unacceptable reindeer to decrease from about 80% to about 40%. No food above intervention levels was permitted to be sold in the open market.

Other intervention levels for cesium:

20 $kBqkg^{-1}$ dry substance of sewage sludge used as fertilizer.

1 $kBqkg^{-1}$ of peat 50% dry substance used as a fertilizer in soil used for vegetables and 3 $kBqkg^{-1}$ if peat is used as fertilizer on grasses etc.

*Other measures and advices (first weeks)*

DK = Denmark, FIN = Finland, N = Norway, S = Sweden

1. Advice to stay out of rain, keep children from playing in rainwater and in sand (FIN).
2. Monitoring landvehicles and people from other contaminated countries (D, FIN, S).
3. Advice to clear transport vehicles at borders (DK, FIN, S).
4. Advice not to travel to potentially contaminated countries (FIN, N, S).
5. Monitoring fishing boats (DK).
6. Advice and information to the public by answering questions by telephone and publishing brochures (all).
7. Advice not to drink rainwater, not to use it for watering livestock and not to use it in the sauna (FIN).
8. Prohibition to let cattle graze outdoors (DK), advice to keep cattle away from grazing outdoors until cleared (FIN, S), restriction on marketing of food (all) see above.
9. Advice not to consume fresh leafy vegetables (S).
10. Advice to wash fresh vegetables (DK, S).
11. Advice to delay planting of early vegetables (FIN).

12. Advice on consumption and cooking of non-cultivated plants and mushrooms (boil and eat max 1-2 times a week) (FIN, S).

13. Prohibition on imports of fresh food from Eastern countries (S).

14. Provision of government compensation for agricultural losses (FIN, N, S).

15. Advice to take protective measures in changing industrial ventilation filters (FIN, S) and in dusty soil cultivation work.

16. Restrictions/prohibition to use sewage sludge (FIN, S) (see above).

17. Recommendation on consumption of freshwater fish (FIN).

*Results of specific measures and methods to reduce internal doses*

The general principles of reducing the internal radioactive impact of environmental contamination are by reducing the transfer factors between soil-plant-animal-man, by increasing the excretion rate in animals, by processing contaminated products to yield a less contaminated product, and by changing habits and other factors that are relevant for activity levels and intake.

Contaminated lakes have been treated with liming and potash treatment. The purpose was to increase the sedimentation possibly in combination with intensive fishing. However, the experiments have not been very successful (T Andersson and M Meili, 1994) maybe some 5% faster recovery of the lake and have cost about 4 million US $ (41 lakes in Sweden, L Håkansson, 1991). The intake of cesium via fish had therefore to be reduced by advice and by intervention levels for sale on the open market.

In agriculture land the uptake of cesium to crops depends mainly on the soil characteristics, content of clay and organic matter. Application of potassium fertilization decreases the uptake of cesium in a way that depends on these factors. The need of fertilization of cultivated soils increases in the following way: clays < loams < sandy soils < peat soils, and consequently the uptake of cesium increases in the same order. Application of potassium fertilizer up to about 200 kg K per ha has reduced the cesium uptake to 10-20% of its normal uptake both on grass and arable land. Further fertilization has not led to further reductions (K Rosén, 1994).

Other methods to reduce the uptake is to change the farming habits and methods. Examples of those used after the Chernobyl accident are:

1. Deep ploughing. A reduction of root uptake by a factor 10 is obtained by ploughing 30-60 cm deep for different crops.

2. Lift cutting level, when harvesting grass, reduces by a factor 10.

3. Postpone harvest.

4. Dispose surface contaminated crops and grass. 25-30% can be caught by crops and grass.

5. Change in crops and varieties cultivated. The uptake can differ 30-40%.

Contamination of sheep, goat and cattle was a significant problem particularly in Norway. Special restrictions were made in allowing cattle graze outdoors (see above) and cattle and sheep were specially fed with uncontaminated food.

Cesium binders like bentonite and AFCF (ammonium iron-hexacyanoferrate (II)) were used. The latter has been found to be some more efficient and brought a reduction of about 90-95% of cesium content of goat milk after being fed 0,5 kg concentrate (1 g $kg^{-1}$) per day for 3-4 weeks (K Hove et al, 1990). Salt licks with 25 g AFCF $kg^{-1}$ for sheep led to 25-75% reduction in body activity after some weeks. However, normally, as far as supplies admitted, cesium free or low level fodder was used for a period of 4-8 weeks for sheep and goats.

In Norway in 1986 about 320 000 sheep (27% of the flock) were in what was called special measure zones where activity in sheep was between 600-2000 $Bqkg^{-1}$ radiocesium. A few percent were banned (> 2000 $Bqkg^{-1}$). The sheep in special measure zones were given special fodder to reduce the activity levels. In 1987 and 1988 corresponding numbers were 280 000 sheep and 360 000 sheep respectively. The high number in 1988 depended on increased intake of contaminated mushrooms, since 1988 was a very good year for mushrooms (P Strand et al, 1990).

The number of cattle in Norway with special food programme was in 1987 45 000 and in 1988 approximately the same.

The contamination of moose was reduced by using salt lick with giese salt applied in 3 areas in Sweden. A reduction of activity concentration in moose of 20-25% was achieved but the results were occasionally higher, sometimes lower depending on whether the salt licks were placed correctly. Change of hunting from October to December resulted in a decrease but could also lead to an increased activity level. So in conclusion, there is no always reliable way to artificially reduce the activity levels in moose (K J Johansson, 1994).

The case of roe deer is somewhat simpler. For instance, by changing hunting from August/September to the spring decreased the activity levels by a factor of 5. Supplementary fodder, with cesium binder, can also be given. However, if there is little snow and above -5°C the roe deer prefer lingonberries, bilberries and heather and all efforts to reduce the activity levels are wasted (K J Johansson, 1994).

The most serious animal contamination in Norway and Sweden was and is of the reindeer. Some studies have been made on reindeer in corrals on the effect of cesium binders like bentonite, zeolite and AFCF in food (2-4%) to increase excretion. 50-90% reduction of body burden was achieved. Addition of potassium to the lick from 0,8 g $d^{-1}$ to 16 g $d^{-1}$ reduced the biological half-life from 22 days to 11 days, same effect by sodium. Another more practical way to reduce the activity contamination in reindeer was to move them to uncontaminated areas, and to postpone slaughtering until the autumn when the reindeer's body burden of radiocesium was the lowest after avoiding lichen during summer time. The intake of cesium by the Laplanders was also reduced by information on cooking methods and restrictions of amount of meet and activity levels. The effects of this information was encouraging and more than 50 % of the Laplanders changed their eating habits. It is concluded that these advices and measures probably reduced the internal doses in average for the Laplanders about a factor of 10 (from 100-150 mSv to 10-15 mSv in 50 years) (SNT 1988).

## Costs and cost-effectiveness

The costs consist of costs for countermeasures, compensation, equipment, information etc. In Norway the total cost is of the order of 50 million US $ and in Sweden of the order of 150 million US $. In Finland and Denmark the costs were much less. The objectives of the measures were various. They were to prevent high individual doses, to retain confidence in the market, to maintain special cultures, to avoid even higher expenses etc. The objectives have generally been reached. The measures taken were sometimes cost-effective in the sense that the cost per saved manSv was quite reasonable as compared with a Nordic agreed value of 100 000 $ per manSv. This is shown from Norwegian data see Table 8 (from P Strand 1994).

**Table 8.** Cost (US $) of countermeasures in terms of manSv saved.

| Countermeasure | US Dollar/manSv |
|---|---|
| Interdiction sheep | 150 000 |
| Interdiction reindeer | 50 000 |
| Special feeding | 35 000 |
| Change of slaughter time | 13 000 |
| Prussian blue boli | 600 |
| Prussian blue concentrate | 150 |
| Dietary advices | 50 |

## The situation today

Even if the contamination and radiation levels have decreased by a factor 2 and more, sometimes much more, there are still areas and food that need special attention. The areas are those most heavily contaminated and the food is reindeer meat, mutton, moose and roe deer meat, fish and mushrooms, particularly from these areas. There are still several thousand lakes with fish containing more radiocesium than allowed for fish in the open market.

The emergency preparedness has been improved in all countries and there is now a much better knowledge of possible consequences and preparedness for taking decisions on countermeasures, making measurements and giving advice. There is also a much improved system for early warning and for exchange of data and information.

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# References


Aaltonen H, Saxén R and Ikaheimonen TK
Airborne and deposited radioactivity in Finland in 1987. Report STUK-A75.
Finnish centre for Radiation and Nuclear Safety, Helsinki:1990.

Aarkrog A, 1979
Environmental Studies on radioecological sensitivity and variability with special emphasis on the fallout nuclides Sr-90 and Cs-137.
Risø-R-437.

Andersson T and Meili M, 1994
The role of lake-specific abiotic and biotic factors for the transfer of radiocesium fallout to fish.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 79-92.

Arvela H, Markkanen M and Lemmelä H, 1990
Mobile survey of environmental gamma radiation and fallout level in Finland after the Chernobyl accident. Rad. Prot. Dos. 32 3 (1990) 177-184.

Becker W, 1987
The diet of the Laplanders after Chernobyl.
Calculations of cesium intake and dadiation doses.
SLV rapport 1987:4, in Swedish.
Statens LivsmedelsVerk, 75126 Uppsala.

Bergman R, Nylen T, Palo T and Lidström K, 1991
The behaviour of radioactive cesium in a boreal forest ecosystem.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 425-456.

Bjørnstad H E, Brittain J E, Saxén R and Sundblad B
The characterization of radiocesium transport and retention in Nordic lakes.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H. Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 29-44.

Blomqvist L, Paakkila O, Rahola T, Rantavaara A, Suomela M and Taipele T
Tjernobyl och Finland, 1987 (in Swedish).
Published by the Finnish Centre for Radiation and Nuclear Safety (STUK).

Broberg A
The distribution and characterization of Cs-137 in lake sediments.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H. Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 45-62.

Carlsson L and Snoeijs P
Radiocesium in algae from Nordic coastal waters.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 105-117.

Dahlgaard H et. al., 1994
General summary and conclusions.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 7-20.

Dahlgaard H, Aarskrog A and Nielsen S P 1988
Chernobyl and the Nordic Seas. 5th Nordic Radioecology Conf.
August 22-25, 1988, Rättvik, Sweden, 7pp (in Danish).

De Geer L-E, Arntsing R, Vintersved I, Sisefky J, Jakobsson S and Engström J-Å, 1978
Particulate Radioactivity, Mainly from Nucleat Explosions, in Air and Precipitation in Sweden Mid-year 1975 to Mid-year 1977.
National Defence Research Establishment.
FOA Report C 40089-T2 (A1) November 1978.

Devell L, 1991
Composition and preperties of plume and fallout materials from the Chernobyl accident.
In The Chernobyl Fallout in Sweden, edited by L. Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 29 - 46.

Devell L, Aarkrog A, Blomqvist l, Magnusson S and Tvelen U, 1986
How the fallout from Chernobyl was delected and measured in the Nordic countries.
Nuclear Europe 11/1986.

Edvarson K, 1991
Fallout over Sweden from the Chernobyl Accident.
The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 47 - 65.

Edvarson K, 1991
External doses in Sweden from the Chernobyl fallout.
In The Chernobyl Fallout in Sweden, edited by L. Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 527-545.

Edvarson K, 1996
Personal communication

Eriksson O, Jones B-E.V and Rannistola T, 1991
Radiocesium contamination and the reindeer.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 401-424.

Evans S
Impact of the Chernobyl fallout in the Baltic Sea ecosystem.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 109-127.

Falk R, Eklund G, Giertz H, and Östergren I, 1991.
Cesium in the Swedish population after Chernobyl: Internal radiation, Whole-body counting.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 547-577.

Finck R, 1991.
Shielding factors for gamma radiation.
Experiments and calculations for Swedish dwellings.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 489-526.

Gaare E and Staaland H, 1994.
Pathways of fallout radiocesium via reindeer to man.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 303-334.

Gudiksen P H, Harvey T F and Lange R, 1989
Chernobyl Source Term, Atmospheric Dispersion and Dose Estimation.
Health Physics 57 (1989) 697.

Hansen H S and Andersson J, 1994.
Transfer of Cs-137 to cows' milk in the Nordic countries.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 197-210.

Holm E
Fallout of transmenium elements in Sweden following the Chernobyl accident.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 67 - 83.

Hove K et al, 1994.
Radiocesium transfer to grazing sheep in Nordic environments.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 211-227.

Hove K, Hansen H S and Strand P, 1990.
Experience with the use of cesium binders to reduce radiocesium contamination of grazing animals.
In Flitton S, KatzE W (eds): Environmental contamination following a major nuclear accident. IAEA-SM-3062:181-189, Vienna 1990.

Håkansson L
Radioactive cesium in fish in Swedish lakes after Chernobyl.
Geographical distribution, trends, models and remedial measures.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 239-281.

Häsänen E and Rahola T
The biological half-life of Cs-137 and Na-24 in man.
Ann. of Clin. Res. 3, 1971, page 236-240

ICRP 89
International Commission on Radioecological Protection.
Age dependent doses to members of the public from intake of radionuclides.
Part I, Publication 56, Annals of the ICRP 20 No.2, 1989.

Johansson K J, 1994.
Radiocesium in game animals in the Nordic countries.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 287-301.

Johansson K J, Bergström R, von Bothmer S and Kardell I, 1991.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 477-486.

Kjelle P E, 1991.
First registration of the Chernobyl accident in the west by the gamma radiation monitoring stations in Sweden.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 21-27.

Kjelle P E, 1996
Personal communication

Meili M
Radiocesium as ecological tracer in aquatic systems - a review
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 127-139.

Moberg L, Falk R, Edvarson K, Mellander H and Nyblom L, 1996
Long term health effects in Sweden from the Chernobyl accident.
Submitted to IAEA conference 8-12 April 1996 on the Chernobyl accident.

Möre H, Becker W, Falk R, Brugård Konde Å and Swedjemark G A, 1995
Foodbasket measurements, autumn 1994.
SSI-rapport 95-22 (in Swedish).
Swedish Radiation Protection Institute, 171 16 Stockholm.

NEA, 1989
Nuclear accidents.
Intervention levels for protection of the public.
Nuclear Energy Agency, OECD, Paris 1989.

Olsen R A, 1994
The transfer of radiocesium from soil to plants and Fungi in semi-natural ecosystems.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 265-286.

Oughton D H and Salby B, 1994.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 165-184.

Risø 1986-1992
Risø National Laboratory.
Environmental radioactivity in Denmark 1985-1991.
Risø Reports No's 540, 549, 563, 570 and 621.

Roos P, Samuelsson C and Mattsson S,1991
Cs-137 in the lichen Cladina Stellaris before and after the Chernobyl accident.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 389-400.

Rosén K, 1994
Studies on countermeasures after radioactive depositions in Nordic agriculture.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P. B. 49, 4000 Roskilde, Danmark, 1994, page 239-259.

Rosén K
Effects of potassium fertilization on cesium transfer to grass, barley and vegetables after Chernobyl.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 305-322.

Saxén R and Aaltonen H
Radioactivity of surface water in Finland after the Chernobyl accident in 1986
Report STUK-A60. Finnish Center for Radiation and Nuclear Safety, Helsinki: 1987

Saxén R
Radioactivity of surface water and freshwater fish in Finland in 1987.
Report STUK-A77. Finnish Center for Radiation and Nuclear Safety, Helsinki: 1990

Saxén R and Koskelainen U
Radiactivity of surface water and freshwater fish in Finland in 1988-1990.
Report STUK-A94. Finnish Center for Radiation and Nuclear Safety, Helsinki: 1992

Saxén R
Transport of Cs-137 in large Finnish drainage basins
In Nordic Radioecology
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P B 49, 4000 Roskilde, Danmark, 1994, page 75 ff.

SNT, 1988
Stråledose fra mat til menneske efter Tsjernobyl.
Statens näringsmiddeltilsyn, Postboks 8187 Dep. 0034 Oslo (in Norwegian).

SSI-88, 1988
Projekt Tjernobyl - Lägesrapport 3.
SSI-rapport 88-13 (in Swedish).

Strand P
Radioactive fallout in Norway from the Chernobyl accident.
Studies on the behaviour of radiocesium in the environment and possible health impact.
NRPA Report 1994:2.
Norwegian Radiation Protection Authority. Postboks 55. N - 1345 Österås. Norway.

Strand P, Brynildsen L I, Harbitz O, and Tvelen K, 1990.
Measures introduced in Norway after the Chernobyl accident.
A cost-benefit analysis.
In Flitton S, KatzE W(eds): Environmental contamination following a major nuclear accident.
IAEA-SM-3062:191-202, Vienna 1990.

Strand T, Strand P and Baarli J, 1987.
Radioactivity in foodstuffs and doses to the Norwegian population from the Chernobyl fallout.
Radiation Protection Dosimetry, Vol. 20 No 4 page 211-220.

Strandberg M
Contamination of annual crops.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P B 49, 4000 Roskilde, Danmark, 1994, page 185-195.

STUK, 1991
Fem år efter Tjernobyl (five years after Chernobyl)
ISBN 951-47-4952-9
Finnish Centre for Radiation and Nuclear Safety (STUK)
Helsinki, Finland.

Sundblad B, Bergström U and Evans S
Long term transfer of fallout nuclides from the terrestrial to the aquatic environment.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 207 - 238.

Suomela M, Blomqvist L, Rahola T and Rantavaara A
Studies on environmental radioactivity in Finland in 1987.
Annual report STUK-A74, Finnish Center for Radiation and Nuclear Safety
Helsinki, Finland (1991)

Suomela M and Rahola T, 1994.
Internal radiation doses to the Nordic population based on whole-body counting.
In Nordic Radioecology.
The transfer of radionuclides through Nordic ecosystems to man, edited by H Dahlgaard, Risø National Laboratory, P B 49, 4000 Roskilde, Danmark, 1994, page 457-471.

Suomela M and Rahola T, 1996
Internal doses of people in Finland after the Chernobyl accident.
Paper IAEA-CN-63/158. To be presented at the international conference One Decade After Chernobyl, Vienna 8-12.4.1996

Suomela M, 1968
Whole body measurements on people in Finland (in Swedish)
Nordic Society for Radiation Protection, Proceedings, 1968.

UNSCEAR 1988
Sources, Effects and Risks of Ionizing Radiation
United Nations Scientific Committee on the Effects of Atomic Radiation.
1988 Report to the General Assembly
United Nations 1988

Wallström E et al, 1991.
Radioecological studies in western Sweden after the Chernobyl accident.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 373-388.

Vintersved J, Arntsing R, Bjurman B, De Geer L R and Jakobsson S
Resuspension of radioactive cesium from the Chernobyl accident.
In The Chernobyl Fallout in Sweden, edited by L Moberg, The Swedish Radiation Protection Institute, 171 16 Stockholm, 1991, page 85 - 106.

Åhman B and Åhman G, 1994
Radiocesium in Swedish reindeer after the Chernobyl fallout: Seasonal variations and long-term decline.
Health Phys. 66(5): 503-512; 1994.

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