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Shortly after the accident, several reseach projects were initiated in Sweden in order to follow the distribution of radionuclides in the aquatic and terrestrial environment. The results, which in many cases are preliminary, show that the recovery of the ecosystem will take several decades.

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CONSEQUENCES IN SWEDEN OF THE CHERNOBYL ACCIDENT

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ABSTRACT

The radiation doses to man in Sweden due to the Chernobyl accident originate mainly from external irradiation from deposited radionuclides and internal irradiation from consumption of radioactively contaminated food stuffs. Inhalation and external irradiation from the passing cloud give only a minor contribution to the total dose. As an average for the Swedish population the individual radiation dose during the first year amounts to about 0.1 mSv, *i.e.* about 10% of the natural background radiation. In the most contaminated areas, however, the individual dose may become 30 times higher than the average dose. The dose committed over 50 years has estimated to be about six times as high as the first year dose. The collective dose for the Swedish population has been estimated to about 1300 manSv the first year after the accident and the corresponding dose over 50 years to 5000 to 7000 manSv. This could lead to 100 to 200 extra fatal cancers. Furthermore, no damages on man that can be related to Chernobyl fallout, *e.g.* pre-natal effects, have so far been observed in Sweden.

Shortly after the accident, several research projects were initiated in Sweden in order to follow the distribution of radionuclides in the aquatic and terrestrial environment. The results, which in many cases still are preliminary, show that the recovery of the ecosystem will take several years.

INTRODUCTION

The release of radioactive material into the atmosphere from reactor four at Chernobyl is the most serious in the history of the civil nuclear industry. The radioactive material released was carried away by the wind and was spread over the world, and particularly over Europe. In Sweden, the fallout was very unevenly distributed and the largest contaminated areas are along the coast of Gulf of Bothnia and the middle inland of Sweden.

Immediately after the accident several measurements were initiated in Sweden in order to get a picture of the situation. These measurements, and others that were initiated somewhat later, were used for calculating radiation doses due to internal and external irradiation. Both individual and collective doses were calculated. Doses due to food consumption were also

estimated by means of whole body measurements of a statistically representative group of people.

The purpose of this paper is to summarise the radiation doses in Sweden due to the Chernobyl accident, give a brief description of the research work that were initiated because of the accident and to give an idea of the future distribution of radioactive caesium in the environment.

2. ATMOSPHERIC TRANSPORT AND DEPOSITION

Sweden was affected by the release from the reactor at two occasions, 27 - 30 April and 8 - 10 May 1986. The calculated trajectories for different altitudes for the first cloud emitted are shown in Figure 1 (ref. 1). Early on 27 April the cloud seems to have split into two parts. The "lower" part of the cloud mainly reached Sweden and Norway whereas the "higher" parts of the cloud mainly passed over Finland. There is a difference in nuclide composition between the lower and higher parts of the cloud that depends on particle size of the debris and volatility of the released components. *E.g.* the cloud that passed Finland and the eastern part of Sweden had a lower content of strontium, ruthenium and plutonium and a higher content of iodine and caesium as compared to the cloud that passed at a lower altitude.

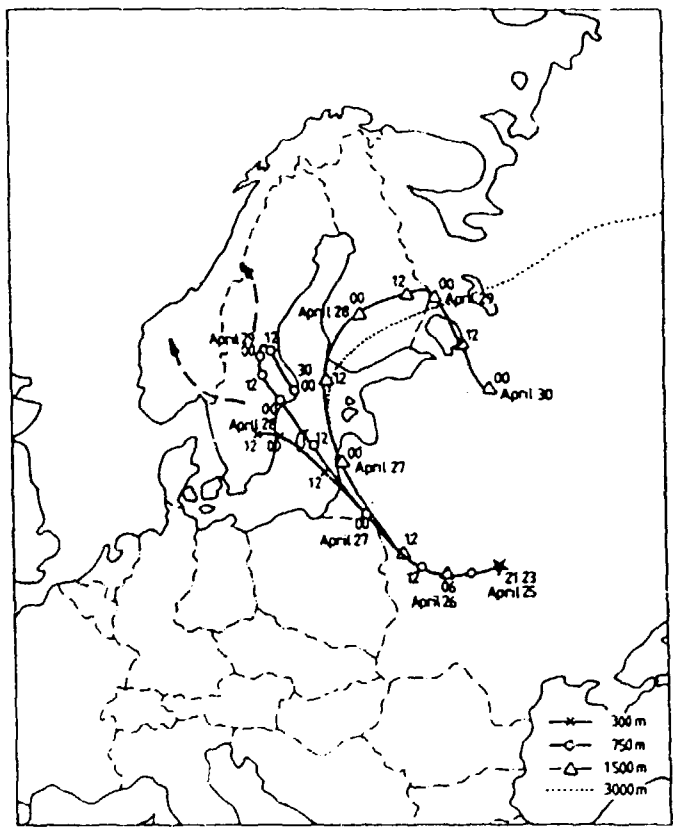

FIGURE 1. The calculated transport path of the cloud first emitted. The dashed lines indicate probable further transport paths.

The Chernobyl fallout was mainly deposited during the period 28 to 30 April, 1986. The radionuclides that first were registrated in Sweden reached ground through fumigation in the most easterly parts of the country and the radionuclides had probably been transported at an altitude of approximately 1500 m (ref. 1). Due to rain fall in the eastern and middle-northern part of Sweden, large amounts of radionuclides were washed out from the cloud and deposited on the ground. The rain fall caused a large and uneven deposition of radionuclides. In most parts of the country, however, the surface contamination was due to dry deposition. A rain fall on the 8 and 9 of May caused a small increase in the ground contamination along the south-west coast of Sweden. This contamination was due to the second cloud emitted from the reactor.

The surface distribution of the fallout was mapped by aerial measurements of *Cs-134* and performed by the Swedish Geological Company during May to June 1986 (ref. 2). The measurement heights were 60 or 150 m and the distance between flight paths were 2, 4, 10 or 50 km depending on deposition density and variation. The reason for measuring Cs-134 rather than Cs-137 is not based on its importance as a contributor to the radiation dose, although this nuclide gives a significant contribution to dose during the first years after the accident, but rather that its gamma energy does not interfere with gamma energies from naturally occurring radionuclides and with Cs-137 originating from nuclear weapon tests. Moreover, it was found that the ratio Cs-134:Cs-137 was 0.6 and almost constant over Sweden, hence making it possible to use Cs-134 as a measure on Cs-137.

From the aerial measurements a deposition map was constructed for Cs-134 and Cs-137 contamination. The map for Cs-137 is shown in Figure 2 and gives the "surface equivalent" ground contamination. This means that the ground deposition was determined as if the nuclides were evenly deposited on a flat surface of infinite area (ref. 3, 4). The "surface equivalent" deposition is always lower than the true deposition. The conversion factor between the two measures depends on the relaxation length, *i.e.* penetration depth, of the radionuclides in the ground. This means that, since ground conditions vary, different correction factors have to be applied for different parts of the country if the true deposition is to be calculated. The real deposition, however, is probably higher by a factor of two which corresponds to a relaxation length of about one centimeter.

New aerial measurements were performed during the autumns of 1987 and 1988 for the most heavily contaminated area, *i.e.* the Gävle area approximately 200 km north of Stockholm. These measurements show that the surface equivalent deposition density, corrected for decay, has decreased by about 20% per year between 1986 and 1988 (ref. 4).

As an average for the country, the "surface equivalent" deposition was determined to about 3 $kBqm^{-2}$ of Cs-134 and 5 $kBqm^{-2}$ of Cs-137. In the five most affected counties (*cf.* Figure 3) the corresponding deposition was about 5 times as high whereas in the most affected county (X-county in Figure 3) the deposition was about 20 times higher (ref. 3). The total deposition in Sweden has been estimated to about 4.3 PBq Cs-137 and 70% is found in wet deposition areas (ref. 4). According to the 1988 UNSCEAR report (ref. 5), the total inventory of Cs-137 was 290 PBq of which 13% was released from the reactor. This means that about 10% of the released caesium was deposited on Swedish territory.

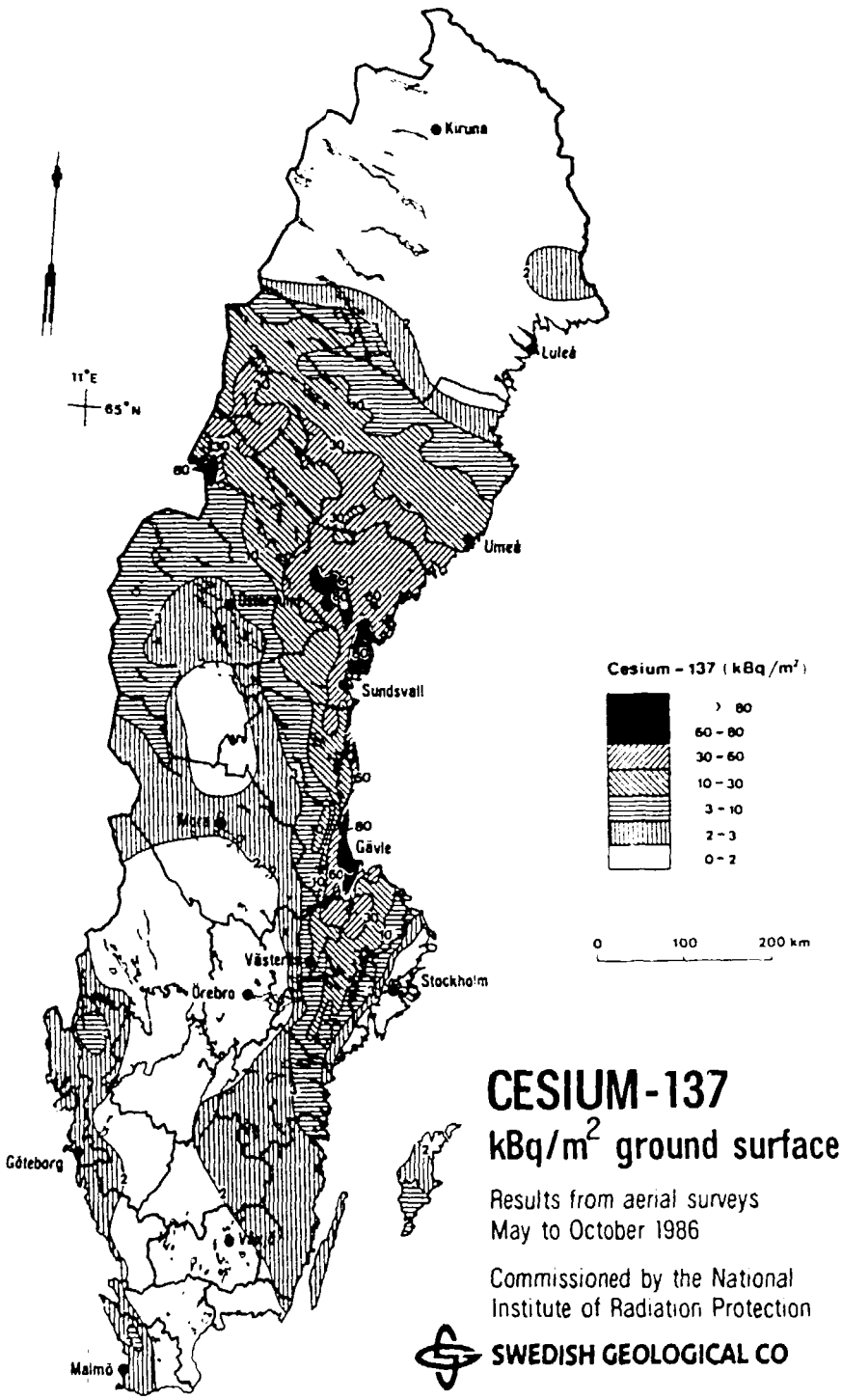

FIGURE 2. Cs-137 deposition density in Sweden

Most affected counties:

C	Uppsala
U	Västmanland
X	Gävleborg
Y	Västernorrland
AC	Västerbotten

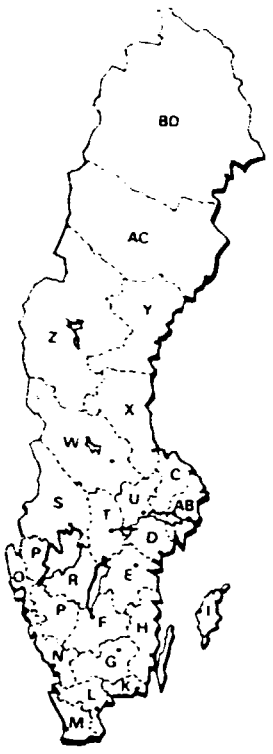

FIGURE 3. Map of Sweden with the geographic location of different counties.

3. RADIATION DOSES TO MAN

The radiation doses from the Chernobyl accident originate from *i*/ external irradiation from the passing cloud and radionuclides deposited on ground and *ii*/ internal irradiation from inhaled radionuclides and ingestion of contaminated food. The average individual effective dose equivalents in Sweden from the Chernobyl accident are summarised in Table 1. Table 2 on the other hand gives the significance of various radionuclides for the different dose pathways (ref. 6).

3.1 External irradiation from the passing cloud

Gamma emitting radionuclides such as iodine and caesium will irradiate persons being under or in the cloud. Radionuclides emitting short range β and α radiation, *e.g.* strontium-90 and plutonium, will not give a significant contribution to the dose in this case. External radiation from the cloud contributes less than one percent to the effective dose equivalent (*cf.* Table 1).

3.2 External irradiation from deposited radionuclides

Radioactive gamma emitters deposited on the ground give rise to the largest dose equivalent (*cf.* Table 1). A typical time evolution of the gamma dose rate from the ground is shown in Figure 4 and the impact of Chernobyl is clearly seen. The background radiation level, which is about 100 nSv/h, is also indicated in the figure. The large dose rate in the beginning is due to short-lived gamma emitters such as I-131, Te-132 and Ba-140. However, when the short-lived

nuclides have decayed, which takes a few months, the ground gamma activity will be dominated by the caesium isotopes. The decrease in dose rate during the winter months is due to shielding by the snow cover.

TABLE 1. Average individual effective dose equivalent in Sweden due to the Chernobyl accident (ref. 3).

Source	Dose equivalent, μSv	
	first year	50-years
Radiation from passing cloud	0.1 - 1	0.1 - 1
Inhalation	1.0 - 20	1.0 - 20
Ground irradiation:		
Sweden (8.3×10^6 people)	100	600
C, U, X and Y counties (1.1×10^6 people)	400	3000
40,000 people in C, X and Y county	1000	10,000
500 people in X county	3000	25,000
Internal irradiation due to food consumption	20 - 60	100

TABLE 2. Important pathways for various radionuclides. Nuclides indicated with × are expected to give relatively large dose equivalents while nuclides indicated with (×) are a minor problem.

Exposure	Iodine	Other short-lived	Caesium	Strontium	Plutonium
Passing cloud	(×)	(×)	(×)		
Inhalation	×	×	(×)	(×)	(×)
Ground	×	×	×		
Food	×		×	(×)	

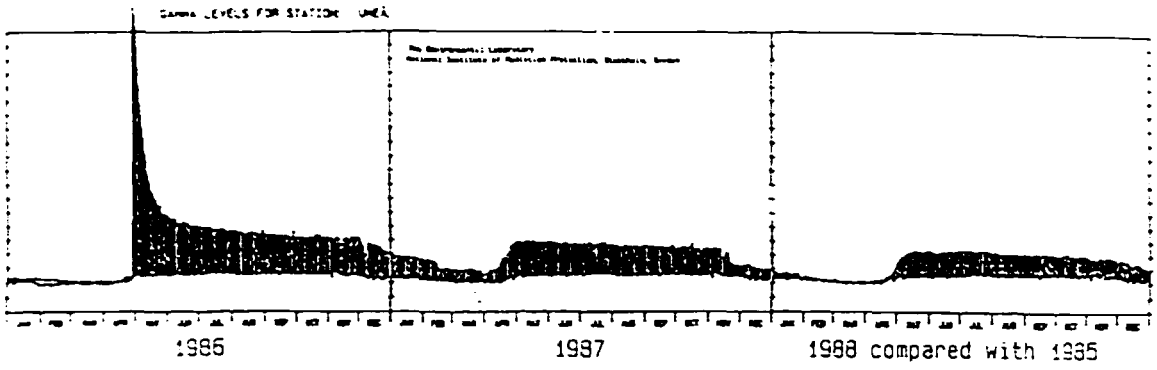

FIGURE 4. The gamma ambient dose rate as a function of time for the monitoring station in Umeå. The lower curve shows the dose rate due to natural radiation. The vertical axis extends from 0 to 1000 nanosievert per hour.

Finck *et al.* (ref. 3, 7) have calculated the effective dose equivalent from ground deposition. The calculations were based on the measured "surface equivalent" deposition (*cf.* Figure 2) and an initial relaxation length of one centimeter and an annual decrease of the "surface equivalent" deposition density by a factor of 0.8 the two first years after the accident. From the third year to year 50 after the accident, the relaxation length is assumed to be constant at three centimeters. However, measurements on migration of caesium in soil indicate that caesium has migrated less than one centimeter in uncultivated soil during the first 18 months after the accident (ref. 8). If this finding is general for Swedish soils, it could mean that the 50 year dose equivalent has been somewhat underestimated. It was also assumed that people spent 15% of their time outdoors and rest of the time in their dwellings. An average protection factor of 0.22 was used for buildings. The estimated radiation doses from ground radiation are compiled in Table 1, however, the 50 year dose might be 10 to 20% too low if the caesium migration have ceased after the first year. In wet deposition areas the first year dose is to about 70% due to caesium whereas in dry deposition areas 50% or more of the dose is due to nuclides such as cerium, iodine, ruthenium, tellurium and zirconium (ref. 3). In both cases, however, the 50-year dose will be dominated by caesium.

3.3 Doses due to inhalation

Inhalation of radionuclides will cause irradiation of the lungs. A fraction of the inhaled nuclides will also be taken up by body liquids and be distributed to different organs in the body. *E.g.* iodine will accumulate in the thyroid, strontium in the bone, caesium in the muscles, hence irradiating these organs.

Radiation doses due to inhalation of particulate material were estimated from air filter samples taken at seven different air sample stations in Sweden (ref. 9). These stations have been in operation for about 25 years, originally for monitoring fallout from atmospheric nuclear weapon tests. Noble gases, gaseous iodine or any other gaseous compounds cannot be

detected. It was assumed that the activity of gaseous iodine was about five times higher than for the detected particulate form. The individual inhalation dose was thereafter estimated to be in the range 1 to 20 μSv. This dose is only a few percent or less of the total long term individual dose.

The estimated inhalation dose also includes the dose from "hot" particles. In principle the "hot" particles can be divided into two main categories (ref. 10). The first category mainly contains radionuclides such as Ru-103 and 106, Ba-140, Ce-141 and 144 and Np-239. The activity of this type of particle is in general about 1000 Bq. The second category is less common, 10-20% of all particles (*cf.* ref. 3), and is mainly composed of Ru-103, Ru-106 and Mo-99 and with an activity of 10 to 50 kBq/particle.

Inhaled "hot" particles may in the lungs give rise to radiation doses over a very limited volume. Several biological experiments have, however, looked for enhanced carcinogenic effects of these particles but have failed to demonstrate such an effect. Therefore, in order to calculate the inhalation dose due to "hot" particles it was assumed that the activity of the particle was distributed evenly over many particles (ref. 11). This means that the dose from *one* "hot" particle of activity N Bq is equivalent with the dose from N particles of 1 Bq each.

3.4 Doses due to food consumption

Ingestion of contaminated foodstuffs lead to irradiation of the gastrointestinal tract. Radionuclides may thereafter be absorbed from the gastrointestinal tract and become distributed among different organs. Estimation of doses due to ingestion of radioactively contaminated foodstuffs has been made both from calculations based on activity in foodstuffs and whole body measurements.

In all of the Nordic countries, except Iceland, whole body measurements have been performed. In Sweden and Finland the measurements were performed on stratified random samples from the total population in the respective countries. This technique should make it possible to get a statistically correct estimate of the average individual dose caused by ingestion of radioactively contaminated foodstuffs.

In the Swedish study, 1000 persons of the age 1 to 75 years were selected and divided into four groups or *strata*. Stratum A refers to "not affected" counties, *i.e.* all counties except C, U, X, Y and AC (*cf.* Figure 3). For the affected counties a division was made between urban area (stratum B) and rural area. People living in the rural area were divided according to age, 1-35 years stratum C and 36-70 years stratum D. The total number of persons in stratum A was 500, in stratum B 360 and in strata C and D 70 each.

From this group, 250 persons were selected and 218 arrived to the National Institute of Radiation Protection (SSI). The measurements were carried out Mars to April 1987, *i.e.* about one year after the accident (ref. 12, 13). The number of measured persons turned out to be sufficient to ensure a standard deviation of less than 15% of the population dose. Approximately one year later, April to May 1988, 109 persons from the group were measured once again. Table 3 summarises the result of the whole body measurements.

The measurements show that the Swedish population during the spring 1987, *i.e.* one year after the accident, had a body content of about 8 Bq/kg of Cs-137. This corresponds to a whole body content of about 500 Bq Cs-137. One year later the body content is about 30% lower. The ratio Cs137:Cs-134 Mars to April 1987 and April to May 1988 was 2.3 and 3.2, respectively. No other radioactive nuclides besides K-40 have been detected.

TABLE 3. Average body content (Bq/kg) of Cs-137 in the Swedish population (ref. 12).

Stratum	Area	1987 (Mars-April)	1988 (April-May)
A	"not affected" counties	5.9	4.4
B	Urban area, affected counties	18.5	12.6
C	Rural area, affected counties (1 - 35 years)	15.7	7.8
D	Rural area, affected counties (36 - 75 years)	11.6	7.9
	Affected counties	17.8	12.0
	Whole country	7.7	5.6

The dose to each individual was calculated as the sum of Cs-134 and Cs-137. Table 4 summarises the annual individual average doses for different groups in Sweden for the first year (May 1986 to May 1987) and second year (May 1987 to May 1988) after the Chernobyl accident.

The doses in Table 4 are average values for both women and men. No significant differences, however, in the radiation dose to women and men exist.

In order to estimate the development of the radiation dose during the years after the accident, one has to know the variation of body content with respect to time. For this purpose, three groups of people representing different living conditions are followed up after the accident, namely: *i*/ the SSI control group; *ii*/ 12 farmers and 12 non-farmers in the Gävle-area which is the most affected area in Sweden; and *iii*/ about 30 Lapps who are dependent on reindeer breeding.

The SSI control group has been followed since 1959 and is composed of 36 persons. The body content of Cs-137 for this group is shown in Figure 5. The first peak in the Figure is due to the nuclear weapon tests fallout while the second peak is due to the Chernobyl fallout.

TABLE 4. Average dose (mSv/year) due to food consumption after the Chernobyl accident.

Stratum	Area	First year	Second year
A	"not affected" counties	0.017	0.021
B,C,D	Affected counties	0.054	0.059
	Whole country	0.023	0.027

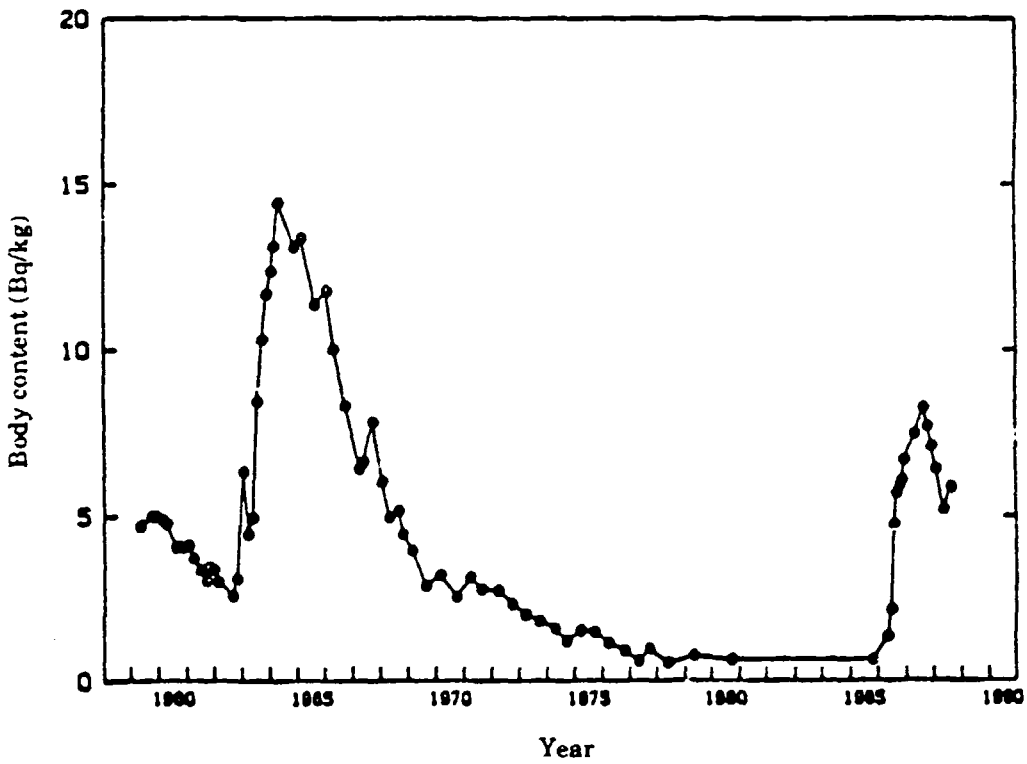

FIGURE 5. Average body content of Cs-137 for the SSI control group.

The radiation dose to the individuals in these groups due to intake of contaminated foodstuffs is shown in Figure 6. The average individual radiation dose for the Swedish population is also indicated in the Figure. As seen, this dose agrees with the dose received by the SSI control group and therefore it is believed that the variation of the dose to the control group reflects the dose to the Swedish population.

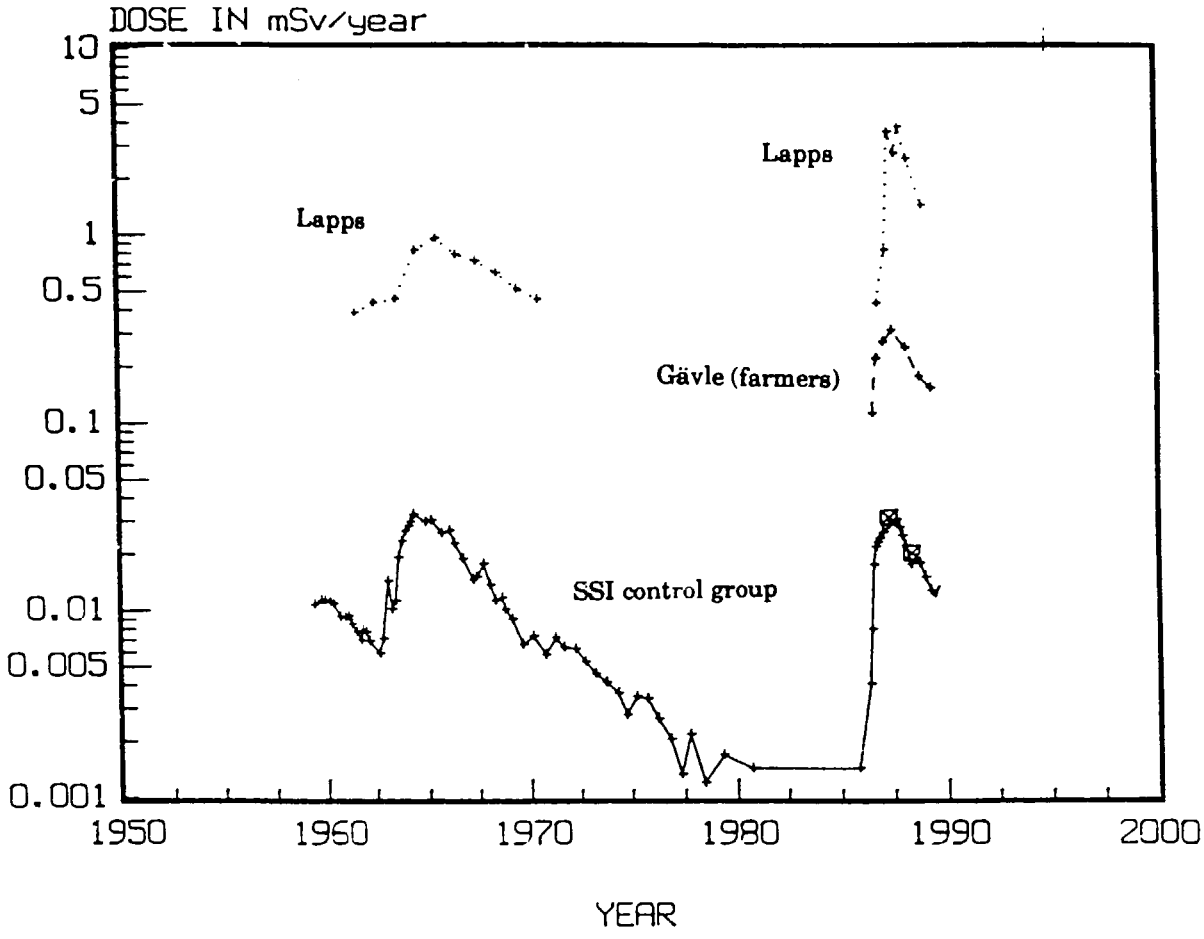

FIGURE 6. Calculated dose to man from whole body measurements of Cs-134 and Cs-137. The average doses for the Swedish population during the first and second year after the accident are indicated with ⊠.

3.5 Collective doses and risk estimates

The collective dose to the Swedish population (8.3 millions) has been estimated to be about 1300 mansievert the first year after the accident. The total collective dose over 50 years will probably be within the range 5000 to 7000 mansievert. As seen from Table 5, ground irradiation is the dominant source.

TABLE 5. Collective doses (manSv) in Sweden (8.3 millions) following the Chernobyl accident (ref. 3).

Exposure	0 - 1 year	0 - 50 year
Cloud irradiation and inhalation	100	100
Ground irradiation	1000	4000 - 6000
Food consumption	200	1000

The collective doses due to the Chernobyl accident can be compared to doses due to other radiation sources which will give about 40,000 mansievert to the Swedish population each year (ref. 3).

Using a risk factor of 0.02 fatal cancers per mansievert, the number of fatal cancers in Sweden due to the Chernobyl accident will be in the range of 100 to 200 fatal cancers over a period of 50 years or more. This figure could be compared with the total number of fatal cancers in Sweden which amounts to about 20,000 per year.

4. RADIOECOLOGICAL RESEARCH PROJECTS

During the first months, several investigations were carried out in order to characterise the fallout, *e.g.* nuclide composition (ref. 10, 14), chemical properties (ref. 15), *e.g.* for iodine, composition of hot particles (ref. 10), *etc.* Beside aerial mapping of the fallout, in-situ gamma spectrometric measurements have been carried out at several locations in Sweden in order to get more reliable dose estimates and to follow the retention of radionuclides in an urban environment (ref. 16, 17).

Most of the research projects, however, aim to get a better understanding of how radionuclides are distributed in the environment and which factors that are of importance for this distribution and the interaction with biota. This relation between organisms or groups of organisms to their radioactive environment we usually call *radioecology*. Before the Chernobyl accident, the knowledge of how radionuclides are distributed in the environment was mainly based on measurements of the fallout from the nuclear weapon tests, normal releases from nuclear power plants and reprocessing plants. The release from the Chernobyl reactor is unique in the way that radionuclides were introduced to the environment over an extremely short period of time, *i.e.* as a pulse. This also gave us a unique possibility to follow the transport of radionuclides in the environment, *i.e.* deposition pattern and factors that affect the deposition, migration in soil, uptake in plants, transport and accumulation in different foodchains. We also have a unique possibility to test and "validate" environmental transfer models.

The radioecology research that is carried out in Sweden, as a consequence of the Chernobyl accident, can be divided into projects that deal with *aquatic* transfer processes and *terrestrial* transfer processes.

Concerning the foodchains there are some basic differences between the aquatic and terrestrial environment. In the aquatic environment we often lack the firm management of the ecosystem, *e.g.* monocultures which are typical for the terrestrial environment. On land, the food chain often ends at a lower trophic level of the ecosystem, *e.g.* vegetables and herbivores. In water, on the other hand, the foodchain often ends with carnivores and as a consequence of this the foodchain becomes longer. This also leads to a more complicated follow-up of the critical pathways.

4.1 Aquatic projects

Radionuclides are present in different physico-chemical forms in water, *e.g.* simple ions, molecules, colloids and sorbed on inorganic or organic particles. The form they are present in affect the distribution pattern in the environment and the bioavailability.

The purpose of the studies is to get a better understanding of the long term behaviour of the radionuclide transfer in the aquatic ecosystem. This include both redistribution between biota, sediment and water as well as the dynamics involved. Figure 7 illustrates schematically the lapse for the concentration of radionuclides, or any other contaminant, in different trophic levels of an ecosystem in the aquatic environment. It can be said that the task is to quantify or determine the scales of the concentration and time axes. Attempts to make such a quantification have been made, however, this is not straightforward and will strongly depend on species involved, lake type, *etc.* (ref. 18, 19).

The lakes investigated are of different types and are located in areas with varying fallout. Some of the lakes are mountain lakes which are poor in nutritious matter and have "simple" foodchains. For many of the lakes data are collected regarding their topography, hydrology, temperature, conductivity, pH, TOC, tot-P, *etc.* (ref. 19-22). Data from one of the lakes are also used in an international study that aim to test models designed for the calculation of environmental transfer and bioaccumulation of radionuclides and other trace substances (ref. 23, 24). The project is called BIOMOVS (BIOspheric MOdel Validation Study) and was initiated by the SSI in 1986. At present 14 organisations from 12 countries have joined the study.

Investigations have also been made in the Baltic Sea, however to a much smaller extent. The caesium activity in sea fish is also considerably lower. Figure 8 summarises schematically the activity in fish living in mountain lakes and forest lakes situated in the fallout area and the Baltic Sea. Fish taken from closely situated lakes, however, can differ largely in specific activity. On the other hand, activity in fish outside affected areas is low.

Before the radionuclides are accumulated in fish they must pass through different trophic levels (*cf.* Figure 7). The main entrance of radionuclides is in the lowest trophic level, *e.g.* planktonic and benthic algae. This uptake occurs directly from the surrounding water, either via absorption or adsorption to the surface of *e.g.* algae (ref. 20, 25). Factors of importance are physico-chemical form of the sorbing nuclide and property of the sorbing cell surface, which

may vary significantly among species. A factor of great importance is also the surface-to-volume ratio (ref. 25), *e.g.* directly after the accident it was found that the specific activity in diatoms was considerably higher than in brown and green algae in the southeastern Bothnian Sea.

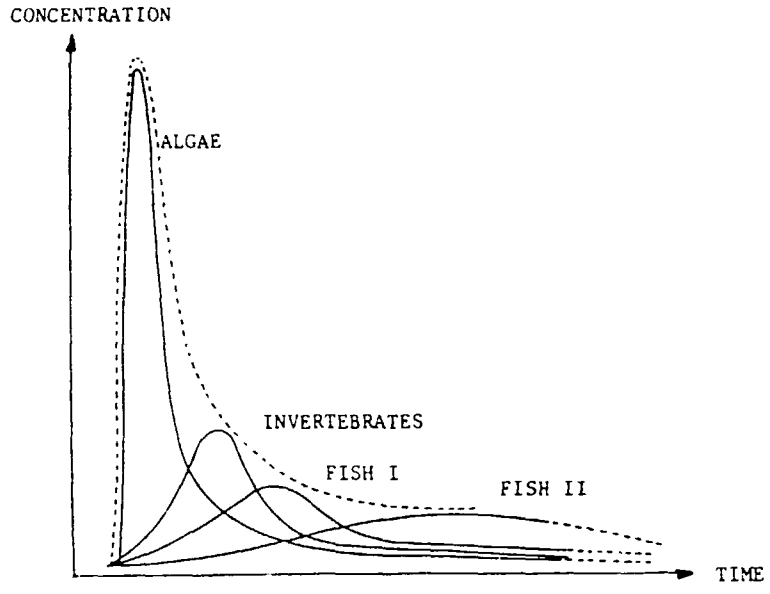

FIGURE 7. Changes in concentrations of radionuclides over time at various trophic levels of the aquatic ecosystem, an example.

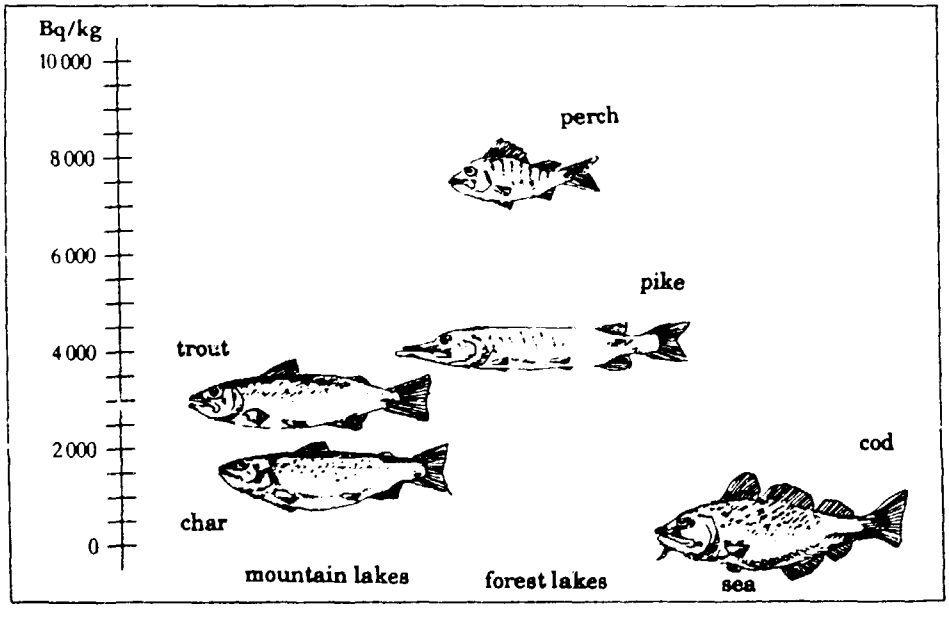

FIGURE 8. Schematic illustration of activity in fish living in contaminated mountain and forest lakes and fish living in the Baltic Sea.

Moving to higher trophic levels, from producers to consumers, the feeding habit of the consumer is an important factor. Sorption may still be of some importance, but is not the dominating accumulation mechanism. The move to higher trophic levels will in many cases lead to a discrimination of some radionuclides, especially when taking the step from invertebrates to fish (ref. 25).

When moving upwards in the trophic levels, the peak value for each level will be delayed in time (*cf.* Figure 7). Investigations in the Bothnian Sea (ref. 25) show that diatoms respond immediately to the fallout and with an uptake maximum about one month after exposure. It was also found that the peak value for perch occur about 200 days after exposure while it will take more than 500 days to reach the maximum concentration in pike. The incorporation of caesium in fishes as well as in other organisms has also been studied in some Swedish forest lakes (ref. 20). Similarly, the lowest trophic level responds immediately, *e.g.* planktonic algae. Bottom living animals and small fishes also respond immediately and the maximum concentration occur about two to four months later. Consumers of bottom animals, *e.g.* 10 to 20 cm perch, have a slower uptake and with a maximum concentration about one year after exposure. For bigger perch and pike the accumulation time is one to three years. It was also found that the size and the drainage area as well as the humic and nutrient content of the lake are of greater significance than the actual deposition of caesium.

The caesium activity in fish of weight 0.1 kg (perch, trout and char) exceeds 1500 Bq/kg *f.w.* in about 16,000 Swedish lakes. Measures for reducing the activity in lake fish are carried out by means of liming lakes, wet lands and whole drainage areas, fertilization of the waters, intensive fishing and addition of potash (ref. 19). A total of 41 lakes are under investigation. So far, however, there are no scientifically tenable evidences that the measures so far taken have a positive effect on the caesium content in fish.

4.2 Terrestrial projects

The terrestrial research projects are mainly directed toward three areas - agriculture, forest and reindeer.

The agricultural research is mainly carried out in the most heavily contaminated areas of C, X and Y county (*cf.* Figure 3). In the investigated areas the fallout occurred at or just before the start of the growing season. The purpose of the research is to follow caesium's transport in the agricultural landscape.

Shortly after the Chernobyl accident about 50 farms in the affected areas were chosen for sampling of pasture, cereal, *etc.* The aim of the study was to investigate how the uptake of caesium depends on soil types and farming practices and how the uptake will vary in time (ref. 26-28).

It was found that transfer from soil to grass is about ten times as high as the transfer from soil to grains. The transfer factors for both groups, however, vary considerably for each year and between the different farms. For grass depends the transfer factor largely on the interception capacity of the plant cover and the dilution by growth. In the second and third year after the accident the value of the transfer factor is reduced by a factor of two to about 10 as compared

to the first year. There is also a variation in root uptake between different soil types and in general is the root uptake from organic soils to grass and grain 5 to 10 times higher than the corresponding uptake from mineral soils (ref. 28). The uptake from sandy and loamy soils is normally larger than for mineral soils but smaller than for organic soils. Addition of potassium fertilizer, however, will reduce the caesium transfer from the soil to grass and grain (ref. 29).

The depth distribution of caesium (ref. 30, 31) and other radionuclides (ref. 31) in soil has also been investigated. The downward migration is slow and still after 18 months 80% of the deposited caesium is found in the upper 20 mm of soil that is rich in organic matter (ref. 30). Only a few percent is found below 50 mm. In most cases, however, the mean depth for various soils was found to be about 10 mm or less during the first 18 months after the accident. These figures refer to soils that not have been ploughed.

Another area of interest is the distribution of caesium in a forest ecosystem. In this ecosystem, the foodchain plant ⇒ elk (*Alces alces)* ⇒ man is of particular interest. This is because consumption of meat from elk, or other game, constitutes the major pathway of radioactive caesium to man in the northern part of Sweden. In order to estimate the resulting dose due to consumption of these food stuffs, knowledge of the long term behaviour of caesium in the forest ecosystem is necessary.

There is today not sufficient basic knowledge for a reliable identification of the dominant processes in a forest ecosystem. This is particularly true for processes that are dominant for longer time periods. A project dealing with water-soil, soil-plant and plant-herbivore interactions has therefore been initiated (ref. 32-36).

An extrapolation of the obtained data in time, say 20 to 30 years into the future, is not realistically since one can hardly claim that data obtained over a period of two to three years are representative for a period that is 10 times longer. An *indication* of expected caesium levels in the forest ecosystem for a 20 year period might, however, be obtained from the cumulative Cs-137 deposition originating from the nuclear weapon tests and the obtained concentrations before the Chernobyl accident. The results of such an exercise indicate that no significant changes from the present day situation are to be expected during the next decade(s) (ref. 33).

It was well known before the Chernobyl accident that radioactive fallout causes high levels of caesium in reindeers. The reason for this is that lichen, which is the most important forage plant for reindeers during the winter, accumulates and holds caesium for very long times. The time dependence of caesium in reindeer will thus be controlled by the time dependence of caesium in lichen. According to pre-Chernobyl data, the effective half-life of caesium in lichen is about 8 years and with a variation of about 3 years.

The research projects that have been initiated aim to get a better estimate of the effective half-life of caesium in lichen and to develop methods for reducing the caesium activity in reindeer meat.

One way to reduce the caesium content in reindeers is to add bentonite to the fodder. The method has shown to be promising, but a side effect is that the reindeer's water balance is

disturbed. If zeolites are used instead of bentonite, no side effects are observed. The effectiveness is, however, less than for bentonite. So far, zeolites have only been used in small scale experiments.

5. LONG TERM CONSEQUENCES

The most severe problem is foreseen in connection with the reindeer breeding in the northern part of Sweden. It is expected that high activities in reindeer meat remain over several decades unless any measures for improving the situation are taken. In general the caesium activity in reindeer meat varies between 1000 to 10,000 Bq/kg *f.w.* and is in most cases considerably higher than 1500 Bq/kg which is the limit for sales. Large economical losses must be taken by people dependent on reindeer breeding, especially Lapps, and this whole industry may be jeopardised. Also the Lappish culture may be at risk.

There are also indications that caesium will remain for a long time in the forest ecosystem, hence leading to increased caesium activities in game, *e.g.* elk and roe-deer.

Another long term problem is related to the high activity in lake fish. As already mentioned, about 16,000 lakes have ceasium activities in fish that exceed 1500 Bq/kg *f.w.* High activities will probably remain over several decades. It is expected that within a period of ten years the activity in lake fish will decrease with about 80%.

The number of cancers induced by the Chernobyl fallout will probably amount to 100 to 200 lethal cases over a period of 50 years or more. This can, however, not be demonstrated statistically since lethal cancers induced by other sources amounts to about 20,000 per year.

The Swedish National Board of Health and Welfare has made an extensive investigation concerning pre-natal effects after the Chernobyl accident. One paid attention to both genetic damage, *e.g.* development of Down's syndrome (mongolism), and direct damage to the fetus, *e.g.* defects and lower birth weights. About 140 children per year are born with Down's syndrome in Sweden and no increase has been observed after the accident. Likewise, no increase is expected to be detected in the future, however, investigations aiming to follow up the effects on man will continue.

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