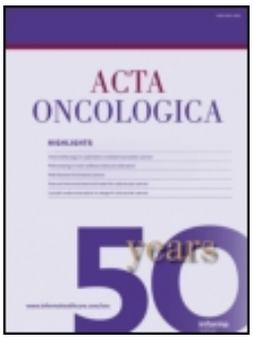

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The Chernobyl Accident: Transport of radionuclides to man living in northern Sweden

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THE CHERNOBYL ACCIDENT

Transport of radionuclides to man living in northern Sweden

L. OLOFSSON and H. SVENSSON

Abstract

The pathways of ^{131}I, ^{134}Cs and ^{137}Cs from the Chernobyl fall-out to man were followed in the county of Västerbotten, Sweden. Reported airplane measurements had shown that the ground deposition of ^{137}Cs was 3–40 kBq/m^2 with hot spots with more than 80 kBq/m^2. Multiplying with a factor of 0.6 gave the ^{134}Cs deposition and an approximate factor of 20 the ^{131}I ground deposition. The effective dose equivalent from ^{131}I became low, <0.1 mSv, as the cows were stabled. The ^{137}Cs activity concentration in different types of food was measured in approximately 8000 samples. The most important sources of Cs intake in man were lake fish, elk (European moose) and reindeer. Variations with time was studied in detail for four types of lake fish. Whole-body measurements on more than 250 persons showed that no group of people on average received more than 1 mSv from food during the first year after the Chernobyl accident. However, single persons eating large amounts of reindeer meat received up to 2.5 mSv. People buying all their food in ordinary provision-shops got less than 0.1 mSv from the food during the first year. The present level of ^{90}Sr activity concentration in man will only give an effective dose equivalent of 0.004 mSv/year, most of it being a result of the atmospheric nuclear bomb tests.

Key words: Chernobyl accident, fall-out in northern Sweden, ^{131}I, ^{137}Cs, whole-body measurements, radionuclide transports.

The fall-out from the Chernobyl accident was fairly high along the Baltic coast from Gävle to Umeå and also in the inland of the counties of Västerbotten, Västernorrland and Jämtland (Fig. 1). Some families in the inland part of this region to a very large extent live on meat from reindeer, elk (European moose), lake fish, and different types of berries. This food had the highest concentration of ^{137}Cs and ^{134}Cs. The problems for the Lapp (Sami) population after the atmospheric nuclear bomb tests were already pointed out by Lidén (13) and extensive research was carried out by his group (4, 14, 16, 20).

At the Radiation Physics Department in Umeå, extensive measurements were carried out already from the first week after the accident. The purpose of these measurements was primarily to secure that the ingestion of radionuclides were below acceptable limits. The number of measured samples was fairly large (approx. 8000) during the first year after the accident. The measurements were not planned for research purposes. However, we soon realized that these data had to be recorded in a way that would make the experience useful for the future. An extensive literature is already dealing with fall-out after the accident (cf. 30).

Material and Methods

Almost all of the measurements except for milk were performed in our low background laboratory in Umeå. The majority of milk measurements were performed in another laboratory.

Radioiodine in the thyroid gland. A 5×2″ NaI(Tl) detector was used at a constant distance from the neck. Using 10 min measuring time gave a standard deviation of about 40 Bq from counting statistics at low activities. A bottle with ^{131}I placed with its centre at 20 mm depth in a polystyrene neck phantom was used for calibration.

^{137}Cs in food. Most of the measurements were performed with a 5×3″ NaI(Tl) detector in a lead shielded measuring chamber. The samples were placed in a 100 ml plastic pot 2 mm above the detector. With 1 min measuring time the standard deviation was about 30 Bq/kg for a 300 Bq/kg sample. Corrections for differences in volume and weight were performed by interpolating in a measuring series with different volumes of ^{134}Cs and ^{137}Cs in water. The corrections could be up to 20%. Milk samples

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were measured in a 2×2″ NaI(Tl) well crystal (20×39 mm hole) using a fixed sample volume. With 20 min measuring time the standard deviation was 20 Bq/l at low activities.

The measurements were also controlled against a HPGE-detector (high purity germanium). As the geometry correction factor was even larger for this measuring system and it also demanded longer measuring times, it was not used in routine measurements. Comparisons with other measuring sites in Sweden has shown good agreement.

^{134}Cs and ^{137}Cs, dissolved in water, achieved from the National Institute of Radiation Protection, Stockholm, were used for calibrations.

^{137}Cs in man was measured with a coaxial HPGE-detector in a chair geometry. The measurement set-up was shielded with lead walls which were 5 cm thick and 150 cm high to even further reduce the background radiation in our low background laboratory. Also the floor was shielded with lead.

Although a HPGE-detector has a lower photopeak efficiency than a large NaI(Tl)crystal this is compensated as the compton scattered photons give a much lower contribution to the narrow (<2 keV) peaks. A large advantage with this type of detector is that peaks lying close in energy is easily resolved, which considerably simplifies the evalution of the spectra. The detector used gave a standard deviation from counting statistics of 87 Bq at a total body content of 3000 Bq. Calibrations were made with a phantom, consisting of polyethene cylinders filled with ^{137}Cs in water. The phantom was borrowed from the National Institute of Radiation Protection, Stockholm.

Intercomparisons were performed using persons travelling between different measuring centres (Table 5). The Umeå data seem to be somewhat low but most comparisons agreed within about 10%.

^{90}Sr in man. Caput femoris from 15 adults receiving hip-joint surgery were used to measure ^{90}Sr. The bone samples were ashed at 800°C and then grinded. The sample was then dissolved in boiling concentrated HCl. The chemical preparation of the sample then followed the method described by Suomola (29), using HDEHP to extract the yttrium from the solution. Finally the sample, in the form of yttrium oxalate was mounted on an aluminum disc and measured with a plastic scintillator, 2″×3/4″ with an aluminated mylar window (6.0 μg/cm^2). The prepared sample was measured overnight with an automatic read-out every 3 hours to ensure that the half-life of the sample was accurate. Counting statistics gave a standard deviation of 5 to 15%. Calibration was done with a ^{90}Sr solution from the National Institute of Radiation Protection.

Results

The pathways of radionuclides to man

Radioiodine. The radioactive cloud reached northern Sweden already about 2–3 days after the accident and included also radionuclides having a very short half-life. In the first period ^{132}Te (→^{132}I) (78 h and 2.3 h half-lives respectively) and ^{131}I (8 days half-life) were dominating. It was shown by the Institut für Strahlenschutz, Munich (7) that the exposure rate in Bavaria was dominated by deposition of ^{132}Te (→^{132}I) during the first week after the accident. The activity of ^{132}Te in air was also measured 3 to 5 days after the accident at different locations in Sweden (6, 26) and in Finland (28) but was lower than that of ^{131}I. Therefore, ^{131}I is of a much greater interest as it has much longer half-life.

The ^{131}I activity could be estimated from the ^{137}Cs activity as the deposition of the latter nuclide was surveyed from aircrafts over the whole of Sweden. The air activity of ^{131}I was measured only at a few locations in Sweden (31). Devell et al. (6) found a ^{131}I/^{137}Cs ratio in air filters of 5, measured 3–4 days after the accident. Only 20 to 25% of the iodine was fixed in these filters as most of the iodine was in gas phase. As ^{137}Cs is fixed on particles, the ratio in air should be about 20. Similar measurements were made by the National Defence Research Institute (FOA) (26) during about the same period. Assuming the same relation between gas and particle phase gives an activity ratio of about 15. The dry-deposition velocity on grass was about 5 times larger for ^{131}I than for ^{137}Cs (1). This may explain why Devell et al. (6) reported an activity ratio for ^{131}I/^{137}Cs of about 40 in grass samples. An extensive measuring programme was undertaken in Finland of the composition and activity in air, rainwater, snow and on ground (27, 28). It seems reasonable that the nuclide composition should not be too different from that in Sweden. The activity *concentration in air* of different nuclides was measured almost hour by hour in Helsinki (at Konala). During the first 2 days of fall-out (28th and 29th of April, 1986), when the major part of fall-out took place, a concentration ratio of ^{131}I/^{137}Cs of about 35 was found. This ratio increased the following days to about 40–50. (About 2 times higher values were found in single samples). The *deposition on the ground* of different radionuclides was determined at stations belonging to the Finnish nationwide monitoring programme. At most stations a ratio of ^{131}I/^{137}Cs lower than about 20 was determined. These results seem to be contradicting those by Devell at al. (6). However, a much larger part of the nuclides in Finland was deposited through rain. In Studsvik, Sweden, it was very little rain and therefore dry-deposition was important. For a very approximate estimate the ^{137}Cs deposition may therefore be multiplied by a factor of about 20 to receive the total ^{131}I deposition at the first days of May in the wet deposition area.

Iodine is transported very efficiently from grass → cows → milk → humans. Large regions in Sweden seemed to have been contaminated by 500 kBq/m^2 or more. (The factor 20 applied on the ^{137}Cs-map, Fig. 1.) This deposition could have given an activity in milk of 100 kBq/l and about 0.3 Gy in the thyroid to a 1-year-old child consum-

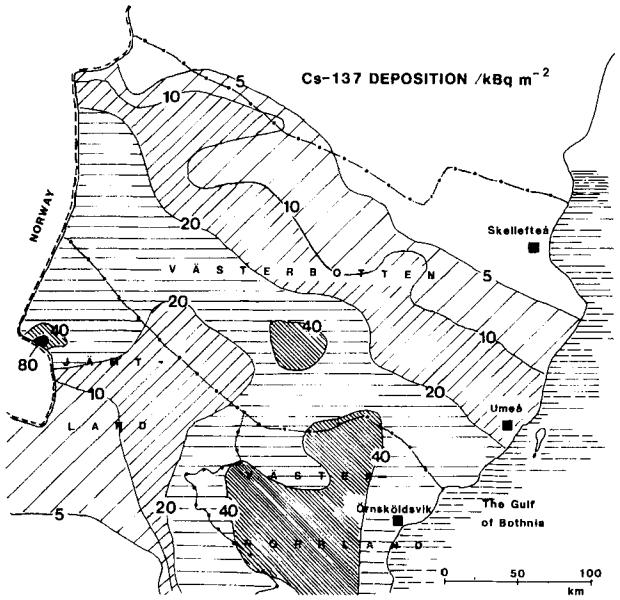

Fig. 1. ^{137}Cs ground deposition in Västerbotten with surroundings. The map is based on air measurements performed in May–October 1986 (31).

Table 1

Activity of ^{131}I in the thyroid gland re-calculated to the 29th of April 1986. The values are given +/− 1 SD of the mean. The maximum values and number of persons are also shown

	Ground deposition less than 10 kBq/m^2 of ^{137}Cs*	Ground deposition 10–40 kBq/m^2 of ^{137}Cs*
People who stayed indoors during the fall-out	0±40 Bq Max=200 Bq No.=8	130±20 Bq Max=800 Bq No.=39
People who worked outdoors	0 Bq Max=0 Bq No.=1	290±100 Bq Max=400 Bq No.=9
People who drank rain-water or melted snow after the fall-out	90±40 Bq Max=150 Bq No.=4	670±320 Bq Max=2400 Bq No.=5

* The ^{131}I deposition was about a factor 20 higher.

ing 1 l (30, 32). However, the cows in northern Sweden were stabled and it was possible to delay the grazing till the ground deposition was below the recommended value for Sweden, i.e. ≤10 kBq/m^2 (25).

It could not be excluded that some groups were directly exposed to ^{131}I from rain-water or from inhalation. Thyroid measurements were therefore carried out on 66 persons during the first month after the fall-out. All measurements were re-calculated to the 29th of April 1986, using an effective half-life of 7.6 days of ^{131}I (8). The population was divided into different groups. Persons with indoor jobs and living in an area with a ground deposition less than 10 kBq/m^2 ^{137}Cs (i.e. less than about 200 kBq/m^2 ^{131}I) had the lowest exposure. The highest values were found in persons who drank rain-water or melted snow after the fall-out (Table 1). The dose levels were, however, low and the maximum measured activity of ^{131}I in the thyroid was 2400 Bq, corresponding to about 3.7 mGy (22) and the mean value was less than 1/10 of that. In order to estimate the effective dose equivalent (in mSv) these factors have to be multiplied by a factor of 0.03 (9).

^{137}Cs and ^{134}Cs. The accumulated deposition of ^{137}Cs (from the nuclear bomb tests) at the end of the 1960's reached about 2 to 3 kBq/m^2 (17). The deposition in Sweden was fairly uniform and it contained almost no ^{134}Cs (19). After the Chernobyl accident the deposition in northern Sweden became very non-uniform, e.g. about 15 kBq/m^2 along the Umeå river, about 30 kBq/m^2 50 km south of the river and about 5 kBq/m^2 80 km north of the river. Hot spots of about 80 kBq/m^2 were found (Fig. 1). All references to ^{137}Cs deposition in this report refers to the aircraft measurement performed by the Swedish Geological Company (SGAB) (31). This gives the equivalent plane surface deposition as an approximation of the true deposition. The deposition also contained about 60% ^{134}Cs (activity ratio). The values were directly related to the amount of rain during the period, when the radioactive cloud passed.

The deposition gradient in the region made it possible to determine the contamination in wild animals and humans as a function of the deposition. Danell et al. (5) found that during the first year after the accident (May 1986 – May 1987) the activity concentration in elk meat, A (Bq/kg), was roughly a linear function of the deposition, d (Bq/m^2), i.e. $A \approx k \cdot d$. For grown-up elks $k \approx 0.013$ (m^2 kg^{-1}) and for calves $k \approx 0.02$ (m^2 kg^{-1}). (The relation is estimated from Fig. 2 in their report.)

The activity in fish (whole fish without guts) from a large number of lakes in the inland was measured (Table 2). The main part of the fish was caught in regions having a deposition between 20 and 40 kBq/m^2. The mean activity concentrations in different types of fish were 1.5 to 4.3 kBq/kg during the first year after the accident. Also the seasonal variations were studied (Figs 2a and b). As the material from a single lake was to small, data has been pooled from several lakes. The activity concentration in fish can vary substantially between different lakes why a weighting procedure was used, in the case of char-fish and salmon trout, to reduce the spread in the data. The activity concentration in the fish in a lake was for each month compared to a reference lake and a mean of these quota, taken over time, was then used as the weighting factor of the lake. The method could only be used when the number of samples from each lake was large enough to give a meaningful comparison between lakes why the data for pike and perch were used without this weighting procedure.

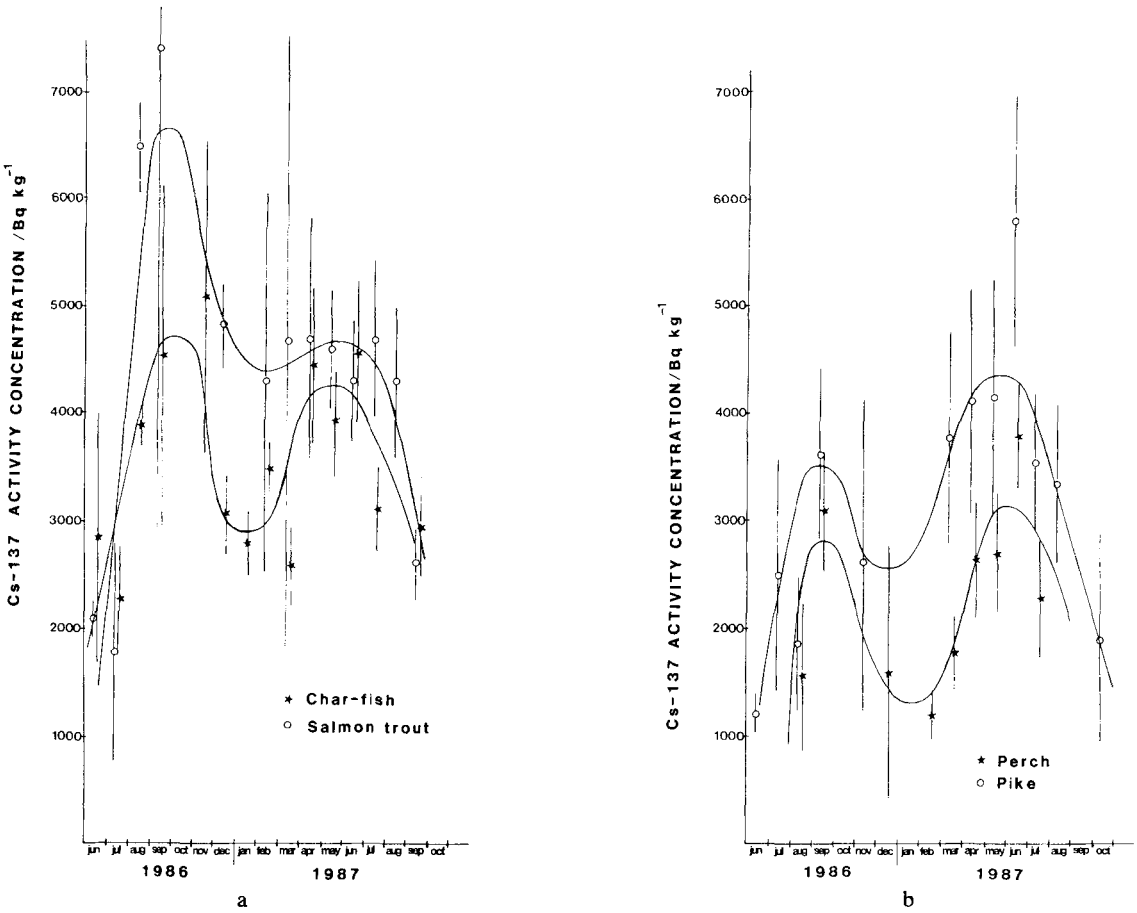

Fig. 2. ^{137}Cs activity concentration in char-fish and salmon trout in an area with ground deposition of 10–40 kBq/m^2. A total of 91 char-fish in 7 lakes and 40 salmon trouts in 6 lakes were used. One standard deviation of the mean is shown, and b) ^{137}Cs activity concentration in pike and perch in an area with ground deposition 10–40 kBq/m^2. A total of 73 pikes in 45 lakes and 114 perches in 74 lakes were used. One standard deviation of the mean is shown.

For the northern areas the lowest value should have been found during the third quarter of the year if ^{137}Cs was available at a constant level in the lake (4). Instead, during 1986 the concentration continued to build up during summer and autumn with a maximum value in September–October. A second maximum was reached in May–June 1987 and the activity then decreased fairly rapidly during late summer and autumn 1987. As the spread in the data is quite large it is difficult to determine the elimination rate of ^{137}Cs in fish. However, it seems to be higher in char-fish and salmon trout than in perch and pike. The activity concentration in char-fish and salmon trout in September 1987 was about half the level in September 1986 while the data of ^{137}Cs in perch and pike seems to be consistent with the half-time (2.1–2.2 years for pike) reported by Carlsson (4) after the fall-out during the 1960's.

^{137}Cs activity concentrations were measured in milk collected from different parts of the region. Generally, one lorry ('line') takes milk from several farms, and the milk from these farms are mixed. Fifty-six of these lines passed farms situated in an area with fairly uniform ^{137}Cs deposition, why the activity concentrations could be related to the deposition. During the summer 1986 (June–September) the ground deposition in kBq/m^2 roughly corresponded to the milk activity concentration in Bq/l (Table 3). The activity concentration remained almost constant in the winter 1986/1987 and then decreased by 20% during the summer 1987. The concentration in the dairy

Table 2

Activity in lake fish from September 1986 to August 1987 in the 'Sami region' of the county of Västerbotten, Sweden. A large part of the samples were taken during the summer 1987

Type	Activity in lake fish (Bq/kg)		
	Mean	Max	No.
Perch	4 300	28 000	374
Pike	2 000	8 300	254
Char	2 700	18 000	302
Whitefish	1 600	9 300	157
Salmon trout	3 300	25 000	235

Table 3

Relation between ground deposition of ^{137}Cs and the activity concentration in milk. Inside the brackets the number of milk collecting 'lines' (see text) are given. Each line was measured 9 times during the period June 1986 to September 1987

Deposition kBq/m^2	Activity in milk Bq/l	
0–10	11	(35)
10–20	21	(16)
20–30	28	(5)

Table 4

Activity concentration of ^{137}Cs for the most used berries and vegetables in the region (Bq/kg). The ^{137}Cs deposition was generally between 10–40 kBq/m^2. The number of samples is given in parenthesis

Time Parameter	Bil-berry	Rasp-berry	Cloud-berry	Lingon	Vege-tables
1986					
mean	60 (42)	78 (48)	205 (140)	83 (29)	21 (56)
max	172	347	2 272	324	207
1987					
mean	70 (52)	32 (13)	298 (47)	89 (13)	7 (15)
max	205	105	746	220	26

Table 5

Whole-body content of ^{137}Cs (kBq). A comparison between different measuring centres

Person	Length cm	Weight kg	Umeå	Stock-holm	Göte-borg	Lund	Malmö	Risø Denmark
A	190	75	1.65		1.81			
B	182	91	0.61		0.77			
C	190	86	0.73		1.07			
C	190	86	1.44	1.63				
D	180	66	1.74	1.80				
E	171	65	1.79	1.91				
F	172	55	0.86, 0.70[2]	0.73		0.74	0.77	0.71[1]
G	190	91	2.92	3.14				

[1] Recalculated from $^{137}Cs+^{134}Cs$ (=1.01 kBq).
[2] Measurements before and after the journey.

milk decreased during the same period by 40%. Large variations in activity concentrations were found in the milk from neighbouring farms due to different types of animal feeding. Activity concentrations in milk from individual farms were measured if the value from a certain line was unexpectedly high.

The activity in dairy milk from the region was on average roughly 20 Bq/l during the first year after the accident. Assuming a consumption of 0.6 l/day, this would in Umeå have given an effective dose equivalent of 0.1 mSv in a year in the average person, using metabolic data from ICRP 30 (10) and the calculations of Snyder et al. (22).

The activity concentrations in the most important berries in the region were also measured (Table 4). Cloudberries contained much higher activities than the other berries. Cloudberries are to a large extent eaten by the Sami population who also have a high intake of reindeer meat. Bilberries and lingonberries had about the same activity level in 1987 as in 1986, while the activity in cloudberries seemed to be somewhat higher in the second year. The increase was about 20% after corrections for sampling from different regions.

In the 1960's it was shown that a ground deposition of about 2000 Bq/m^2 gave an activity concentration in reindeer meat of about 3000 Bq/kg as a maximum during the winter. The minimum values during the summer were about 1/4 of the maximum. In some parts of the region, traditionally used by the Lapps, the deposition after the Chernobyl accident was 5 to 30 times higher than during the 1960's, which thus should have resulted in a very high activity concentration. However, different counter-measures were undertaken why only a few ^{137}Cs-activity concentration values over about 10 kBq/kg were found (2). Unfortunately it is difficult to estimate the mean from published data but a crude estimate gives about 5 kBq/kg in reindeer slaughtered in the autumn 1986 (2) and 0.5 kBq/kg during the same period 1987 (23). This large decrease is explained by the extensive counter-measures performed, e.g. moving reindeers to areas with low contamination or taking lichen from such areas to feed the reindeers before slaughtering.

The whole-body content of ^{137}Cs in people from the region was measured during the period July 1986 to October 1987. Mainly 3 different types of food contributed considerably to the concentration of ^{137}Cs in humans, namely lake fish, and meat from elks (European moose) and reindeers. Other types of food had activity concentrations of the same magnitude (e.g. cloudberries) but these

Table 6

Grouping of people according to their eating habits and the ^{137}Cs deposition in the area where they gather their food

Group No.		No. of persons
1	Elk (European moose), lake fish and reindeer less than once a month	43
2	Elk and lake fish more than once a month but reindeer less than once a week	54
3	Elk and lake fish more than once a month and reindeer more than once a week Less than 10 kBq/m^2 of ^{137}Cs	39
4	Elk and lake fish more than once a month and reindeer more than once a week 10–40 kBq/m^2 of ^{137}Cs	87

Table 7

Activity concentration ratio females/males for ^{137}Cs in total body. The spread in the ratio, between different time periods, is calculated and ±1 SD of the mean ratio is given

Group	Females/males
1	0.6±0.1
2	0.8±0.2
3	0.5±0.2
4	0.5±0.1

are not eaten in the same quantities. Reindeer meat can contribute with both large quantities and high concentrations of ^{137}Cs, especially among the Sami people (12). However, also other persons than the Sami people consume considerable amounts of reindeer meat. The material was therefore divided into 4 different groups in relation to the consumption of ^{137}Cs-containing food, group 1 having the lowest intake and group 4 the highest (Table 6).

After the A-bomb tests in the 1960's it was shown that the activity concentration in males was about twice as high as in females (14). Similar observations were made in the present study (Table 7). Therefore, the data were analysed separately for the two sexes and a weighted mean was then calculated, corresponding to an equal number of males and females.

In Fig. 3 the ^{137}Cs activity divided by the total body mass as a function of time is shown for the 4 groups. Groups 1 and 2 had an activity concentration slowly varying with time while in group 3 the time variations is very uncertain. In group 4 a sharp rise was observed in March 1987, probably mainly due to the fact that people after the first year became less concerned about the activity in food. Part of the explanation could also be the fact that reindeers slaughtered in the spring have a higher ^{137}Cs concentration because they eat more lichen in the winter. In June 1987 a more constant value was reached. This can be understood by observing individuals measured at several occasions (Fig. 4). It seems obvious that most persons with a high activity concentration changed back to a more cautious behaviour after information of the measurement result. Also here, the seasonal variations in reindeer meat plays a role. Contrarily, persons with low activity concentrations probably tended to increase their intake of contaminated food. This makes it difficult to foresee the levels in the future. However, the mean activity concentration in reindeer meat from the slaughter in August 1987 is expected to give about the same level of concentration as is now present in group 4, but if the level is to be kept low then the present large efforts to reduce the ^{137}Cs in reindeer meat must continue for a long period

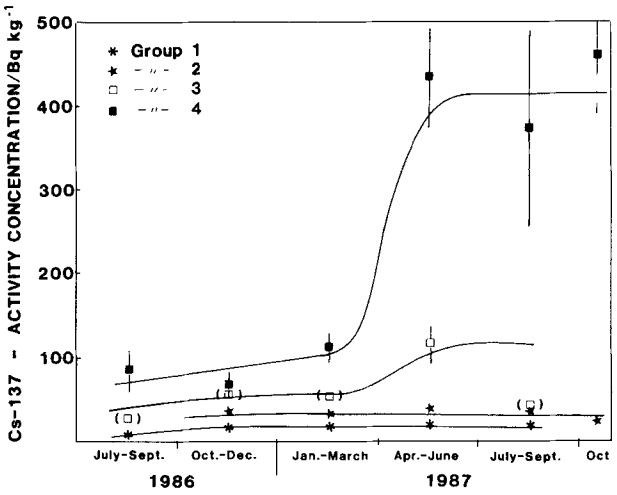

Fig. 3. ^{137}Cs activity in total body divided by body weight. The mean value during each period of time is presented. Group 1 lowest intake—group 4 highest intake of ^{137}Cs in food (see Table 6). One standard deviation of the mean is shown. Markers in parenthesis correspond to a few measurements only.

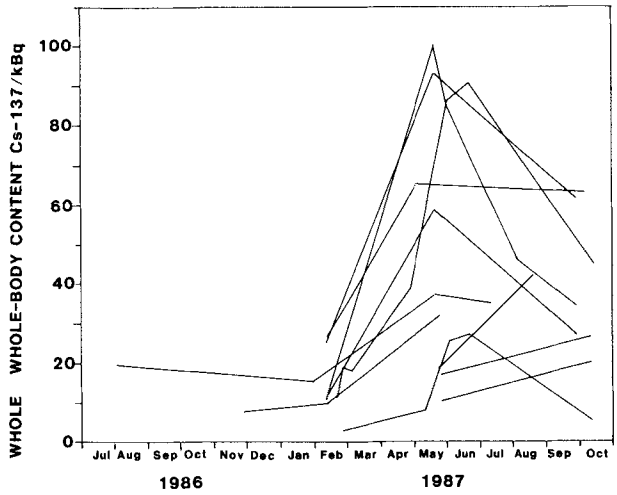

Fig. 4. Whole-body content of ^{137}Cs for some of the individuals in group 4, measured at several occasions.

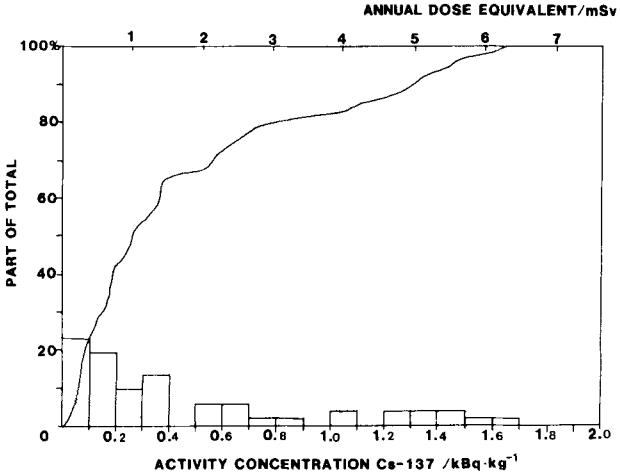

Fig. 5. ^{137}Cs activity concentration in April–June 1987 in group 4. Both the integral and differential distribution are shown. The annual dose equivalent is calculated including the dose from ^{134}Cs.

Table 8

Dose equivalent (mSv) per year in different groups (see Table 6) from whole-body ^{137}Cs and ^{134}Cs. The maximum dose equivalent is given in parenthesis

Time-period	Group 1	Group 2	Group 3	Group 4
July 1986–June 1987	0.07 (0.1)	0.14 (0.4)	0.26 (1.0)	0.70 (2.5)
July 1987–June 1988	0.07 (0.1)	0.2 (0.4)	0.4 (0.7)	2 (7)
1987–2037			0.07 (0.4)	0.3 (1)

of time. It is interesting to notice that the ^{137}Cs activity concentration in the Sami population is now on about the same level as in the 1960's (14).

Finally, Fig. 5 shows the spread in the activity concentrations of ^{137}Cs for people with high intake (group 4) from April to June 1987. More than 50% of the measurements are below 0.3 kBq/kg but there is quite a long high activity tail in the distribution reaching 1.65 kBq/kg.

As the ^{134}Cs activity was not determined during the whole-body measurements the ^{134}Cs/^{137}Cs activity fraction had to be estimated in the dose calculations. In the following calculations a value of 0.43 (^{134}Cs/^{137}Cs) is used. This value was determined on the 25th of May 1987 in a sample of rain-water taken during the fall-out. It has been confirmed that this sample was representative of the ^{134}Cs/^{137}Cs ratio in this region both by measurements on meat and on man to within 10%. Lower activity ratios will of course be found in samples having a large contribution of ^{137}Cs existing before the Chernobyl fall-out. As ^{134}Cs decays faster than ^{137}Cs this fraction will decrease with time (t in days) by

$$\frac{A134}{A137}(t) = 0.43 \cdot \exp\left[\frac{-\ln 2 \cdot t}{810}\right]$$

This expression is integrated to give mean values over longer time periods. Caesium can be assumed to be almost uniformly distributed in the body. This means that the dose equivalent and the effective dose equivalent will be the same. In order to calculate the absorbed dose in the body, conversion factors given by Snyder et al. (22) were used, i.e. $5.7 \cdot 10^{-6}$ mGy/kBqh for ^{134}Cs and $3.8 \cdot 10^{-6}$ mGy/kBqh for ^{137}Cs. No correction for the differences in body weight have been used in these calculations.

In Table 8 the average dose equivalents from whole-body ^{137}Cs and ^{134}Cs in the 4 different groups of ^{137}Cs intake (Table 6) are shown. During the first year (July 1986 – June 1987) the dose equivalent was calculated from the measured mean activity concentrations of ^{137}Cs. All groups had mean values below 1 mSv but some individuals could have received effective dose equivalents up to 2.5 mSv. The estimated dose equivalent from milk, i.e. 0.1 mSv/year, seems to cover all of the dose equivalents estimated for group 1 in Table 8. This is probably partly due to the very conservative biological half-time (110 days) used for caesium (10) which probably has a much shorter real biological half-time (15). Anyhow, milk was an important source of ^{137}Cs intake in this group. For the second year a constant level of ^{137}Cs, equal to that of the end of the year before, was assumed. In group 4 (highest intake), this will give an increased dose equivalent relative to the first year and a mean of 2 mSv/year was estimated. It must be stressed that changes in consuming habits could give a new increase in the whole-body activity. It is therefore important that the measurements be continued.

If the effective dose equivalent is averaged over a 50-year period it will, even in group 4, be below 1 mSv/year as a mean. Yet a very pessimistic biological half-life of 15 years in food (mostly reindeer) has been assumed. Mattsson (17) gives a value of 8–14 years in undisturbed lichen and 5–6 years in grazed areas.

^{90}Sr-contamination. The activity release of ^{90}Sr at the Chernobyl accident was about 20% of that of ^{137}Cs (33). Measurements in Finland (28) indicated that the deposition of ^{90}Sr was about 1 to 10% of that of ^{137}Cs. Therefore, the ^{90}Sr contamination should not be very high in humans. Measurements on 15 adults showed that the mean ^{90}Sr content was 0.05±0.01 Bq/g Ca in caput femoris (one standard deviation for a single measurement is given). This value can be compared with results from 1961–1963 by Bengtsson (3) who found about 0.02 Bq/g Ca from 14 femur samples, and from 1966 by Salo & Uotila (21) who found a mean value of 0.06 Bq/g Ca. In Denmark about 0.03 Bq/g Ca was found in 1980, and in USSR 0.06 Bq/g Ca in 1978 (see compilation in UNSCEAR (32)). All these data concern adults. It can be concluded that the mean value (0.05 Bq/g Ca) is in the high end of the reported data if one assumes that the ^{90}Sr only originates from the bomb tests in the 1960's. There ought to have been a slow decrease in the ^{90}Sr content since the literature values were measured (20).

The absorbed dose from ^{90}Sr will be very low and a continouos level of 0.05 Bq/g Ca would give approximately 0.02 mGy/year to the red bone marrow and 0.04 mGy/year to the bone lining cells (32). This gives an effective dose equivalent of only 0.004 mSv/year (9).

Discussion and Conclusions

It is well known that ^{131}I is a very critical radioisotope in contamination directly after a reactor accident (24). However, it was remarkable that fairly large areas in Sweden, at a distance of 1400 to 1500 km from Chernobyl, had a contamination as high as about 500 kBq/m^2. Grazing of cows is in Sweden not recommended above 10 kBq/m^2 (25). This was a minor problem as the cows in the largest contaminated area still were stabled at the end of April 1986. Their grazing was then delayed, in some regions till the end of June. The problem seemed to have been greater in central Europe, even if the ^{131}I deposition generally was below 100 kBq/m^2, as the cows were grazing at the time of the accident. Different counter-measures were undertaken (18). Thyroid measurements on people in Västerbotten, a county in northern Sweden, showed that the effective dose equivalent from ^{131}I also in the heaviest exposed group was very low, the maximum determined value being 0.1 mSv.

The ^{137}Cs deposition in the studied region was about 5 to 20 times larger than the total deposition from A-bombs. In spite of this, the mean whole-body activity of ^{137}Cs in the heaviest contaminated group in the 'Sami-region' was about the same as in 1965 (14). During the first year, July 1986 – June 1987, the mean effective dose equivalent in the most exposed group was 0.7 mSv, from ^{137}Cs and ^{134}Cs in food, with a highest individual value of 2.5 mSv. The prognosis for next year (1988) is about 3 times these values as more and more of the Sami population will probably go back to their usual food habits. The effective dose equivalent in people having an ordinary intake of 'wild products' was during 1986/1987 about 0.1 mSv, and a slow decline can be expected.

The contamination with ^{90}Sr from the Chernobyl accident was insignificant. An effective dose equivalent of less than 0.004 mSv/year was determined. Most (or almost all) of the ^{90}Sr activity in the region still derives from the A-bomb fall-out.

From all the sources of *ingestion*, the mean effective dose equivalent during 1986/1987 should be about 0.1 mSv with values of a few mSv for the heaviest exposed individuals. The *external* irradiation from the fall-out, at a ^{137}Cs deposition level of 10–30 kBq/m^2 (typical in the region) caused about 0.6 mSv during the first year after the accident and the effective dose equivalent during 50 years can be assessed at about 3 mSv (0.06 mSv/year) (11). The external irradiation thus gives the largest dose contribution from Chernobyl to most people (groups 1 and 2 in Table 6).

The effective dose equivalent 1986/1987 from both internal and external radiation for the population in Västerbotten can, as a mean, be estimated at about 0.7 mSv/year. Very few persons received more than 2 to 3 mSv. This should be compared with the natural background irradiation (from external irradiation, radon in houses, medicine) which as a mean gives an effective dose equivalent of 6 mSv/year. The value can be more than 10 times higher for people living in 'radon-houses' (30).

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