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AREAL AND DEPTH DISTRIBUTION OF RADIOCAESIUM IN ESTONIAN NATURAL SOILS

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Abstract. Areal and depth distribution of radiocaesium in Estonian natural (uncultivated) soils in 1991—1992 was studied using laboratory low-background gamma spectroscopy. The average mass and linear relaxation length values for ^{134}Cs and ^{137}Cs were determined from the analysis of about 25 soil core profiles from 110 sampling sites throughout Estonia. The average mass relaxation length values, ϱ/α, are 42 ± 15 kg·m^{-2} for ^{134}Cs from the Chernobyl fallout and 110 ± 30 kg·m^{-2} for ^{137}Cs from the atmospheric nuclear weapons test fallout. A sublinear dependence for the mass relaxation length on time, $\varrho/\alpha \sim t^{0.57}$, was found. The radiocaesium concentration inside a soil profile was found to be proportional to the relative drying loss of the soil sample, while its in-layer activity correlates reasonably well with the inverse density of the corresponding soil layer at a certain depth below the surface. Both these findings seem to support the assumption that the amount of radiocaesium and the content of organic matter are proportional along a soil profile. Two different average linear relaxation length values, α^{-1}, 0.015 ± 0.004 m and 0.046 ± 0.007 m, with about equal probability of occurrence were found for ^{134}Cs from the Chernobyl fallout, while only one value, 0.10 ± 0.01 m, is characteristic of an older ^{137}Cs deposition from the nuclear weapons test fallout.

The areal distribution of the Chernobyl radiocaesium deposition is extremely uneven. About 60% of the total (integrated) fallout, namely 50 TBq of ^{134}Cs and 90 TBq of ^{137}Cs, was deposited in 2 out of the 15 Estonian counties (Lääne-Virumaa and Ida-Virumaa). The highest deposition at Narva-Jõesuu (Ida-Virumaa) reached about 40 kBq·m^{-2} (^{137}Cs) in 1986, while the country-wide mean deposition was estimated to be 1.9 kBq·m^{-2}. So, on the average the Chernobyl fallout in Estonia approximately doubles the ^{137}Cs mean deposition from the nuclear weapons tests (2.2 kBq·m^{-2} in 1986). A map showing the areal distribution of ^{134}Cs (Chernobyl) deposition in Estonian soils is presented.

Key words: radiocaesium, mass and linear relaxation length, low-background gamma spectroscopy, areal distribution, Estonian soils, atmospheric nuclear weapons test fallout, Chernobyl fallout.

INTRODUCTION

Radiocaesium is one of the major artificial radioactive pollutants in soils. Because of its relatively long half-life and its incorporation into ecosystems, radiocaesium persists in the environment for several decades, contributing to the dose rate through both external and internal irradiation. The radiocaesium contamination of soil originates from two global fallouts of different character and age: atmospheric nuclear weapons tests (NWT) and the Chernobyl-4 reactor accident (CRA). NWT carried out during 1945—80 released about 10^{18} Bq of (fission) ^{137}Cs, the areal distribution

of which was relatively even (2.9 kBq · m^{-2}) over the Northern Hemisphere (UN, 1982; UN, 1988; OECD, 1987). In contrast, the fallout from CRA in April—May 1986, consisting of ^{134}Cs ($0.55 \cdot 10^{17}$ Bq) and ^{137}Cs (10^{17} Bq), was extremely inhomogeneously distributed mainly over Europe and Asia (OECD, 1987; Anspaugh et al., 1988). The presence of the "light" radiocaesium ^{134}Cs ($t_{1/2}=2.06$ yr) in the CRA fallout with the $^{134}Cs/^{137}Cs$ activity ratio of 0.55 ± 0.05 (Devell et al., 1986) enables to distinguish between the two contributions.

Data on the radiocaesium contamination in Estonian environment, incl. soil, are scarce. The accumulation of ^{137}Cs in the indicator species, such as mushrooms (Parmasto & Liiva, 1988) and microlichens, and a few soil samples (Martin et al., 1991) has been studied. Only recently some results of an air-borne ^{137}Cs scanning for part of Estonia were published (Saar, 1991).

This paper presents the results of a study of the areal and depth distribution of radiocaesium (^{134}Cs and ^{137}Cs) in soil samples collected at more than 110 different locations throughout Estonia in 1991—92. This study is part of a project of investigating the radionuclide contents in the near-surface soil and the corresponding gamma dose rate levels in Estonia. The main objectives were (a) the assessment of the Chernobyl radiocaesium contamination of soil in different parts of Estonia, (b) the investigation of depth distributions of radiocaesium in natural (uncultivated) soils, and (c) a search for correlations between radiocaesium concentrations and some properties of soil.

Using the laboratory gamma spectroscopic analysis the integral deposition of both radiocaesium isotopes in about 27 soil cores and their concentration depth distributions were determined to the depth of 15—20 cm. The corresponding relaxation lengths were calculated assuming an exponential depth distribution. The derived mean values were used to evaluate the integral depositions in the samples for which only the uppermost core section (0—3 cm) was analysed. A map showing the areal distribution of radiocaesium from the CRA fallout in Estonia is presented.

MATERIAL AND METHODS

Soil samples

The sampling sites were preselected on the map to cover the territory of Estonia with a relatively uniform grid. As a rule, the samples were collected in natural (uncultivated) grass-covered flat areas without or with minimal sheltering vegetation (trees, bushes, etc.) at least 20 m from the roads. The sites with a possible wind or water erosion, flooded areas, and soils rich in stones were avoided. The average distance between the neighbouring sampling sites was about 25 km. Additional samples were collected in Ingerimaa (Ingria) and Petserimaa regions (at present under Russian subordination).

At each site a single soil core (for a few sites, 2 cores) to the depth of 15—20 cm was collected by using a coring tool with a 9 cm diameter. At the site the core was pressed out of the tool into a plastic bag of a matching diameter. At the laboratory the cores were cut into 3 cm thick sections, dried in an oven at 105 °C for 12—24 hours, homogenized in a mortar, and put into cylindrical 105 cm^3 beakers. The bulk densities of all dried samples were determined. For sections of several cores the drying weight losses were also calculated. The hermetically sealed beakers were stored for a minimum of three weeks to ensure an equilibrium of radon and its daughter isotopes before a gamma-spectroscopic analysis.

Gamma spectrometry

A low-background gamma spectrometer consisting of a Ge(Li) detector with a resolution of 2.5 keV and an efficiency of 7% at 1.33 MeV placed into a 2 mm Cu-lined 10 cm thick lead housing was used. The detector signals were amplified by a spectrometric linear amplifier, analysed by a 8 K pulse height converter, and stored in a 4 K multichannel analyser coupled to an AT computer. The energy and efficiency scales of the spectrometer were calibrated by using the IAEA standard powder sources (U, Th, K, soil, whey powder) and the "Multigamma" solution (K.-L. Sjöblom, Finnish Centre for Radiation and Nuclear Safety) in the measurement beakers. The background of the spectrometer was checked periodically. The measurement time varied in the range from 10 to 72 hours. The spectrum analysis was performed by using the GAMMA-83 gamma spectrum analysis software (Sinkko, 1981; Sinkko & Aaltonen, 1985) and, partly, a corresponding software prepared in the Institute of Physics of the Estonian Academy of Sciences. The concentrations of the radiocaesium isotopes ^{134}Cs and ^{137}Cs as well as the natural radionuclides ^{40}K, ^{208}Tl, ^{214}Bi/Pb, ^{226}Ra, ^{228}Th/Ac, ^{235}U in the soil samples were determined. The results concerning natural radionuclides will be discussed elsewhere.

RESULTS AND DISCUSSION

Depth distribution of radiocaesium

Relaxation lengths. It is typical to assume that after some time the depth dependence of the fallout radionuclide concentration in soil approaches an exponential distribution (Beck et al., 1972): $S(z)=S(0)\exp[-(\alpha/\varrho)\varrho z]$, where $S(0)$ is the radionuclide surface concentration in Bq·kg^{-1}, $S(z)$ is the concentration at the depth z, and α/ϱ is the reciprocal of the relaxation length in m^2·kg^{-1}. The total (integrated) fallout radionuclide activity per unit soil surface area, S_A, is given by the expression $S_A=S(0)/(\alpha/\varrho)$. From the last expression it follows that the mass relaxation length, α/ϱ, can be calculated using the expression $\varrho/\alpha=S_A/S(0)$.

In the present study the S_A value has been calculated by adding the radiocaesium activities of all the measured sections in the profile and the extrapolated contribution from deeper layers and by dividing the result by the cross-section area of the core. Instead of the values $S(0)$ and ϱ/α, the concentration of the uppermost section, S_0, and the ratio $R=S_A/S_0$, respectively, have been used. For a number of the profiles analysed the depth distribution of the radiocaesium concentration deviates from the expected exponential distribution, e.g. fallouts of two different ages for ^{137}Cs, near-surface "anomalies" (subsurface maxima) in the case of ^{134}Cs. At the same time the analysis of the 3 cm sections gives a relatively rough approximation for the true radiocaesium depth distribution from the CRA fallout. For these reasons the ratio R can be considered only as an average mass relaxation length.

Considerable site-to-site variations in the soil type, its density, organic matter and moisture content, as well as in the relative amount of radiocaesium isotopes were observed. As a result, for the 27 profiles analysed, collected at the sites shown in Fig. 1, the evaluated R values vary in a wide range from 15 to 86 kg·m^{-2} for ^{134}Cs and from 16 to 128 kg·m^{-2} for ^{137}Cs.

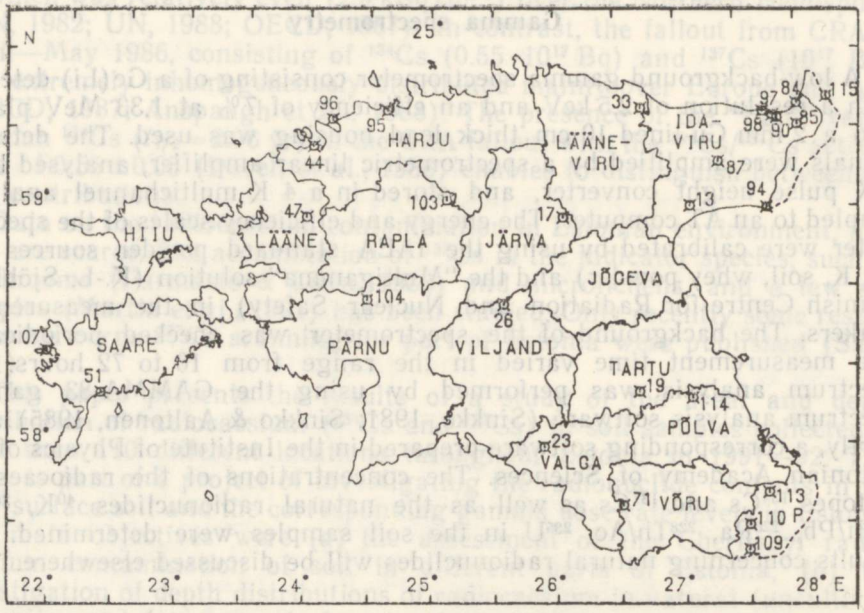

Fig. 1. Sampling sites in Estonia for which the depth distributions of radiocaesium in soil cores were analysed. The present names and borders (solid line) of Estonian counties (*maakond*) are given. P—Petserimaa, I—Ingerimaa.

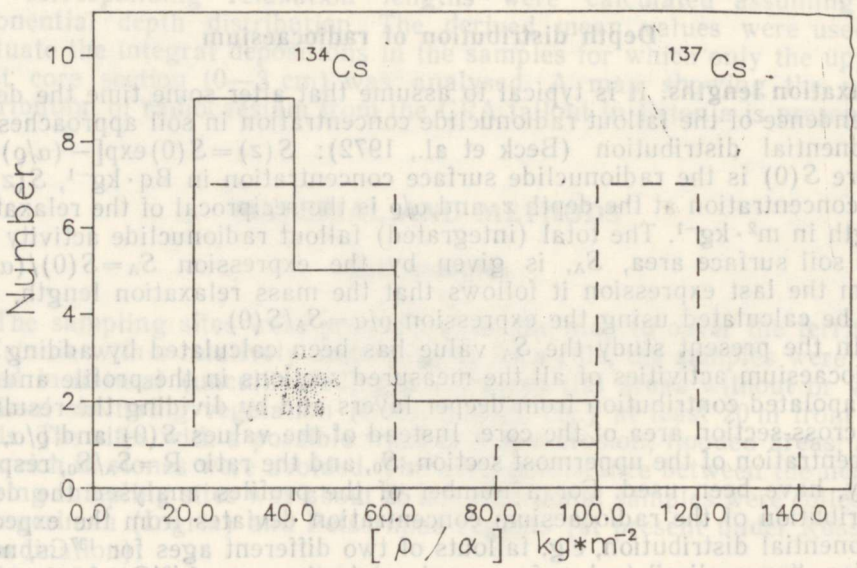

Fig. 2. Frequency distributions of the average mass relaxation lengths, ϱ/α, for ^{134}Cs and ^{137}Cs in Estonian soil profiles.

Fig. 2 demonstrates the frequency distribution of the R values for ^{134}Cs and ^{137}Cs in the soil profiles studied. Two distinct regions of values peaking at about 40 and 110 kg·m^{-2} can be seen. The larger values are typical of ^{137}Cs from the sampling sites with a minimum CRA fallout and are characteristic of an older NWT fallout. The weighted average value for this contribution, $R = 110 \pm 30$ kg·m^{-2}, was estimated. The same procedure

with the ^{134}Cs data yields $R = 42 \pm 15$ kg·m^{-2} for the Chernobyl fallout. These values were used to calculate S_A for the samples where only the upper (0—3 cm) sections were analysed.

The universal use of the two R values for the estimation of S_A can be justified by the following circumstances: (1) no systematic region-specific variation of the R values has been found; (2) the frequency distributions of the soil densities for the 0—3 cm sections are nearly identical for the profiled samples and for the entire set of samples. Both distributions can be approximated by Gaussians with the halfwidths of about 400 kg·m^{-3} and with the mean values of 850 and 890 kg·m^{-3} (dry weight), respectively.

Linear relaxation lengths. An attempt was made to correlate the ^{134}Cs and ^{137}Cs linear relaxation length values, $\alpha^{-1} = R/\varrho$, and the surface soil densities, ϱ_0 (Fig. 3). The α^{-1} values should be considered as the averaged ones, similar to the R values. For the NWT fallout ^{137}Cs the α^{-1} values group is a comparatively narrow region from 0.08 to 0.11 m with a mean value of 0.10 ± 0.01 m and it shows no apparent dependence on the soil type and density. This is valid also for both radiocaesium isotopes from the CRA fallout. Here two distinct regions of values were found in the range of α^{-1} from 0.01 to 0.02 m and from 0.035 to 0.05 m with the mean relaxation length values of 0.015 ± 0.004 m and 0.046 ± 0.007 m, respectively. At the same time the probability of their occurrence is practically equal for all different soil regions in Estonia. This suggests the existence of two soil types, differing in the migration rate of radiocaesium in their near-surface layers. The problem needs additional systematic radioecological studies.

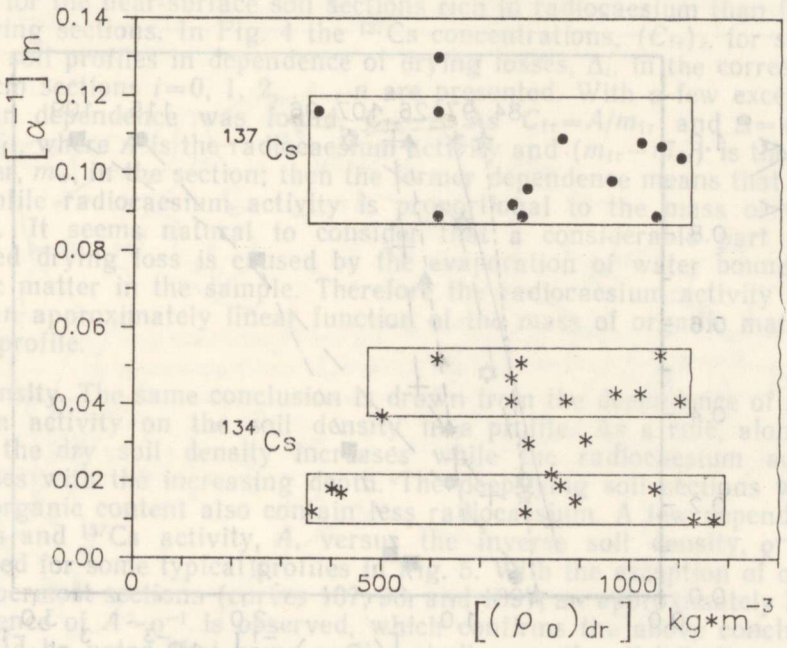

Fig. 3. The dependence of the average linear relaxation length, α^{-1}, for ^{134}Cs and ^{137}Cs on the surface soil density (0—3 cm, dry), $(\varrho_0)_{dr}$. Two density-independent regions of relaxation length values dominate for ^{134}Cs from the CRA fallout. The values for ^{137}Cs from the NWT fallout group around 0.10 m.

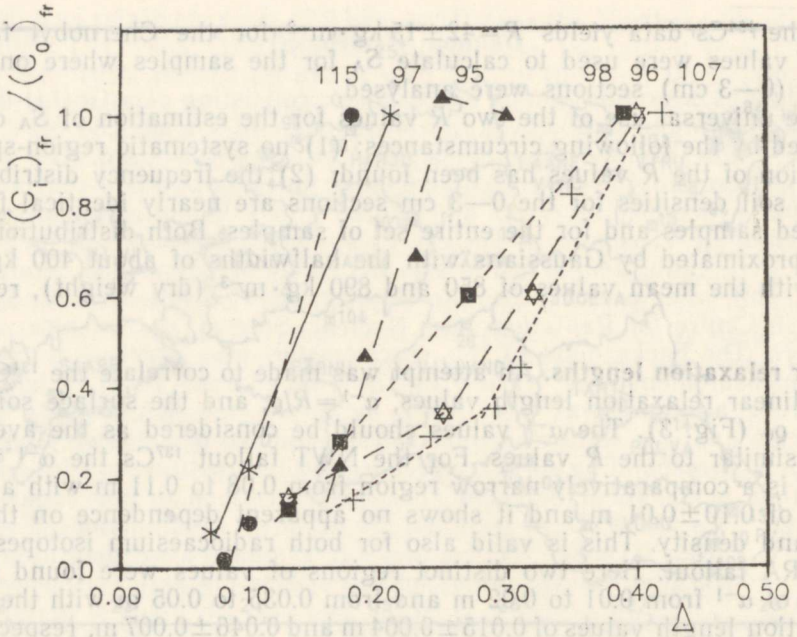

Fig. 4. Dependence of the relative radiocaesium concentration in 3 cm soil core sections, $(C_i)_{fr}/(C_0)_{fr}$, on the relative weight loss of the corresponding section i, $\Delta=(m_i)_{fr}-(m_i)_{dr}/(m_i)_{fr}$. $(C_i)_{fr}$ is the radiocaesium concentration (fresh weight) of the section i; $i=0$ corresponds to the core section from the depth of 0—3 cm; $i=1$, to the 3—6 cm section, etc.; $(m_i)_{fr}$ and $(m_i)_{dr}$ are the fresh and dry masses of the section i, respectively. Data are presented for the following sampling sites (see Fig. 1): Nos. 97 and 115 for ^{134}Cs, and Nos. 95, 98, and 107 for ^{137}Cs.

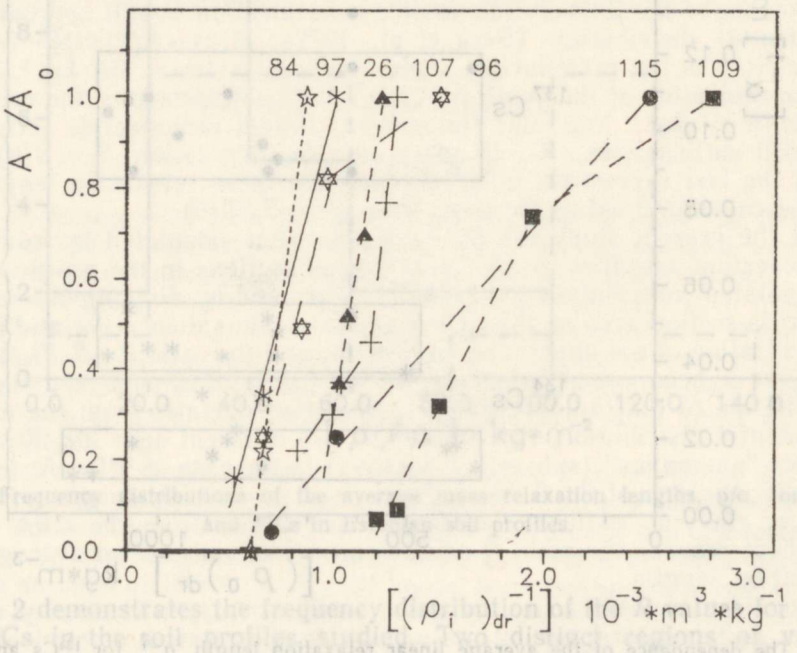

Fig. 5. Dependence of the relative activity of the 3 cm core sections, A_i/A_0, on the inverse dry density of the corresponding section i ($i=0, 1, 2, \ldots$) for 7 soil cores from sampling sites Nos. 97, 107, and 115 for ^{134}Cs, and Nos. 26, 84, 96, and 109 for ^{137}Cs (Fig. 1).

Time dependence of the mass relaxation length. It follows from the mean relaxation length that in 1991, five years after the Chernobyl accident, about 60% of its radiocaesium deposition was bound in the upper 3 cm soil layer. Fallouts of two different ages give a possibility to evaluate an approximate time dependence of the mass relaxation length in Estonian soil. Measurements in the Nordic countries have shown that more than 85% of the radiocaesium deposition from the NWT fallout occurred in 1957—65, with the maximum in 1963 (see, e.g. Hove & Strand, 1990). Using the last date and the determined R data, 42 ± 15 kg·m^{-2} for the 5.2-year-old CRA fallout and 110 ± 30 kg·m^{-2} for the 28-year-old NWT fallout, the sublinear time dependence of $R(t)$ can be approximated by an empirical relation:

$$R = at^n,$$

where t is the time interval in years that has passed since the fallout, $n = 0.57$, and $a = 16.5$ kg[m^2(yr)^n]$^{-1}$.

Errors in the values of the parameters a and n are big (of the order of 50%); nevertheless, the obtained dependence confirms that in natural soils a considerable amount of radiocaesium from the Chernobyl fallout will remain in the near-surface layers and be bioavailable for many decades. Effective half-lives for ^{137}Cs in the range from 20 to 27 years have been reported for natural "soil—plant" systems (Hove & Strand, 1990).

Drying losses. The importance of organic matter in holding radiocaesium in the near-surface soil follows indirectly from the dependence of the ^{137}Cs concentration on the relative drying loss, $\Delta = (m_{fr} - m_{dr})/m_{fr}$, along a soil profile. Here m_{fr} and m_{dr} are the fresh and dry masses of the corresponding core section, respectively. In general, the drying losses Δ are bigger for the near-surface soil sections rich in radiocaesium than for the deep-lying sections. In Fig. 4 the ^{137}Cs concentrations, $(C_{fr})_i$, for several typical soil profiles in dependence of drying losses, Δ_i, in the corresponding 3 cm sections $i = 0, 1, 2, \ldots, n$ are presented. With a few exceptions a linear dependence was found, $C_{fr} \sim \Delta$. As $C_{fr} = A/m_{fr}$ and $\Delta = (m_{fr} - m_{dr})/m_{fr}$, where A is the radiocaesium activity and $(m_{fr} - m_{dr})$ is the mass of water, m_w, in the section; then the former dependence means that along the profile radiocaesium activity is proportional to the mass of water, $A \sim m_w$. It seems natural to consider that a considerable part of the observed drying loss is caused by the evaporation of water bound with organic matter in the sample. Therefore the radiocaesium activity seems to be an approximately linear function of the mass of organic matter in a soil profile.

Soil density. The same conclusion is drawn from the dependence of radiocaesium activity on the soil density in a profile. As a rule, along the profile the dry soil density increases while the radiocaesium activity decreases with the increasing depth. The deep-lying soil sections with a small organic content also contain less radiocaesium. A few dependences of ^{134}Cs and ^{137}Cs activity, A, versus the inverse soil density, ϱ^{-1}, are presented for some typical profiles in Fig. 5. With the exception of one or two uppermost sections (curves 107, 96, and 109), an approximately linear dependence of $A \sim \varrho^{-1}$ is observed, which confirms the above conclusion. It should be noted that some earlier studies on the distribution of the NWT fallout have also established a significant positive correlation of the ^{137}Cs concentration with the amount of organic matter and a negative correlation with soil density (Cawse & Horrill, 1986). So the present study on the depth distribution of radiocaesium in soil supports these conclusions.

Areal distribution of the Chernobyl fallout

Total ^{134}Cs depositions determined for about 110 sampling sites throughout Estonia were used to draw a map describing the areal distribution of the Chernobyl radiocaesium fallout (Fig. 6). A cubic spline method for interpolating the values between the sampling sites enables to calculate the surface activity isopleths. The reference date for the activity values is 1 July 1991. The distribution of the Chernobyl deposition is extremely uneven, demonstrating complex meteorological conditions during the passage of the clouds containing contaminants. In this respect there is a significant similarity between the areal distribution of the fallout in Estonia and in the neighbouring countries, Finland (Arvela et al., 1989) and Sweden (Persson et al., 1987). In Estonia the total ^{134}Cs depositions vary at different sampling sites in the range from practically zero deposition to 3.9 $kBq \cdot m^{-2}$ (or from 0 to 21 $kBq \cdot m^{-2}$ calculated back to 1 May 1986). The corresponding area-weighted mean depositions are 0.18 $kBq \cdot m^{-2}$ and 1.05 $kBq \cdot m^{-2}$, respectively. By using the CRA fallout $^{134}Cs/^{137}Cs$ activity ratio of 0.55 ± 0.05 (Devell et al., 1986), the maximum and mean deposition values for ^{137}Cs were evaluated. The maximum deposition of 40 $kBq \cdot m^{-2}$ was found near Narva-Jõesuu (NE Estonia), while the mean country-wide deposition was only 1.9 $kBq \cdot m^{-2}$ (1986). According to the estimate, about 60% of the total radiocaesium amount from the CRA fallout in Estonia was deposited in two of the 15 counties (Fig. 1), namely in the coastal regions of Ida-Virumaa and Lääne-Virumaa, NE Estonia. Table 1 presents some data reflecting the distribution of radiocaesium in different Estonian counties. Statistical errors of the data are usually of the order of 10—20%, but real errors, probably, can be much bigger due to the high inhomogeneity of the CRA fallout and the limited number of sampling sites.

Fig. 6. Areal distribution of ^{134}Cs (the Chernobyl fallout) in Estonia. The isopleths of total ^{134}Cs deposition (in $kBq \cdot m^{-2}$) are given for the reference date of 1 July 1991. All the sampling sites used to calculate the isopleths are shown.

Table 1

Mean, minimum, and maximum depositions (in kBq · m^{-2}) and total depositions (in TBq) of ^{134}Cs and ^{137}Cs in 15 Estonian counties and in Ingerimaa and Petserimaa (1 July 1991)

County (number of samples)	^{134}Cs			^{137}Cs		
	mean	min—max	total	mean	min—max	total
Lääne (5)	0.19	0.1—0.24	0.45	3.3	2.1—4.8	7.9
Harju (10)	0.13	0—0.44	0.56	3.3	1.9—5.9	14.3
Hiiu (2)	0.07	0.04—0.1	0.07	2.7	1.6—3.8	2.8
Jõgeva (5)	0.01	0—0.06	0.03	1.8	1.4—2.6	4.7
Saare (14)	0.09	0—0.17	0.26	2.7	1.4—4	7.8
Järva (2)	0.18	0.17—0.19	0.47	3.4	2.2—4.5	8.9
Ida-Viru (16)	1.26	0.13—3.9	4.3	14.0	2.4—37	47.0
Põlva (5)	0.14	0—0.48	0.3	4.0	1.5—7.8	8.7
Pärnu (8)	0.02	0—0.12	0.1	2.7	1.5—3.7	13.0
Lääne-Viru (8)	0.32	0.11—1.0	1.1	4.4	2.0—10.5	15.0
Rapla (3)	0.23	0—0.4	0.68	4.3	2.0—6.3	13.0
Tartu (7)	0.06	0—0.19	0.18	2.0	1.5—2.8	6.2
Valga (2)	0	0	0	1.5	1.3—1.8	3.1
Viljandi (6)	0.03	0—0.18	0.11	2.7	1.2—5.0	9.6
Võru (6)	0.09	0—0.23	0.21	2.6	1.6—4.1	5.9
Petseri (5)[a]	0.09	0—0.15	n.c.	3.3	1.3—5.6	n.c.
Ingeri (1)[a]	5.5	—	—	55.0	—	—
Estonia (99)	0.20	0—3.9	8.8	3.85	1.2—37	168.0

[a] not taken into account in calculating the mean and total values for Estonia;
n.c. — not calculated.

Table 2

Mean values of ^{137}Cs depositions from the NWT and CRA fallout and ratios of maximum to mean depositions of the latter for different countries in 1986

Country	Area, 10^3 km^2	NWT mean, kBq · m^{-2}	CRA mean, kBq · m^{-2}	CRA Max/mean
Denmark	43	2.4[b]	1.1	2.7
Estonia[a]	45	2.2	1.9	20
Finland	340	1.9	5.6	13
Norway	390	1.1[c]	7.1	>9
Sweden	450	2	7.6	23

[a] Results of the present work;
[b] Aarkrog et al., 1988;
[c] Hove & Strand, 1990 (for near-surface soil levels).
The other data have been taken from OECD, 1987, and Anspaugh et al., 1988.

The area with the ^{137}Cs deposition bigger than 10 kBq·m^{-2} is comparatively small, only about 1.5% of the whole Estonian territory. Even higher deposition (about 60 kBq·m^{-2} in 1991) has been measured in a sample collected in Ingerimaa (east of the Narva River). The ^{137}Cs depositions higher than 2 kBq·m^{-2} have been found also in Lääne County (NW Estonia), in northern parts of Pärnu and Järva counties, and in Põlva County (SE Estonia, near Lake Pihkva).

The mean deposition from the NWT fallout in 1986 has been estimated to be equal to 2.2 kBq·m^{-2}, a value slightly bigger than that caused by the CRA fallout. Thus in Estonia the Chernobyl fallout approximately doubled the ^{137}Cs inventory. The estimated value for the NWT deposition is in good accordance with the corresponding values available for the neighbouring countries (Table 2).

Basing on the derived mean deposition values, the total radiocaesium inventory from the Chernobyl fallout reached in Estonia (with an area of 45 200 km^2) $50 \cdot 10^{12}$ Bq = 50 TBq of ^{134}Cs and 90 TBq of ^{137}Cs in 1986.

In comparison with the Nordic countries the mean Chernobyl radiocaesium deposition in Estonia is about 3—5 times smaller than in Finland, Norway, and Sweden, but similarly unevenly distributed. At the same time the much more homogeneous deposition in Denmark is characterized by a 1.7 times smaller mean value.

CONCLUSIONS

The present work is the first systematic study of the depth and areal distributions of radiocaesium deposition in Estonia, constituting useful baseline data. The ^{134}Cs analysis has been extremely useful in distinguishing between the depositions from the NWT and the CRA fallouts even 5—7 years afterwards. The results obtained show that the ^{137}Cs concentrations vary in a wide range due to an extremely uneven deposition from the Chernobyl accident. At the same time its environmental impact for most of the Estonian territory has been relatively small. The need for detailed systematic and long-term radioecological investigations in North-East Estonia, where the Chernobyl deposition reached the levels comparable to the most polluted areas in the neighbouring countries, has been confirmed.

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REFERENCES

Aarkrog, A., Bøtter-Jensen, L., Chen Qing Jiang, Dahlgaard, H., Hansen, H., Holm, E., Lauridsen, B., Nielsen, S. P., Søgaard-Hansen, J. 1988. Environmental Radioactivity in Denmark in 1986. Report Risø-R-549. Risø National Laboratory, Roskilde.

Anspaugh, L. R., Catlin, R. J., Goldman, M. 1988. The global impact of the Chernobyl reactor accident. — Science, **242**, 1513—1519.

Arvela, A., Markkanen, M., Lemmelä, H., Blomqvist, L. 1989. Environmental Gamma Radiation and Fallout Measurements in Finland, 1986—87. Supplement 2 to Annual Report STUK-A74, Report STUK-A76. Helsinki.

Beck, H. L., DeCampo, J., Gogolak, C. 1972. In situ Ge(Li) and NaI(Tl) Gamma-Ray Spectrometry. Report HASL-258. US Atomic Energy Commission, New York.

Cawse, P. A., Horrill, A. D. 1986. A Survey of Caesium-137 and Plutonium in British Soils in 1977. Report AERE-R 10155, Harwell.

Devell, L., Aarkrog, A., Blomqvist, L., Magnusson, S., Tveten, U. 1986. How the fallout from Chernobyl was detected and measured in the Nordic countries. — Nuclear Europe, 11, 16—17.

Hove, K., Strand, P. 1990. Predictions for the duration of the Chernobyl radiocaesium problem in non-cultivated areas based on a reassessment of the behaviour of fallout from nuclear weapons tests. In: Environmental Contamination Following a Major Nuclear Accident. Proceedings of a Symposium, Vienna, 16—20 October 1989, Vol. 1. IAEA, Vienna, 215—223.

Martin, L., Nifontova, M., Martin, J. 1991. Radionuclide variation in macrolichens in Estonia after the Chernobyl accident. — Proc. Estonian Acad. Sci. Ecol., **1**, 1, 42—51.

OECD. 1987. The Radiological Impact of the Chernobyl Accident in OECD Countries. Nuclear Energy Agency, OECD, Paris.

Parmasto, E., Liiva, A. 1988. Seeneaeg! Seeneaeg? Kas meie söögiseened on radioaktiivselt saastunud? — Eesti Loodus, 8, 519—522.

Persson, C., Rodhe, H., De Geer, L.-E. 1987. The Chernobyl accident—a meteorological analysis of how radionuclides reached and were deposited in Sweden. — Ambio, **16**, 20—31.

Saar, J. 1991. Radiation. In: Estonian Environment 1991. Environmental Report 4. Environment Data Centre, National Board of Waters and the Environment, Helsinki, 37—38.

Sinkko, K. 1981. Gammaspektrien tietokoneanalyysi näytemittauksissa. Licentiate Thesis, Department of Physics, University of Helsinki.

Sinkko, K., Aaltonen, H. 1985. Calculation of the True Coincidence Summing Correction for Different Sample Geometries in Gamma-ray Spectroscopy. Report STUK-B-VALO 40, Helsinki.

UN. 1982. UNITED NATIONS. Ionizing Radiation. Sources and Biological Effects. Report to the General Assembly. UNSCEAR Official Records, Suppl. No 45(A/37/45). UN, New York.

UN. 1988. UNITED NATIONS. Sources, Effects and Risks of Ionizing Radiation. Report to the General Assembly. UNSCEAR Official Records, Suppl. No 45(A/43/45). UN, New York.