

Variation of Cs-137 deposition in soil and lakes in the north-east region of Poland

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Abstract The aim of this work is to assess the current deposits of Cs-137 in soil and in lakes in the Poland's north-eastern lake district (12–14 years after Chernobyl accident) a dozen or so years after the Chernobyl disaster. Caesium content in soil and sediment was determined by gamma spectrometry (in 79 soil samples and 136 sediment samples). The soil samples were collected in summer 1998–2000. The sediment from 29 lakes were sampled in May 2000. The highest surface activity was found in soil with the arithmetic mean (AM) 3.13 kBq m^{-2} and median (M) 2.37 kBq m^{-2} . For lakes, the respective values were 1.89 kBq m^{-2} and 1.5 kBq m^{-2} . In the case of the soil samples collected from forests, fields, lake shores and arable lands, the mean values of caesium deposition were: 4.37, 2.56, 3.42 and 1.52 kBq m^{-2} , respectively. The mean surface activity of sedimentary samples (AM) from oligotrophic, mesotrophic and eutrophics lakes was found to have different values, viz. 1.48, 2.23 and 1.86 kBq m^{-2} , respectively.

Key words Chernobyl accident • environment contamination • lake sediment • radiocaesium

Introduction

Radionuclides of caesium have been introduced into the environment as a result of nuclear tests and nuclear power station accidents. The largest accident of the latter type that took place at Chernobyl in 1986 led to caesium contamination which depended on the distance from the site of the accident, amount and pattern of rainfall and on local topography.

As a result of direct fallout on the lake surface, certain amount of caesium entered lake water and some was deposited in the catchment area, from where it can still migrate towards lakes. Caesium which directly got into the lake water was adsorbed by suspended organic and inorganic particles and subsequently fell down to the lake bottom to create sediments. It has been estimated that in 1990, 99% of the radioactive caesium coming from direct fallout on the surface of lakes was present in lake sediments [2]. Some observations reveal that caesium is particularly well adsorbed on micaceous clay minerals [7]. The highest Cs-137 accumulation was observed in underwater meadows [18]. Radioactive caesium from the sediment can again return to the water phase by resuspension from the lake bottom due to the intake of water plants through their root system or enter the food chain of bottom organisms, fish and, in the end, man [8, 10].

Thus there are two opposite processes that bring about changes in the radiocaesium pool in lakes: washing out of Cs-137 deposited in sediments and its transport with rivers water to the sea, and the supply of Cs-137 to water reservoirs from river basins. The aim of this work is to assess the current deposits of Cs-137 in soil from four types

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of environment: forests, non-arable fields, lake shores and arable lands and sediments of lakes in the Polish north-eastern lake district 12–14 years after the Chernobyl disaster.

Materials and methods

The studies employed the protocol for sampling and measurement procedures recommended by the International Atomic Energy Agency (IAEA) [5]. Caesium content in soil was determined by gamma spectrometry. Soil samples were taken during field studies in 1998–2000 by means of a steel sampler iron 15 cm in diameter, which made it possible to obtain core of soil samples. Samples were subsequently packed in small plastic bags and delivered to the Laboratory. After having been dried at 105°C to constant weight, they were sifted through a sieve <2 mm so as to remove rocks and vegetable material, and then placed in 600 cm³ measuring vessels with a Marinelli beaker. The radioactive decay of Cs-137 was taken into consideration and all activities were calculated on May 31st 2000.

The sediments were sampled in May 2000 from 29 lakes in north-eastern part of Poland. The samples were collected by the diver from places located outside litoral zone of the lake, where vegetation was not present. The depth of diving was limited by a visibility in water. For oligotrophic lakes the depth reached 20 m. In lakes with a smaller visibility the samples were taken from the depths from several to about 10 m. Plastic containers in the form of tubes 10 cm in diameter and 30 cm high were used. The containers were sunk vertically in the bottom and, after being blocked at both ends, were brought back to the surface. The semi-liquid material obtained in this way was dried up for several weeks until it became solid. Next, the upper part of the dried sediment cylinder was powdered to be suitable for gamma-spectrometry measurements. The volume and mass of the powdered sediments were estimated.

The surface activities A_{surf} (kBq m⁻²) in sediments were calculated from the activity concentrations A_{conc} (Bq kg⁻¹) for radiocaesium accordingly to the following formula:

$$A_{\text{surf}} = A_{\text{conc}} V \rho S^{-1}$$

where: ρ – bulk density (kg m⁻³), V – the analysed volume of dry mass sediment (m³), S – the internal area of the probe pipe (m²).

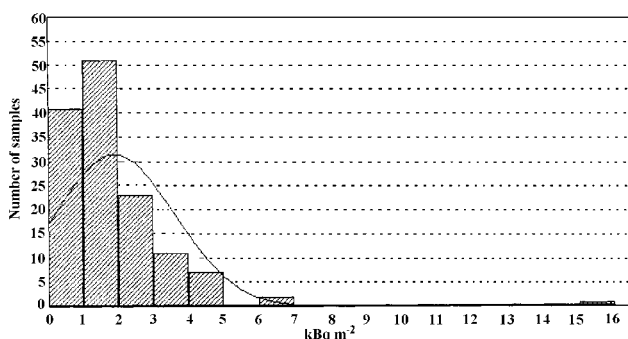


Fig. 1a. The surface activity (kBq m⁻²) distributions in soil.

The measurements of Cs-137 concentrations in the samples thus prepared were made using the method of solid-state gamma spectrometry with a coaxial germanium detector (relative efficiency: 15%) and a Canberra system S-100 for collecting, storing and analysing spectrum (IBM PC with a compatible MCA board, hardware and Microsoft Windows compatible software). The lower detection limits of the spectrometric system were determined by the L.A. Curie method [4]. The minimum detectable concentrations (MDC) for the 661.67 keV line (Cs-137) in the soil and sediment samples (average density of 1 g cm⁻³) were measured in the Marinelli geometry and was found to be 0.3 Bq kg⁻¹.

Results and discussion

79 soil and 136 sediment samples from the north-eastern region of Poland were measured spectrometrically. Soil samples were obtained from four types of environment: forests, non-arable fields, lake shores and arable lands. Sedimentary samples were collected from the 17 eutrophic lakes (Brożane, Dębniak, Guzki, Jałówek, Jędzelewo, Krzywe, Lepaki, Oleckie, Pobjono, Rajgrodzkie, Rogale, Sawinda Wielka, Tajty, Tobołowo, Toczyłowo, Wizajny, Zawadzkie) and from the 9 mesotrophic lakes (Dobskie, Gaładuś, Garbaś, Jałowo, Płaskie, Serwy, Sumowo, Sunnowo, Selmeł Duży) and from the 3 oligotrophic lakes (Aszryynis, Białe Filipowskie, Białe Wigierskie). Distributions of surface radiocaesium activities in the sediment samples are shown in Figs. 1a and 1b.

Higher concentrations were found in soil. The arithmetic mean (AM) and median (M) values of the surface activity was equal to 3.13 kBq m⁻² and 2.37 kBq m⁻², respectively. In the case of the lakes, the corresponding values were: AM=1.89 kBq m⁻² and M=1.5 kBq m⁻². The soil samples collected from forests, fields, lake shores and arable lands the AM values were: 4.37, 2.56, 3.42 and 1.52 kBq m⁻², respectively. The mean surface activities of sedimentary samples from oligotrophic, mesotrophic and eutrophic lakes showed different values, viz. 1.48, 2.23 and 1.86 kBq m⁻², respectively.

The distributions of Cs-137 surface activity in soil and sediments shown in Figs. 1a and 1b are asymmetrical, and positively skewed for which AM>M>M₀ (mode). Therefore, Table 1 gives parameters, which may be used to describe these distributions. The variation in radiocaesium

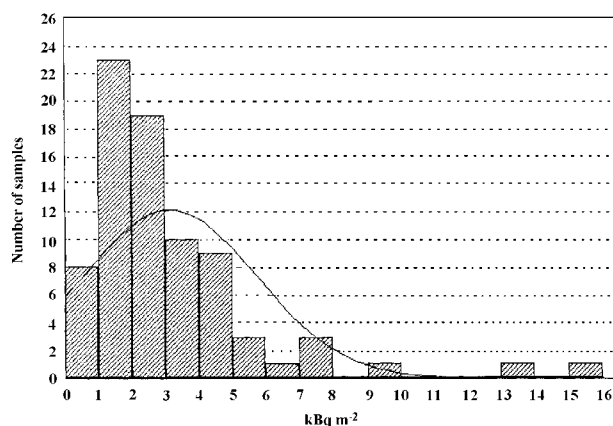


Fig. 1b. The surface activity (kBq m⁻²) distributions in sediments.

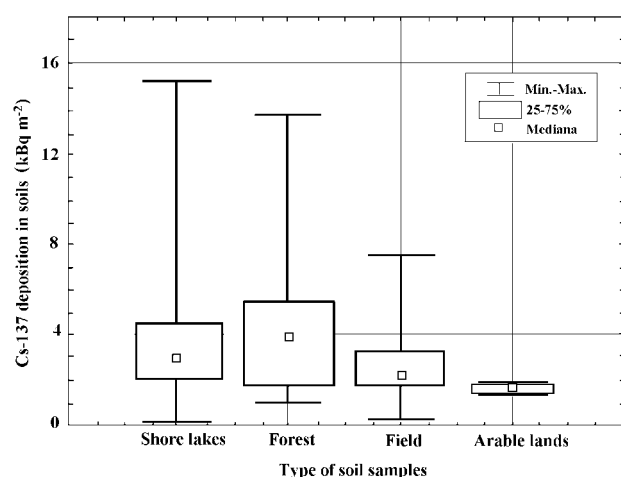
Table 1. The surface activity (kBq m^{-2}) in soil and sediments.

	Number of measurement (N)	Minimum (Min.)	Maximum (Max.)	Arithmetic mean (AM)	Standard of deviation (SD)	Median (M)	Geometric mean (GM)
Soil	79	0.17	15.20	3.13	2.60	2.37	2.32
forest	18	0.93	13.66	4.37	3.27	3.80	3.38
field	36	0.20	7.41	2.56	1.46	2.13	2.15
lake shore	20	0.17	15.20	3.42	3.38	2.56	2.11
arable lands	5	1.31	1.77	1.52	0.20	1.60	1.51
Sediment	136	0.06	15.30	1.89	1.72	1.50	1.39
oligotrophic lakes	24	0.30	3.33	1.49	0.90	1.12	1.15
mesotrophic lakes	37	0.20	6.76	2.23	1.39	1.73	1.59
eutrophic lakes	75	0.06	15.30	1.86	1.87	1.50	1.39

deposits in soil and lake sedimentation is shown in Figs. 2a and 2b.

The analysis of Poland's radiological maps [13] reveals that the investigated area does not display any anomalous levels of caesium contamination. The mean caesium content in the soil, determined in 1995 by the Institute of Geology in this area, was about 5.7 kBq m^{-2} ($\text{SD} = \pm 2.6 \text{ kBq m}^{-2}$ [14]). Our results, recalculated for May 2000, provide a lower value of AM equal to $3.13 \pm 2.6 \text{ kBq m}^{-2}$. The mean radiocaesium deposition in lake sediments in the above area was found to be $1.89 \pm 1.72 \text{ kBq m}^{-2}$, which amounts to 83% of the deposits on the soil of source.

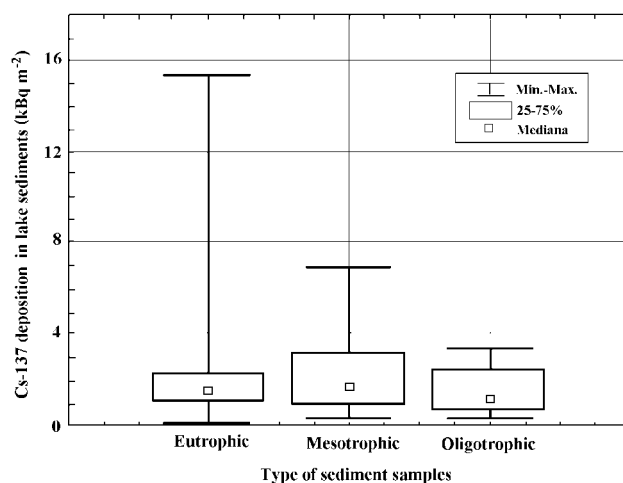
A fraction of the caesium fallout on the lake surface in 1986 was not deposited in bottom sediments because in free flowing lakes it could drift away with the water current [11]. In Poland, the Cs-137 transport with the current was shown to occur, for example, in the river Bug [3]. In addition, caesium deposited in sedimentations can be continuously released to the adjacent water layers. More than 90% of particles suspended in the bottom layer is a result of re-suspension from the lake bottom [10]. In the winter months a continuous process takes place of dissolution of caesium content in sediments [8]. Over the period of over a dozen of years this process reduces radiocaesium levels in the bottom layers of lake sediments. A fraction of caesium can still be found in the lake water, and its concentration, for example, in the Sawinda Wielka lake 9 years after the Chernobyl accident was estimated to be 8 Bq m^{-3} [19].

**Fig. 2a.** The variation in radiocaesium deposits in soil (in kBq m^{-2}).

The higher Cs-137 deposition observed in the forest soil ($\text{AM}=4.37 \text{ kBq m}^{-2}$) is a result of the presence of a large amount of leaves and conifer needles absorbing the caesium fallout [6]. This effect does not occur in the non-arable areas, therefore, the mean value was lower ($\text{AM}=2.56 \text{ kBq m}^{-2}$). The soil samples taken from the lake shores show the largest spread of values (maximum value of $\text{AM}=15.2 \text{ kBq m}^{-2}$). This type of soil is representative for an area of rain and thaw water run-off and also washed with lake water. The arable soil samples show the lowest deposition of caesium in the surface layer ($\text{AM}=1.52 \text{ kBq m}^{-2}$) with the lowest value of $\text{SD}=0.2 \text{ kBq m}^{-2}$, which is due to a good homogenisation of the surface soil resulting from cultivation procedures.

Differentiation of caesium deposition values in the lake bottom sediments may be due to the different morphometric parameters of the lakes [15, 17] and the type of the drainage area [9]. In the present work, the caesium deposition was correlated with the trophic levels of lakes. We found that the poorest oligotrophic lakes have shown the lowest value of deposition $\text{AM}=1.49 \text{ kBq m}^{-2}$, whereas the richer mesotrophic and eutrophic lakes revealed a higher depositions of $\text{AM}=2.23$ and 1.86 kBq m^{-2} , respectively.

Lakes are eutrophic due to excessive input of nutrients. Eutrophication has a significant impact on biota above aquatic sediments because of large increases in phytoplankton density. These changes lead to increased inputs of detritus to aquatic sediments. If oxygen is

**Fig. 2b.** The variation in radiocaesium deposits in lake sedimentation (in kBq m^{-2}).

abundant, the sediment microbial activity is stimulated [16]. However, with increased detritus inputs to the sediments deoxygenation can occur. Under anoxic conditions, changes in microbial biodiversity occur and reductive microbial processes, such as methano-genesis and nitrate ammonification may dominate [12]. The process of redissolution of Cs ions from sediment due to ion exchange with biogenic ammonia ions was already observed [1]. Slight differences in Cs-137 amounts deposited in sediments in lakes of various trophic levels seem to point to a slight influence of lake trophic level on radiocaesium deposition in sediments. It can be assumed that oxydoreductive processes have a small influence on caesium release from sediments.

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