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The Impact of the Chernobyl Accident – an Evaluation from the German Perspective

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Abstract

During the years 1991-1993, from May till October, specialists from the Federal Republic of Germany investigated both the environmental and population radiation exposure due to the accident at the Chernobyl nuclear power plant in the former Soviet Union. In the survey an area of approximately 100,000 km^2 extending from Tula in Russia to Rovno in the Ukraine was investigated. This area included regions of low and high contamination levels in the three post-Soviet republics of Russia, Belarus and Ukraine

Up to seven environmental measuring vans from different German institutions took part in the field study on the determination of the local dose rates and of the burden of environmental samples and foodstuffs. The results agreed fairly well with those obtained by Russian scientists.

The body content of radioactive cesium of more than 300,000 individuals was examined. Up to 7 mobile whole-body counting laboratories were operational with a total number of 20 counters. Each person examined received an official certificate giving the result of the measurement. Observed activity levels were generally much less and in no case higher than the annual limit of intake acceptable for professional radiation workers. Only for about 2 % of the people examined did the results suggest the necessity of further observation by health physicists and physicians similar to the regular medical examination professional radiation workers undergo.

The project was supplemented by the individual measurement of external doses. About 7000 thermoluminescence dosemeters were distributed and evaluated in the years 1992 and 1993. The results show a dependence on social and location factors.

To complete the programme, blood samples from 100 Ukrainian individuals were investigated in 1993. The results of biological dosimetry show remarkable deviations from the normal distribution.

The project was funded by the German Ministry of the Environment, Nature Conservation and Nuclear Safety. It was coordinated by the Research Centre Jülich (KFA) and supported by other companies and scientific centres. The support from the Research Centre Rossendorf and TÜV Dresden from the federal state of Saxony was of special importance due to their experience with the social and political structures and with the Russian language. Thanks are due to Dr. H. Braun, R. Kassebaum and Dr. W. Hake for their support in the project organization, and to Dr. P. Sauermann, Dr. U. Wolf and Dr. I. Uray for their important help in the scientific evaluation.

Contents

1. The German Measuring Programme: Aims and Realization

The reports on the consequences of the reactor accident at Chernobyl in late April 1986 have been extremely contradictory as yet. In the post-Soviet republics affected by the accident there was a general lack of reliable information concerning the levels of radioactive contamination of the environment and of foodstuffs. Where data were given, they were insufficiently supported and explained.

Not least, these circumstances led to considerable disquiet and concern amongst the public even at a great distance from the site of the accident. Many remedial programmes were untenable because of anxieties and mistrust on the part of the population. Thus, first and above all, the situation in the areas affected by the Chernobyl accident represented a psychological and an information problem.

In order to assist in informing those affected and above all to determine their actual radiation exposure, the Federal Republic of Germany performed a measuring campaign in the three post-Soviet republics between Moscow and Kiev in the years 1991-1993. For this purpose, the Federal Ministry of the Environment made available funds of about DM 12 million.

After consultation with the German Radiation Protection Commission, a measuring programme was drawn up and the Research Centre Jülich entrusted with its implementation and organization. Starting in 1991 a total of 22 vehicles with 27 measuring assemblies and more than 150 staff, recruited on a voluntary basis from various institutions in the Federal Republic of Germany, were engaged in the first measuring campaign lasting from mid-May to early October 1991 [1]. In the following two years the programme was slightly reduced. The measuring locations can be seen from Figures 1.1 and 1.2.

Great store was set by the comprehensive information of the population. In addition to general information through the media two leaflets were distributed to all visitors to explain the radiological situation. Each person undergoing measurements or bringing food samples received a form with the results and a short explanation in Russian. This was favourably accepted by the public.

1.1 Operation of the Environmental Measurements

The measuring programme of the Federal Republic of Germany started with environmental measurements in Russia in the period from 21 May to 11 June 1991. The following scientific institutions from the Federal Republic of Germany were involved with their environmental measurement vehicles and operating personnel:

- Institute for Radiation Hygiene (BfS), Neuherberg
- Institute for Radiation Protection (GSF), Neuherberg
- Hahn-Meitner Institute, Berlin
- Monitoring Station for Radioactivity (University of the Saarland), Homburg
- Institute for Hygiene of Water, Soil and Air (BGA), Berlin
- Federal Civil Defence Office, Warning Station, Linnich
- Central Office for Safety Technology, Düsseldorf
- Research Centre Jülich

In the two subsequent years this concept was changed. Only one environmental measurement vehicle was occupied over the whole period from May till September. The scientific team was still recruited from different scientific institutions and rotated every 3 weeks.

In the first year, the vehicles with the sensitive measuring instruments were transported by aircraft from the Russian military airport at Templin, north of Berlin, to Moscow and from there they were driven to the regions where they were to be employed: Tula, 200 km south of Moscow, Kaluga, 250 km southwest of Moscow, and Bryansk, 500 km southwest of Moscow. The measuring campaign was coordinated from Bryansk. In the subsequent years, the vehicles were transported from Germany to the sites of action in the three republics and back by road.

The measuring locations were selected by us or by our partners on the National Committees for Remedying the Consequences of the Chernobyl Reactor Accident (Goskom Chernobyl) of the three republics. These committees also undertook to supply the detectors with liquid nitrogen, the vehicles with fuel and the German teams with board and lodging.

The environmental investigations had the following objectives:

- collecting basic data for the interpretation of the whole-body measurements
- supplementing the whole-body measurements by measuring the external radiation (area dose rate)
- providing factual information for the public on their uptake of radioactivity from their diet by measuring the radioactivity of their food.

In each settlement the measuring programme began by determining the area dose rate and the soil contamination. Subsequently, basic foodstuffs, such as water, milk, meat, potatoes, bread and cereals, were examined by gamma spectrometry as an aid to interpreting the whole-body measurements.

All foodstuff supplied by the public was measured. Each participant was given the contamination result measured for his food sample and a brief explanation in Russian on a specially designed form. Thus, members of the public were able to understand the meaning of the given figure for the contamination of their food. In order to cope with the great demand for food measurements, it was often necessary to work into the night without any free weekends.

A total of more than 4000 food measurements, more than 500 soil examinations and more than 1000 area dose rate determinations were carried out [2], [3], [4].

1.2 Operation of the Incorporation Measurements

The second component of the measuring programme consisted mainly of whole-body measurements. The first campaign started in June 1991, after the environmental measurements had finished.The following German institutions provided personnel for this programme:

- Central Institute of Nuclear Research (ZfK), Rossendorf
- Federal Civil Defence Office, Warning Stations, Bonn
- Nuclear Auxiliary Services (KHG), Eggenstein
- Technical Control Board for Saxony (TÜV), Dresden-Chemnitz
- Hahn-Meitner Institute (HMI), Berlin
- Federal Agency for Materials Research (BAM), Berlin
- Research Centre Geesthacht (GKS)
- Brenk Systemplanung (BS), Aachen
- Research Centre Jülich (KFA)
- Technical University (TU), Dresden
- Polytechnic (FH), Jülich
- Federal Agency for Public Health (BGA), Berlin
- Technical Control Board for Bavaria (TÜV), Munich
- other institutions and private individuals.

In 1991 three semitrailers with 4 measuring assemblies each and 4 box-type delivery vans with 2 measuring assemblies each were employed for the whole-body measurements. This permitted flexible operation. The delivery vans enabled measurements to be carried out at settlements with populations ranging from several hundreds to a few thousands even if the roads were not suitable for heavy vehicles, and the semitrailers were parked in towns or large villages where they remained stationary for a few weeks thus permitting a high throughput.

It was to be expected that in 1991, i.e. 5 years after the accident, the detectable incorporation of the population would only consist of the gamma-emitters Cs-137 and Cs-134. Even in 1990 other international projects, especially the IAEA project, could only find these radionuclides in human bodies. Our measurements at the Research Centre Jülich for some of the reactor and emergency personnel on duty in block 4 of the Chernobyl power station during and after the accident confirmed this. On the basis of the environmental and food measurements, a significant contamination of the population with Sr-90 and Pu-239 was finally ruled out.

Therefore, an important criterion for selecting the measuring systems was a sufficient detection sensitivity for the isotopes Cs-137 and Cs-134 in a short measuring time appropriate for the measuring task. It should be possible to detect a Cs activity of 1 kBq in the human body within a measuring time of 1 minute. This requirement was fulfilled not only by spectroscopic, nuclide-resolving measuring systems but also by simpler systems with plastic scintillators. We therefore decided on a mixed concept. Apart from spectroscopic systems, instruments operating radiometrically with plastic scintillators were employed which required knowledge of the isotopic composition of the incorporated activity. This mixed system permitted mobility and cost savings since the instruments operating radiometrically were much lighter and considerably cheaper. On the other hand, more efficient spectrometric measuring systems were available for more complicated measurements and if the highest reliability was required. The lighter QBM1 monitors[1] with plastic scintillators were employed in the small, mobile vans whereas the heavy instruments of the Fastscan[2] spectrometric system with NaI detector and the H 13010[3] type with plastic scintillator were installed on the semitrailers. Since 1992 the semitrailers were additionally equipped with a special whole-body counter for babies and toddlers. Also since 1992 the number of vans was reduced to one using a H 13010 incorporation monitor.

The vans were used in towns in order to reach kindergardens, factories, offices, schools and suburbs. In the country, they were used to take measurements in villages, small towns, collective farms, state farms, convalescent homes and holiday camps. The semitrailers were stationed mainly in administrative centres. They were usually parked at hospitals or health centres.

For intercalibration purposes, comparative measurements were carried out each year with a whole-body phantom made available by our Russian partners from the Scientific Research Institute for Hygiene at Sea of the Ministry of Health of the former USSR, which had already successfully participated in an IAEA intercomparison [5] and which had also been used to calibrate the Russian measuring assemblies. Human beings of different weight and size can be simulated in a standing, lying or sitting position using individual polyethylene bricks, 1 kg in weight. Two tubes with a Cs-137 standard were embedded in each brick in order to simulate the distribution of cesium in the body.

The results of the most extensive intercalibration measurements from 1992 are given in table 1.1. Phantoms of 4 different sizes ranging from a 3-year-old child (12 kg) to the standard adult (70 kg) were used to test the performance of counters of the German measurement campaign, as well as of

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[1] Quick Body Monitor, Nuclear Enterprises, Edinburgh, UK, supplied by MAB, Munich

[2] Fastscan Model 2250, Canberra Packard GmbH, Frankfurt

[3] Herfurth H 13010-WB, Hamburg

Russian mobile and stationary measuring systems. For the Fastscan system the agreement between our measured data and the reference activities was very excellent. Only the burden of the small child was overestimated by about 3 % and the burden of the standard adult was underestimated by just 8 %.

Both the Herfurth and QBM-1 incorporation monitors were calibrated in Russia using real people contaminated by incorporated Cs activities of more than 100 nCi (as determined with a Fastscan) as phantoms. Table 1.2 compares Cs body burdens measured both with the Fastscan system and Herfurth incorporation monitors. The agreement for real persons is much better than expected from the intercalibration with phantoms.

type	phantom size	activity (AP) kBq	measured results (AR) kBq		relation AR/AP	
			A	B	A	B
QBM1A	F1	3.55	2.77	3.60	0.78	1.01
	F2	7.14	4.20	8.40	0.59	1.18
	F3	15.0	4.92	11.56	0.33	0.77
	F4	20.8	6.87	17.86	0.33	0.86
FASTSCAN	F1	3.55	3.67		1.03	
	F2	7.14	7.13		1.00	
	F3	15.0	15.0		1.00	
	F4	20.8	19.2		0.92	
HERFURTH	F1	3.55	4.60		1.30	
	F2	7.14	6.36		0.89	
	F3	15.0	10.2		0.68	
	F4	20.8	12.8		0.62	
MASTER-GEMINY	F1	3.55	12.0		3.38	
	F2	7.14	16.5		2.31	
	F3	15.0	17.3		1.15	
	F4	20.8	22.3		1.07	
SIB-2	F1	3.55	4.19		1.18	
	F2	7.14	7.66		1.07	
	F3	15.0	14.6		0.97	
	F4	20.8				
Robotron-20046	F1	3.55	4.10		1.15	
	F2	7.14	8.07		1.13	
	F3	15.0				
	F4	20.8	19.7		0.95	

Table 1.1: Russian-German intercalibration with an official polyethylene phantom
A-QBM1A with factory calibration
B-QBM1A with a correction of the factory calibration as used by KFA
Phantom size: F1 infant, F2 child, F3 female adult, F4 male adult

weight [kg]	Fastscan [nCi]	Herfurth [nCi]
86	83	75
35	128	158
70	1186	1260
55	556	543
76	106	91
70	109	95
77	203	186
57	470	382
67	244	260
91	191	228
65	106	100
73	376	381

Table 1.2: Comparison of the Cs body burden of persons as determined both with Fastscan Systems and Herfurth monitors

The measuring campaign was guided by the project management in Germany. The project management provided the personnel and the technical support for the operating teams. The operating experience was continuously evaluated by the project management and used to modify the instruments and procedure. A coordination group was set up in order to guide and supervise operations on site. This group was particularly involved in the following tasks:

- liaison office with the project management in Jülich
- preparation of operations in the settlements
- contacts with Russian officials, such as Goskom Chernobyl or local authorities
- information for the media and scientific institutions
- catering, counselling and instruction for the measuring personnel
- instrument and vehicle servicing
- transportation of measuring personnel between the airports and the operating areas.

A well-equipped office belonging to the Goskom Chernobyl in the centre of Bryansk was made available for the coordination staff. This integration into the offices of our Russian partners promoted the cooperation. In the first two years Bryansk proved to be a suitable site since it was situated at the centre of the operating area with good transport routes to Moscow. In the last year, 1993, the measuring teams were scattered over an extended area with the main emphasis on Belarus and Ukraine. Thus, the office in Bryansk was no longer so important as a service and communication centre and the coordination staff moved routinely between Belarus, Russia and the Ukraine in a weekly cycle.

The measuring staff at each location formed one measuring team consisting, depending on the situation in the operating areas, of 4-10 members. Some of them were able to speak enough Russian so that it was possible to communicate with the visitors in every measuring vehicle. Our colleagues from Eastern Germany were of great assistance in this respect.

The daily throughput for the semitrailers was conservatively planned as an average of 200 persons/day. This goal was exceeded by far and as a rule a daily throughput of 500 persons was reached (peak value 820 persons on July 4, 1991 in Ludinovo).

The goal for the vans was a daily throughput of 100 persons/day. During the operating period the standard number was closer to 300 persons/day (peak value 562 persons/day on August 8, 1991 in the Chvastovitchi rural district). This rate was only possible since the crew voluntarily worked much longer than planned to cope with the great demand.

All those examined were given a certificate of the measurement results in Russian. Our Russian partners also received a carbon copy of this certificate. In general this was stored at the health authorities in the districts. We kept a carbon copy in Germany for our own purposes. Russian assistants helped us to fill in the numbered forms, recording the personal details required and entering the measuring date. The record number, year of birth, weight and height were included in the data sets at the time of measurement.

The measured data were entered on the certificate by the measuring staff and authenticated by a rubber stamp and signature. At the same time the measured data were evaluated according to three categories of the whole body activity.

In order to achieve our goal, namely of informing the public about their true radiation exposure and explaining it and thus reducing unnecessary fears and reestablishing social stability, as many people as possible had to be measured. Our minimum goal was to examine 100,000 persons in the first year. We expected that about one third of the population in the operating areas would take part in the measuring campaign. Our expectations were clearly exceeded and in the first year 163,000 people were measured. In the subsequent years this number decreased to 90,000 in 1992 and 64,000 in 1993 due to reduced funding of the project.

These high numbers were possible only on the basis of the willingness of the measuring staff to work overtime, and the suitability and serviceability of the measuring technology. However, this good performance was also due to the fact that our campaign was readily accepted by the authorities and the public so that in some areas more than 70 % of the population made use of our services.

We operated in more than 240 settlements in the ten regions of Bryansk, Kaluga, Orel, Tula in Russia, Gomel, Mogilev, Brest in Belarus and Kiev, Shitomir, Rovno in Ukraine. The most important of these settlements are shown in the map (fig. 1.1 and fig. 1.2). In the first year, 1991, all the measurements were undertaken in Russia, about 50 % in the Bryansk district, where the greatest number of measuring assemblies were deployed, but almost 32 % of the measurements were made in the Kaluga district, where only two vehicles were in operation. This good result can be attributed to the great interest on the part of the public in the Kaluga district and the excellent organization by the Russian partners. In 1992 about 55 % of the measurements were undertaken in Russia, 31 % in Belarus and 14 % in the Ukraine. Most of the measurements in 1993 were carried out in the Ukraine (57 %), fewer in Russia (23 %) and Belarus (20 %) [6], [7], [8].

1.3 Operation of the Personal Dose Measurements

In 1992 and 1993 the measuring programme was supplemented and completed by personal dose measurements of the external dose. They were performed in collaboration with colleagues from the Novosybkov branch of the Institute for Radiation Hygiene in St. Petersburg. The personal dose was mainly measured in Russia near the western border of the Bryansk region in the districts Novosybkov, Slinka, Klimovo, Krasnaya Gora, Gordeyevka, Klincy and Starodub. In 1992 a minor part of these measurements (about 10 %) was done in the Gomel Region in Belarus in the districts Narovlya, Dobrush, Chechersk and Korma.

Personal dose measurements were carried out in about 100 selected settlements with varying Cs ground contamination from less than 74 kBq/m^2 to about 3700 kBq/m^2. In some of these settlements decontamination measures had been previously taken. The settlements also varied due to size and economy (agricultural, industrial). Within the settlements we tried to differentiate the population according to occupational aspects.

The dosemeter was a tissue equivalent LiF thermoluminescence detector of the type TLD-100*) (1/8 x 1/8 x 0.035"). The detector was covered with light-preventing black polyethylene and welded in a cylindrical polyethylene box (height 33 mm, diameter 14 mm, wall thickness 1 mm). This box was worn round the neck on a thin chain.

The dosemeter was evaluated in a device of the Solaro**) type. To achieve an extremely low detection limit special preparation of the detectors was neccessary. Immediately before evaluation the TLDs were rinsed for 2 minutes in a solution of 0.2 % hydrofluoric acid in methanol. Disturbing chemoluminescence effects during the evaluation of the dosemeter were thus excluded [9] and the detector background was reduced to less than 10 μSv.

For the dose evaluation, only the main peak of the glow curve was used. The other peaks of the glow curve and the infrared photons were eliminated by the computer code Fastglow. The result of this very careful TLD evaluation was that dose measurements were possible down to the range of 10 to 15 μSv with an error of ± 15 %.

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*) Manufacturer: Harshaw Bicron, Solon, USA
**) Manufacturer: Nuclear Enterprises, Edinburgh, UK

Before starting the measuring campaign the TLDs were calibrated individually and classified in groups of efficiency differing by not more than ± 3 %.

Our calibration was controlled by the Research Centre of Medical Radiology in Obninsk for two groups with 10 TLDs each. The set value of the dose was 5000 μSv ± 4 %. The measured values were (5004 ± 150)μSv and (5026 ± 164)μSv, respectively. Thus, the accuracy of our TLD measurements in this dose range was about ± 3 %.

The precise measurements of doses in the range of 10 to 15 μSv made it possible to determine the personal dose with sufficent accuracy even for relatively short exposure periods of only 7 to 10 days. These short periods made it much more certain that the dosemeters were carried and treated in a proper manner by the individuals exposed.

The transport dose was assessed for every dosemeter shipment. In general, it was between 1 and 3 μSv.

The dosemeter was distributed together with an information sheet and a questionnaire for statistical reasons. The questionnaire asked about individual data like address, sex, age, occupational circumstances and type of home.

The measurement of the personal dose was carried out in summer 1992 and 1993 between May and September. A supplementary measuring campaign took place in winter 1993/94. In 1993 every person being monitored received a second dosemeter to determine the area dose rate in their home.

The results of the personal dose measurements were converted to average dose rates taking the exposure time into consideration. The average dose rate, known as the individual dose rate $\dot{H}_p$, was the basis for our statistical investigations [20]. For comparison with other authors the annual and monthly doses are also given in some cases. However, these values must be treated carefully because the personal dose depends on the season and other circumstances.

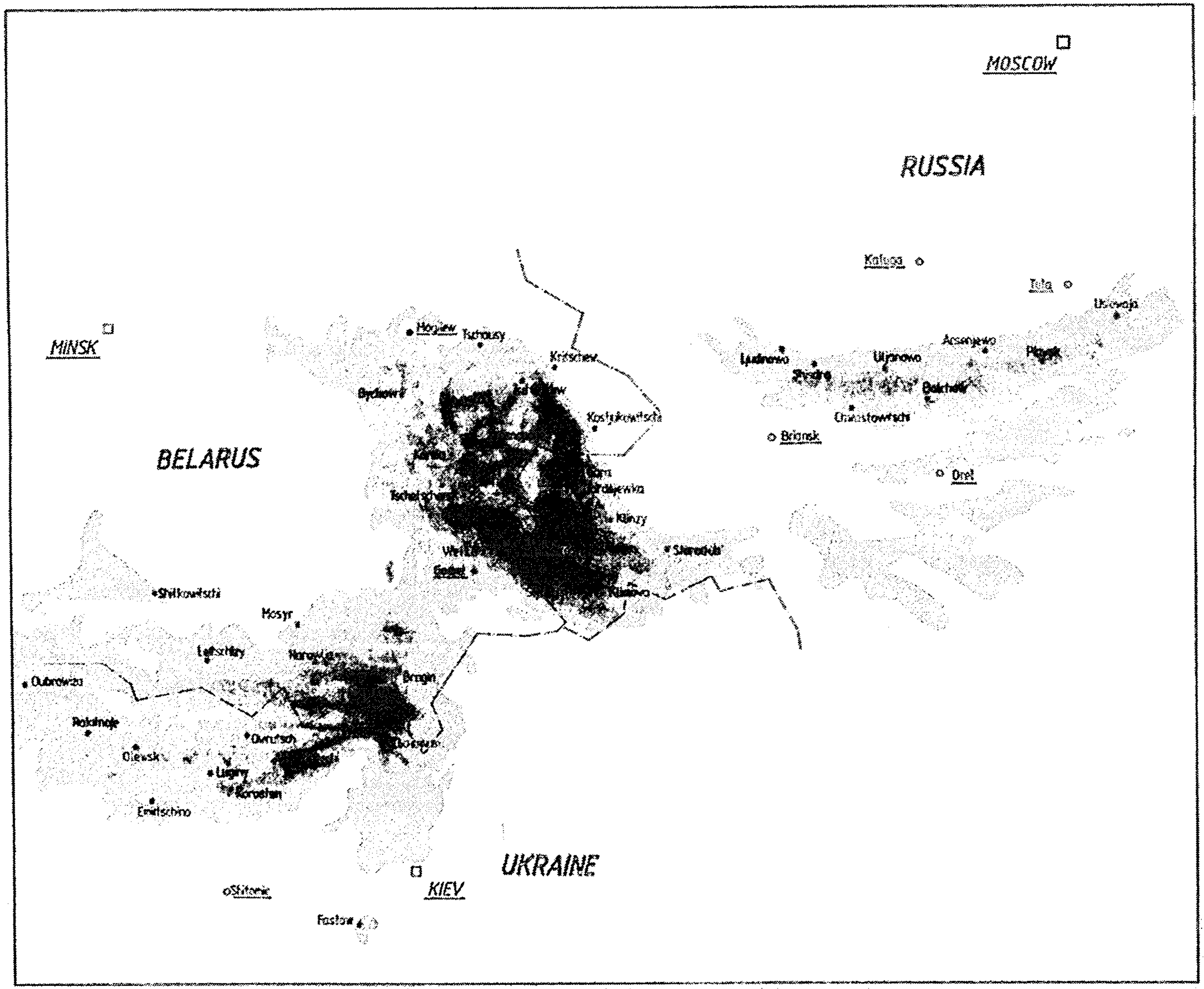


Fig. 1.1: Contaminated areas with district towns where measurements were carried out for 4 contamination (Cs-137) classes:

1st	> 40	Ci/km^2 (dark grey)
2nd	15 - 40	Ci/km^2
3rd	5 - 15	Ci/km^2
4th	1 - 5	Ci/km^2 (light grey)

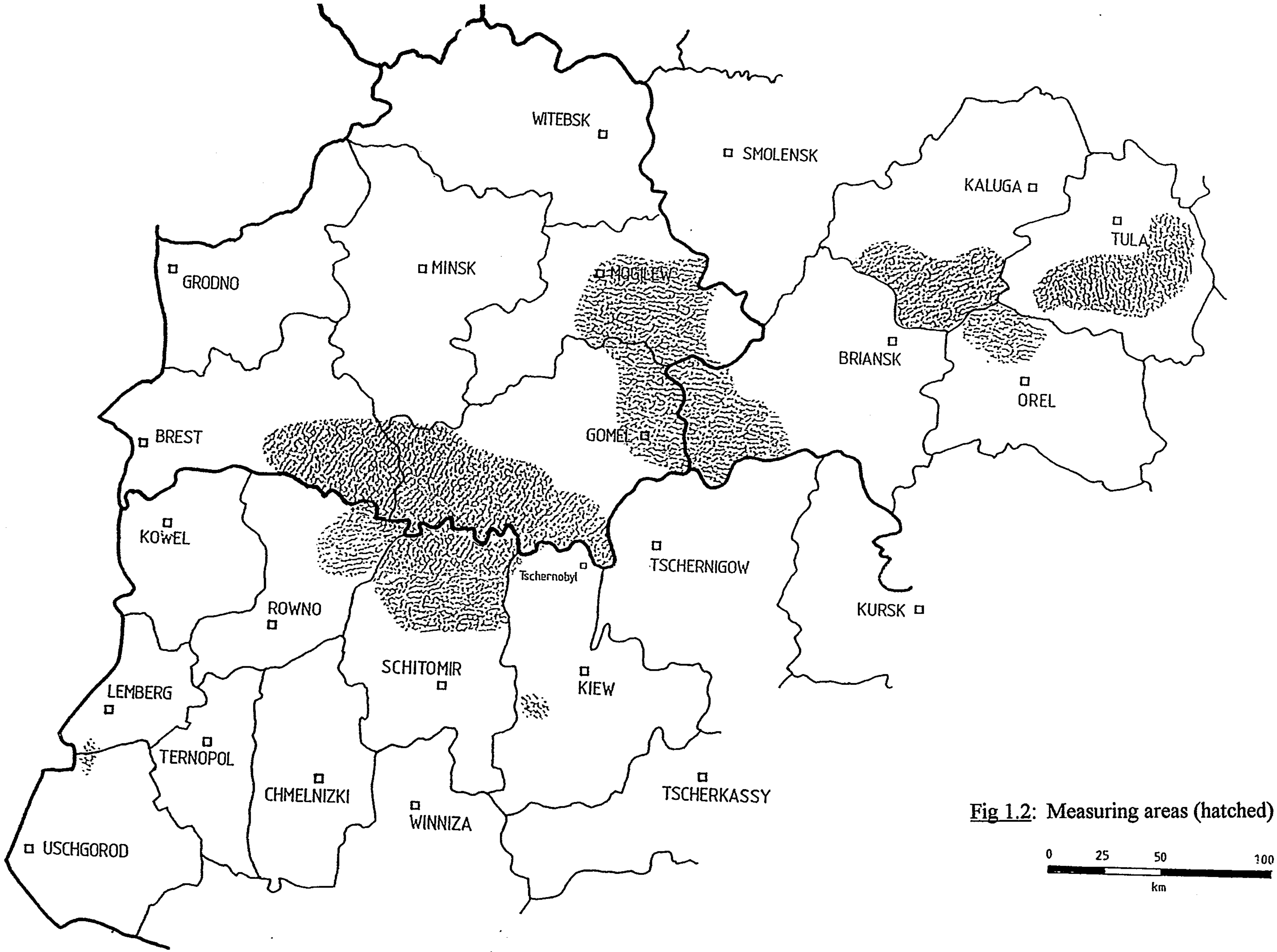


Fig 1.2: Measuring areas (hatched)

2. Contamination of the Environment and Diet

2.1 Soil Contamination and Area Dose Rate

The reactor accident at Chernobyl on April 26, 1986 caused a high release of radioactive material into the environment. The numbers given by Russian scientists are especially 2.3 MCi Cs-137 (85 E15 Bq) and 1.2 MCi Cs-134 (45 E15 Bq).This corresponds to about 1/3 of the respective core inventory [11].

The cesium activity was dispersed in the form of aerosols. About 10% of the emitted cesium activity was released in particle form of a larger size with high deposition velocities (5-10cm/s). These particles were created during the first few days after the explosion. Most of the cesium activity was released as small and easily transportable aerosols, which were deposited only at great distances due to their low deposition velocities of between 0.15 and 1 cm/s.

The larger particles from the first, the dispersion phase precipitated in the inner zone within a distance of less than 30 to 50 km round the reactor site. Outside this zone the contamination was mainly formed by the deposition of aerosols created in the second, the evaporation phase. These aerosols were dispersed over the European territories. About one half of all the cesium released (1.3 MCi or 50 E15 Bq of Cs 137) contaminated the European part of the former Soviet Union.

Due to their higher sublimation temperature the release of plutonium and strontium isotopes from the destroyed reactor only took place during the dispersion phase. 60 kCi (2.2 E15 Bq) of strontium and 1 kCi (0.04 E15 Bq) of plutonium were released mainly in the form of heavy fuel particles after the initial explosion or resuspended by the graphite fire during the next few days. These particles precipitated with a high deposition velocity of about 10 cm/s and thus contaminated predominantly the inner zone. Outside a circle with a radius of about 30 km the surface contamination by strontium and plutonium was very low.

Therefore, only the two cesium isotopes 134 and 137 are responsible for the long-term radioactive contamination outside the inner zone. Only this area is important for the population dose because the inner zone is almost identical with the prohibited zone from which the normal population was evacuated. The relation between the two cesium isotopes changed from 1:2 at first to 1:10 for 1991 when we started our measurements. In 1993 at the end of our measurement campaign we frequently found relations between Cs-134 and Cs-137 of nearly 1:20. After that even Cs-134 was no longer of any importance.

The total activity of 1.3 MCi (50 E15 Bq) Cs-137 which was deposited on the European part of the former Soviet Union would lead to an average surface contamination of about 0.2 Ci/km^2 (7 GBq/km^2)if the contamination were distributed homogeneously. In fact, it is distributed extremely inhomogeneously. The highest surface contamination outside the inner zone is above 50 Ci/km^2 (1850 kBq/m^2).

The area of higher contamination with more than 1Ci/km^2 (37 kBq/m^2) is twice the size of Switzerland (about 100,000 km^2). This area is called the Observed Area (OA) where certain measurements but in general no protective measures have been carried out. The area above 15 Ci/km^2 (555 kBq/m^2) of average surface activity is called the Controlled Area (CA) where certain protective measures like prohibition of the consumption of locally produced food have been taken. The size of this area is about 10,000 km^2 with about 272,000 inhabitants.

A look at the official contamination map (fig. 1.1) shows that there are mainly three large areas of higher contamination:

- The central spot round the reactor site including the prohibited and evacuated zone and neighbouring areas of Belarus and the Ukraine in a northern and western direction from the site

- The Bryansk-Gomel spot at the Russian/Belarussian border round the towns of Klincy, Novosybkov and Gomel

- The Tula-Kaluga-Orel spot in Russia where towns like Uzlovaya and Ludinovo are more highly contaminated.

The contamination of the central spot was mainly caused by the explosion on April 26 and dispersed by eastern winds in western and northwestern directions. It includes so-called forest districts (Polesy region) round Korosten in the northwest Ukraine and near the southern border of Belarus. Three factors were decisive for the high contamination of the two other spots at a distance of more than 100 km from the reactor site [11]:
- The cloud reaching these areas on April 27 and 28 was created before April 27 when the emission from the reactor was highest due to the uncovered graphite fire of the first few days
- The release and transport took place at an altitude of 1000 m where the wind speed was about 10 m/s directly above the reactor and slowed down to 1-2 m/s above the contaminated areas thus reinforcing the fallout processes
- At that time there were several local atmospheric pressure subsystems above the areas of contamination, a near-surface cyclone was located south of Gomel, creating thunderstorms with local precipitations of 3mm/d on April 27 near Gomel, of 2 and 7 mm/d on April 28 in the Bryansk and Tula region, respectively, resulting in enhanced activity washout.

The result was that even at distances of several hundred kilometres the contamination levels were nearly as high as in the central zone.

During our measuring campaign in 1991 to 1993 we determined the soil surface contamination in several settlements in the Controlled and Observed Areas of the three post-Soviet republics by soil sampling and γ-spectrometry or by in-situ γ-spectrometry. Fig.2.1 - 2.3 give an overview of the mean values of the soil contamination by Cs-137 at all the places where measurements were made. The map is underlaid with the official contamination distribution as published by the IAEA [12].

It can easily be recognized that our results adequately correspond to the official data given by the underlaid background. Nevertheless, there are some remarkable details:

1. We observe striking differences for the local mean values of our soil measurements in the highly contaminated areas in subsequent years for some places. It must be pointed out that in general the mean values are based on different numbers of measurements and different measuring points from year to year. Even if all these conditions were unchanged a fluctuation by a factor of 2 may be possible.
2. All measurements were carried out for undisturbed and disturbed soil (arable land). For the same initial cesium contamination the area dose rate 1 m above ground is about a factor of 2 higher over undisturbed ground in comparison to fields. The lowest area dose rate was found above streets where the rain washed off the initial contamination.

Altogether, 300 soil contamination measurements were carried out in 60 settlements. Tab. 2.1 gives special results in some more highly contaminated villages in Russia.

village/district	1986*	1991			1992			1993		
	M	N	M	MM	N	M	MM	N	M	MM
Beresovka/Klincy	-	-	-	-	7	238	206/266	2	204	170/244
Nikolayevka/Kr.Gora	-	-	-	-	3	1280	776/1732	-	-	-
Schiryayevka/Gord.	-	-	-	-	3	1203	717/1783	-	-	-
Unecha/Klincy	600	-	-	-	4	519	362/662	3	431	297/607
Ushcherpye/Klincy	800	9	566	141/913	4	409	358/1579	2	1079	964/1208
Veprin/Klincy	1100	4	1001	519/1304	4	1030	866/1921	2	855	815/898

Table 2.1: Cesium soil contamination in villages of the Bryansk Region in Russia,
N number of measurements, M mean value, MM min./max. values in kBq/m^2
* from [11]

Most of these measurements were carried out over uncovered and virgin ground (meadows). Only a few measurements were made over arable land (fields) with a distinct depth distribution of the activity compared with the exponential distribution of undisturbed soil. Deviations from the exponential depth distribution required special care especially for the in-situ measurements but did not affect the results for the surface-related soil contamination. The contamination over concrete or asphalt covered ground was significantly lower due to natural and man-made decontamination processes.

The soil contamination within the settlements varied by a factor of about 2. In some cases, like the Russian village of Ushcherpye, the distribution of the surface activity was more inhomogeneous with a factor of 4 between the minimum and the maximum values. Fig. 2.4 gives a typical distribution of the soil contamination in the village of Partizan. These measurements were performed by Russian scientists [13] with a special measuring device called CORAD, which was developed at the Kurchatov Institute, Moscow, and which was also employed during our measurements.

CORAD has a NaI detector measuring the Cs-137 contamination within a radius of 0.8 m by in-situ measurements with the 3 energy channel technique. The measuring time is only about 10 min. Due to this and to the fact that it does not depend on knowledge of the depth distribution it is very suitable for large-scale measurements such as the surface activity distribution in settlements.

During our measurements we carried out intercomparison tests with other devices. In tab 2.2 all the results from the CORAD system and the other systems are compiled. In the last two columns the mean values of all measurements with and without the CORAD system are given. The agreement is good and a systematic deviation of the CORAD results cannot be observed. The results cover a range from 90 to 3600 kBq/m^2.

			soil contamination (kBq/m^2)				mean values	
location	district	data	layer sampling$^{1)}$	mix samplers$^{2)}$	In situ$^{3)}$	CORAD$^{4)}$	$^{5)}$	$^{6)}$
Perevos	Novosibkov	30.06.93	151	---	322	179	217 ± 91	236 ± 121
Gorodok	Korma	06.07.93	292	393	371	317 ± 18	343 ±47	352 ± 53
Otor	Chechersk	07.07.93	209	192	198	256 ± 86	214 ± 29	200 ± 9
Sebrovichi	Chechersk	07.07.93	455	771	894	947 ± 197	766 ± 220	707 ± 226
Yaseni	Bragin	12.07.93	187	162	142	88 ± 14	145 ± 42	164 ± 23
Savichi	Bragin	12.07.93	2853	3275	7128	3566 ± 813	4205 ± 1970	4419±2356
Soboli	Bragin	13.07.93		294 109		142 ± 64	182 ± 99	202 ± 130

Table 2.2: Intercomparison measurements with the CORAD system

$^{1)}$ Summation of activity in all layers and extrapolation of the sampling cross-section from 9.6 cm^2 to 1 m^2

$^{2)}$ Evaluation of mixing sample over all layers and extrapolation from 9.6 cm^2 to 1 m^2

$^{3)}$ In-situ gamma spectrometry for exponential depth profile, infinite area

$^{4)}$ CORAD results, finite area

$^{5)}$ Mean value of all results

$^{6)}$ Mean value of results except CORAD

The mean values are given in figure 2.1 - 2.3. The 3-year values in fig. 2.1 do not show any systematic decrease with time. The physical decay of Cs-137 causes an activity decrease of about 2.5 % per year, but this cannot be observed in the results presented in fig. 2.1 - 2.3 due the scattering of the measurements by a factor of 2. Other efficient reduction processes like migration, resuspension or transfer into the biosphere obviously do not prevail.

In addition, from the second column of table 2.1, where the mean values of the cesium soil contamination in 1986 are given in kBq/m^2 for three Russian villages according to [11], a reduction could at most be observed in the magnitude of the physical decay.

To confirm the assumption made above that the strontium and plutonium contamination outside the inner zone is negligible, some soil samples from Russia were also evaluated in German radiochemical laboratories, mainly in Jülich, to determine these nuclides.The results of the strontium analysis are summarized in table 2.3. Even though the picture is not very uniform it seems that the ratio of the activity concentration of Cs-137:Sr-90 is smaller for arable land than for meadows and pastures. This may be related to the fact that the more mobile strontium has already migrated to a greater extent out of the sampling layer down to the untouched soil.

district	ground	number of samples	MM Cs-137 Bq/kg	MM Sr-90 Bq/kg	MM ratio Cs/Sr
Klinzy	meadow	9	80/7500	5/150	24/97
Ludinovo	meadow	1	1100	10	111
Uzlovaya	arable	9	160/460	7/25	14/33
Ulyanovo	meadow	4	1200/2400	1/10	150/1800

Table 2.3: Sr-90 content in soil samples from Russia, 1991/1992,
MM= min./max. value

Due to different transfer factors from soil to plants, the ratio between Cs-137 and Sr-90 alters in the biosphere. In three grain samples from the Klincy district this ratio was below 7 while in dried mushrooms from the same district we found ratios of up to 55 000.

In addition to the measurements of the surface contamination the area dose rate was determined in the same places. The results for all measuring periods are presented in fig. 2.5 - 2.7.

The dose rate measurements strongly depend on the soil characteristics. The results are higher above undisturbed ground as in Nikolayevka and Shiryayevka where the measurements were made above pastures. The results are lower above arable land and covered sites where translocation or decontamination processes took place. Nevertheless, there is a strong relation to the official ground contamination data underlaid in shades of grey, but again there is no evidence for time dependence.

The strong relation between the cesium ground contamination and the area dose rate equivalent dose in tissue is confirmed by our measurements. Fig. 2.8 presents this correlation for the results above pastures and fields in the Klincy district. Although a strong relation is apparent, one has to keep in mind that there are some disturbing factors such as alterations in the Cs-134 - Cs-137 relation and in the natural background radiation from the uranium and thorium decay chain. The latter effect can be assessed by the constant part of the correlation line which corresponds to the segment on the Y-axis. The gradient of the correlation line for undisturbed ground (pastures) corresponds to a dose rate-ground contamination relationship of about 1 µSv/h for 1000 kBq/m^2 of Cs contamination on soil. The regression line can be presented in the form $y = b + m\,C$, where y is the area dose rate in µSv/h, C is the soil contamination in kBq/m^2, b = 0.2 µSv/h and m = 0.001 $(\mu Sv/h)/(kBq/m^2)$. The correlation coefficient is r = 0.89. For arable land (fields) the regression line is characterized by the numbers b = 0.2 µSv/h, m = 0.0005 $(\mu Sv/h)/(kBq/m^2)$ and r = 0.89.

The most important factor one has to observe for dose rate and for in-situ measurements is the depth profile of the cesium distribution. As already mentioned, this depends mainly upon the question of ground cultivation. For undisturbed soil there is an exponential depth distribution while for arable land the distribution is more homogeneous. However, for undisturbed soil some problems may arise if the depth distribution is not exponential-like. Fig. 2.9 presents two profiles from the year 1992 , one is normally shaped while the other one shows a deviation at a depth of 10 cm which is hard to explain.

Summer 1993 was extremely wet in Russia. Thus, the soil humidity increased resulting in an enhancement of the soil density of about 10 %. At the end of the 1993 campaign we measured area dose rates and soil contaminations in the previously flooded areas of Rovno. These values are presented in table 2.4 and are incorporated in the correlation between area dose rate and soil contamination (Fig. 2.8). There was no significant deviation from the other values.

Surface water was only slightly contaminated. From 96 drinking water measurements in 1991, 12 samples were identified with more than 18 Bq/l which is the official limit value in Russia. These samples were not taken from the official supply system but from lakes and wells. In 1992 only one result from 44 measurements was above 100 Bq/l. One further result was 21 Bq/l, all the other concentration values for surface water were below the official limit.

village	district	Cs soil contamination, kBq/m^2	area dose rate $\mu Sv/h$	type
Ozera	Dubroviza	11	0.13	schoolyard
Shachi	Dubroviza	28	0.20	schoolyard
Cheremel	Dubroviza	78	0.22	meadow
Cheremel	Dubroviza	30	0.15	meadow
Budimlya	Dubroviza	8	0.23	meadow
Budimlya	Dubroviza	25	0.23	meadow
Velyun	Dubroviza	53	0.45	meadow
Dubroviza	Dubroviza	100	0.19	stabilized
Staroye Selo	Rakitnoye	50	0.17	meadow
Drosdyn	Rakitnoye	54	0.16	schoolyard
Drosdyn	Rakitnoye	48	0.10	meadow
Drosdyn	Rakitnoye	12	0.13	arable land
Rakitnoye	Rakitnoye	13	0.13	meadow

Table 2.4: Measurement of the soil contamination and the area dose rate in the Rovno region in October 1993

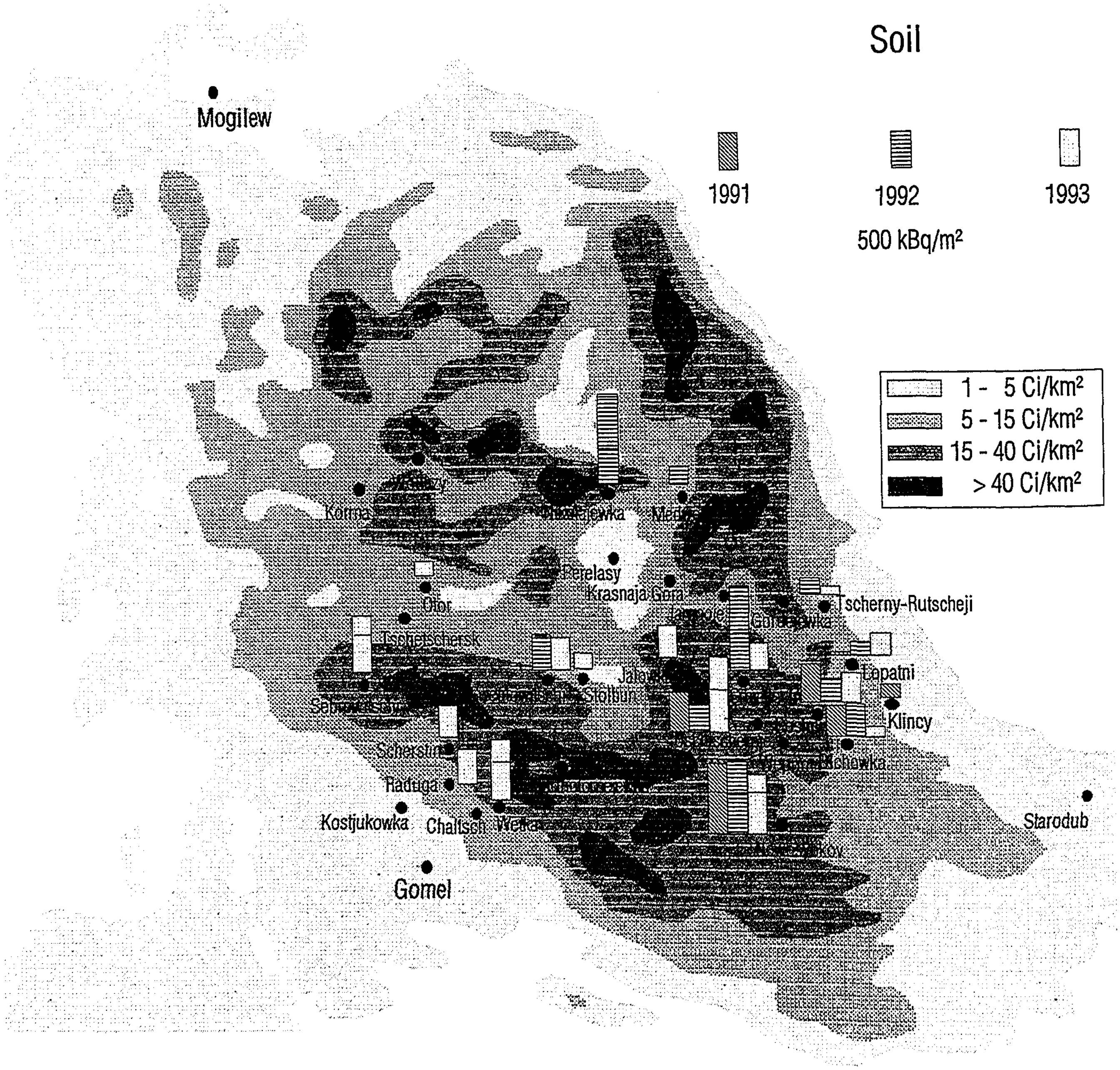


Fig. 2.1: Soil contamination in the Bryansk-Gomel spot in 1991-1993

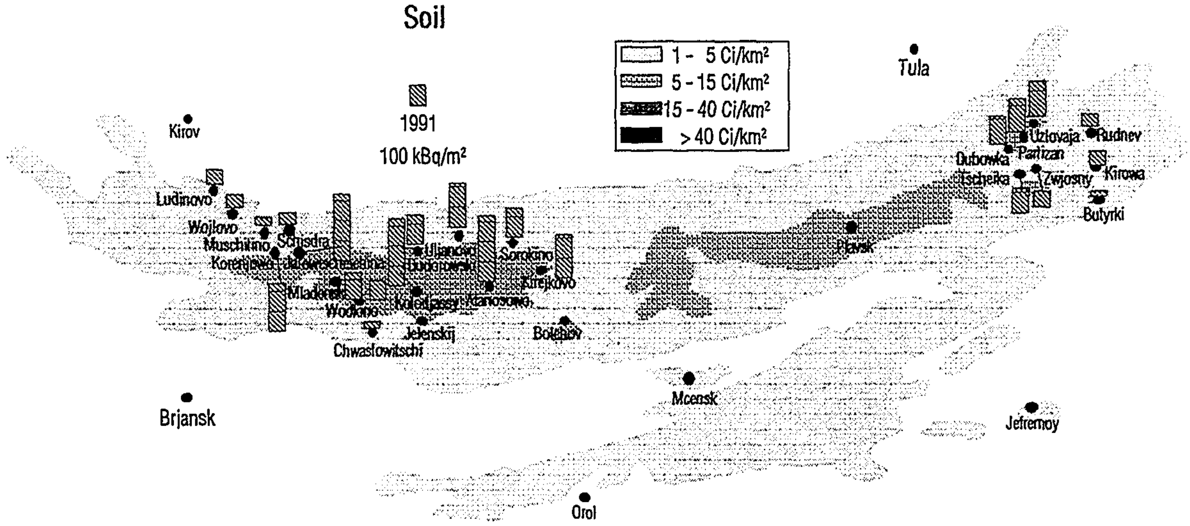


Fig. 2.2: Soil contamination in the Tula-Kaluga-Orel spot in 1991

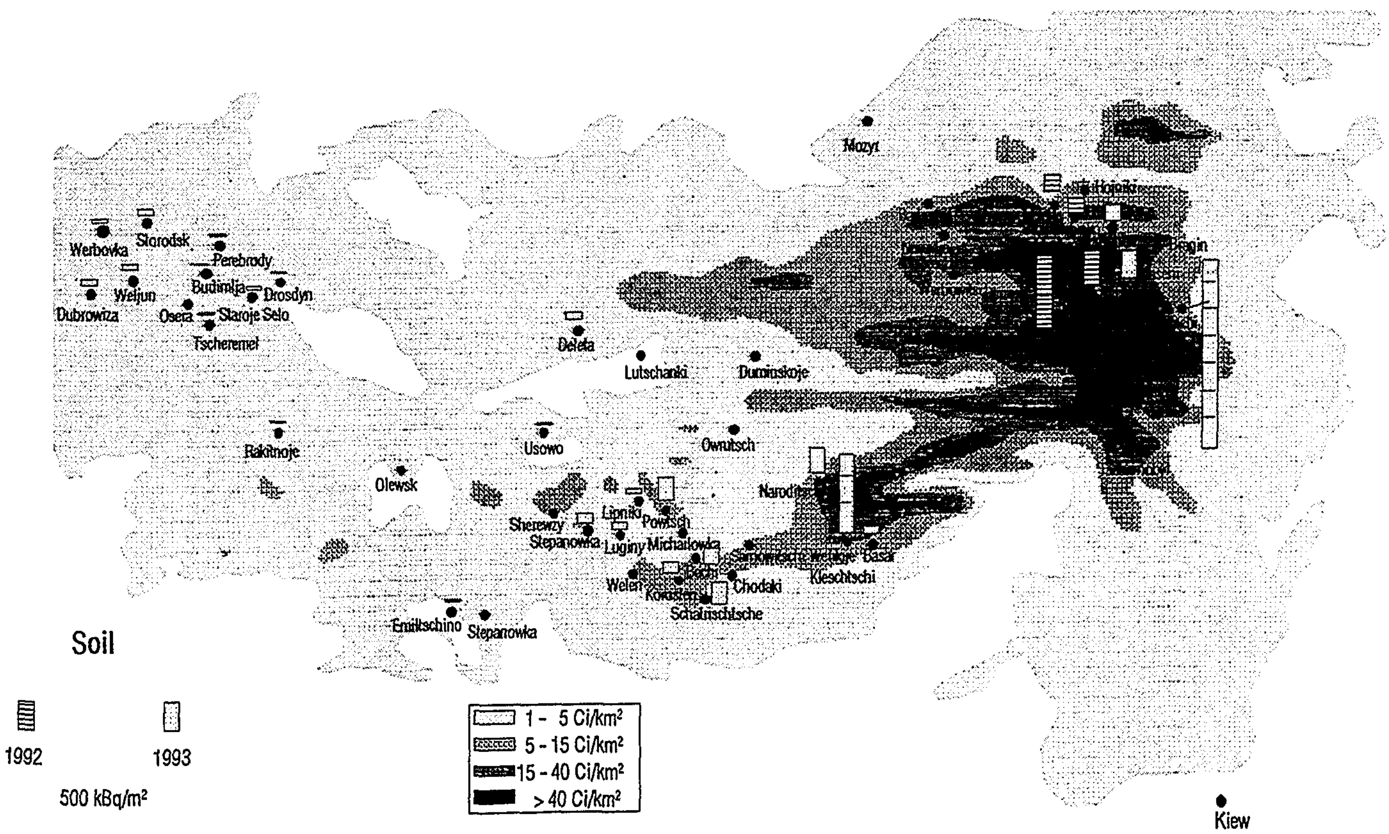


Fig. 2.3: Soil contamination in the central spot in 1992-1993

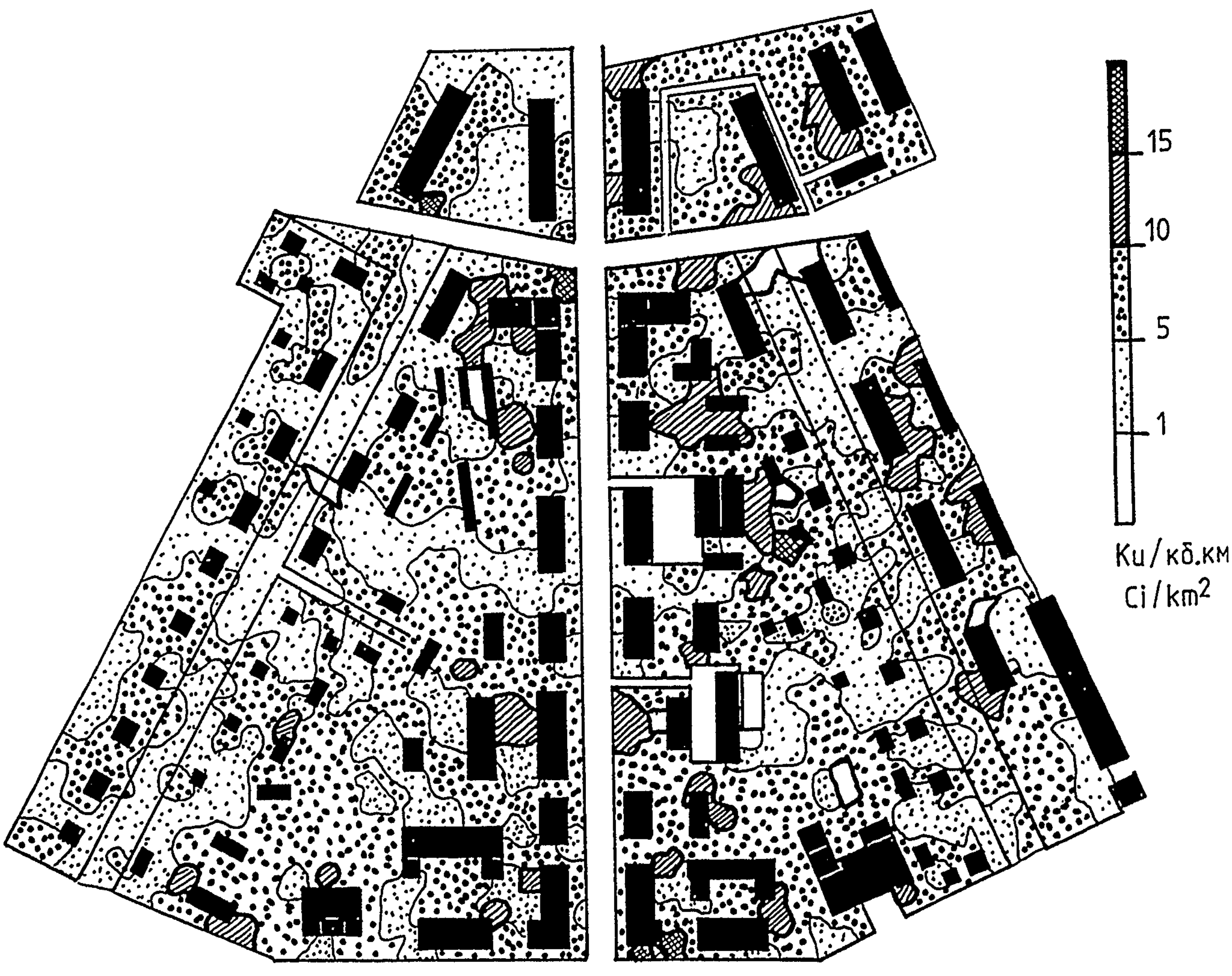


Fig 2.4: The contamination pattern of Partizan measured with the CORAD system by Urutskoev [13]. (The black spots indicate buildings with undetermined contamination)

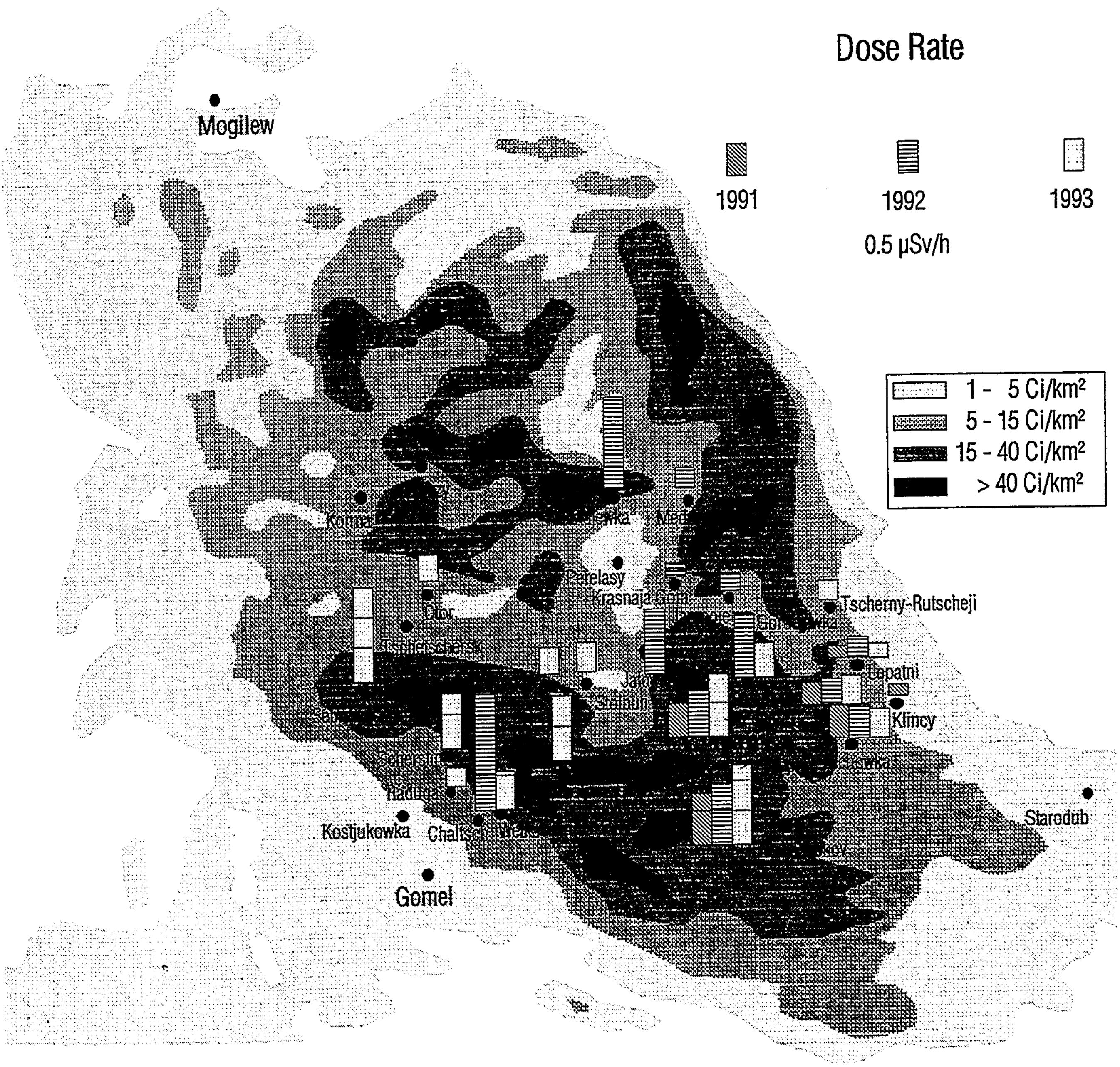


Fig. 2.5: Area dose rate in the Bryansk-Gomel spot in 1991-1993

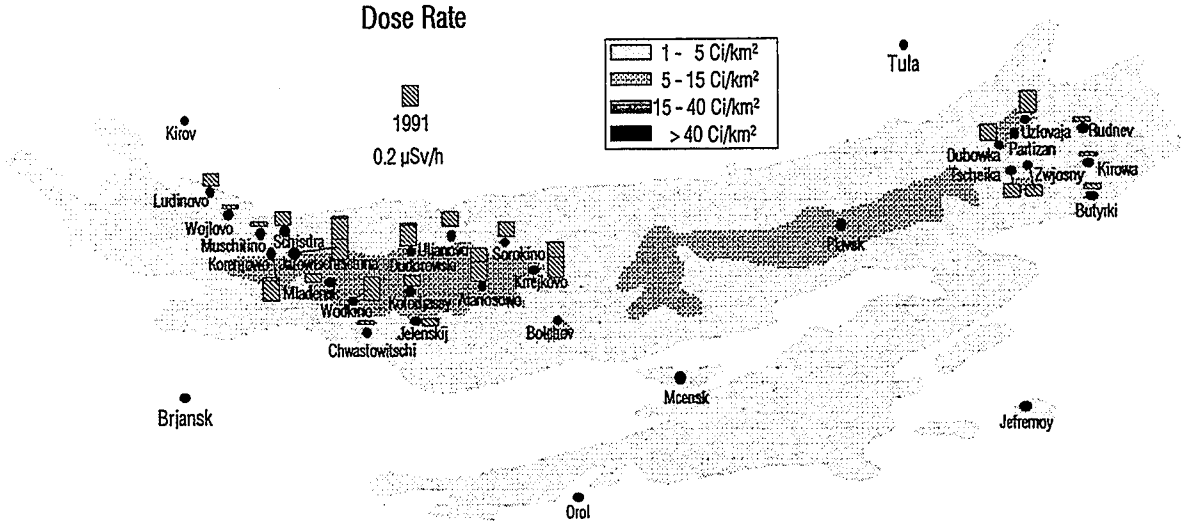


Fig. 2.6: Area dose rate in the Tula-Kaluga-Orel spot in 1991

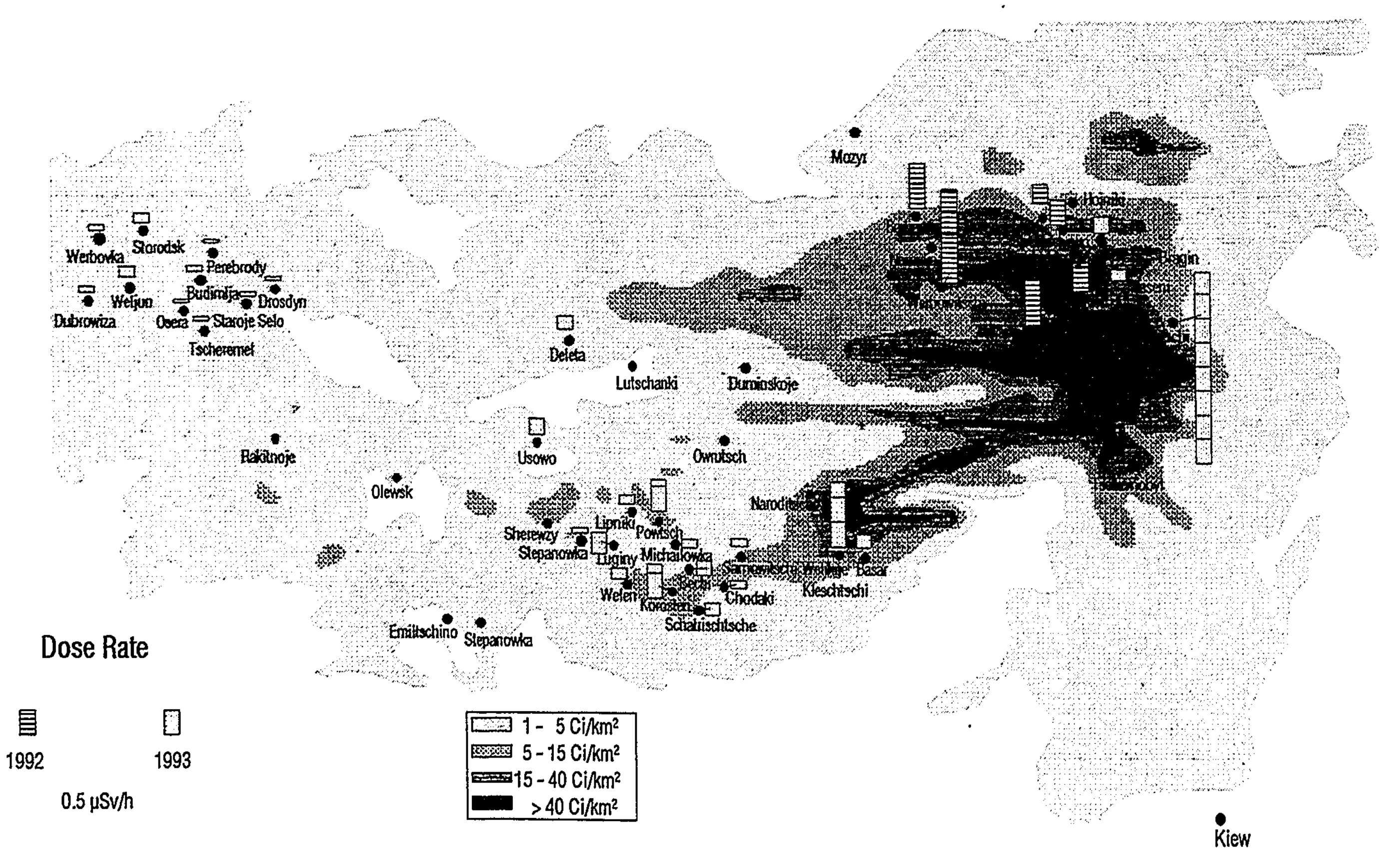


Fig. 2.7: Area dose rate in the central spot in 1992-1993

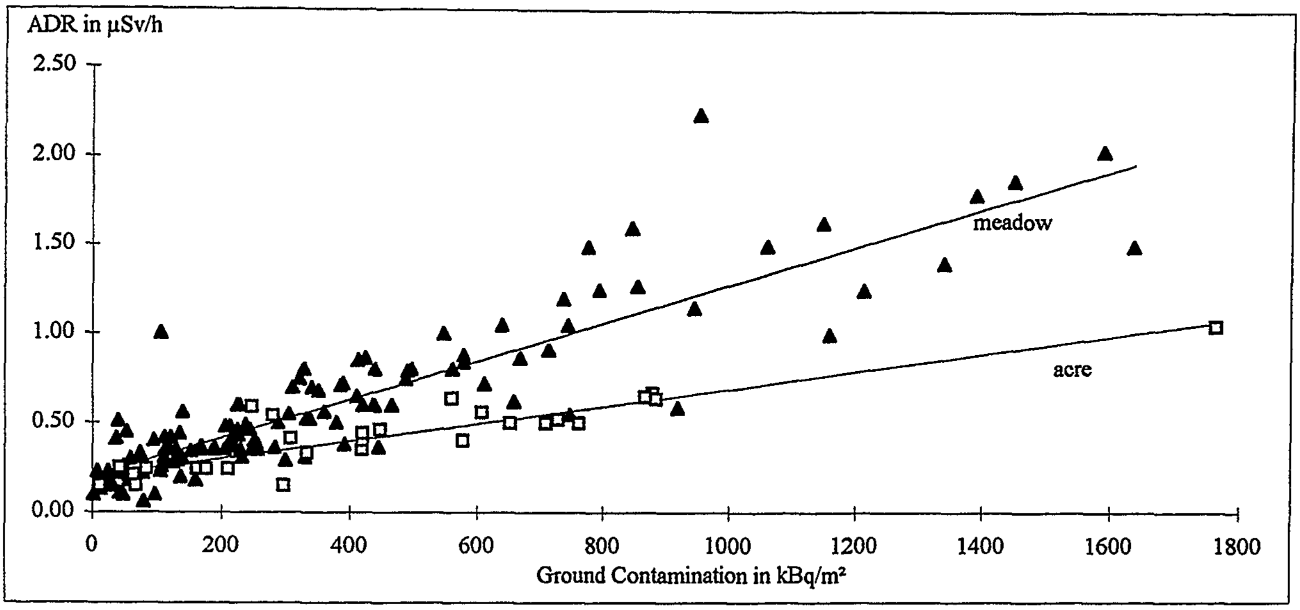


Fig. 2.8: Relation between area dose rate (ADR) and ground contamination measured in 1992 and 1993

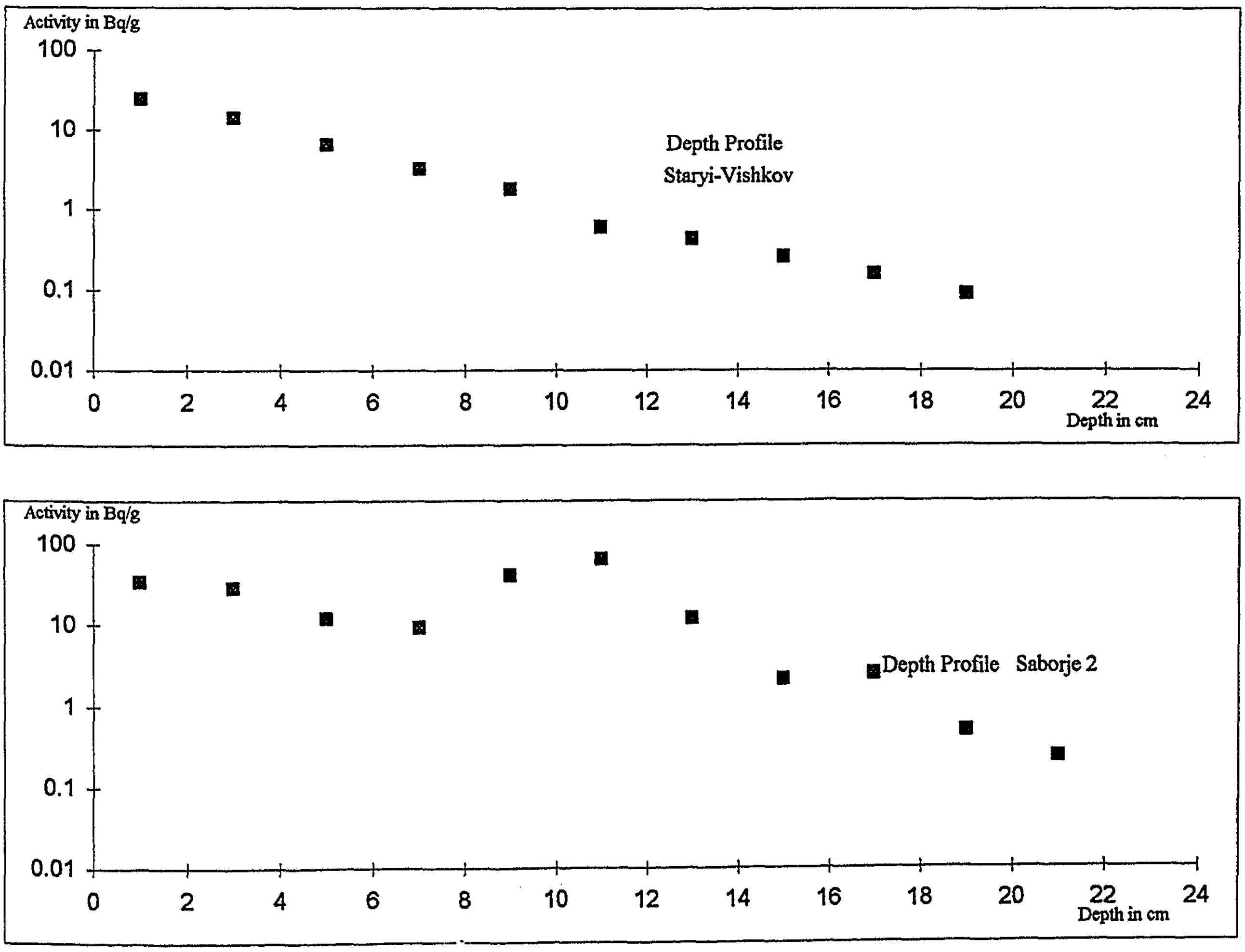


Fig. 2.9: Depth profiles of Cs activity in undisturbed soil

2.2 Transfer to the Biosphere and Food Contamination

The transfer of soil contamination into the biosphere mainly depends on two factors, the activity concentration and the soil characteristics. Therefore, the activity in food or in the human body may differ significantly between areas which are almost equally contaminated. In general, the highest contamination levels in the biosphere were found in areas where sandy soils with low clay content (e.g. turf-podzol, peat soils and sandy loam) predominated. Such soil types prevail in the western part of the Tula-Kaluga-Orel spot around the town of Ludinovo, in the Bryansk-Gomel spot round Klincy and almost outside the central spot in the districts of Rovno (Ukraine) and Brest (Belarus). In these areas the body activity is higher than expected with respect to the soil contamination. On the other hand, the internal contamination of the population in Uzlovaya near the city of Tula was very low, although the soil contamination there is comparable to that of Ludinovo. This is due to the fact that in the area round Uzlovaya chernozem soils and soils with high lime content prevail.

The accident occurred during spring and, consequently, the surface contamination accumulated on grass and early crops like lettuce and vegetables. On areas with a Cs-137 contamination of 1000 kBq/m^2 Russia scientists measured an activity for this kind of biomass of between 1000 and 500,000 Bq/kg in May 1986. By the end of summer 1986, the levels of surface contamination on the biomass had fallen by two or three orders of magnitude because of weathering and biomass growth [11]. Thereafter root uptake and resuspension became progressively more important.

From autumn 1986 till spring 1987 the Cs-137 content in crops and vegetables was between 200 and 300 Bq/kg in the more highly contaminated areas (1000 kBq/m^2) of the western districts of the Bryansk region. The contamination of fruit and greenstuff in these areas was about 500 Bq/kg, forest berries had an average of 900, mushrooms and local fish about 1100 Bq/kg, according to Russian measurements reported in [11]. This contamination changed only slightly with time. In spring 1988 a slight decrease was observed in crops (100 Bq/kg), vegetables (200 Bq/kg) and greenstuff (400 Bq/kg). A stronger decrease was found in local fish (500Bq/kg) and forest berries (630 Bq/kg). The contamination of fruits (600 Bq/kg) and mushrooms (1300 Bq/kg) increased slightly.

In principle, this tendency could be confirmed by our measurements from 1991 till 1993. The village of Veprin in the Klincy district is a typical example. According to tab. 2.1 in section 2.1, we found an average soil contamination of about 1000 kBq/m^2 there. In 1991 the activity in fruit (strawberries, redcurrants, cherries) was between 0 and 1000 Bq/kg, vegetables and greenstuff were contaminated by up to 500 Bq/kg and our results for forest berries (3 samples of raspberries, 2 samples of blueberries) were between 930 and 2900 Bq/kg. The Cs-activity of one sample of local fish was 7600 Bq/kg, two samples of pickled or dried mushrooms had more than 12,000 and about 66,000 Bq/kg, respectively.

The results in 1992 and 1993 were significantly lower. Three measured samples of local fish and samples of berries (strawberries, raspberries) were contaminated below 110 and 510 Bq/kg, respectively. Tab. 2.5 gives a survey of all our food measurements in three districts of Russia, in Belarus and in the Ukraine.

region	mushrooms and forest produce (Bq/kg)	garden produce (Bq/kg)	grain and grain products (Bq/kg)	meat (Bq/kg)	fish (Bq/kg)
Klincy	< 18 - 12320	< 2.5 - 1270	79	4 - 4590	23 - 1170
Gordeyevka	89 - 18000	< 4 - 514		< 20 - 820	71 - 86000
Krasnaya Gora	119 - 16900	1.5 - 758	< 3 - 8	27 - 500	28 - 2925
Belarus	< 10 - 64000	< 4.1 - < 300	16 - 25	23 - 7311	61 - 607
Ukraine	< 7.6 - 304	< 1 - < 10		< 1	

Table 2.5: Range of foodstuff measurements 1991 - 1992

Potatoes, Vegetables:	592 Bq/kg
Meat	740 Bq/kg
Milk	370 Bq/l
Drinking Water	18 Bq/l
Oil	185 Bq/kg
Bread, Flour, Sugar	370 Bq/kg
Mushrooms, Wild Fruits	1480 Bq/kg
Dried Fruit	2960 Bq/kg
Dried Mushrooms, Tea	7400 Bq/kg

Table 2.6: Contamination limits for foodstuff in the CIS (former Soviet Union)

In tab. 2.7 - 2.9 the results of food measurements are given for the three republics according to the classification 1 to 3:

Class 3 was highly contaminated foodstuff with activity concentrations above the official limits for the three republics given in tab. 2.4. Class 2 was foodstuff moderately contaminated above 50 Bq/kg but below the official limit. Class 1 involves slightly contaminated foodstuff below 50 Bq/kg. For drinking water only two classes were defined:

class 3 > 18.5 Bq/l and
class 2 < 18.5 Bq/l.

Table 2.7: Classification of measurements on the radioactivity in foodstuff in Russia

	1991				1992				1993			
	class 1	class 2	class 3	samples	class 1	class 2	class 3	samples	class 1	class 2	class 3	samples
Potatoes, Vegetables	89.6 %	10.4 %	0 %	183	57 %	43 %	0 %	100	76.9 %	15.4 %	7.7 %	13
Meat	43.9 %	48.8 %	7.3 %	41	25.9 %	66.7 %	7.4 %	27	33.3 %	66.7 %	0 %	3
Milk	40.2 %	50.1 %	9.7 %	702	19.1 %	54 %	27 %	504	3.3 %	34.1 %	62.6 %	91
Drinking Water		87.4 %	12.6 %	95		80 %	20 %	5		100 %	0 %	3
Fats, Oil	50 %	50 %	0 %	2	66.7 %	22.2 %	11.1 %	9	33.3 %	66.7 %	0 %	3
Bread, Flour, Sugar	100 %	0 %	0 %	4	0 %	0 %	0 %	0	0 %	0 %	0 %	0
Mushrooms, Wild Fruits	14.3 %	68.4 %	17.3 %	133	3.7 %	69.5 %	26.8 %	82	12.5 %	50 %	37.5 %	8
Dried Fruit	0 %	100 %	0 %	3	0 %	0 %	0 %	0	0 %	0 %	0 %	0
Dried Mushrooms, Tea	0 %	66.7 %	33.3 %	15	0 %	100 %	0 %	3	0 %	0 %	0 %	0
< detection limit				377				263				36
Without classification				502				324				40
Total				2057				1317				197

Table 2.8: Classification of food measurements in Belarus

	1992				1993			
	class 1	class 2	class 3	samples	class 1	class 2	class 3	samples
Potatoes, Vegetables	89.4 %	10.6 %	0 %	66	66.7 %	33.3 %	0 %	6
Meat	14.3 %	28.6 %	57.1 %	7	0 %	0 %	100 %	1
Milk	71.4 %	28.6 %	0 %	21	72.7 %	27.3 %	0 %	22
Drinking Water		83.3 %	16.7 %	6		75 %	25 %	4
Fats, Oil	0 %	100 %	0 %	4	0 %	0 %	0 %	0
Bread, Flour, Sugar	0 %	0 %	0 %	0	0 %	0 %	0 %	0
Mushrooms, Wild Fruits	46.7 %	53.3 %	0 %	15	4.4 %	29.4 %	66.2 %	68
Dried Fruit	0 %	100 %	0 %	2	0 %	0 %	0 %	0
Dried Mushrooms, Tea	0 %	0 %	100 %	1	0 %	41.7 %	58.3 %	12
< detection limit				71				66
Without classification				74				67
Total				267				246

Table 2.9: Classification of food measurements in the Ukraine

	1992				1993			
	class 1	class 2	class 3	samples	class 1	class 2	class 3	samples
Potatoes, Vegetables	100 %	0	0	2	91.1	8.9	0	101
Meat	0 %	0	0	0	33.3	66.7	0	12
Milk	0 %	0	0	0	52.8	41.3	5.9	489
Drinking Water		0	0	0		100	0	4
Fats, Oil	0 %	0	0	0	0	100	0	1
Bread, Flour, Sugar	0 %	0	0	0	0	0	0	0
Mushrooms, Wild Fruits	0	100	0	1	9	68.1	22.9	144
Dried Fruit	0	0	0	0	33.3	33.3	33.3	3
Dried Mushrooms, Tea	0	0	0	0	0.9	26.1	73.0	115
< detection limit				21				216
Without classification				11				170
Total				35				1255

The contamination of milk and milk products is most important for the internal exposure of humans. Since 1986, the concentration of cesium in milk has had a cyclically decreasing character [11]. Maximum values occurred during summer periods, when animals were grazing on pastures, and minimum values were measured during the winter periods, due to the consumption of less contaminated food, including silage, root crops, and hay. On average, the levels of contamination in privately produced cows' milk was 1.2 times higher in summer and 2 times higher in winter than the levels on collective farms.

For the long-term decrease of the cesium transfer from soil into grass and, subsequently, into milk a half-life of 1.2 years is given from Russian measurements according to [11] for the first 4-year period of observation. In 1986 this effect was superposed by the process of natural weathering which caused a greater decrease in the first few days after the accident with a half-life of 22 days. Thus, in Novosybkov the Cs-137 content in milk from private farms was reduced from 11,000 Bq/kg in May 1986 to 1300 Bq/kg in August 1986 and 700 Bq/kg in summer 1987. For the same period, the milk from state shops decreased from 700 Bq/kg to 400 Bq/kg and then stayed on this level till summer 1987.

In 1991 we found that the average milk concentration east of Novosybkov in the Klincy district was about 130 Bq/kg, on the basis of 227 milk samples from private farms. But maximum values were still high, the highest was 3700 Bq/kg. In this district, the official limit of 370 Bq/kg was exceeded for 25% of all results. Tab. 2.10 gives a summary of the results of our milk measurement in Russian settlements in 1991. A survey of the average milk contamination in all settlements visited by our measuring teams from 1991 to 1993 is given in fig. 2.10 - 2.12.

district	number of measurements	number of results				maximum value Bq/l
		above 370 Bq/l Cs 137		above 2000 Bq/l Cs 137		
		absolute	relative	absolute	relative	
Uzlovaja	22	0	0%	0	0 %	20
Ludinovo	442	9	2 %	0	0%	973
Klincy	227	57	25 %	10	4.4 %	3720

Table 2.10: Summary of the 1991 milk measurements in Russia

The Klincy district was most intensively investigated during our measuring campaign. Therefore, many special results can be evaluated from these measurements. Table 2.11 gives the mean values of the area dose rate and the cesium contamination of soil and milk for the settlements investigated in this district. Comparing the annual results in the villages of Guta Korezkaya, Lopatni, Olchovka, Rozny, Ushcherpye and Veprin, where measurements were performed in all three years we find no time trend for the area dose rate and the soil contamination. The variation of the mean value per year is 7 %, which is much less than the variation of the single measurements and their errors. In

1993 the area dose rate measured in this district varied between 0.16 and 1.30 µSv/h, values which can be found in Germany due to natural radiation.

A linear regression calculation confirms that the milk concentration decreases with time. According to fig. 2.13 the gradient of the regression line for eight villages of the Klincy district from 1991 to 1992 is 0.53, which confirms the half-life of 1.2 years according to [11]. The correlation coefficient is 0.99. In 1993 the gradient was lower. In the three Russian villages of Blizna, Guta Korezkaya and Lopatni the milk concentration even increased from 1992 to 1993. The reason for this effect may be found in the following items:

- The sampling conditions were not completely identical.
- Due to the higher precipitations the Cs-transfer into the grass was increased in 1993.
- Due to the worse economic situation in 1993 the population did not use so much fertilizer and consumed more home-produced milk even in areas where it was prohibited.

village	ADR (µSv/h)			soil (kBq/m²)			milk (Bq/l)		
	1991	1992	1993	1991	1992	1993	1991	1992	1993
Beresovka		0.29	0.16		238	204		357	274
Blizna		0.25	0.18		176	130		461	707
Dushkino	0.25			229			104		
Golubovka	0.06	0.46		24	286				
Gulyovka	0.26			377					
Guta Korezkaya	0.53	0.56	0.50	962	625	443	666	535	868
Kivai	0.33			374			135		
Klincy	0.18			193			130		
Lopatni	0.19	0.33	0.23	35	195	319	212	382	666
Maliy Topal	0.22			239			90		
Martyanovka	0.08			76			93		
Olchovka	0.53	0.55	0.48	438	465	147	117	165	77
Pablichi		0.13		8	32		65		
Pervoye Maya	0.22			110			212		
Pestshanka	0.09	0.16		71	66		122	78	
Roshny	0.36	0.42	0.48	540	340	424	561	388	161
Satishe	0.08			68					
Smolyevitschi	0.14			166					
Smotrovaya Buda	0.16			116			355		
Sossnovka	0.08			79					
Teremoshka		0.56			496				
Tulu Kovshina		0.45	0.47		345	381		604	258
Turosna	0.43	0.41		350	273		107	139	
Unetsha		0.5	0.62		519	431	2062	1201	1101
Ushcherpye	0.55	0.78	1.03	566	409	1079	884	519	278
Velekiy Topal	0.21			253			43		
Veprin	0.84	1.0	1.30	1001	1030	855	2128	1169	343

Table 2.11: Mean values of the area dose rate (ADR) and the cesium contamination of soil and milk in villages of the Klincy district, 1991-1993

In the Klincy district, a correlation was found between the milk concentration and soil contamination of pastures. The milk concentration in relation to the contamination of pastures for the years 1991 and 1992 is given in fig 2.14. The correlation is weak but evident. In 1993 the correlation could not be confirmed. This is another hint that in 1993 the situation was somewhat different due to the reasons mentioned above.

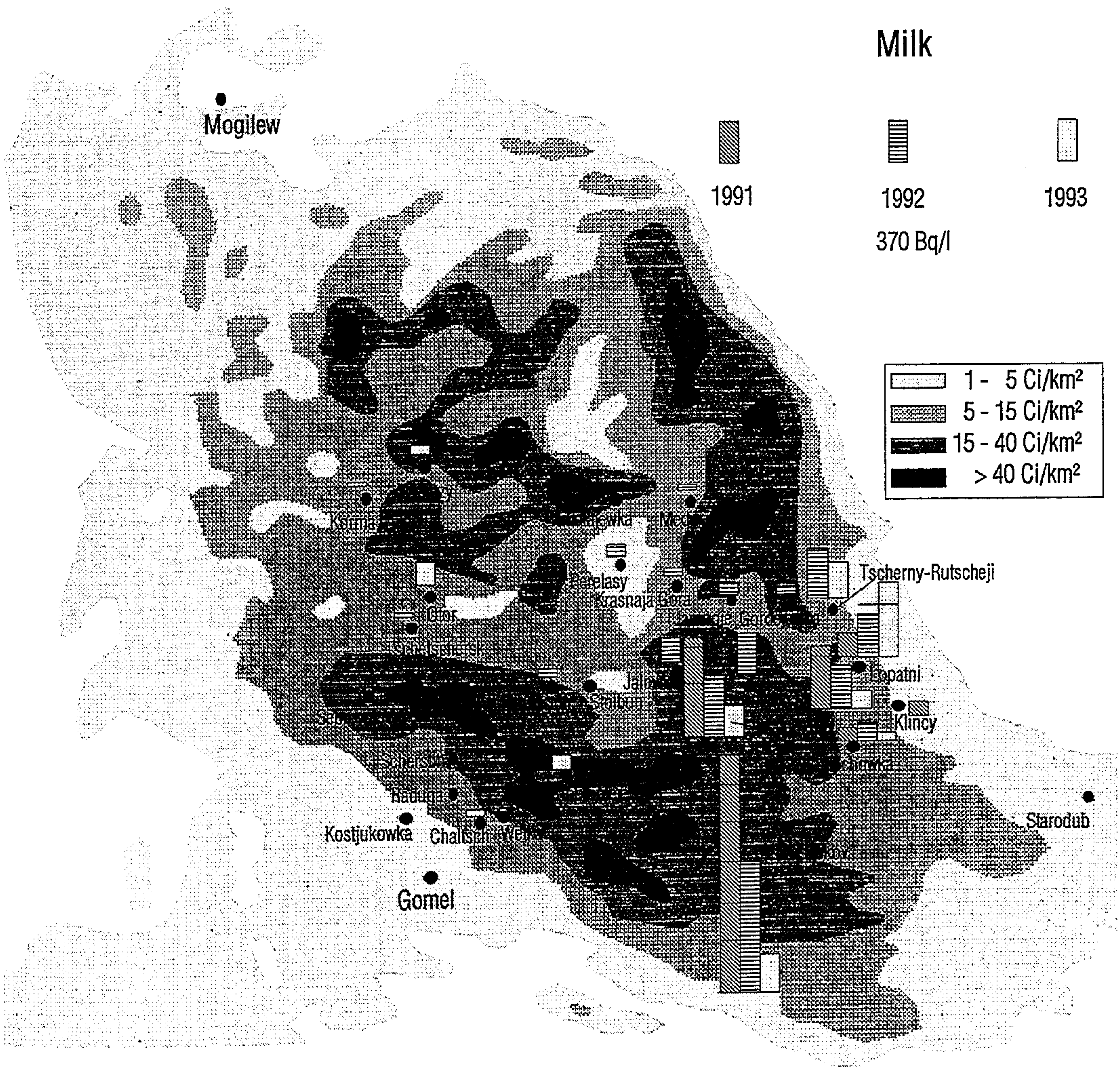


Fig 2.10: Milk contamination in the Bryansk-Gomel spot in 1991-1993

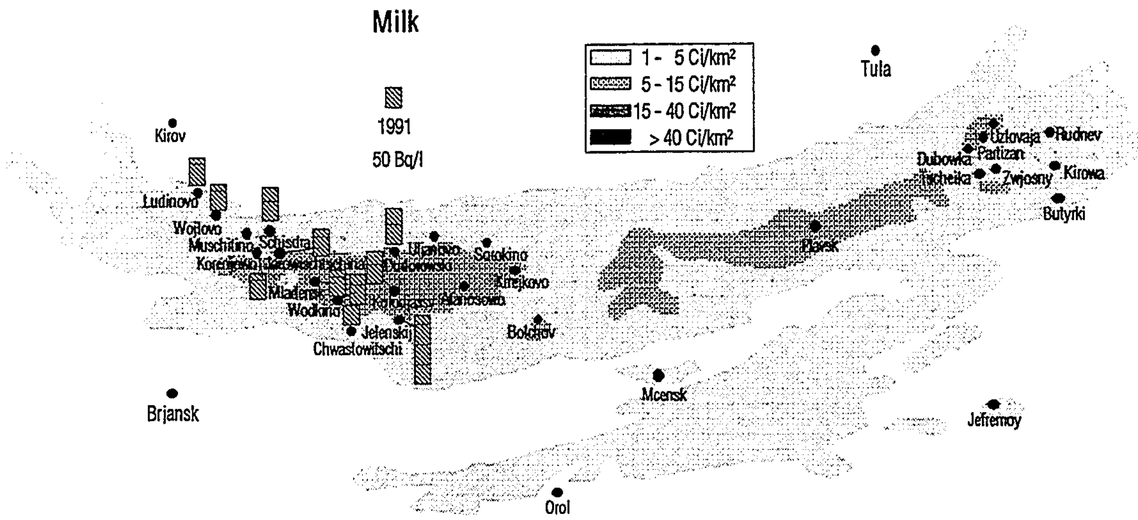


Fig 2.11: Milk contamination in the Tula-Kaluga-Orel spot in 1991

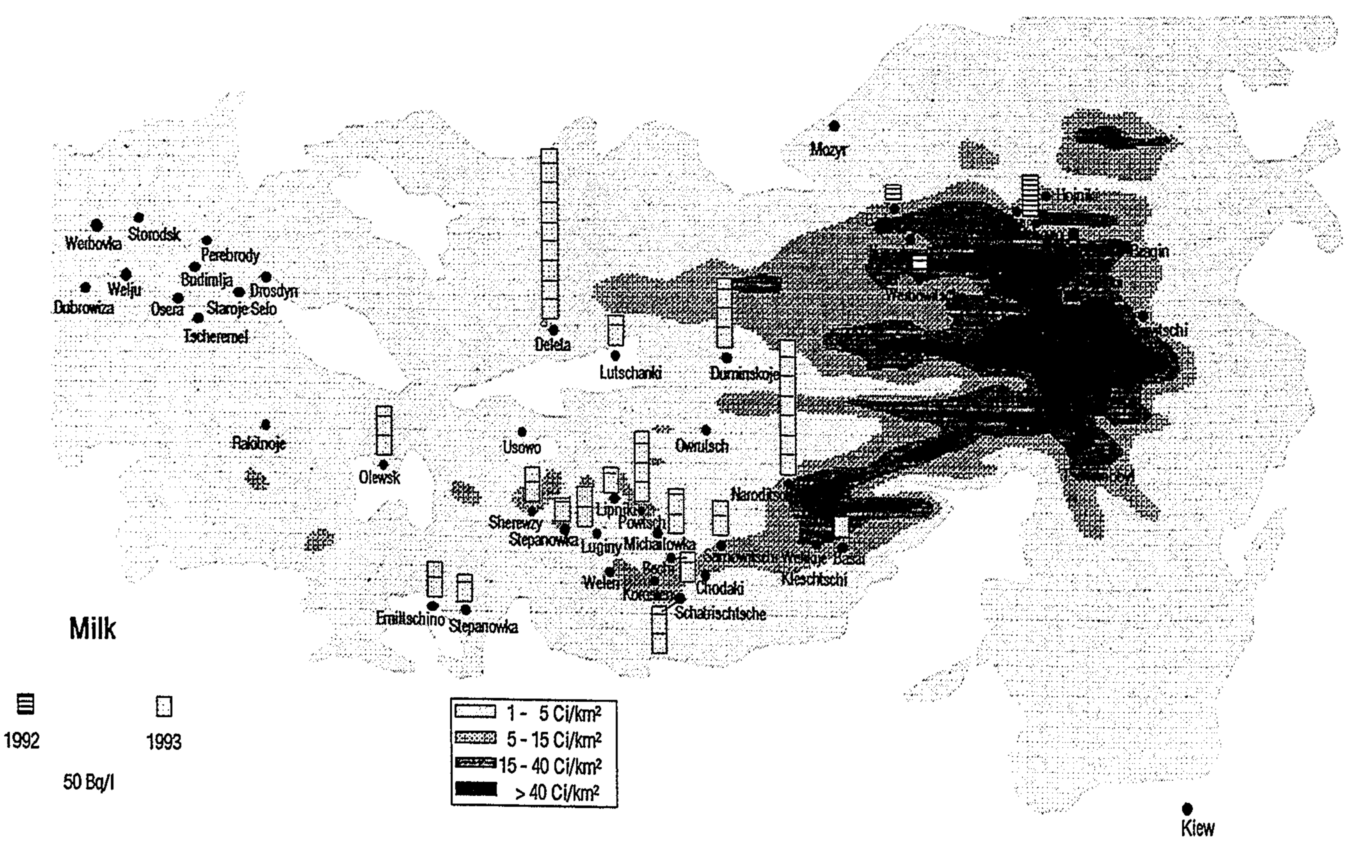


Fig 2.12: Milk contamination in the central spot in 1992-1993

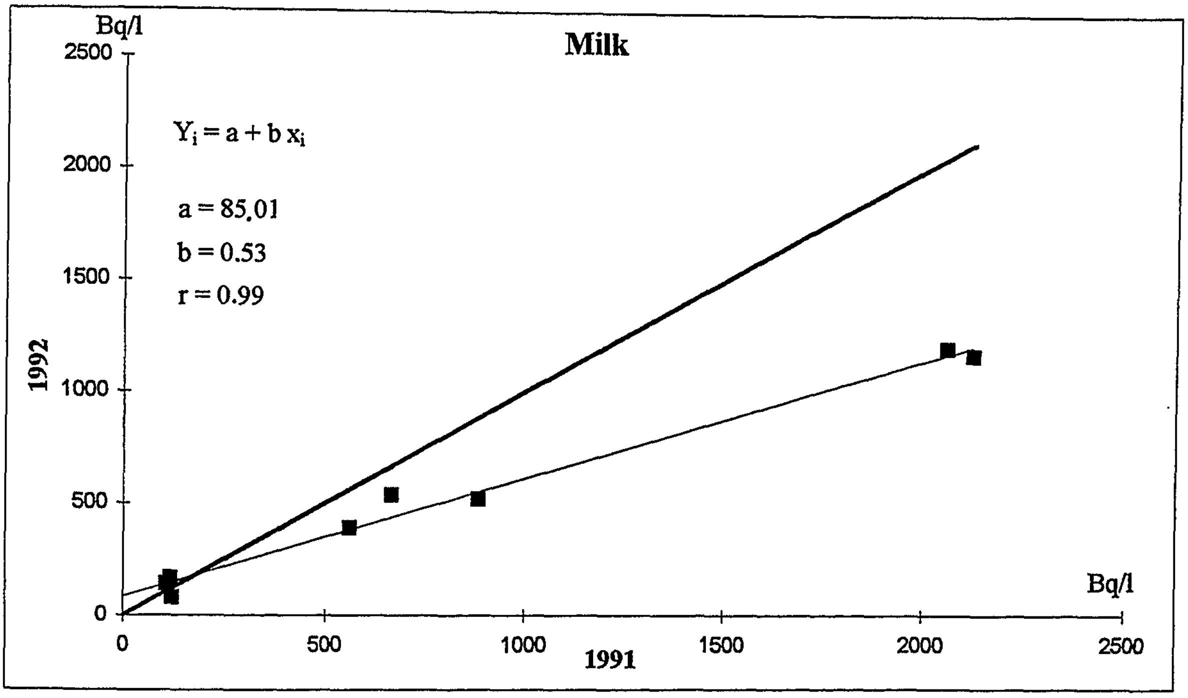


Fig 2.13: Correlation between the 1991 and 1992 results measured in the villages of Guta Korezkaya, Lopatni, Olchovka, Pestshanka, Rozny, Ushcherpye, Turozna and Veprin in the Klincy district

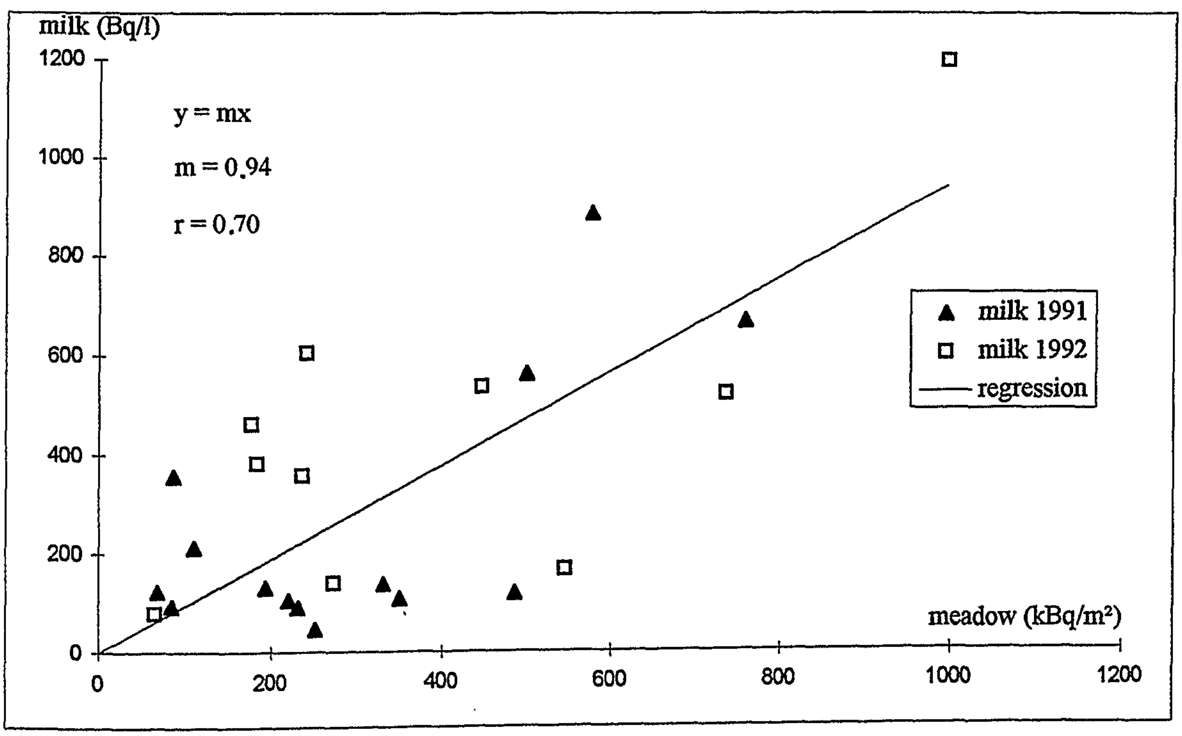


Fig. 2.14: Relation between Cs contamination in milk and soil contamination on meadows in 1991 and 1992

3. Radiation Exposure to the Population

3.1 Internal Contamination

A considerable part of the population in the European territories of the former Soviet Union received high radiation doses as a result of the Chernobyl accident. We have to distinguish four groups of exposed people:

- Group A- individuals who helped mitigate the accident (liquidators)
- Group B- former inhabitants of evacuated areas
- Group C- inhabitants of significantly contaminated areas who were not evacuated
- Group D- the remainder of the population.

Group A consists entirely of adults who worked within the 30 km zone. This category includes 600 000 individuals.

Altogether, 116 000 individuals were evacuated. They belong to group B. Among these were 49 000 inhabitants of the town of Pripyat evacuated on April 27 and 42 000 inhabitants of 83 villages in the 30km zone evacuated between May 4 and May 7.

About 4 million people belonging to group C live on a territory of more than 100 000 km^2 with Cs-137 surface activity levels exceeding 37 kBq/ m^2 (1 Ci/km^2). According to section 2.1 this is the so-called Observed Area (OA). Approximately 270 000 of these individuals live in the so-called Controlled Area (CA) , a territory of about 10 000 km^2 with Cs-137 surface activity levels exceeding 550 kBq/m^2 (15 Ci/km^2).

The remainder of the population in group D numbers approximately 280 million people with a relatively insignificant exposure from local contamination.

Our measurement campaign was mainly concerned with group C people. Since the accident in 1986 this group has been continuously exposed to external and internal radiation. The main contributors to the radiation dose were external exposure from deposited radionuclides and the ingestion of iodine or cesium isotopes, while direct external exposure from the plume and inhalation exposure in the plume or from resuspension were insignificant.

For the group C people living in the Observed and Controlled Area temporary dose limits were implemented after the accident. For the first year the dose limit adopted was 100 mSv. In 1988 and 1989 the limits were lowered to 30 mSv and 25 mSv, respectively. For the following years a concept was proposed limiting the whole lifetime dose after the accident to 350 mSv. However, this limitation was not implemented due to strong objections from the public. Today there is no general dose limit for the population. There is only a dose criterion for evacuation measures. If the annual dose is greater than 5 mSv/a evacuation is required, between 1 and 5 mSv/a evacuation must be performed according to the optimization principle.

Since the accident a large number of measurements have been carried out to determine the Cs content in humans. The Cs content is exclusively responsible for the internal dose. Fig. 3.1 presents the results of several hundred body counter measurements in the town of Novosybkov in the highly contaminated area with an average ground contamination of 800 kBq/m^2. In July 1986 the highest value of the internal contamination was measured for male adults with an average body activity of 120 kBq. Since then the body activity had a cyclically decreasing character with slightly higher values in the summer period due to the higher consumption rate of locally produced agricultural and forest produce [11].

In addition, fig. 3.1 shows similar but significantly higher curves for the small villages of Veprin and Unecha in the Bryansk region. Veprin has an average ground contamination of 1100 kBq/m^2 and the highest average body activity was measured in August/September 1986 for 72 individuals as 330 kBq for male adults. To complete the data from the early phase after the accident two curves from the Kaluga district measured for individuals from the villages are given in fig. 3.2 showing the development of the body activity in areas with a lower ground contamination.

These curves present the annual mean from 9600 and 14000 (in total) measurements in the villages of Ulyanovo and Shvastovichi, respectively [11]. But all these measurements were not clearly defined: the individuals and the measurement conditions were not identical for different years. Identical measurement conditions were only given for German measurements from the Berchtesgaden region in the southwest of Germany. From 1986 to 1989 three groups - male, female, children - were monitored continuously by the Institute for Radiation Hygiene on a whole body counter in Neuherberg near Munich. The results as presented in fig. 3.2 are a well-defined basis for dose reconstruction purposes. In addition, analogous results from adults in Jülich are given in fig. 3.2.

As a general rule, all these data make clear that the body activity in the more highly contaminated areas of Russia decreased more slowly than those in the less contaminated regions. From 1986 to 1990 the body activity decreased in Jülich by a factor of 50, in Berchtesgarden by a factor of 20 and in the Kaluga region by a factor of 5. The data from Kaluga seem to be uncertain. Data from Belarus given in [14] suggest a decrease by a factor of about 10.

In June 1991 when we started with whole body measurements the average level of internal contamination was already below 10 kBq. We were convinced that the body activity of more than 90% of these group C people would not exceed 7 kBq for adults and 4 kBq for children. This activity does not involve any significant radiation risk. It corresponds to an effective dose of 0.3 mSv/a , thus lying within the variation of the natural background. Therefore, we evaluated all measured data according to three categories. The breakdown of these categories can be seen from table 3.1. A differentiation was made between adults and children. The limits for children were selected in such a way that for 5- to 12-year-olds the respective dose corresponds to the limits for adults. For babies a special counter was developed and put into operation in 1992. For this group the limits were reduced once more.

	adults	children
category 1	up to 7 kBq	up to 4 kBq
category 2	7 kBq to 25 kBq	4 kBq to 15 kBq
category 3	over 25 kBq	over 15 kBq

Table 3.1: Division of the measured data into categories

On the basis of [38] dose factors were derived for continuous uptake by ingestion (tab 3.8 in section 3.3). The limit of 7 kBq thus actually results in an effective dose of 0.3 mSv/a and the activity of 25 kBq leads to an effective dose of 1 mSv/a. Slightly different doses could result if other dose factors specified in the literature are used [10]. Compared with the respective dose limits for evacuation measures mentioned above, both internal doses have a sufficient margin for the observance of the external dose component. Therefore, these categories adequately correspond to the official evacuation dose limits of 1mSv/a and 5mSv/a.

Measurement results in the first category do not give any cause for concern. Body activities within the second category are also sufficiently safe, but in this case it was recommended to limit the intake of food known to be highly contaminated (e.g. mushrooms, game). The values in the third category do not in any case exceed the permissible limits of intake according to ICRP 30, i.e. they were in the range corresponding to dose values permissible for persons professionally exposed to radiation. For continuous uptake of Cs-137 the annual dose limit of 50 mSv/a corresponds to a body burden of more than 1000 kBq. The highest measured results are given in table 3.2. Obviously, they are below this value.

district	year of birth	sex	date	activity (kBq)	int. dose (mSv/a)
Narovlya	1949		1993	770	29
Narovlya	1931		1993	630	24
Narovlya	1940		1993	510	19
Narovlya	1983		1993	270	19
Krasnaya Gora	1932	m	1993	480	18
Narovlya	1940		1993	480	18
Gordeyevka	1965	m	1992	410	16
Gordeyevka	1936	m	1992	350	13
Narovlya	1945		1993	350	13
Narovlya	1928		1993	340	13
Narovlya	1944		1993	310	12
Narodichi	1947	m	1993	310	12
Narodichi	1956	m	1993	310	12
Krasnaya Gora	1931	m	1991	290	11
Gordeyevka	1932	m	1992	290	11
Narodichi	1936	m	1993	280	11
Slinka	1937	m	1991	280	11
Rokitno	1942	f	1993	280	11
Slinka	1956	f	1993	280	11
Gordeyevka	1955	m	1991	270	10
Narovlya	1969		1993	270	10
Gordeyevka	1943	m	1992	260	10
Narodichi	1947	m	1993	260	10
Slinka	1956	m	1993	260	10
Slavgorod	1937	m	1992	250	10

Table 3.2: List of individuals with a whole body activity of more than 250 kBq Cs measured during the period 1991 - 1993

In general, our recommendation for people in category 3 was to repeat the whole body measurement some weeks later and to have an annual medical check-up in the same manner as for professional exposed workers. The implementation of any other protective measure was the responsibility of the local authorities.

A summary of all body counter measurements carried out by German experts from 1991 to 1993 is given in table 3.3. A total of 317,011 measurements were performed in the three republics Russia, Belarus and Ukraine. The total number of measurements decreased from 163,000 in 1991, to 90,000 in 1992 and 64,000 in 1993 due to our continuously reduced funding and a reduction in the interest of people who in the previous year had been found to have low exposure. Nearly 90 % of all results are within the lowest category 1, about 8 % are in category 2 and only 2 % must be assigned to category 3.

	number	cat.1	cat.2	cat.3
1991 Total Russia	163 033	93.7 %	5.3 %	1.0 %
1992				
Total	90 460	90.6 %	7.9 %	1.5 %
Russia	49 858	85.8 %	11.7 %	2.5 %
Ukraine (Fastov)	11 373	100 %	0 %	0 %
Belarus	29 229	95.2 %	4.4 %	0.4 %
1993				
Total	63 518	81.0 %	14.6 %	4.5 %
Russia	14 836	70.0 %	22.9 %	7.1 %
Ukraine	36 126	84.5 %	11.9 %	3.7 %
(incl.Rovno	2 773	38.6 %	34.9 %	26.5 %)
Belarus	12 556	83.8 %	12.4 %	3.8 %
1991-1993 Total	317 011	90.3 %	7.9 %	1.8 %

Table 3.3: Summary of all German body counter measurements 1991-1993

Some particularities can be observed for the different republics and for different years. Thus, the category 3 fraction increased over the 3 years from 1 % to 1.5 % and 4.5 %, probably due to the reduced interest of people with low exposure in a repetition of their measurements.

The highest fraction of category 3 results was found in 1993 in the district of Rovno where 26.5 % had to be assigned to the highest category. In Rovno in the western part of the Ukraine the ground contamination is not high. Thus, the high body burden in these inundated areas demonstrates again the overriding importance of transfer factors. In tab. 3.4 the results from our measurements and from [15] are given for some villages in the Rovno district, which always show the same relation: low ground contamination but high body burden.

The results from Rovno verify that besides the soil contamination three site factors are most decisive: the first depends on the soil quality and determines the transfer soil-plant (transfer factor), the second depends on the proximity of forest or lakes (forest factor) and the third factor is called the cow factor and describes the degree of self-sufficiency with milk produced locally and depends on the number of cows per capita.

If one of these factors becomes enhanced the importance of the internal dose would be higher than the external. In Beresovka, an isolated village in the Klincy district, we found high body burdens: 54 % of the population had an internal contamination of more than 25 kBq Cs. But the soil contamination was only 1-5 Ci/km^2 (37-185 kBq/m^2). Locally-produced milk was a major source of nutrition.

In the Rovno district two of these three site factors are given. This area was flooded before in 1993. The inhabitants were dependent on locally produced food, meat, milk products and wild produce, especially fish from the lakes.

In 1991 some 10,000 measurements were carried out in regions of the Observed Areas where the ground contamination was relatively high but almost all individuals could be assigned to category 1 or 2. In the Tula region in Russia only 3 persons of 28,000 individuals checked were assigned to category 2. A similar situation was observed in 1992 in the district of Fastov south of Kiev (s. tab. 3.3). These measurements with results of 100 % in the first category mainly had a political background to calm the fears of the population. From a scientific point of view the low results are not very interesting. Fig. 3.7-3.9 give overviews of all results in the different villages and districts visited by our measuring teams.

village	district	German results		results from [15]	
		body activity [kBq]	soil cont. [kBq/m^2]	body activity [kBq] adults	soil cont. [kBq/m^2]
Chemerel	Dubrovitza	57.5	30	-	-
Veshitza	Rokitno	55.0	25	69.4* (26-112)	96 (37-174)
Staroye-Selo	Rokitno	45.0	55	59.2 (12-106)	48 (37-174)
Drozdyn	Rokitno	42.5	25	40.5 (17-63)	52 (37-174)
Vebovka	Dubrovitza	25.0	60	-	-

* 1992

Table 3.4: Comparison of 1993 average results in the Rovno district for German measurements and measurements according to [15]

According to general experience, only the smaller part of the human dose is due to incorporated radionuclides from food consumption while the predominant part is due to external irradiation from Cs activity deposited on the ground.

Nevertheless, there are many settlements where the internal dose may be much more important. There are significant local variations in the relation between external and internal dose, as is shown in fig. 3.3, which was determined by Belarussian scientists [16]. In settlements near the forest, where people are very dependent on highly contaminated forest produce, the internal dose is higher than the external dose and may reach nearly 80 % of the whole dose. If both factors - proximity to forest and high transfer factors due to sandy soil - coincide the internal dose may be significantly higher, even if the area is less contaminated. This is shown in fig. 3.4 for the settlements of Haltch and Grebeny.

The third of the most decisive components among the highly contaminated foods is milk from private cows. As our colleagues from Belarus have clearly demonstrated [16], the internal dose distributions are separated for the population groups with and without this kind of food (see fig. 3.5). To endorse this, fig. 3.6 shows the significant dependence on each one of these factors. Recommendations have been derived from this knowledge. The group who did not comply with the official advice received about 50 % of the whole collective dose.

Upon closer inspection of individual villages, fig. 3.10 and 3.11 present special data in Veprin (Russia) and Kirov (Belarus). The age distributions in Veprin 1991 and Kirov 1993 show that the highest burdens are individual cases only. Such maxima were observed mainly for elderly people. Some of them have little money to spend on contamination-free foodstuffs from the state-owned shops and hence they rely on food produced by themselves. A few younger people showing enhanced values eat a lot of fish caught in the local lakes or are hunters or supplement their diet by a relatively high amount of mushrooms and berries. Obviously some age groups are missing in the results presented from Veprin, and also from other settlements. This reflects the migration of workers as well as children, who are at summer holiday camps.

Finally, fig. 3.12 and 3.13 give a summary of all results in the Klincy district and the Gomel region. In the Klincy district comprehensive measurements were performed for all three years. Therefore, fig 3.12 gives a good impression of the time dependence of our results. A comparison between fig. 3.12 and 3.13 reveals the surprisingly lower results in Belarus. We think this is due to the fact that we were not allowed to visit small villages.

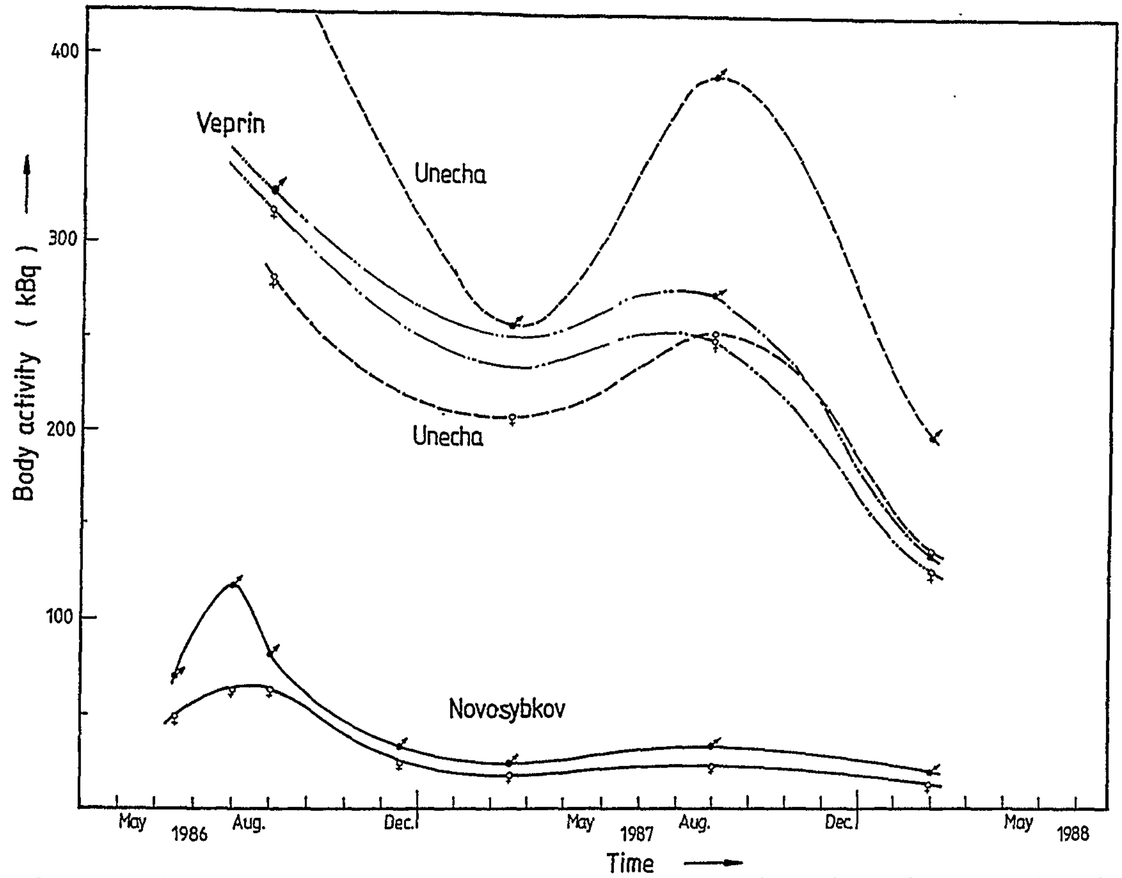


Figure 3.1: Body Counter Measurements in Novosybkov, Unecha and Veprin (Bryansk region)
♂ male, ♀ female

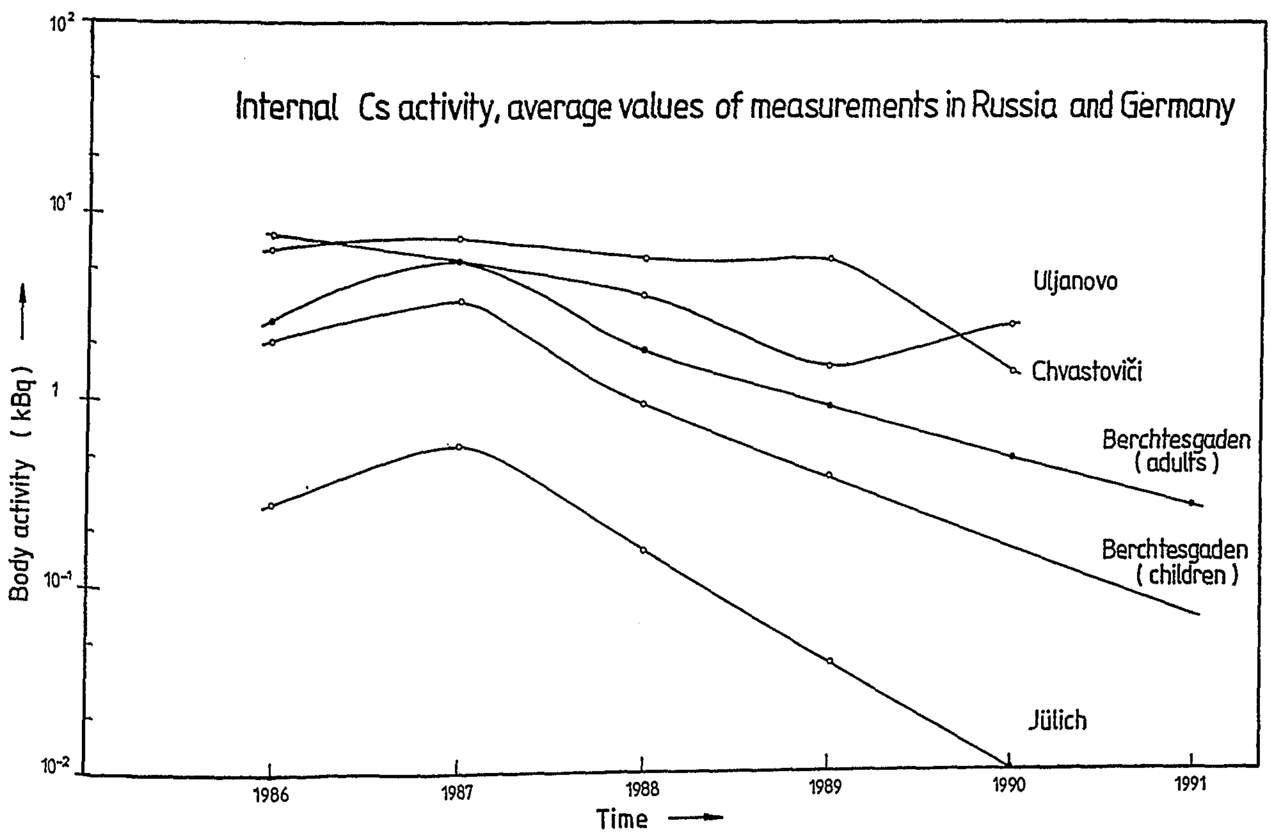


Figure 3.2: Body Counter Measurement in the Kaluga region and in Germany

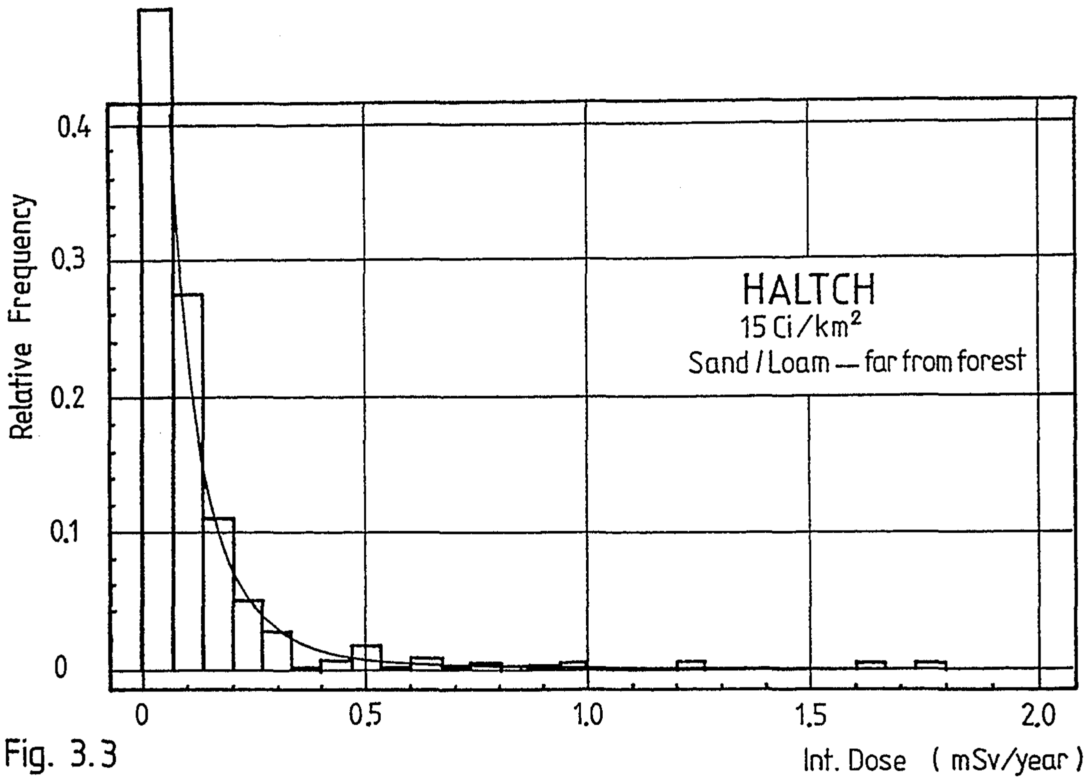


Fig. 3.3

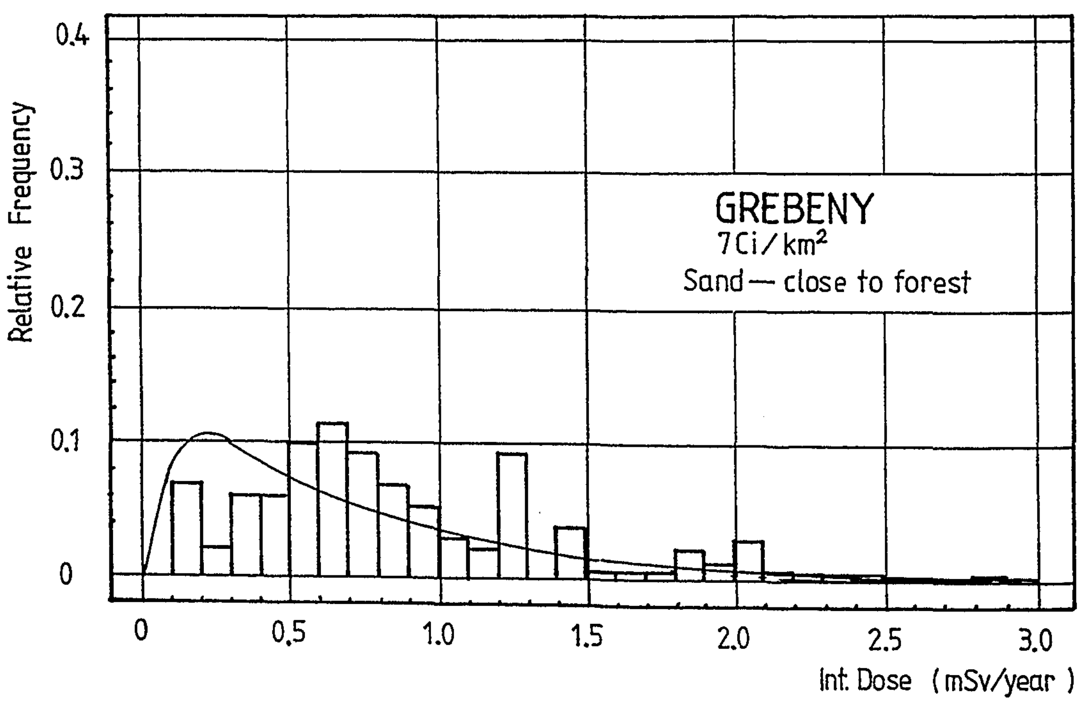


Fig. 3.4

Frequency Histogram

Int. dose from Cs-137 incorporation in two villages of the Gomel region,

(from Skryabin [16])

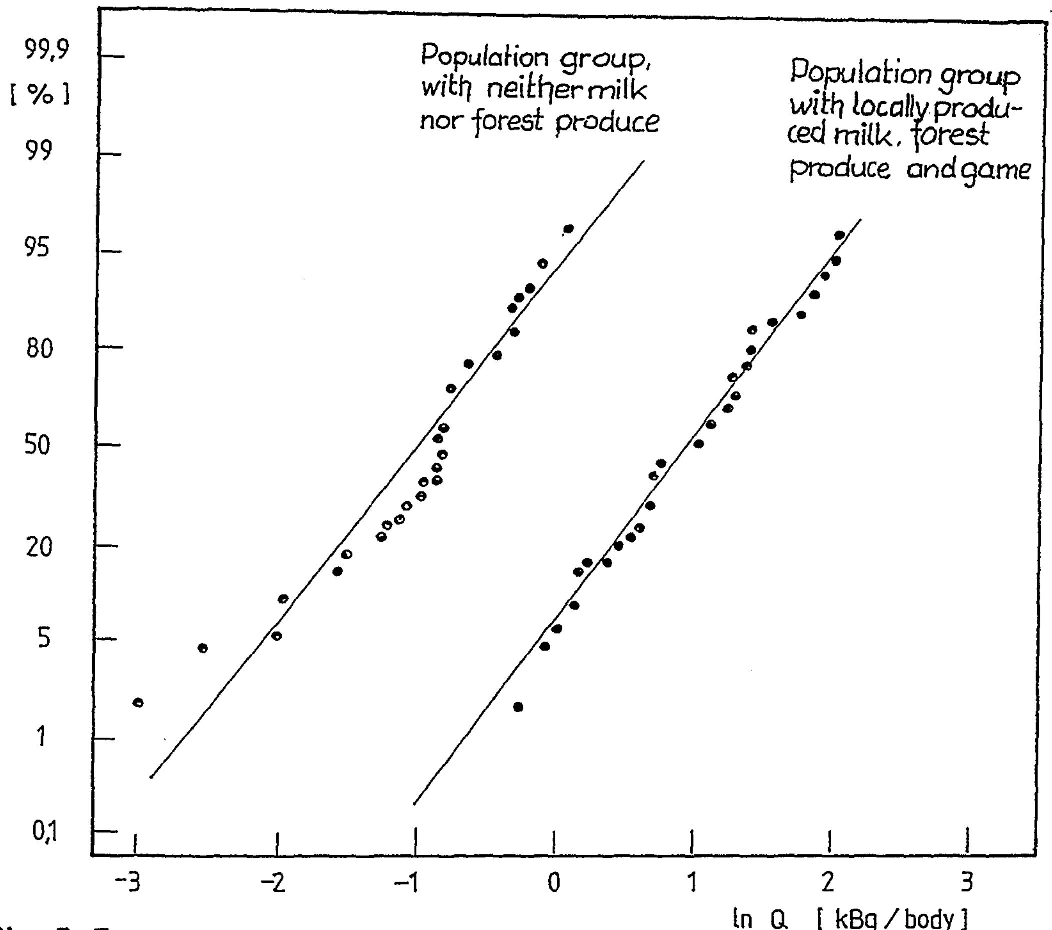
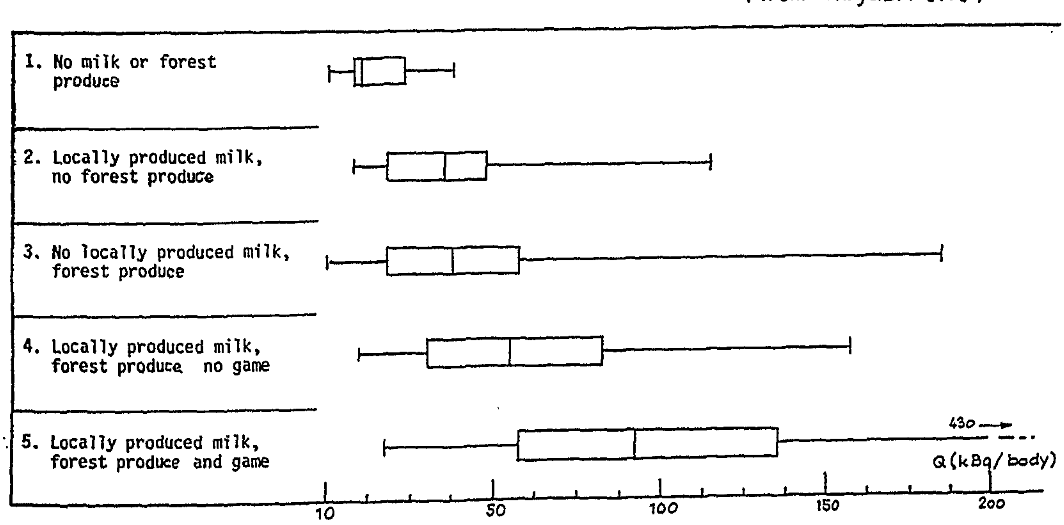


Fig. 3.5

Accumulated frequency distribution of Cs-137 body burdens Q in population groups differing in dietary habits (from Skryabin [16])

Fig. 3.6

Cs-137 body burdens in relation to living conditions and/or dietary habits at Kirov, Belarus, 1993, based on questionnaires and whole-body counting. (from Skryabin [16])



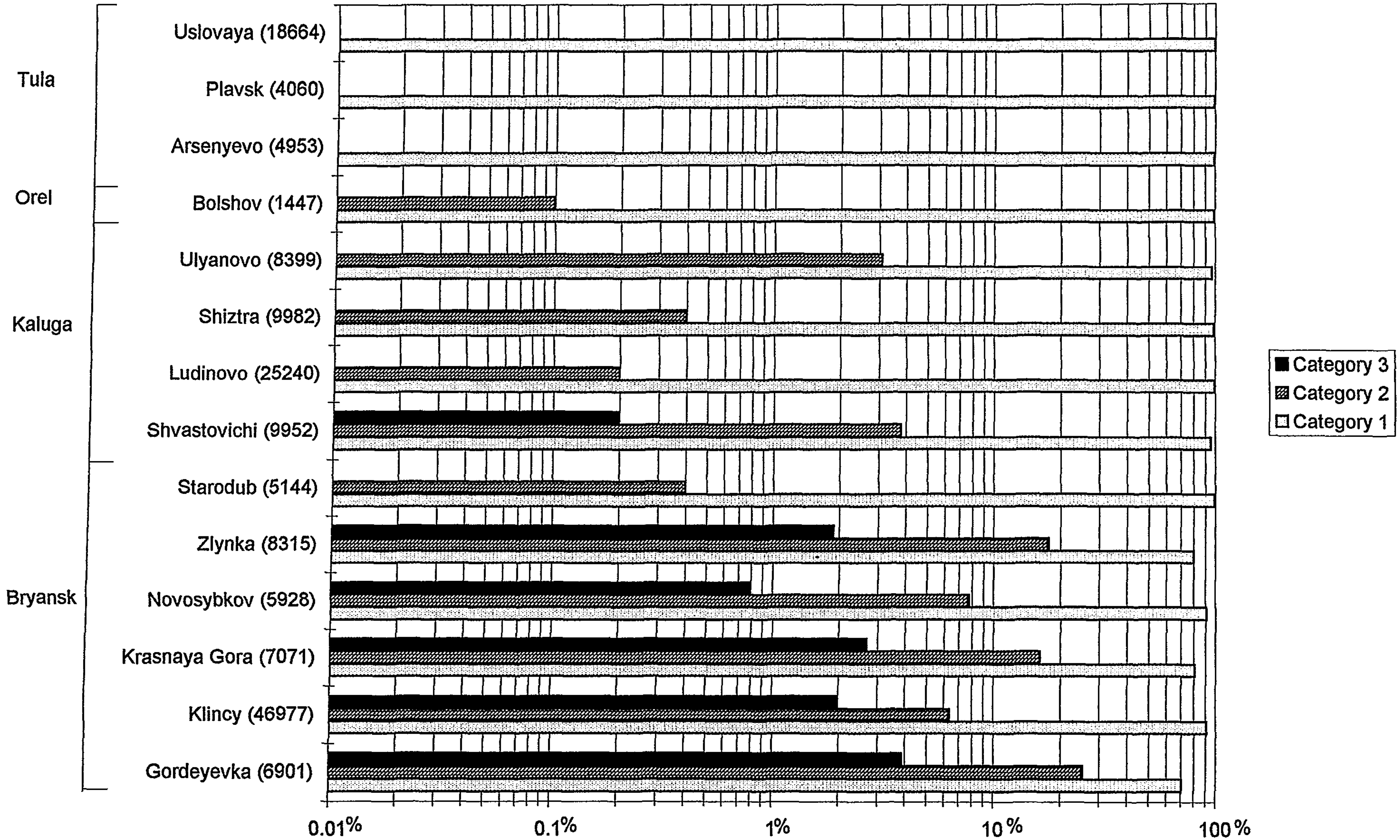


Fig. 3.7: Results of whole body measurements in Russia 1991, in brackets: total number of measurements

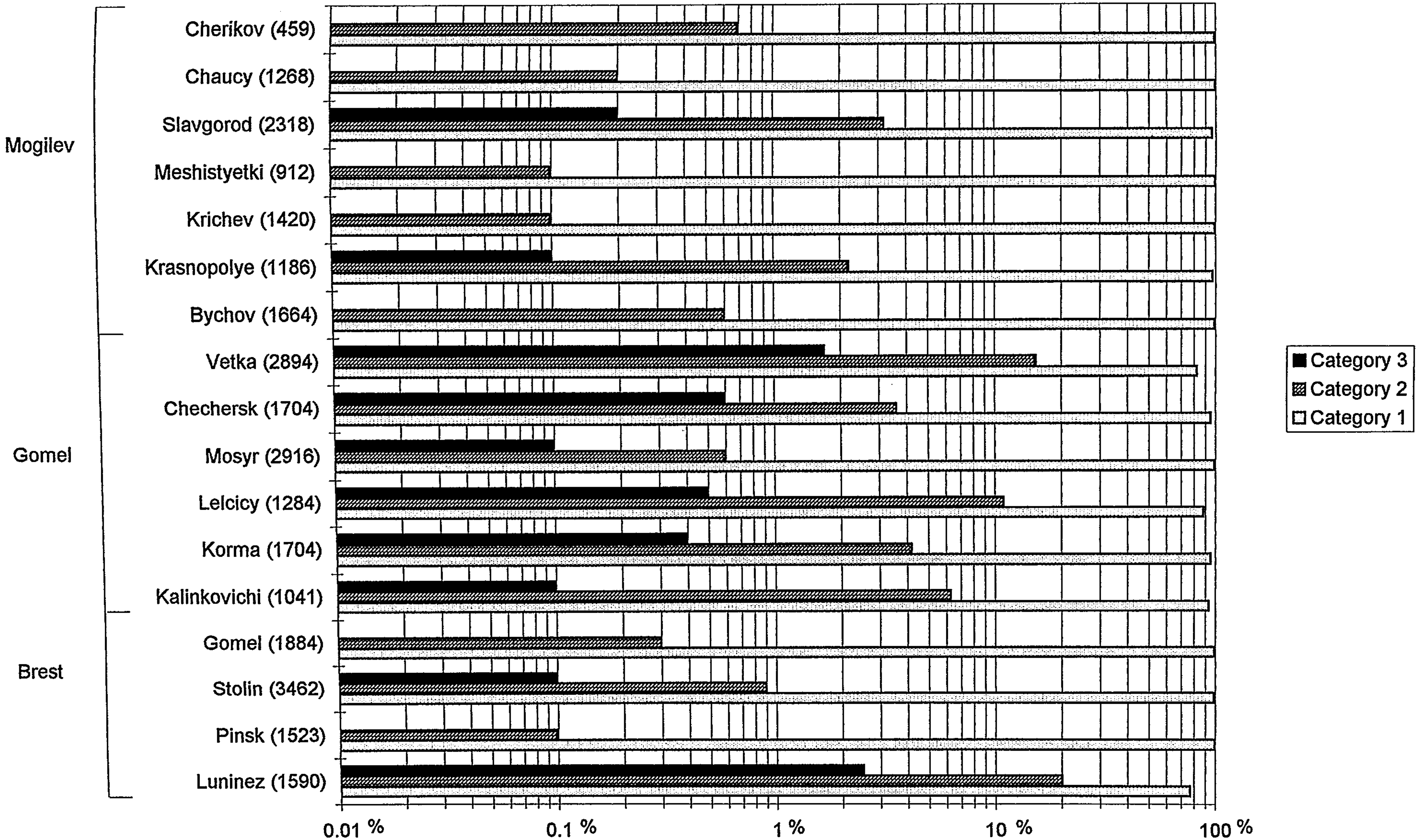


Fig. 3.8: Results of whole body measurements in Belarus 1992, in brackets: total number of measurements

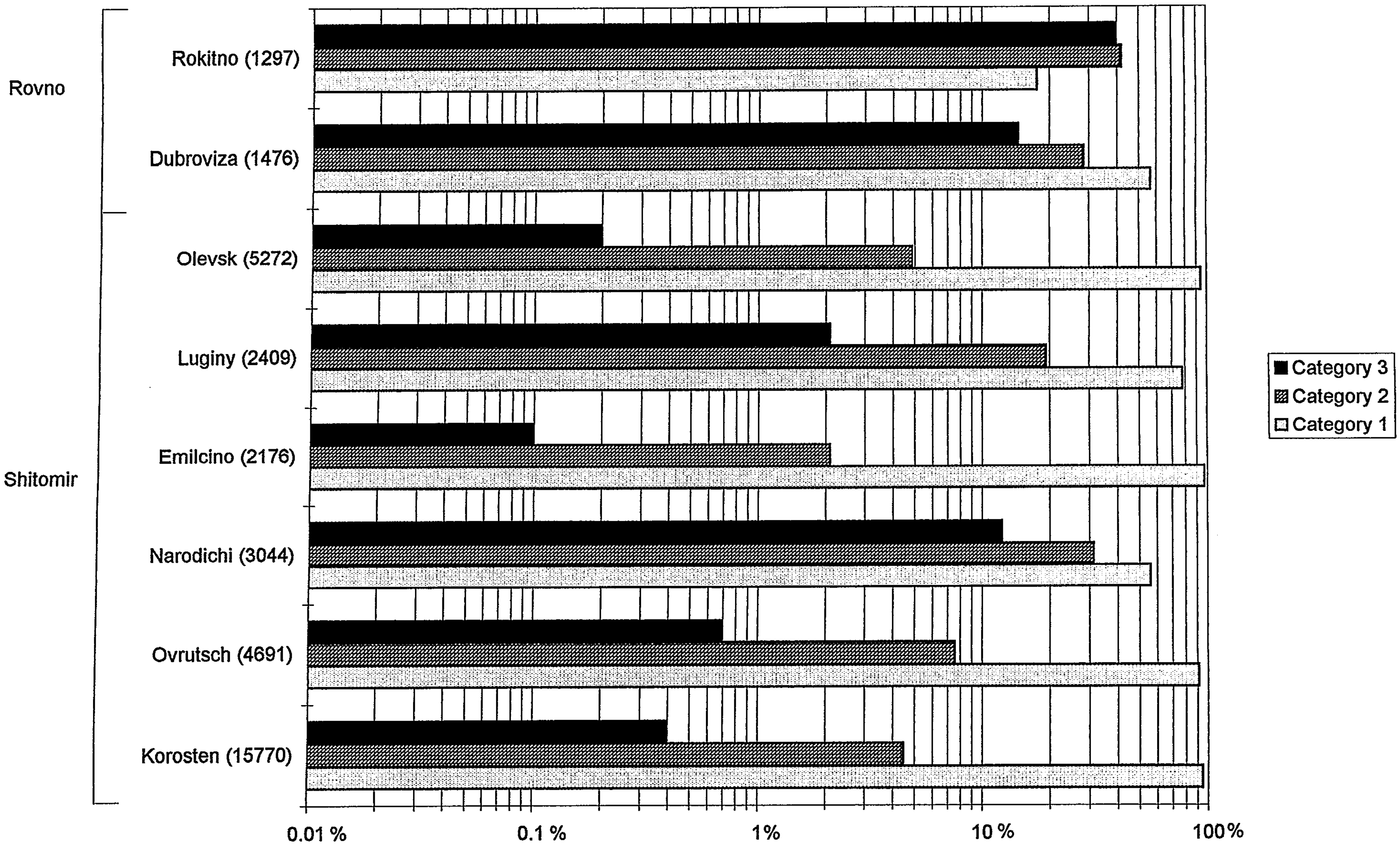


Fig. 3.9: Results of whole body measurements in Ukraine 1993, in brackets: total number of measurements

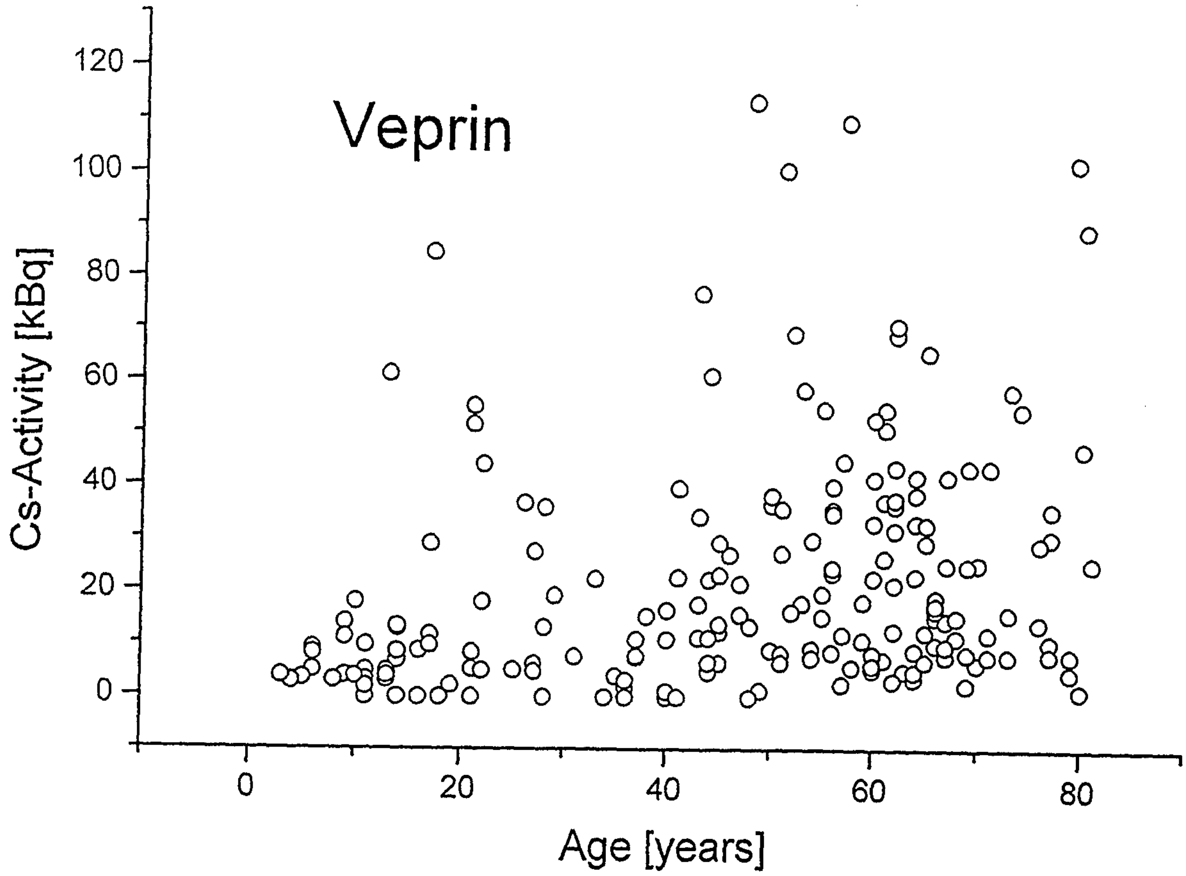


Fig. 3.10: Age distribution of Cs body burdens in Veprin 1991

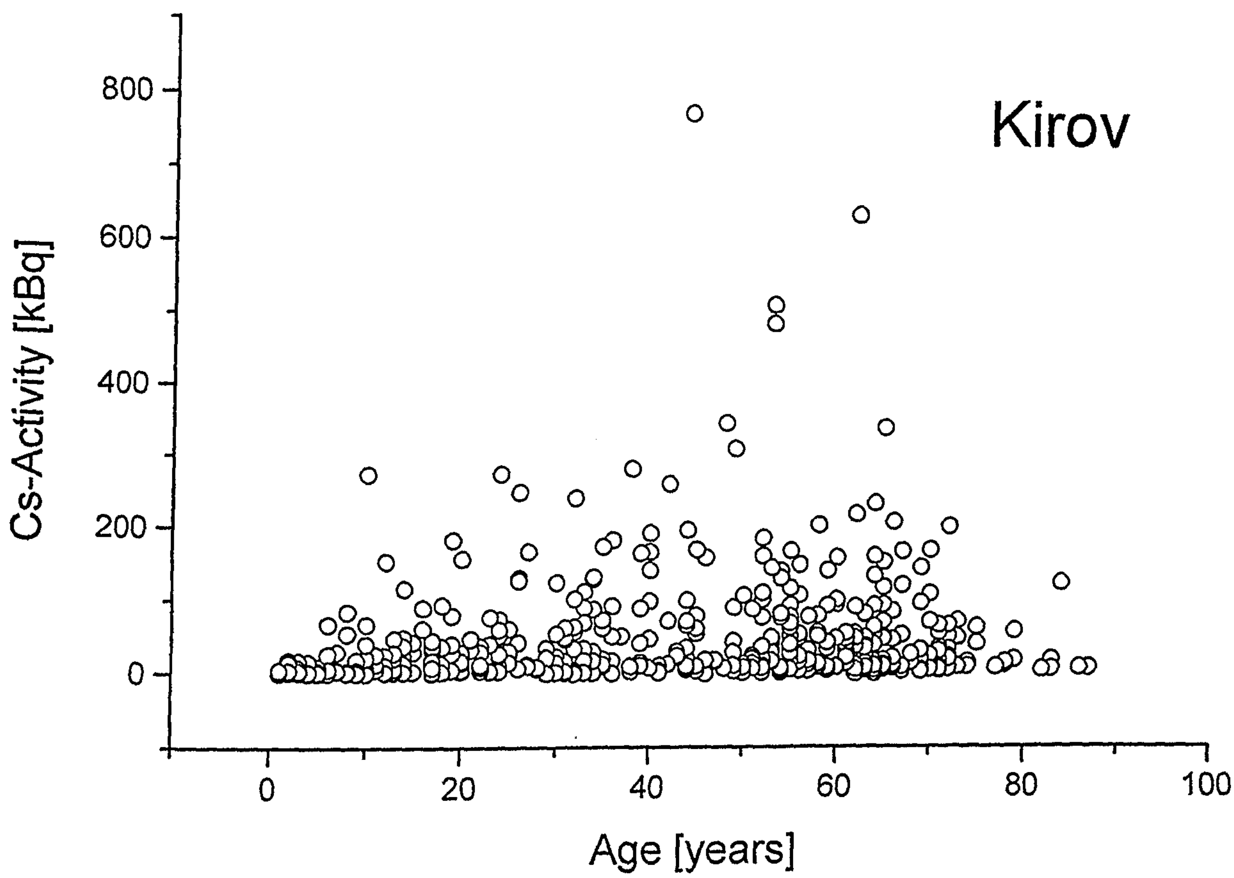


Fig. 3.11: Age distribution of Cs body burdens in Kirov 1993

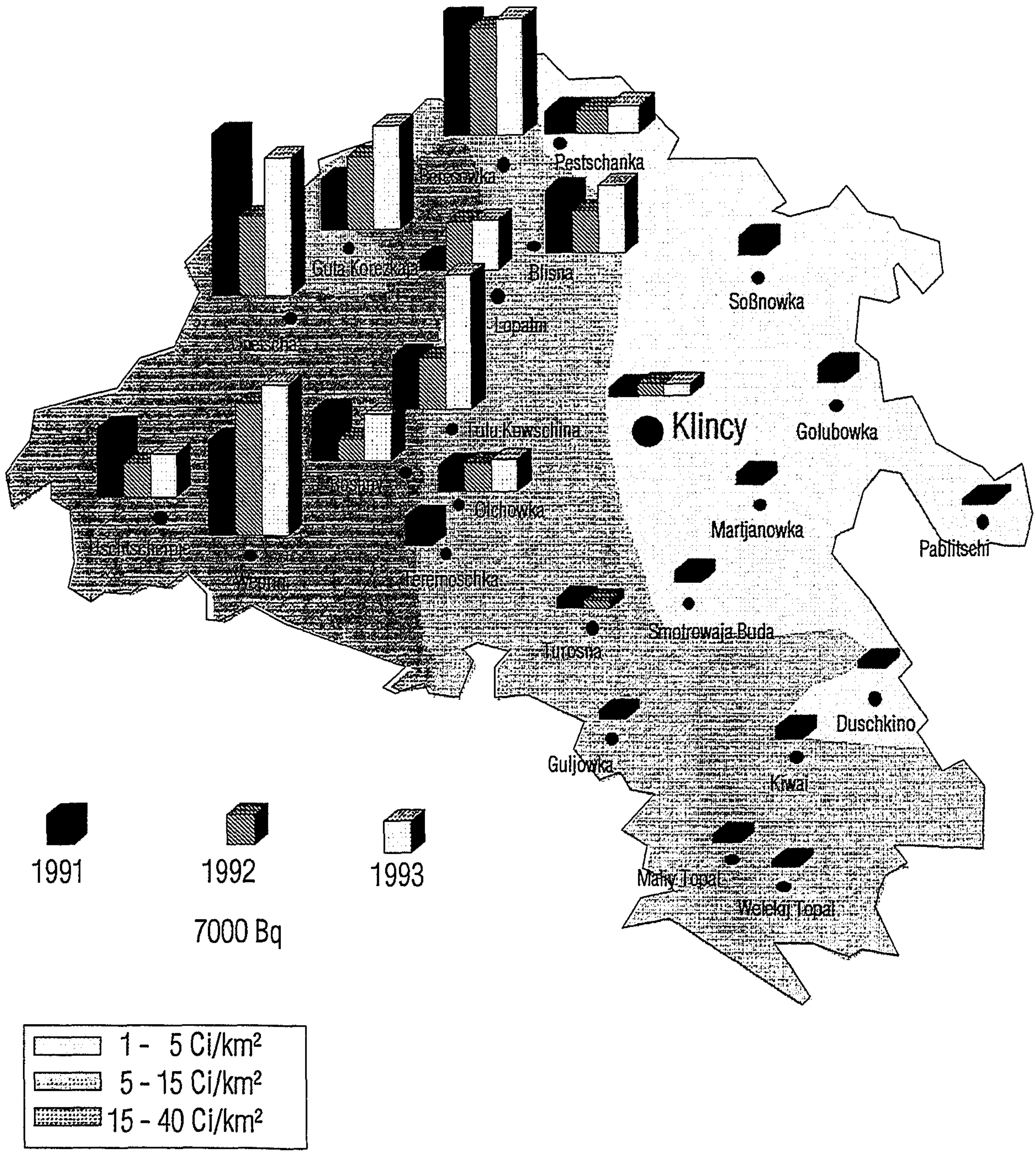


Fig. 3.12: Incorporated activity in the Klincy district (average per village)

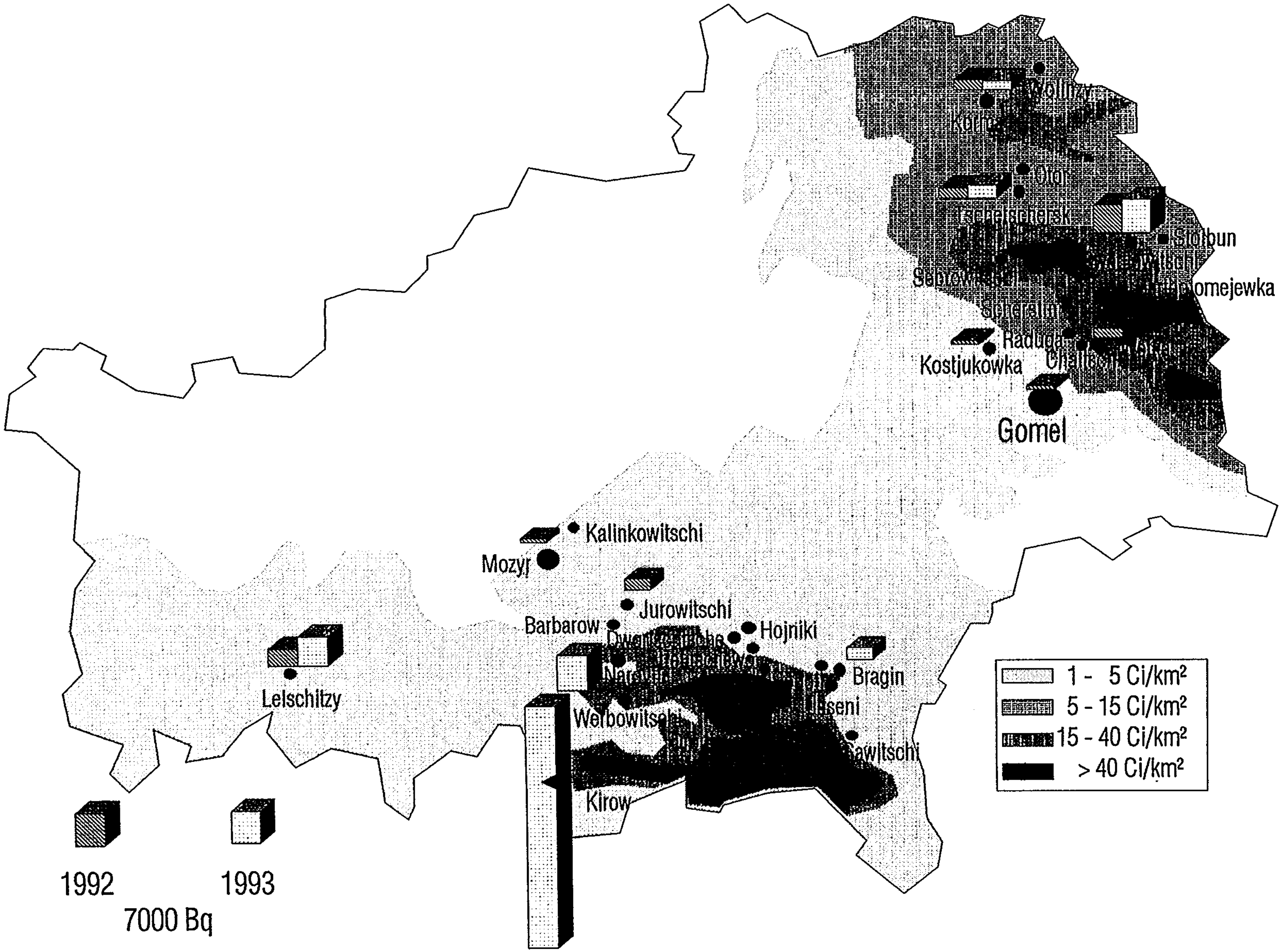


Fig. 3.13: Incorporated activity in the Gomel region (average per village)

3.2 External Exposure

External gamma radiation from ground contamination is the most significant contributor to the long-term population dose. Therefore, the evaluation of the radiation exposure for people in contaminated areas, and the estimation of and information about the resulting risks would be incomplete without a thorough estimation of both contributors, i. e. internal and external exposure.

In the first few days after the Chernobyl accident, the external dose rate from the ground surface contamination was caused by different nuclides. The main contributors were cesium, iodine, lanthanum and ruthenium isotopes. Fig. 3.14 gives the time dependence for the relative contribution of the most important radionuclides based on calculations according to the release given in chapter 4 and in [11]. It should be pointed out that the short-lived isotope I-132 and its precursor Te-132 were absolutely dominant for the first 10 days.

Twenty days after the accident the cesium isotopes became more and more dominant. From that time on the area dose rate was dependent, apart from anthropogenic factors, only on the soil contamination and the depth distribution for the cesium isotopes 134 and 137. The decrease of external exposure over the last few years is a result of changes in these three factors: the anthropogenic factor by man-made decontamination, the soil contamination by physical decay and the depth distribution by migration processes in the upper soil layer. The dashed line in fig 3.14 gives the decrease of the total relative dose from the ground contamination due to the physical decay of all radionuclides setting the origin as 100 % 20 days after the accident when most of the short-lived isotopes had lost their dominance. From that time till 1991 the area dose rate decreased by more than 80 %. Considering only the Cs isotopes 134 and 137, the decay during that period is 52 %, and the Cs-137 decrease during that 5-year period is slightly above 10 %. Russian measurements in the Bryansk region of the average dose rate from Cs gamma radiation indicate that the dose decreased by 78 % from 1986 to 1991 as a result of both radioactive decay and vertical migration. The dose rate in air at a height of 1m above a plane with a Cs surface activity of 1 kBq/m^2 decreased from 2.36 nGy/h in 1987 to 0.92 nGy/h in 1991. That is a reduction of 61 %. These results were derived from field investigations over virgin soil in the western districts of the Bryansk region and they are given in [11].

Our own investigations (s. section 2.1) resulted in a dose rate-ground contamination conversion factor of about 1 nSv/h for 1 kBq/m^2 of Cs contamination over undisturbed ground (pastures) for 1992. The resulting mean value for the dose rate in air from Cs-137 only is 0.68 nGy/h. For 1986 in [11] a dose rate of 1.82 nGy/h in air for this Cs-137 contamination (initially covered by approximately 0.5 g/cm^2 of matter) is suggested. Thus, we find a dose rate reduction due to Cs migration in soil of 63 % during this period.

In principle, the determination of the annual external dose and the dose reconstruction can be performed in three ways:

a. Determination on the basis of area dose rate measurements

b. Estimation of the external dose from ground contamination

c. Dose assessment by personal dosemeter, especially TLDs

During our measurements from 1991-1993 enough data were collected for external dose evaluations according to these three methods.

a. <u>Area Dose Rate Measurements</u>

In more than 150 towns and villages mean area dose rates (ADR) were measured. On the assumption that, on average,

- people stay outside for 6h (2h in winter, 10h in summer),
- the shielding factor s is 2 for wooden houses (rural population) and 8 for brick houses (urban population)
- approximately 5% of the house surfaces are windows without any shielding
- the geometry factor g - describing the deviation from an infinite contaminated plane - is less than 1 because our measurements were always made over free areas
- the conversion factor is 0.7Sv/Gy

the effective dose (H_{eff}) due to external radiation can be calculated by the formula

$$H_{eff} = ((0.95/s+0.05)\cdot g\cdot 0.75\cdot 8760 + 0.25\cdot 8760)\cdot ADR\cdot 0.7. \qquad \text{(I)}$$

In principle, the geometry factor g is a distance factor, i.e. reflecting the fact that inside houses there is a certain distance from a person to the next contaminated area outside. The geometry factor inside single-family houses has been calculated as 0.5.

b. <u>Ground Contamination Measurements</u>

The assessment of the external dose from the ground contamination can be done in two different ways. The first is the official method of the State Committee for Hydrometeorology [17] that derived area dose factors (ADF) from more than 10 000 TLD measurements in the years 1989-1990. From this, ADFs for 1991 of between 0.13 and 0.06 $mSv\cdot km^2/Ci\cdot a$ were recommended for villages and district towns, respectively. For a given area activity (C) in Ci/km^2 the annual external effective dose can be calculated by the formula

$$H_{eff} = C \cdot ADF. \quad \text{(II)}$$

A similar procedure has been developed by [18] from contamination measurements in Bavaria for different rural and urban settlements and environments. The area dose factors (ADF) are distinguished for fallout and washout as 0.1 and 0.07 $mSv \cdot km^2/Ci \cdot a$, respectively. The results of this method are in good agreement with those according to the official Russian procedure.

The second way explicitly considers the vertical migration. As mentioned above, in 1992 we measured a dose rate of 0.68 nGy/h above soil contaminated with 1 kBq/m^2 Cs 137. Thus, we obtained a reduction factor for 6 years of 63 % corresponding to a relaxation length of 6 cm [19] for meadows and undisturbed soil. For the actual contamination with both isotopes (Cs-134 and Cs-137) the dose rate from [11] was 0.92 nGy/h for 1991, corresponding to a reduction of 61 %. The agreement is fairly good.

The result is a dose rate for free air over contaminated soil with a realistic exponential depth distribution. For the determination of the corresponding effective dose equivalent, it has to be divided by shielding factors and multiplied by social factors, geometry factors and conversion factors as mentioned above.

In rural and urban areas the depth distribution is in many cases not exponential. Compared with the dose for free air over virgin land the dose over dirt, disturbed areas, asphalt or concrete will be lower by a factor of about two [11]. Our measurements over arable land in 1992 resulted in a dose rate conversion factor of 0.44 nSv/h for 1 kBq/m^2 ground contamination which is - compared to 1 nSv/h according to section 2.1 - an endorsement for the factor of 2.

c. Dose assessment by TLD

Finally, for the evaluation of external exposure, a number of dose assessments by personal TL dosemeter were carried out in 1992 and 1993 under the supervision of I. Uray from the Institute of Nuclear Research in Debrecen, Hungary [20], [21]. In these years, during the summer period and once in winter 1993/1994, the dosimetric control of about 6 000 individuals was carried out. About 100 settlements in Russia (Bryansk region) and Belarus (Gomel region) of different characters and contamination levels up to 4000 kBq/m^2 were investigated.

The doses to individuals measured in villages were higher than those in towns with the same soil contamination. The results of personal dose monitoring were therefore separately analysed for towns and villages. The values from villages with approximately the same soil contamination were combined into a group, and the different groups were then subjected to a linear regression procedure. For soil contaminations the official data were used. Except for the locations with maximum contamination where decontamination work had been carried out, the linear regression technique showed a strong correlation between the dose rate determined from the personal dosemeter (person-related dose rate $\dot{H}_p$) and soil contamination. The correlation coefficients for the arithmetic mean, median, 5 % and 95 % quantile are above 0.95.

Figure 3.15 shows the correlation between person-related dose rate and soil contamination for the 1992 and 1993 measurements. The deviation of the values for high soil contamination is to be attributed to decontamination measures since the contamination values specified apply to the original contamination prior to decontamination.

The linear regression technique gives

$$\dot{H}_p = 0.122 + 0.220 \cdot C \qquad \text{(III)}$$

for the person-related dose rate in 1992 in the villages, where C is the soil contamination in MBq/m^2 and $\dot{H}_p$ the person-related dose rate in µSv/h.

For 1993 the linear regression technique gives

$$\dot{H}_p = 0.121 + 0.182 \cdot C \qquad \text{(IV)}$$

The contribution from natural radiation is about the same in both years, while the contamination-dependent fraction decreased by 17 % in 1993.

Equations (III) and (IV) allow an estimate of the personal dose. In order to determine the external radiation exposure of the population, the personal dose must be converted into the effective dose equivalent because the personal dosemeters were calibrated for photon dose equivalent. This can be done according to German calculation principles for the determination of body doses from external radiation exposure due to photon emitters from a plane source [22]. The ratio of personal dose to area dose is 0.75 for 660 keV photon radiation and the ratio of effective dose equivalent to area dose is 0.63. This gives a ratio of effective dose equivalent to personal dose of 0.84. This ratio only depends marginally on the photon energy in the energy range from 0.15 to 1.0 MeV.

The effective dose equivalents estimated according to the different methods are compiled in table 3.5 for a soil contamination of 1 kBq/m^2. The contribution from natural radiation exposure was not taken into consideration. The effective dose equivalent in lines 1 and 2 was calculated according to eq. I (see above) from measured area dose rates. The effective dose equivalent in line 3 is calculated from the soil activity according to the official method of the State Committee for Hydrometeorology [17] with a derived area dose factor ADF = 0.13 $mSv \cdot km^2 \cdot Ci^{-1} \cdot a^{-1}$. Line 4 is based on personal dose measurements by the Institute of Radiation Hygiene, St. Petersburg [11], and the effective dose equivalents in lines 5 and 6 are calculated from our person-related dose rate measurements according to eq. (III) and eq. (IV).

As was to be expected, there is not complete agreement between the effective doses calculated according to the different methods. The values derived from the area dose rate are higher than those from personal dose measurements. It should be noted that the calculation of the effective dose from the area dose rate was based on some very generalized assumptions. Moreover, the area dose rate over untilled soil was used. In comparison, the area dose rate over tilled soil with the same soil contamination is only about 45 % of the value over pastures, and untilled soil cannot always be assumed in the vicinity of buildings. The values derived from personal dose measurements should be most reliable since they take precise account of the presence of persons at differently contaminated locations.

line	method	year	H µSv/a
A	**from the area dose rate**		
1	assuming 0.92 $\mu Gy \cdot h^{-1} \cdot MBq^{-1} \cdot m^2$ (p. 56) in eq. I	1991	2.52
B	**from soil contamination**		
2	official method of the State Committee for Hydrometeorology	1991	3.51
3	KFA measurements over pasture	1992	2.66
C	**from individual dose measurements**		
4	Institute of Radiation Hygiene	1991	2.10
5	KFA Jülich for villages	1992	1.62
6	KFA Jülich for villages	1993	1.34
7	KFA Jülich for towns	1992	0.77

Table 3.5: Comparison of effective dose equivalent determined according to different methods for a soil contamination of 1 $kBq \cdot m^{-2}$

The personal dose measurements of Erkin and Lebedev [11] lead to an average annual decrease of the contamination-dependent fraction of the effective dose of 0.3 $\mu Sv \cdot h^{-1} \cdot MBq^{-1} \cdot m^2$. Erkin et al. found an annual decrease of the contamination-dependent fraction of the personal dose by a factor of between 0.72 and 0.94 (average 0.8) for personal dose measurements carried out at four locations in two consecutive years. This agrees extremely well with the annual decrease of the values determined from personal dose measurements in table 3.5 (line 4, 5, 6). However, the values determined from the area dose rate in table 3.5 are to be regarded as conservative estimates. In addition to the strong correlation between person-related dose rate and level of soil contamination given in fig. 3.15 for village inhabitants, the evaluation of this information yielded further interesting correlations.

There is also a correlation between person-related dose rate and level of soil contamination for the inhabitants of towns. The following applies to town dwellers according to the 1992 measurements:

$$\dot{H}_p = 0.123 + 0.105 \cdot C \qquad (V)$$

The correlation coefficient is 0.859. The correlation for the towns is not so strong as for the villages since the towns examined have areas of a village-like structure of differing extent with wooden houses and also urbanized regions of varying sizes. The towns are populated by different social groups with very different external radiation exposure.

The external dose consists of two fractions, a natural and a contamination-related. The contribution from natural radiation is practically the same in towns and villages. But the average contamination-dependent dose fraction in towns is only 48 % of that in villages. Due to the poorer correlation between person-related dose rate and soil contamination, great differences result for individual towns. In Klincy, the contamination-dependent dose fraction is 90 %, in Klimovo it is only 15 % of the average value for villages with the same soil contamination [20].

In the town of Novosybkov a comparison was carried out between our body counter measurements and our TLD measurements. Figure 3.16 shows that the external dose is completely independent of the body burden. A weak correlation can be observed between the external dose and the home dose.

The radiation exposure also depends on the type of building. If the natural background is not considered, the radiation exposure in brick and concrete buildings is only 58 % of that in wooden houses for the same soil contamination. This value is slightly below that of 70 % according to Erkin and Lebedow [11]. Erkin et al. [39] also find the slightly lower average value of 67 % and show that this ratio can be very different for different locations, ranging between 45 and 94 %. If these large local differences are taken into account, our results are in satisfactory agreement.

The dependence of radiation exposure on the type of occupation was examined separately for different types of buildings. If natural radiation exposure is not included, the exposure in wooden houses with the same soil contamination is 23 % lower in the case of activities in the building than for outdoor activities or mixed activities outdoors and indoors. For persons living in wooden houses, i.e. basically the rural population, no difference was found for radiation exposure between outdoor and mixed activities. For persons living in brick or concrete houses, radiation exposure with activities in the building is 44 % lower than in the case of outdoor activities and 17 % lower for mixed activities. The difference in radiation exposure ascertained by us for in- and outdoor activities for persons living in wooden houses is in agreement with the data of Erkin et al. [39]. However, the data provided by Erkin et al. are not restricted to persons living in wooden houses, but apply irrespective of the type of dwelling.

External radiation exposure also depends on age. It is lowest for 20-to-50-year-old persons and increases for children and old people. The maximum radiation exposure was found in the age group from 60 to 69 years. The contamination-dependent fraction of radiation exposure for this age group is about 60 % higher than for the age group between 30 and 39 years.

The measurements in winter 1993/94 were intended to provide a contribution to the dependence of radiation exposure on the season. The expected significantly lower radiation exposure in winter, which was also observed by other authors, was not found. The dose reduction in winter was very small, and a slight increase of the average person-related dose rate was even measured at two locations. Due to an unusually damp summer in 1993 - the area dose rate over wet soil is lower than over dry soil - and because the winter measurements were carried out during a period with relatively little snow, no significantly lower radiation exposure was found in winter.

A better agreement of the results developed by individual dose measurements with results from the area dose rate assessment could be achieved only if the social factor in equation I were improved, which is just a very rough figure because it only considers the time spent indoors and outdoors. It should be distinguished for special professional groups, i.e. hunters, herdsmen, schoolchildren.

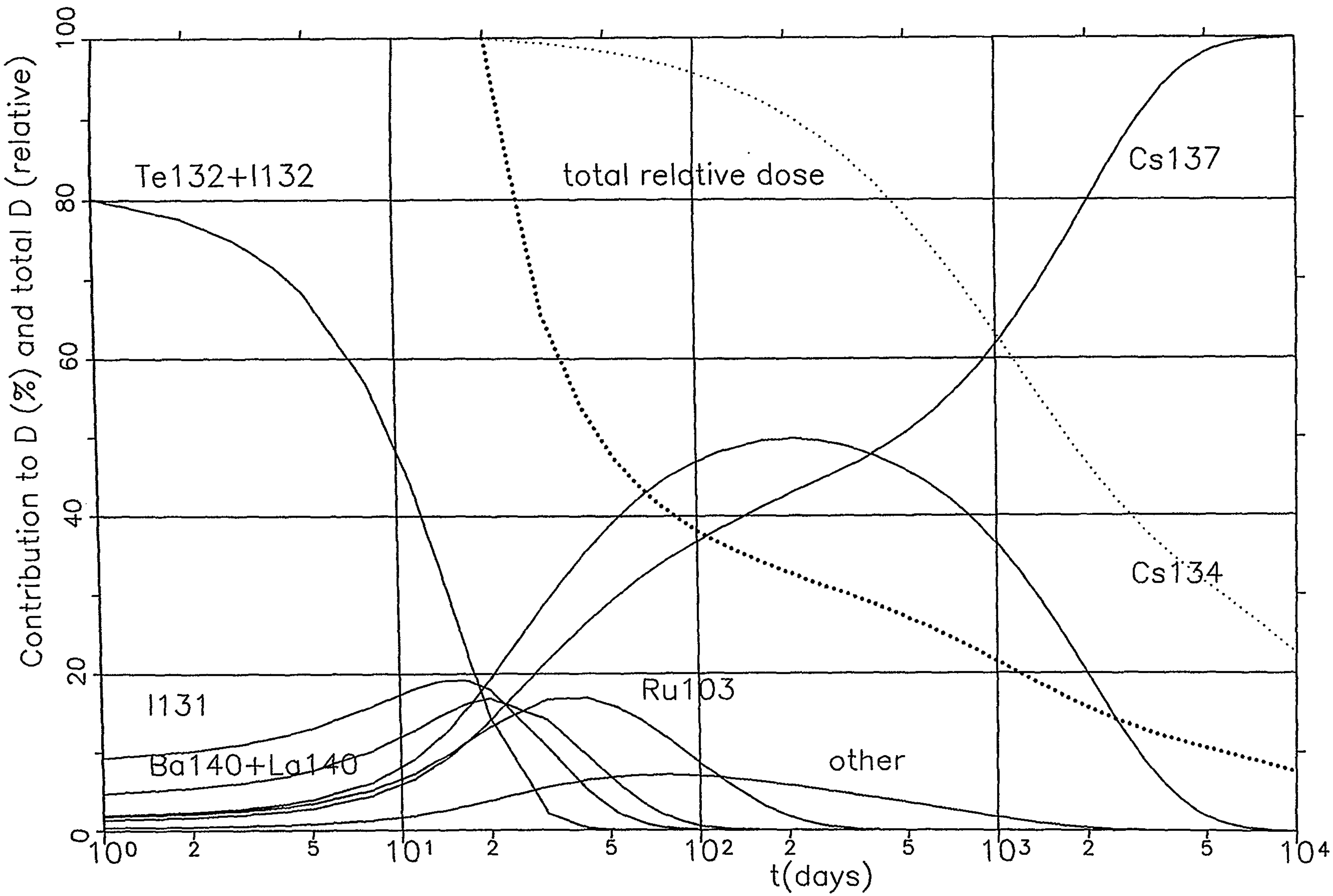


Fig. 3.14: Relative contribution of the single radionuclide to the whole activity released from a reactor accident and decrease of the total dose starting after 20 days (100 %)

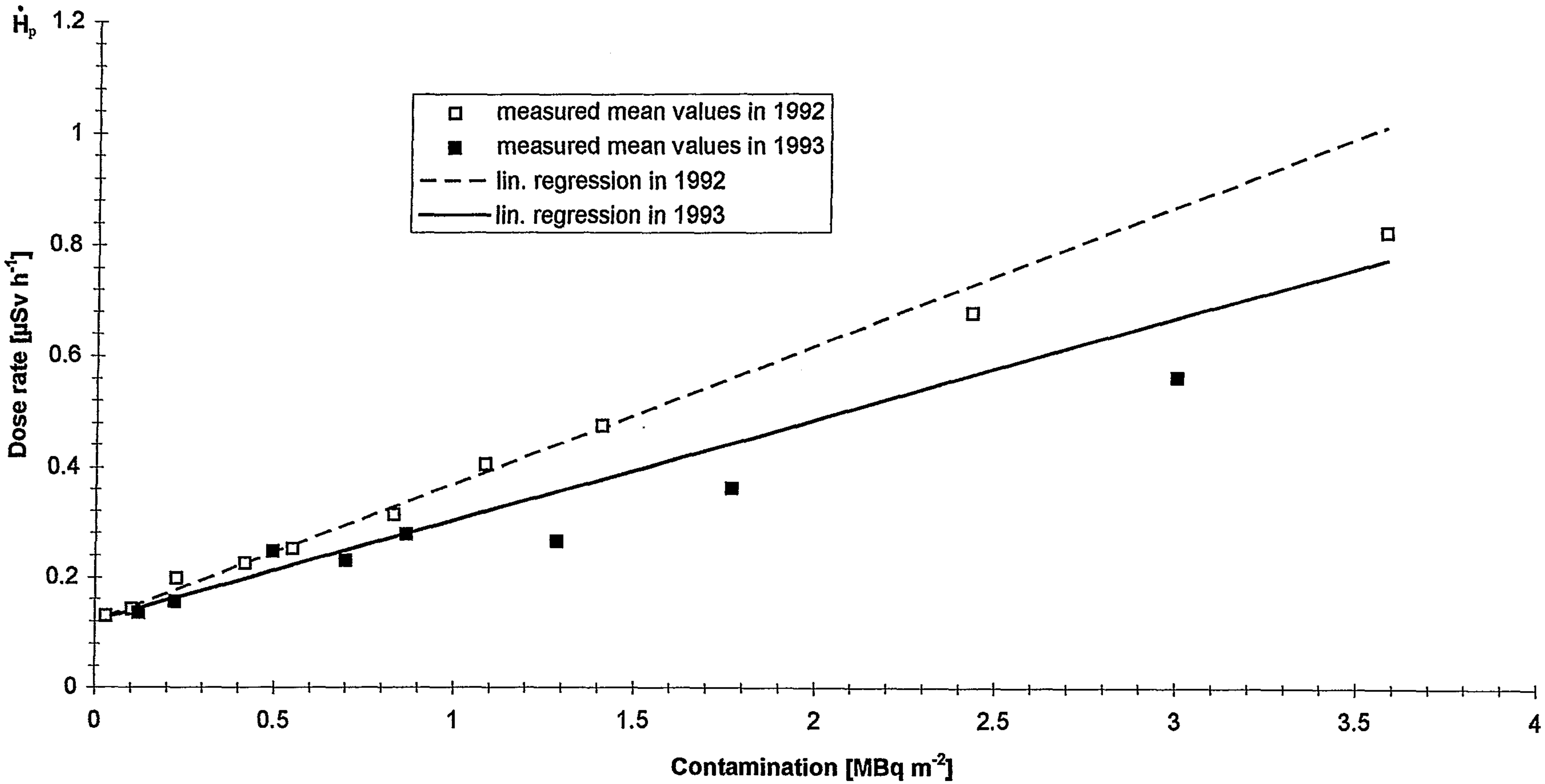


Fig. 3.15: Correlation between the person-related dose rate $\dot{H}_p$ and the soil contamination in villages

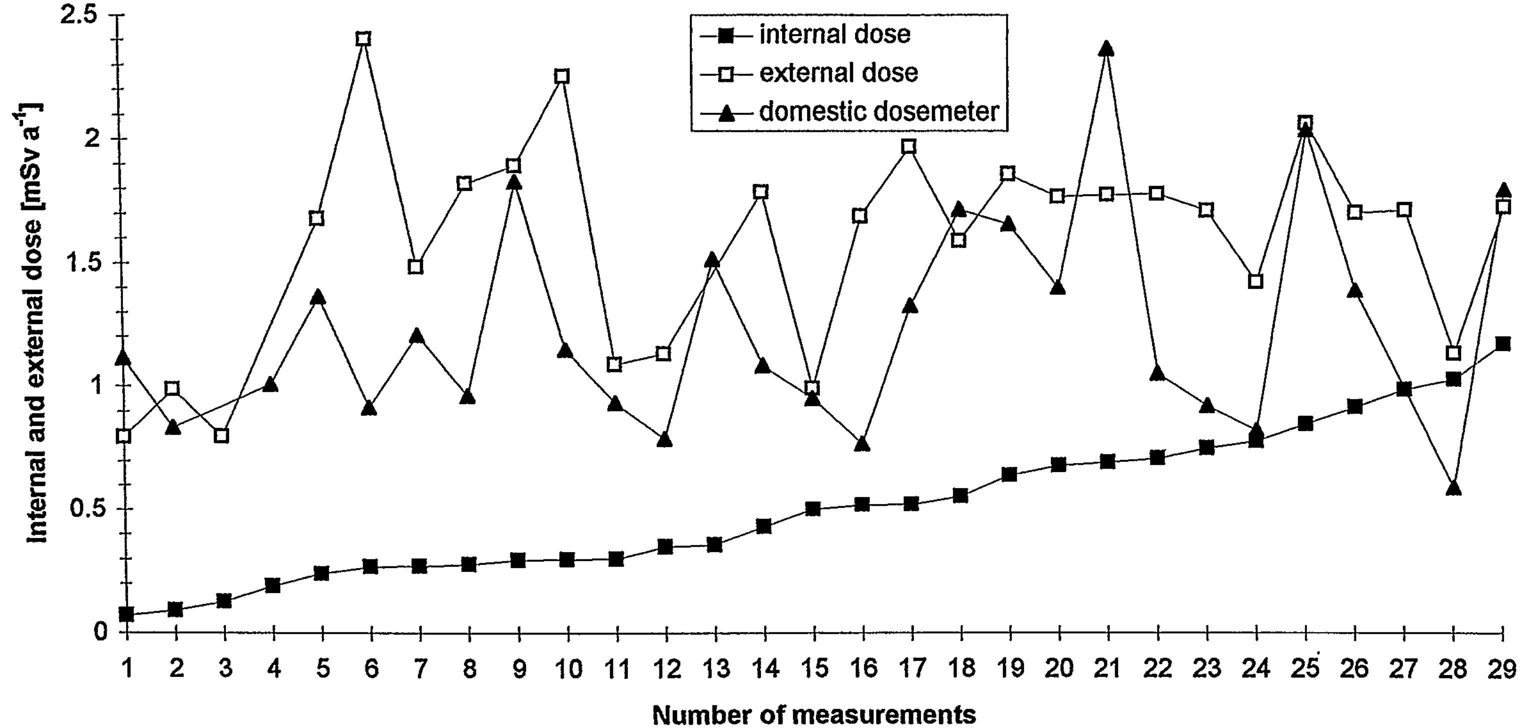


Fig. 3.16: Comparison of the internal and external dose in Novosybkov for certain individuals

3.3 Biological Dosimetry

Biological methods of examining the radiation exposure and medical checks supplemented our measuring programme for the population. These measurements were carried out by scientists of the Federal Health Office, Clinical Diagnostic Branch in Berlin under the supervision of U. Wolf with evaluation by H. Fender [23].

In summer 1992 a medical subprogamme was started consisting of two components. First, sonographic investigations of 187 individuals in two settlements (Klincy,Pestshanka) of the Bryansk region were performed by a portable ultrasonic device to investigate a possible increase of thyroid diseases for children (45) and adults (142) [24]. 37 of these individuals had actual cesium burdens above 7 kBq. The results were insignificant for children but 43% of the adults were striking, mainly due to thyroid enlargement and nodules. In this region there is usually iodine deficiency.

Secondly, a chromosomal analysis was carried out in lymphocytes of the peripheral blood. About 80 blood samples were taken in these settlements of the Bryansk region and prepared in the research centre at Obninsk. But this trial failed because the samples were spoiled.

As part of the measurement programme in the CIS regions affected by the Chernobyl reactor accident and on the basis of the plan agreed upon with the Ukrainian Chernobyl Ministry, chromosome analyses were carried out again in 1993 on selected children from West Ukraine in addition to physical incorporation measurements and medical surveys in close cooperation with the Research Institute for Paediatrics, Obstetrics and Gynaecology for the Ukrainian Academy of Sciences in Kiev.

Chromosome analysis is currently among to the most sensitive biological methods used for monitoring mutagenic and carcinogenic noxae in the case of exposure. The methodology of chromosome analyses of peripheral blood lymphocytes practised internationally for many decades is used for this purpose. This cytogenetic examination procedure searches for unstable chromosome aberrations, such as dicentric and ring-shaped chromosomes as well as acentric fragments whose spontaneous mutation rate is very low (approx. 0.4 - 1 dicentric chromosomes and 2 - 4 acentric fragments in 1000 cells examined).

At elevated acute whole-body doses ($\geq$ 50 mGy of loosely ionizing radiation) this method is quite suitable for use as a biological dosemeter in the individual case, e.g. for acute overexposures due to radiation accidents. On the other hand, at lower chronic radiation exposure no information can be obtained for the individual case, but - if at all - only for the population, e.g. within the framework of so-called "population monitoring". This can be done by observing the increase in the mean frequency and the deviation of the distribution of specific forms of aberration in the total number of cells in comparison with control persons not exposed to radiation. The diagnostic value of the above-mentioned forms of aberration is restricted in so far as they are a typical consequence of a radiation exposure incurred, but do not have any radiation specificity. Identical or similar chromosomal modifications are also found after exposure to a variety of potentially genotoxic pollutants (e.g. substances contained in cigarette smoke, alcohol, some drugs, certain chemical substances in the working and natural environment). A prerequisite for an etiologically valid evaluation of the results of chromosome analyses is therefore a survey of the vital status and of the individual case history.

The aim of chromosome analyses in Ukraine was to clarify whether there is a correlation between physical dose measurement and biological findings. An individual correlation was no longer to be expected due to the low actual dose. However, correlations on a collective basis in the sense of the above-mentioned population monitoring were not to be excluded.

There have been a number of similar investigations by other scientific groups. In [25] information is given on a comprehensive study in Russia and Belarus. It was carried out mainly in 1991 and comprised 1855 individuals classified into 10 cohorts of children in different settlements in the districts of Kaluga, Bryansk and Gomel, additional charts for adults, evacuees and two controls. The doses were between 8 and 102 mSv for the 5 years since the accident. The highest dose of 102 mSv was evaluated for the town of Zlynka in the district of Bryansk. In this town the average ground contamination is about 1000 kBq/m^2 Cs-137. Only there could a significant increase of dicentric chromosomes be observed. The value was 1.38 ± 0.34 (confidence level 90 %) for children and 1.86 ± 0.31 for adults. For the controls the corresponding values were 0.30 ± 0.14 and 0.50 ± 0.19.

In general, the dose in the investigated areas of the Ukraine was lower than in Zlynka. Therefore, less significant results were to be expected.

The surveys were made for 97 children/adolescents from the region around Bazar in West Ukraine in September 1993.

The individual selection of the children/adolescents envisaged for special medical examinations was made on the basis of measurement results obtained by Fastscan whole-body counters in the in-situ measurement trailer in Bazar. These measurement results of actual radiation exposure formed the basis for classifying the children/adolescents examined into exposure groups.

Using a cytogenetic questionnaire tailored to the examination project, relevant case history and additional exposure data of the children/adolescents were recorded by the Ukrainian physicians within the scope of blood sampling. Particular attention was paid to the following questions:

- residence prior to and after the nuclear accident
- family history (especially hereditary and cancerous diseases)
- present diseases
- passive or active smoker status
- current medication and previous treatment with cytostatic drugs, hormones etc.
- iodine blocking of the thyroid by potassium iodine during the early accident phase
- previous medical radiation exposures
- virus infections in the past four weeks

Clinical peculiarities of the children/adolescents were recorded in the case history.

These surveys were made immediately prior to blood sampling carried out by paediatricians on the 97 children/adolescents in the period from 02.09. to 04.09.1993 either at the Bazar hospital or in some cases in a school at Losnitza.
For chromosome analyses 4.5 ml blood was taken in each case with the parents' consent from the cubital vein using Sarstedt monovettes (anticoagulant: Li heparin). The individual monovettes were provided with a coded personal identification number.

The transport of the blood samples for processing in Berlin was started on 06.09.1993. The transport vehicle only arrived in Berlin on 09.09.1993.

In total, blood samples from 97 children/adolescents were available for chromosome analysis. Due to the long time which elapsed between blood sampling and the preparation of blood cultures it was not possible to obtain any usable results for 22 test persons so that only a total of 75 (37 boys, 38 girls aged 7 to 17 years) were included in the evaluation (see distribution of age and sex in table 3.6).

The children/adolescents came from West Ukraine, more particularly from the regions around Bazar, Narodici and Losnitza, with a level of Cs soil contamination from 37 to about 1480 kBq/m^2. The breakdown based on individual incorporation measurement data was as follows:

- category 1 (< 4kBq):	29 test persons
- category 2 (4 to 15 kBq):	23 test persons and
- category 3 (> 15 kBq):	23 test persons

The cytogenetic examinations were carried out in the form of a blind study, i.e. in a coded form until all analytical results were available. For more details see [23].

age	male	female	total
7	1	0	1
8	1	4	5
9	3	3	6
10	2	3	5
11	8	15	23
12	6	2	8
13	5	8	13
14	6	1	7
15	2	0	2
16	0	2	2
17	3	0	3
total	37	38	75

Table 3.6: Age and sex distribution of the test persons

The preparations were evaluated by optical microscopy with 1000-fold magnification. As far as possible, 1000 cells (metaphases) per test person were analysed for structural aberrations, separately according to chromatid and chromosome aberrations.

For the chromatid aberrations it is known that only one chromatid of a chromosome or several chromosomes is modified. The forms of chromatid aberrations are:

achromatic gap	gap
chromatid break	ctb
chromatid exchange	cte

In the case of chromosome aberrations, on the other hand, both chromatids of a chromosome or several chromosomes are aberrant. Among the chromosome aberrations are:

atypical monocentric chromosomes	at
excess acentric fragments	ace
dicentric chromosomes	dic
polycentric chromosomes	poly
ring chromosomes	r

If polycentric chromosomes can be identified, they are evaluated according to their centromere number (n) as (n-1) dicentric chromosomes, e.g. a tricentric chromosome corresponds in the evaluation to two dicentric chromosomes. A ring chromosome is counted as a dicentric chromosome.

As already mentioned, chromosome analyses results were only determined for 75 children (= 77 %). 22 test persons (23 %) were not incorporated in the evaluation, i.e. for 8 test persons preparation was no longer possible and for 14 test persons the number of analysable cells was much smaller than 100 so that no evaluation was carried out.

Usable results were obtained for 75 test persons, i.e.

for 42 test persons with	1000 cells
for 11 test persons with	> 500 cells and
for 22 test persons with	> 100 to < 500 cells

The total number of evaluated cells is 57,880. Table 3.7 contains the results obtained for the 75 children and shows the average aberration rates.

cate-gory	mean incorpor-ated activity [Bq]	number of persons	age [a]	sex m/f	number of cells	mean aberration rates ± SEM per 1000 cells		
						dic	ace	at
Σ		75	11.7±2.2	37/38	57880	4.6±0.3	7.2±0.4	0.7±0.1
1	1950	29	10.7±1.5	12/17	20105	4.2±0.5	9.4±0.7	0.5±0.2
2	8950	23	12.0±2.4	11/12	17480	5.3±0.6	10.9±0.8	0.6±0.2
3	24200	23	12.8±2.1	14/9	20295	4.3±0.5	7.5±0.6	0.8±0.2

Table 3.7: Mean aberration rates in the peripheral blood of Ukrainian children as a function of the radiation exposure categories:

Cat. 1: < 4000 Bq, Cat. 2: 4000 Bq - 15000Bq, Cat. 3: > 15000 Bq.
SEM standard errors of mean (Poisson statistics)
dic = dicentrics, ace = acentrics, at = atypical monocentrics

The average rates of chromatid aberrations are in the normal range, and it was found that these results can be classified within the lower reference range for all children examined.

However, the average rates of chromosome aberrations are perceptibly increased. The Berlin laboratory control values determined for men aged 18 to 48 years (total number of cells: 38,600) revealed an average rate of 0.4/1000 cells for dicentric chromosomes and 1.7/1000 cells for excess acentric fragments [26]. These control values have also been confirmed by other authors, or similar values have been specified.

Taking the three incorporation categories defined into consideration, there is no correlation between the average activity values and the average aberration rates for the children/adolescents examined by us. Moreover, the groups examined do not differ with respect to age distribution and sex ratio.

It is striking to note for the examination data on West Ukrainian children/adolescents reported here that a total of 43 cells with more than two dicentric chromosomes per cell (multiaberrant cells or overdispersion) were identified. In contrast, other examinations of 95 children from Volodarka only revealed 2 cells of this type and examinations of 30 children from the Kiev region only one cell of this type. Neel et al. [27] as well as Bochkov et al. [28] also frequently observed such multiaberrant cells in their chromosome analyses of persons from Ukraine and Russia independent of the level of radiation exposure. However, these data cannot be directly compared quantitatively with our results since both groups of authors evaluated a significantly smaller number of cells per person (≤ 200 and on average approx. 300). According to [29] 15 people from Belarus and one person from Kiev were investigated for chromosome abberations. In five people a statistically significant increase of dicentric chromosomes was observed. For four people the author also observed a significant increase of the dicentric yields and a highly significant overdispersion of the intercellular distribution of dicentrics. The cause of this kind of departure from the Poisson distribution by over-representation of cells with several aberrations was thought to be either viral infections (rogue cells) or incorporated radionuclides inducing an extremely heterogeneous exposure to ionizing radiation, i.e. hot particles.

On the basis of the questionnaire data and incorporation measurements it was possible to estimate the individual annual doses including external and internal radiation exposure for the 75 children/adolescents examined in 1993. The internal radiation exposure results from the measured body activity multiplied by the dose factor (DF). Tab. 3.8 gives the age-dependent dose factors for Cs-137 and Cs-134 as well as for the mixture (19:1) predominating at that time.

age [a]	DF Cs-137	DF Cs-134	DF mixture
0	0.45	0.55	0.456
1	0.19	0.25	0.194
5	0.11	0.15	0.113
10	0.071	0.090	0.072
15	0.045	0.064	0.046
20	0.038	0.053	0.039

Table 3.8: Whole-body dose factors (DF) for different age groups in mSv/kBq·a

The external dose had to be estimated from the soil contamination at the place of residence. The average soil contamination was known for four places of residence of the children/adolescents: Bazar 360 kBq/m^2, Narodici 365 kBq/m^2, Losnitza 757 kBq/m^2, Rudnaya-Bazarskaya 275 kBq/m^2. For some other places (Minski, Brodnik, Suchariovka, Mechinska, Babici, Lobarka, Osonka) the exact average soil contamination was unknown. But these were only very small places with a very limited number of test persons (n=15). So they were neglected in the individual dose estimation. Multiplication of the soil contamination at the place of residence by the area dose factor of 0.1 mSv $a^{-1}/Ci\ km^{-2}$, which ranges between those specified by the State Committee for Hydrometeorology for rural and urban areas [17], permitted a calculation of the external radiation exposure of the examined children/adolescents.

The results are compiled in fig. 3.17. It shows the number of dicentric chromosome aberrations plotted against the individual total dose as the sum of internal and external radiation exposure. It can be seen that a correlation between radiation dose and chromosome aberrations is not given even for this plotting. Therefore, it can only be assumed that other noxae have a dominant influence on the significant increase in chromosomal alterations in West Ukraine.

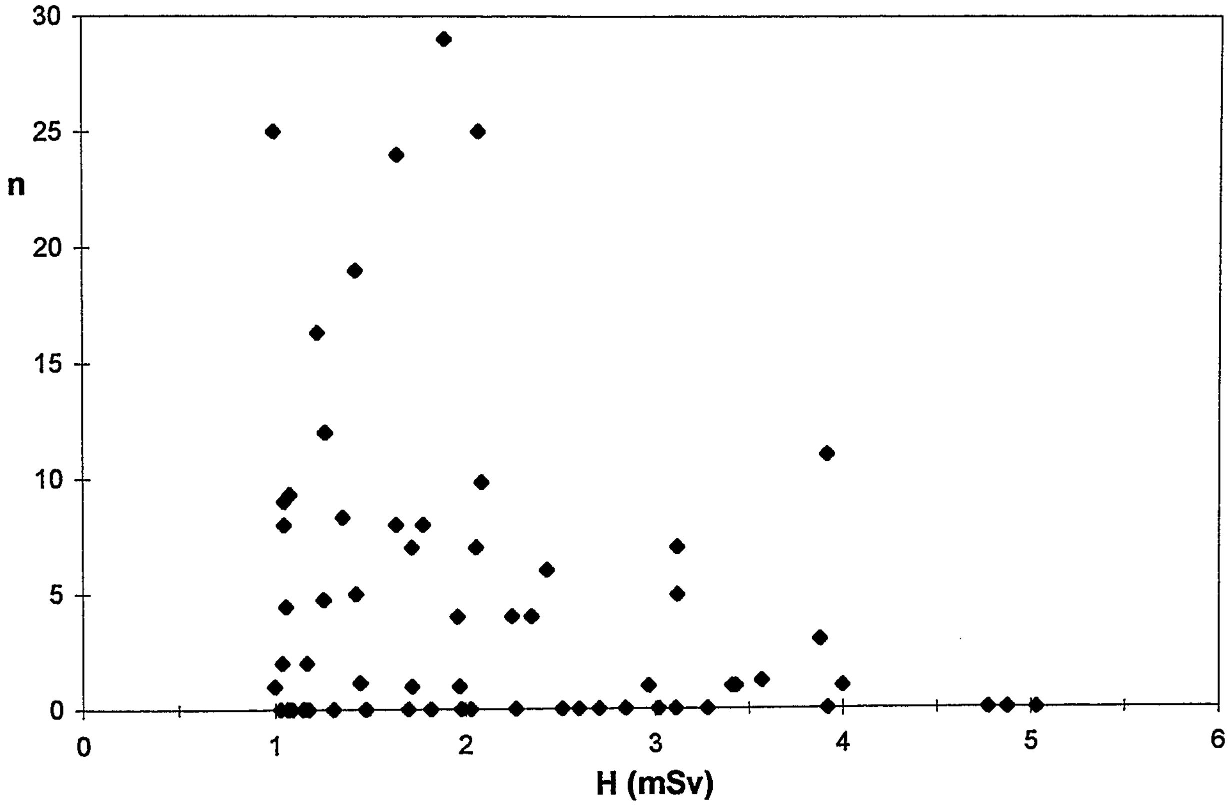


Fig. 3.17: Number of dicentric chromosome aberrations per 1000 cell divisions as a function of the effective dose equivalent due to internal and external radiation for the villages of Narodici, Bazar, Losniza and Rudnaya-Bazarskaya

4. Reconstruction of Thyroid Doses

The thyroid doses in the early phase after the accident are of special interest due to the fact of significantly increasing malignancy in the thyroids of children in the region of Gomel, Belarus, in Russia and in the Ukraine. On behalf of the Commission of the European Communities an international expert group formed a Panel of Thyroid Experts in order to evaluate the situation concerning reported increased rates of thyroid cancer in children. In their 1993 report [30] they came to the conclusion that in the years since the Chernobyl accident, there has been a substantial increase in the incidence of thyroid carcinoma in children in Southern Belarus and, as reported recently, in Northern Ukraine and Russia. It is unlikely that greater awareness and detection would account for this increase. The Panel concludes that the radiation exposure with average thyroid doses of 2 to 5 Gy in these territories may cause some effect on the thyroid of the children. On the basis of the life time risk estimate of ICRP 60 about 400 additional thyroid cancers in 100 000 children can be anticipated at a dose of 5 Gy to the childhood thyroid.

An explanation for these phenomena must be found in a new dose evaluation assuming that the official dose estimations are inadequate. On this basis, efforts have been started for a comprehensive reconstruction of thyroid doses especially in the early phase due to iodine isotopes.

According to [11], on the day of the accident the reactor inventory after 1100 days of operation included about 3 EBq I-131, 5 EBq I-133 and 3 - 4 EBq Te-132. About 50% of the iodine core inventory was released into the environment. Te-132, the precursor of I-132, may be released in the order of 10%. The radioactive plume that was responsible for the iodine contamination of the Ukraine and the Bryansk-Gomel spot was released from the reactor on about April 27 and finally reached the Russian contamination areas on April 28-29.
Wet fallout in combination with a varying wind velocity formed extremely localized iodine depositions differing by as much as one order of magnitude within a given settlement.

Looking at the reactor inventory it becomes clear that the situation is not changed fundamentally by the short-lived isotopes I-132 and I-133 because their activity is similar to that of I-131. Due to their dose factors, their total amount and their release rates the influence on the public radiation exposure by inhalation was about one order of magnitude higher than the I-131 thyroid dose immediately after the accident. But on April 27 and 28 the decay time was already more than 24 hours, thus reducing the influence of the short-lived isotopes in the Gomel district and in Russia by several orders of magnitude making them completely unimportant with the minor exception of Te-132 and its daughter I-132. Therefore, the majority of the thyroid doses to inhabitants was attributable to I-131 intake with milk and other contaminated food products. In the long term, the dose from the inhalation pathway was less important, especially under conditions of wet deposition. But even with only dry deposition the total thyroid dose from inhalation of I-131 was less then 10%, if no protective action was taken.

In the Controlled Areas (CA) of Russia, especially in the western districts of the Bryansk region, during the first half of May 1986, the I-131 concentration in milk was up to 100 kBq/l. After May 15 the radioiodine concentration in milk decreased with an effective half-life of approximately 5 days. This was observed for milk from collective farms.

In May 1986 the I-131 thyroid content was monitored by Russia scientists for several thousand inhabitants of the contaminated areas, among them many children. High values were found in many villages of the contaminated areas. On May 16-17 in the village of Barsuki in the Bryansk region the mean value for 7- to 11-year-old children was 55 kBq (27 individuals), from 12 to 17 years it was 159 kBq (4 individuals). The maximum values were up to 300 kBq for some individuals. The Cs-137 soil contamination in this village was nearly 3000 kBq/m^2 [11].

In the town of Novosybkov the I-131 content in the thyroid was considerably lower. The respective value for 7- to 11-year-old children was 5 kBq for 13 individuals. This was attributable not only to lower levels of radioactive contamination, but also to the earlier implementation of protective actions. For example, the delivery of local dairy products to elementary schools and kindergartens was discontinued from May 4 to 6. Several days later, Novosybkov was supplied with milk brought in from uncontaminated districts.

Based on the measured I-131 activity in the thyroid, the doses were calculated on the assumption that the daily iodine intake into the body was constant for the first 15 days and the biological half-lives of iodine secretion from the thyroid are between 16 days for babies and 100 days for adults. The Bryansk region is an iodine deficiency area with an estimated iodine intake of 70-80 µg/day and an average thyroid mass for adults of 26.7 g (normal 200 µg/d and 20 g, respectively).

From this, according to [11], the average thyroid dose in a village like Plavsk (Tula) with a Cs ground contamination of 450 kBq/m^2 was between 0.35 and 0.7 Sv for infants under 3 years, while the dose from external radiation due to Cs surface contamination was 50 mSv in the first year. In Nikolaevka, Krasnogorsk district, and in some other villages of the Bryansk region the average thyroid doses for infants were up to 2.7 Sv due to the higher ground contamination. In these villages protective measures were launched later compared with towns like Novosybkov. In general, the doses in the towns was comparably lower by a factor of 2 to 3.5. The average thyroid dose for infants in Novosybkov was only 0.1-0.2 Sv.

In May-June 1986 measurements of the thyroid radioactivity were carried out on 120 000 individuals in Belarus and 150 000 in the Ukraine including a high percentage of children. In the Ukraine, especially in Kiev, only the sale of green vegetables was strictly prohibited during the first few days after the accident. The I-131 content in milk was controlled in the dairies [31]. The averaged results of the measurements in Kiev showed an I-131 activity concentration of 1100 Bq/l during 25 days in May 1986. The consumption of this kind of milk was the main source of iodine intake into the thyroid. The measured iodine activity in the thyroid for inhabitants of Kiev showed maxima between the 20th and 30th day after the accident of 20 kBq (<7a), 70 kBq (8-11a) and again 20 kBq for older people.

According to [32] the measured activity data for eight heavily contaminated Ukrainian districts and the town of Pripyat resulted in thyroid mean doses which were highest in the district of Narodichi (0.5 Gy for adults >16a) and in the town of Pripyat (2.8 Gy for children <7a).

The collective thyroid dose in the Ukrainian districts was equal to 64,000 man-Sv, leading to the possibility of 300 additional thyroid cancers and a 1.4-fold increase above the spontaneous thyroid cancer morbidity for children. For the western districts of the Bryansk region a similar increase must also be expected. A population of 112000 persons received a collective thyroid dose of 22000 man-Sv resulting in 102 additional thyroid cancers with an expected excess over the spontaneous level of 20-40 % [11].

The need for dose reconstruction results from the lack of early thyroid measurements in many villages of the contaminated area. In addition, many data are inadequate. The failure of officials to reveal the extent of the accident and to enlist radiation protection specialists prevented systematic measurements during May 1986. As a result, the thyroid doses - mean dose values and individual doses - must be reconstructed today from very limited information.

At present, different techniques for dose reconstruction seem to be possible:
1. Linear regression analysis of the relationship between the mean thyroid dose and environmental parameters like cesium soil contamination or iodine milk concentration in May 1986.
2. Correlation between the mean thyroid dose and the mean content of cesium in the bodies of adults during June-August 1986, using the fact that the excretion of cesium from the body of an adult occurs slowly (half-life 100 d).
3. Determination of the I-129/I-127 relation in soil and, thus, reassessment of the fallout and washout of iodine in the early phase after the accident.

The first and second method were examined in [11]. A clear correlation was found between all the factors mentioned above. These investigations were made for the age group of 3-6 years in the Bryansk and Tula region. The correlation coefficient between thyroid doses and cesium soil contamination ranged from 0.86 to 0.95. The correlation between thyroid dose and cesium content in the body could be proved if a distinction was made between different soil types in the Tula and Bryansk region (chernozem soils and podzol soils, respectively). Then the correlations were similarly strong. Therefore, these two techniques were used to estimate the mean thyroid doses in villages where measurements were unavailable. For the estimation of individual doses the correlation between thyroid dose and individual milk consumption was investigated. A statistically significant relation was found with correlation coefficients of 0.6-0.8 for all age groups.

The third method is based on the fact that the long-lived iodine isotope I-129 ($T_{1/2}$=1.6E7 a) was created in the reactor and released after the accident in a fixed activity relation to I-131 of about 5.5E-8. The average residence time in the upper soil layer is very long. Thus, even after several years, I-129 can be used as a tracer for the deposition of the short-lived but radiologically much more important isotope I-131. The natural background contamination of I-129 is due to cosmic rays and spontaneous fission of uranium. The relation to the stable isotope I-127 is nearly constant and is in the order of about 6E-13. In the northern hemisphere this relation is enhanced due to atomic weapons tests. Here, the relation for typical environmental samples is in the order of E-9 to E-8. After the Chernobyl accident, in many European countries this relation increased to the order of E-6. Two results from human thyroid gland samples from Slovenia measured after the Chernobyl accident by [33] are in this range, 1.02E-6 and 1.26E-6.

Higher relations were found in the vicinity of nuclear facilities in the United States [34]. In the neighbourhood of the nuclear reprocessing plant in Hanford and in the nearby city of Richland the I-129/I-127 weight relation was between 0.3E-5 and 4E-5 in soil samples taken between 1972 and 1979. For grass samples the results were similar. On the other hand, in foodstuff these levels were significantly lower. The relation between the I-129 levels in foodstuff to the level in the surface soils used to produce the food was between 0.1 and 0.5 for normal sites and between 0.01 and 0.02 near the Hanford facility. For the stable isotope I-127 such a depletion has not been observed. Thus, the reduction of the relation I-129/I-127 in food is of the same order. Near the Hanford site in Richland,WA, weight ratios between 0.3E-6 and 5E-6 were found in milk and vegetables.

The authors [34] did also find that the I-129 level from the discharge of a nuclear reprocessing plant differs between grass and forest communities. Forest communities were more efficient in accumulating I-129 from the atmosphere. This is attributed to the larger volume of air filtered by forest areas. The weight ratios in forest communities were higher by a factor of 15 to 20 in the litter and by a factor of 3 to 8 in the soil.

The reconstruction of thyroid doses by determining the I-129 concentration in environmental samples depends on many parameters which may be uncertain. As shown above, the determination of the atomic ratio I-129/I-127 of the normal background before deposition from the accident took place is one of the parameters that must be evaluated carefully. Today it can only be assessed by background measurements in areas which are comparable but not affected by the accident.

In 1994 we started a measuring project*) to reconstruct the radiation dose received by the thyroid gland from the short-lived iodine nuclides by measuring the soil contamination with the long-lived

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*) This project was supported by the European Union

nuclide I-129. The soil samples were taken from undisturbed ground around villages or towns according to the I-131 soil measurements in the year 1986. It was proved by measurement of the soil contamination profile that the ground was really undisturbed. Since the general distribution of I-129 with depth was unknown a depth-dependent determination of the soil contamination was necessary. This means that the soil samples have to be divided into layers of 2 cm in thickness.

Soil samples were taken from undisturbed ground around 13 villages or towns in the western Bryansk region (Russia), see fig 4.1. To determine the I-129 soil contamination caused by Chernobyl only, one has to subtract the background soil contamination due to atomic weapons test explosions in the atmosphere. Therefore, soil samples were also taken from low-contaminated areas similar in the kind of soil and climate. In the background areas the Cs-137 soil contamination was less by three orders of magnitude. The radioactivity, mainly of the nuclide Cs-137, was determined by gamma spectrometry. According to the radioactivity content of the samples the measuring time changed from 5 to 900 minutes. The concentrations of I-129 and I-127 were measured by neutron activation analysis.

For each village or town the normalized depth profiles for Cs-137 were determined including the "background" villages, see fig. 4.2 and 4.3. We observed that the mean slope of the "background" depth profiles is smaller than the others. The so-called relaxation length is 7.5 cm in the "background" areas compared to about 4 cm in the regions contaminated by the Chernobyl reactor accident. This may be explained by the longer contamination time of the atomic bomb tests. The results are in good agreement with the Russian average values and with Russian measurements in the year 1986. The good correlation of the Cs-137 depth profiles indicate that there are only negligible uncertainties associated with soil sampling.

The results of the retrospective dose reconstruction are given for the two settlements Novozybkov and Nikolaevka. The I-129 concentration in soil for the town of Novosybkov was determinated by neutron analysis for the 2 cm layers from 0 to 10 cm in depth. For the two layers 6 - 8 and 8 - 10 cm the I-129 concentration has the constant value of about 1 mBq/kg. The soil measurements of the nearly uncontaminated village of Kostenichi for the 0-2 cm layer give about the same value, which is therefore regarded as the background value. Subtracting this background the I-129 contamination is assessed as:

Novosybkov	0.481 Bq I-129/m^2
Nikolaevka	0.446 Bq I-129/m^2

Assuming the Chernobyl reactor had been in operation for about 2 years before the accident the I-131 soil contamination results in:

Novosybkov	8.7 (0,603/4,76)	MBq I-131/m^2
Nikolaevka	8.0 (3,18/25,1)	MBq I-131/m^2.

For a longer operating time of the reactor the I-131 soil contamination will decrease. The values in the brackets are Russian measuring values. The first one is a measurement of one single soil sample for each settlement on 22.05.1986. The dispersion calculation shows that the main contamination by washout happened on 28.04.1986. The correction for a 24-day decay of I-131 results in the second value in the brackets. These results show that our assessments give at least the same order of magnitude for the I-131 contamination as the Russian measurements of the soil contamination at the end of May 1986.

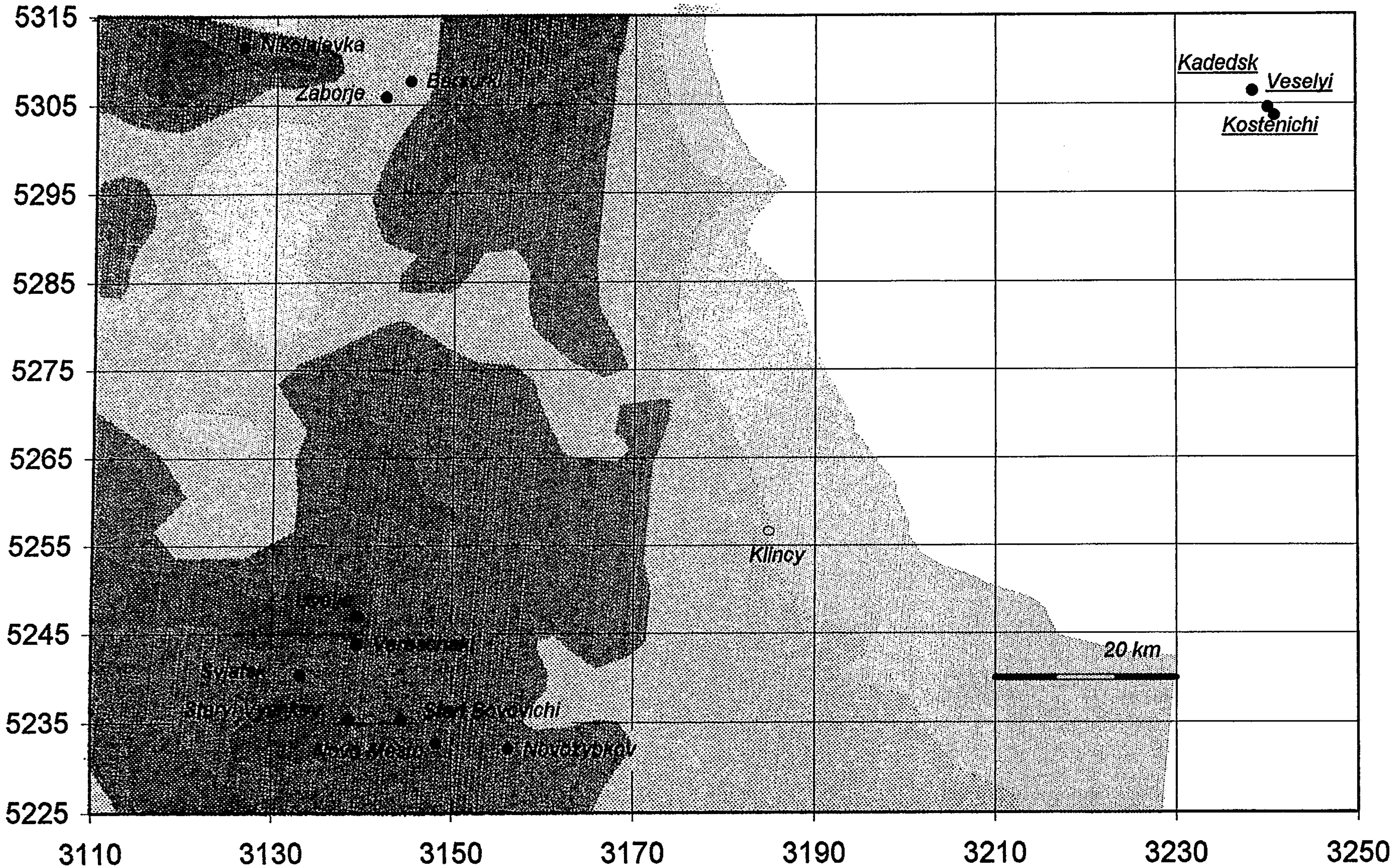
By model calculation without countermeasures we obtain the following integrated thyroid doses for infants caused by ingestion of I-131-contaminated food:

Novosybkov 1.0 Sv
Nikolaevka 0.9 Sv.

Due to countermeasures the average doses are lower by one order of magnitude compared with the results from Russian scientists given above.

The I-129 activities found in the soil samples are listed in fig. 4.4. There is no clear dependence of the iodine contents with depth. A strong decrease of the concentration with depth in the upper layers was to be expected but this was not found by our results. This makes it necessary for more I-129 analyses to be done down to greater depths where the I-129 concentration strongly decreases.

Fig. 4.1 Soil sampling locations



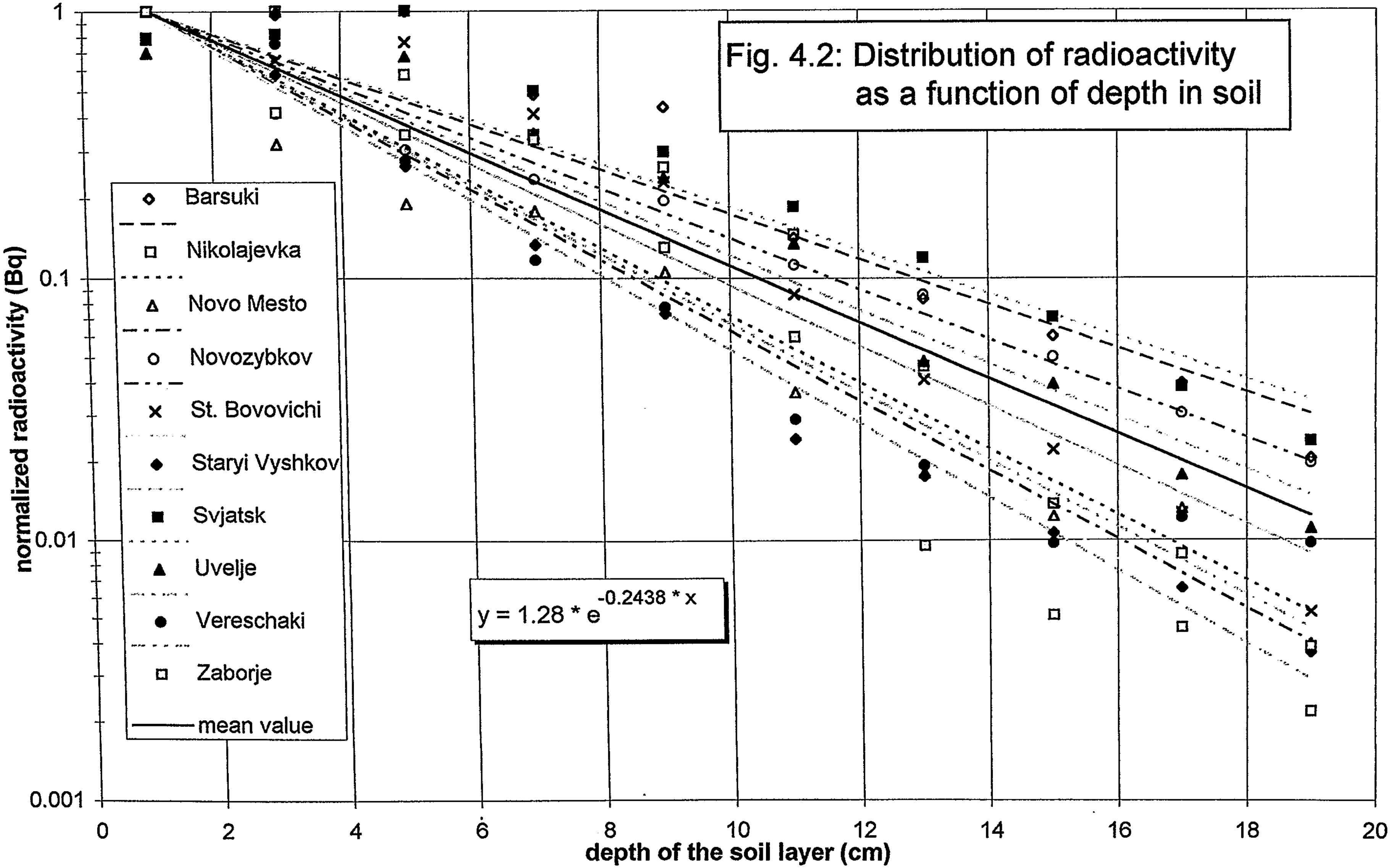


Fig. 4.2: Distribution of radioactivity
as a function of depth in soil
normalized radioactivity (Bq)
depth of the soil layer (cm)
1
0.1
0.01
0.001
0
2
4
6
8
10
12
14
16
18
20
Barsuki
Nikolajevka
Novo Mesto
Novozybkov
St. Bovovichi
Staryi Vyshkov
Svjatsk
Uvelje
Vereschaki
Zaborje
mean value
y = 1.28 * e^(-0.2438 * x)

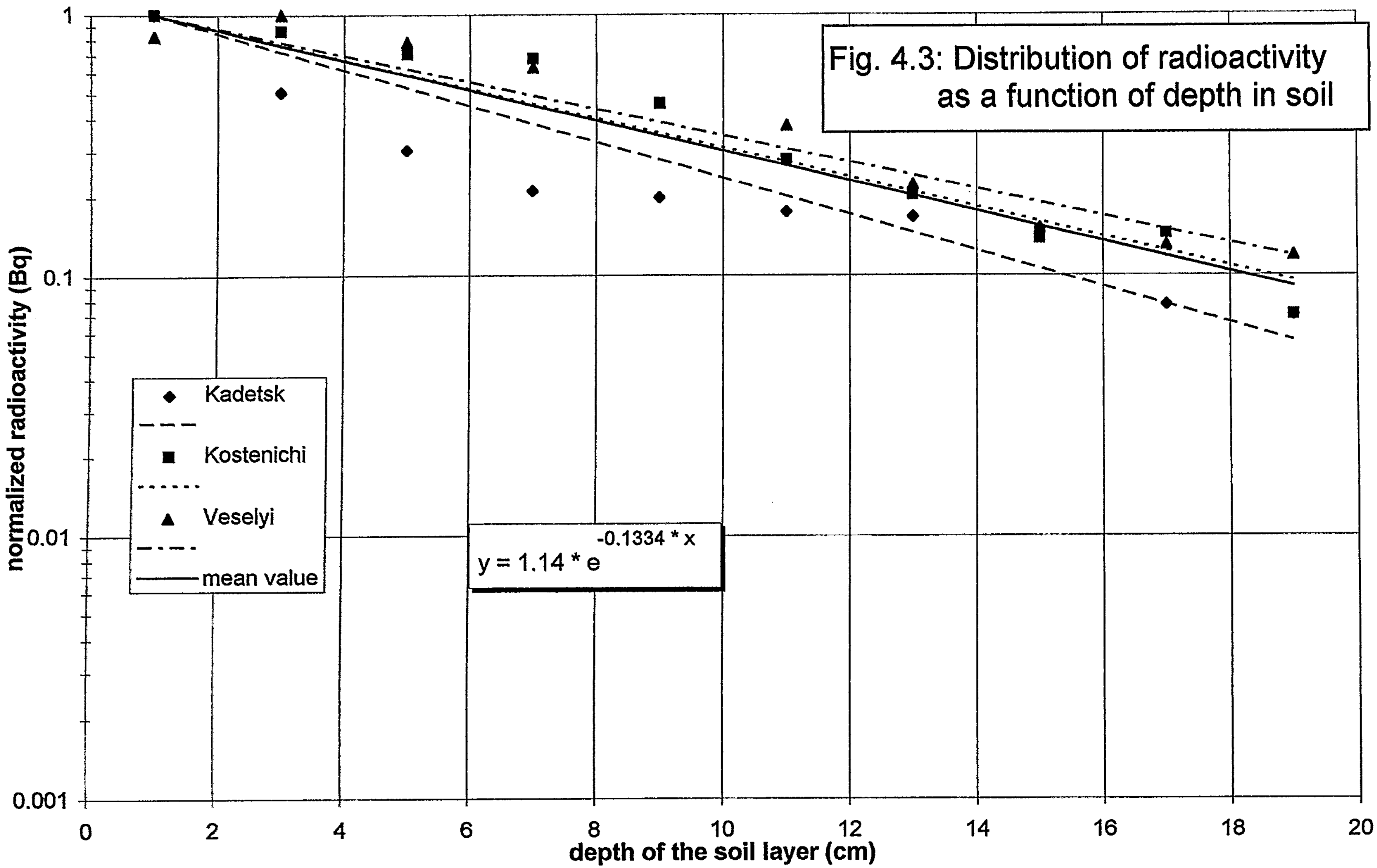
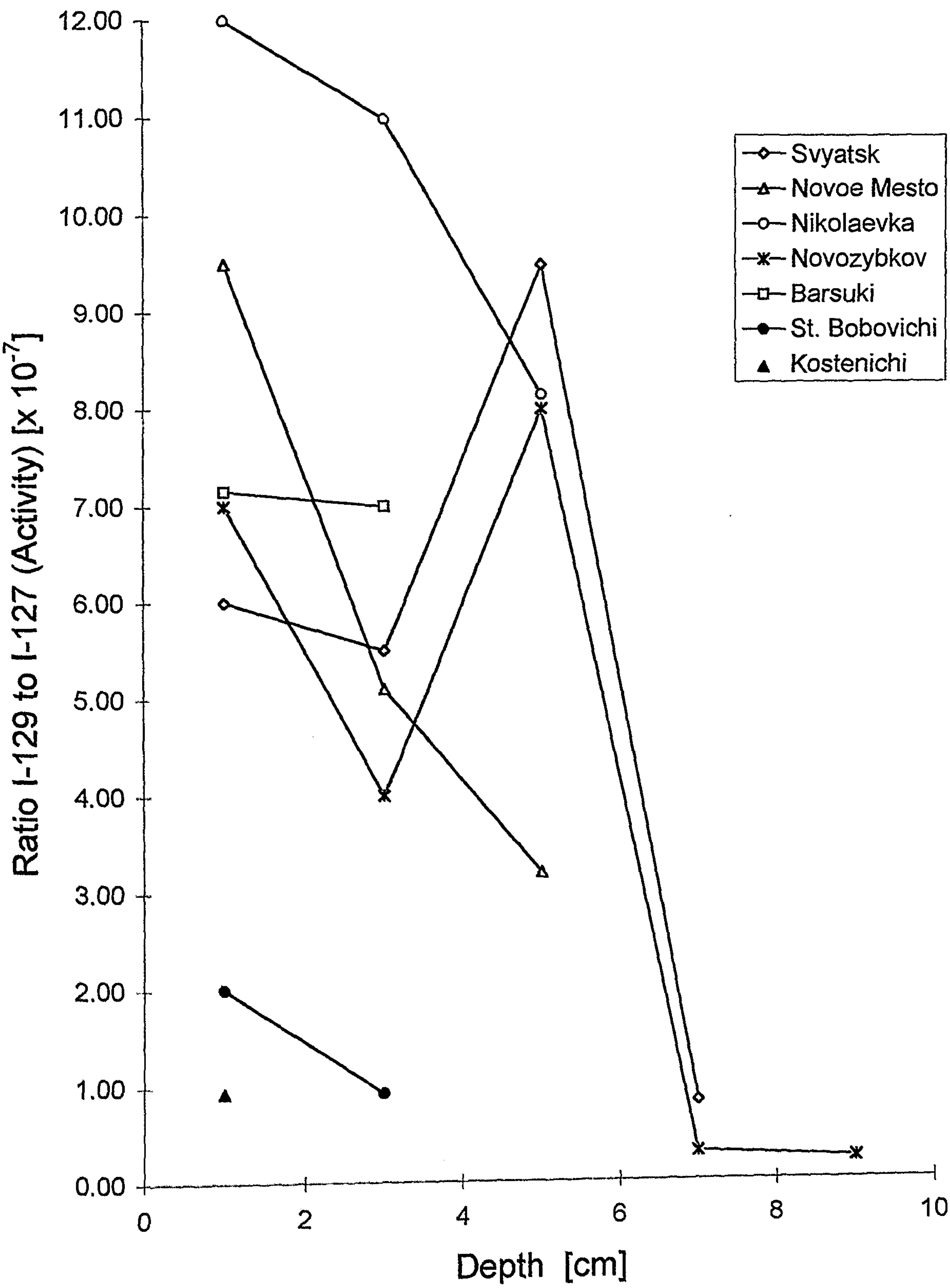


Fig. 4.3: Distribution of radioactivity as a function of depth in soil

Fig. 4.4

I-129 in Soil Samples



5. Assessment of the Population Dose and Dose Reconstruction

An assessment of the total dose from internal and external exposure is necessary from two aspects:

- The actual annual dose must be known as a decision basis for remedial measures, i.e. relocation.
- The lifetime dose must be calculated and reconstructed for risk evaluations.

In this section the annual doses and the 10-year dose commitment will be calculated on the basis of the measurement results from our measurement project and compared with other results.

The dosimetric evaluation of our measurements in the year 1991 for some villages in the Bryansk region (Russia) is given in tab. 5.1. On the assumption that the cesium body burden is constant over the whole year, an ingestion dose factor of 0.04 mSv/kBq·a for adults >20 a applies, (see tab. 3.8 in section 3.3).

Further, tab. 5.1 presents the results of the external dose calculation for the mean soil contamination of each village. These were partly official data because our own contamination measurements were not complete for all villages. However, both data sets were in good agreement. According to section 3.2 the calculation of the external dose equivalent from soil contamination in 1991 was performed applying our area dose factors measured in 1992 and 1993 by TLD measurements (s. section 3.2, tab. 3.5). The 1991 value is calculated from these results. Taking migration effects and a higher Cs-134 fraction into consideration the dose rate per unit contamination was higher by 14 % than in 1992. Thus, the area dose factor for 1991 is ADF = 1.85 mSv·m^2/MBq·a.

The next tab. 5.2 gives corresponding results for villages and towns in the Gomel region (Belarus) for the years 1992 and 1993. In this region a total of 23,000 measurements were carried out. The internal exposure is calculated from our whole body measurements, the external exposure from the measured Cs-137 soil contamination complemented by Belarus data [36]. Tab. 5.2 gives the impression that the internal dose is even more important than the external dose for some villages. But in towns and villages where the supply of low-contaminated food has almost been ensured the internal doses were reduced to a fraction of not more than 20 %. In rural areas this is still a problem, and in many villages the population have to supply themselves with garden or forest produce.

We found the highest current doses in Kirov in the Narovlya district. This is a settlement near to the forest with a high degree of self-sufficiency.

In the Gomel region we were not allowed to use the very mobile vans to make measurements. Therefore, measurements in remote villages were not possible and we can expect that there are many more villages like Kirov and Svetilovichi in the Gomel region with a relatively high population dose.

Tab. 5.3 completes the data for some locations in the Shitomir and Rovno region in the Ukraine in 1993. In the Rovno region internal doses are high, though the soil contaminations are low. This shows once more the importance of additional factors (closeness to forest, soil transfer factors, nutrition habits) for the amount of incorporated activities.

The ICRP dose limit for the general population of 1 mSv/a is exceeded in most of the villages given in tab. 5.1, 5.2 and 5.3, but in no case was the dose limit for occupationally exposed workers (50 mSv/a) reached. The highest individual dose during the whole measuring campaign was reached by a man with a body burden of about 770 kBq in the Narovlya district (see tab 3.2 in section 3.1). The resulting dose was about 30 mSv/a. All results in tab. 5.1-5.3 are given for adults. For children the doses are in the same range, because, in general, they had significantly lower contamination. This compensates the higher biological sensitivity. This can be seen from table 5.4 where for some of the major settlements the mean ingestion doses are given for different age groups.

settlement	mean body burden kBq	Cs ground contamination kBq/m^2	int.dose mSv/a	ext.dose** mSv/a	total dose mSv/a
Unecha	41.5	519*	1.7	0.96	2.7
Beresovka	32.5	238*	1.3	0.44	1.7
Sarechye	26.4	-	1.1	-	-
Veprin	24.7	1001	1.0	1.85	2.9
Usherpye	18.1	566	0.7	1.05	1.8
Blizna	16.2	176*	0.6	0.33	0.9
Tulu Kovshina	13.7	345*	0.5	0.64	1.1
Guta Koretskaya	13.1	962	0.5	1.78	2.3
Roshniy	13.1	540	0.5	1.00	1.5
Lopatni	9.7	195*	0.4	0.37	0.5

* 1992 ** ADF = 1.85 mSv·m^2/MBq·a

Tab. 5.1: Average annual doses of adults in villages of the Bryansk region (Russia) 1991

settlement	mean body burden kBq		Cs ground * contamination kBq/m²	int.dose mSv/a		external dose** mSv/a		total dose mSv/a	
	1992	1993		1992	1993	1992	1993	1992	1993
Bragin	-	3.2	810	-	0.13	1.31	1.09	-	1.22
Yurovichi	3.0	-	296	0.12	-	0.48	0.40	0.60	-
Kirov	-	57.5	1400	-	2.30	2.27	1.88	-	4.18
Korma	2.4	2.1	700	0.10	0.08	1.13	0.94	1.23	1.02
Leltshitzy	4.1	7.4	90	0.16	0.30	0.15	0.12	0.31	0.42
Mozyr	1.1	-	65	0.04	-	0.11	0.09	0.15	-
Narovlya	-	8.9	710	-	0.36	1.15	0.95	-	1.31
Shitkovichi	-	2.7	135	-	0.11	0.22	0.18	-	0.29
Svetilovichi	7.0	8.3	850	0.28	0.33	1.38	1.14	1.66	1.47
Vetka	2.4	-	740	0.10	-	1.20	0.99	1.30	-

* from [36] ** ADF=1.62 mSv·m²/MBq·a for 1992 and 1.34 mSv·m²/MBq·a for 1993

Tab.5.2 Average annual dose of adults in towns and villages of the Gomel region of Belarus in 1992 and 1993

settlement	mean body burden kBq	Cs ground Contamination kBq/m²	int.dose mSv/a	ext.dose mSv/a	total dose mSv/a
Bazar	6.8	360	0.27	0.48	0.75
Emilcino	1.4	37	0.06	0.05	0.11
Korosten	1.3	189	0.05	0.25	0.30
Luginy	5.6	120	0.23	0.16	0.39
Narodichi	14.7	365	0.59	0.49	1.08
Olevsk	4.8	-	0.19	-	-
Ovruch	2.2	-	0.09	-	-
Drosdyn	44.4	25	1.78	0.03	1.81
Shachi	19.6	-	0.78	-	-
Cheremel	66.6	55	2.66	0.08	2.74
Verbovka	26.3	57	1.05	0.08	1.13
Veshiza	59.2	30	2.37	0.04	2.41

* ADF= 1.34 mSv·m²/MBq·a

Tab. 5.3 Average annual dose of adults in towns and villages of the Shitomir and Rovno region (Ukraine) in 1993

district	settlement	1993 [mSv/a]				
		1-5 years	5-10 years	10-15 years	15-20 years	> 20 years
Orel	Bolchov	< 0.05 *)	< 0.03*)	< 0.02 *)	< 0.01*)	< 0.01 *)
Bryansk	Klincy	0.16	0.17	0.16	0.13	0.12
	Starodub	0.06	0.03	0.03	0.04	0.03
	Klimovo	0.25	0.19	0.15	0.14	0.18
	Gordeyevka	0.38	0.36	0.29	0.35	0.36
	Krasnaya Gora	0.27	0.33	0.42	0.28	0.42
	Mirny	0.18	0.33	0.34	0.46	0.50
	Novosybkov	0.20	0.40	0.40	0.29	0.32
	Zlynka	0.36	0.34	0.51	0.54	0.50
Gomel	Bragin	0.08	0.12	0.13	0.13	0.12
	Kirov	1.10	(1.7)+)	2.20	(1.9)+)	2.20
	Korma	0.08	0.07	0.08	0.08	0.08
	Lelchizy	0.14	0.21	0.23	0.25	0.28
	Narovlya	0.31	0.22	0.34	0.35	0.34
	Shitkovichi	0.09	0.08	0.09	0.10	0.10
	Svetilovichi	0.38	0.31	0.48	0.34	0.31
Mogilev	Chechersk	0.12	0.12	0.16	0.14	0.12
	Bychov	0.06	0.03	0.03	0.03	0.04
	Kricev	< 0.05 *)	0.03	0.02	0.01	0.02
	Meshisetki	0.05	0.03	0.02	0.02	0.02
	Palush	(0.11)+)	(0.18)+)	(0.26)+)	(0.38)+)	0.34
	Slavgorod	0.06	0.03	0.05	0.05	0.08
	Chaucy	0.05	0.04	0.03	0.03	0.04
	Cherikov	0.06	0.04	0.03	0.03	0.05
	Viduychi	0.14	0.33	0.28	0.24	0.36
Shitomir	Bazar	0.29	0.33	0.52	0.32	0.26
	Emilcino	0.05	0.05	0.04	0.04	0.05
	Korosten	0.06	0.05	0.04	0.04	0.05
	Luginy	0.24	0.19	0.21	0.25	0.21
	Naroditshi	0.47	0.58	0.71	0.55	0.56
	Olevsk	0.20	0.18	0.17	0.21	0.18
	Ovrutsh	0.12	0.09	0.10	0.08	0.09

*) results mostly below limit of detection

+) only single measurements in this age group

Table 5.4: Mean annual doses in 1993 due to ingestion of radiocesium given for different age groups

Effective doses have also been determined from our TLD measurements of the external radiation exposure. For some settlements the mean external doses in 1993 are shown together with the mean internal doses in figure 5.1. The value of the internal dose for Saborye is taken from [40]. The value of the external dose for Kirov is taken from the year 1992. For most of these settlements the external dose is more important than the internal dose. The mean external doses estimated from ground contamination agree sufficiently well with the mean external doses from direct measurements.

The method of external dose reconstruction for a five-year period from 1986 to 1991 is based on the official results [17] of more than 10,000 direct measurements of external radiation exposure with TLDs in 1989 and 1990. This method makes use of the so-called area dose factors (ADF(i)) which are given in tab. 5.5 for different years i and settlement types. Neglecting the additional dose rate from short-lived nuclides during the first month after the accident the external exposure D_5 for the early period till 1991 can be determined by the formula

$$D(i) = ADF(i) \cdot C, \qquad D_5 = \sum_{i=1}^{5} D_{(i)}$$

where C is the soil contamination in kBq/m^2 in 1991. For the time before 1991 the ADF values already consider the physical decay and an exponential depth distribution for the soil contamination.

	1987	1988	1989	1990	1991
village	9.2	8.4	6.2	4.1	3.5
urbanized village	6.3	5.8	4.2	2.8	2.4
district town	4.2	3.8	2.8	1.9	1.6

Tab. 5.5: Area dose factors for the first few years after the accident in $mSv \cdot m^2/MBq \cdot a$, according to [17]

Starting from our own measurements with area dose factors for 1992 and 1993 according to tab. 3.5 in section 3.2, our external dose reconstruction makes use of the fact that the decrease in the dose rate of the Cs-137/Cs-134 mixture due to the physical decay, considering the higher area dose factor for Cs-134 for a 10 a period since 1986, is about 65 %; 18 % in the first year, 4 % in the tenth. In addition, migration effects must be considered. Our measurements in 1992 over undisturbed ground yielded a mean value of 0.69 nGy/h over soil contaminated with 1 kBq/m^2 Cs-137. This is a reduction of 63 % compared with the 1986 value of 1.82 nGy/h (see section 3.2). Thus, migration effects are in the same order of magnitude as physical decay. For our calculation we considered all these effects with the consequence that an annual dose in 1993 resulted in an 18 times higher 10 a dose.

The reconstruction of the internal dose on the basis of our whole body counter measurements needs some knowledge about the development of the activity level in humans during the first few years after the accident. This information is given in fig. 3.1 and 3.2 from section 3.1. In general, the average activity in male inhabitants of villages in the Bryansk region decreased by a factor of about 2 within 600 days starting from August 1986. In the town of Novosybkov this factor was about 5. As mentioned above (section 3.1), the results from Belarus [14] reveal a decrease by a factor of 10 from 1986 to 1991. But unfortunately, this information does not seem to be very reliable due to the fact that the measurements were not carried out systematically over the years always with the same persons. Therefore, the variation for these results is great.

Systematic monitoring of a fixed group of persons over the first few years after the accident was performed in Germany [35]. Fig. 3.2 in section 3.1 gives the results of measurements made with a group of adults and children from the Bavarian region of Berchtesgarden. This area is of a rural type. It was contaminated with a cesium activity of about 50 kBq/m^2 and thus it would range at the lower limit of the so-called Observed Area (OA). After an initial increase in 1986, fig. 3.2 shows an exponential decrease since 1987 with a factor of about 5 for the first 600 days. It is thus similar to that of Novosybkov. Considering the fact that the life style in rural German areas seems to be more urbanized compared with Russian villages and that protective measures in Germany were taken as early and comprehensively as in Novosybkov this can be interpreted as a confirmation of Russian results. Therefore, we suggest for dose reconstruction purposes the Berchtesgarden curves for towns with a decrease by a factor of 20 from the end of 1986 to 1991. For villages we suggest an exponential decrease by a factor of 2 after 600 days without distinction between men and women, children and adults, and a decrease by a factor of 15 from the maximum at the end of 1986 to 1991, being between the German and the Belarussian experience.

Fig. 5.2 presents the internal and external dose distribution for a town and a village based on measurements in 1991 but reconstructed and extrapolated for a period of 10 years after the accident. They show the relations between external and internal doses and the increasing importance of the ground radiation as time proceeds. According to fig. 3.12 and 3.13 in section 3.1 we did not observe such a rapid decrease after 1991 as is indicated in Fig. 5.2 from extrapolation of the first 5 years. Our measurement results for the year 1991-1993 are nearly constant. They are also indicated in fig. 5.2.

The sum of the annual doses as indicated in fig. 5.2 for the first 5 a, but assuming constant body activities after 1991, was calculated for a period of 10 years after the accident. In tab. 5.6 the results of the ten-year dose equivalent for the external and internal doses are listed for those villages and towns of the three republics which had the highest mean body activities. The different 10 a total doses for 1991-1993 in villages of the Klincy district give a good impression of the variation and uncertainty of our dose evaluation (see tab. 5.6 last column).

To compare our results with those of other organizations in fig. 5.3 the doses of the official estimates and those from the IAEA project [12] in the years 1990-1991 are given together with a summary of our results.

Obviously, our external doses are significantly lower than those from the other estimates. This is due to the shorter time period of 10 a and to our area dose factors from tab. 3.5 in section 3.2, which are about 50 % of the official estimates from [17]. But our factors seem to be very reliable because they are in good agreement with those from Erkin and Lebedev [11] and they were obtained by sensitive TLD measurements under realistic conditions.

The internal doses are also lower than the official estimates, but they are in sufficient agreement with the IAEA results considering that they made many fewer measurements. Thus, our total doses are in the lower part of the IAEA results.

village	district	year of measure-ment	mean body activity kBq	Cs ground contamin. kBq/m^2	10 a int. dose mSv	10 a ext. dose mSv	10 a total dose, mSv
Cheremel	Rovno	1993	66.4	55	104	1	105
Veshiza	Rovno	1993	59.2	30	92	1	93
Kirov	Gomel	1993	57.5	1400	90	34	124
Staroye Selo	Rovno	1993	46.0	51	72	1	73
Drosdyn	Rovno	1993	44.4	25	70	1	71
Unecha	Klincy	1991	41.5	(519)	68	14	82
		1992	21.3	519	34	14	48
		1993	32.1	431	51	11	62
Veprin	Klincy	1991	24.7	1001	40	26	66
		1992	34.9	1030	56	27	83
		1993	39.2	855	62	21	83
Beresovka	Klincy	1991	32.5	(238)	53	6	59
		1992	29.5	238	47	6	53
		1993	33.9	204	53	5	58
Sarechye	Klincy	1991	26.4	-	42	-	~52
		1992	27.1	-	43	-	~53
		1993	34.0	373	53	9	62
Tulukovchina	Klincy	1991	13.7	(345)	22	10	33
		1992	13.8	345	22	10	32
		1993	29.2	381	46	10	56
Uvelye	Krasnaya Gora	1992	28.3	-	45	-	~74
		1993	29.2	(1190)*	46	29	75
Verbovka	Rovno	1993	26.3	57	41	1	42
Saborye	Krasnaya Gora	1993	26.0	(2300)*	41	56	97

Tab. 5.6: 10-year doses for adults in settlements with the highest body activities measured in 1991-1993

* Recalculated from a mean external dose of 3.1 mSv/a measured by TLDs.

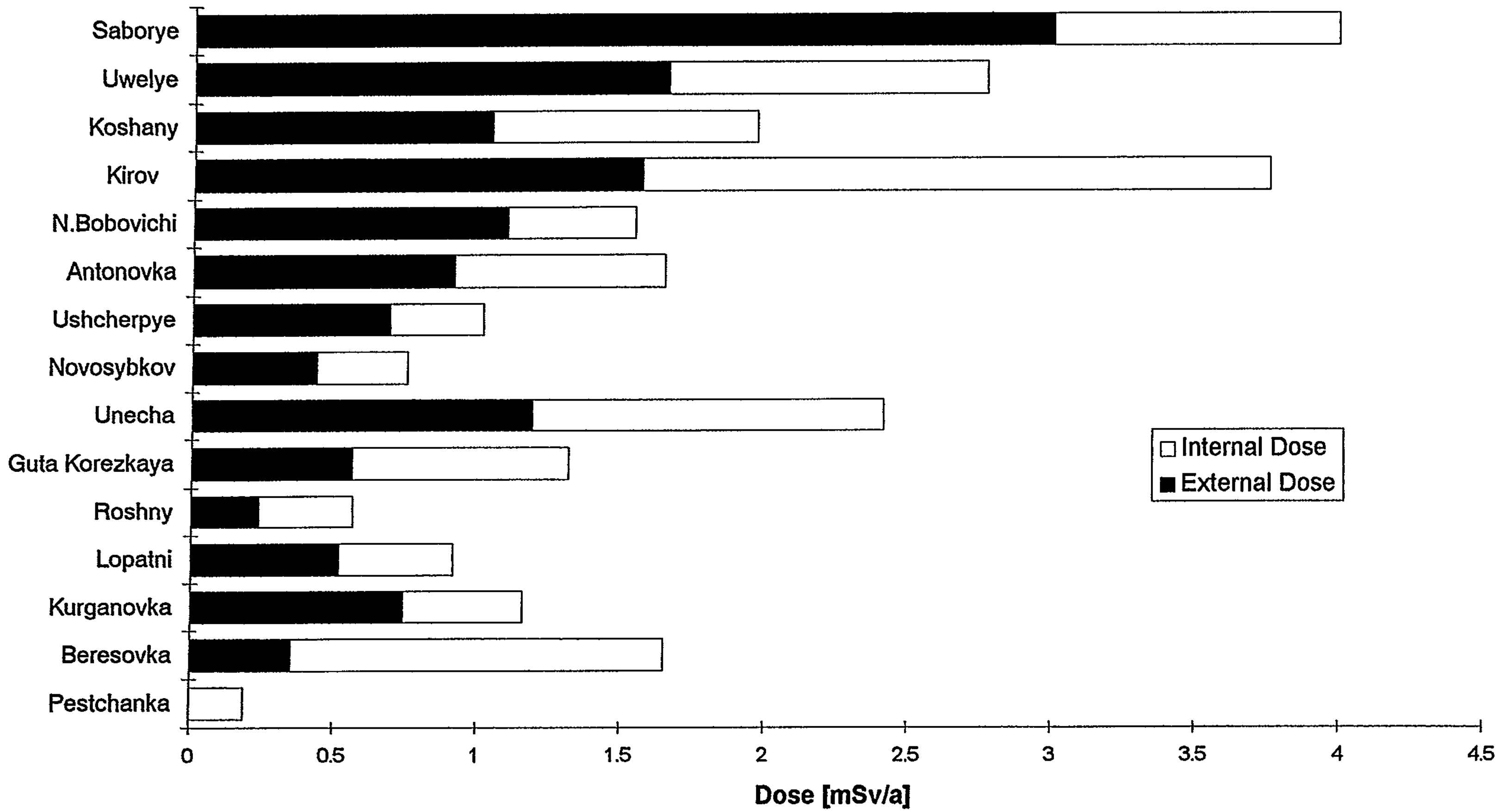


Fig. 5.1: Average internal and external exposure in 1993 for different places

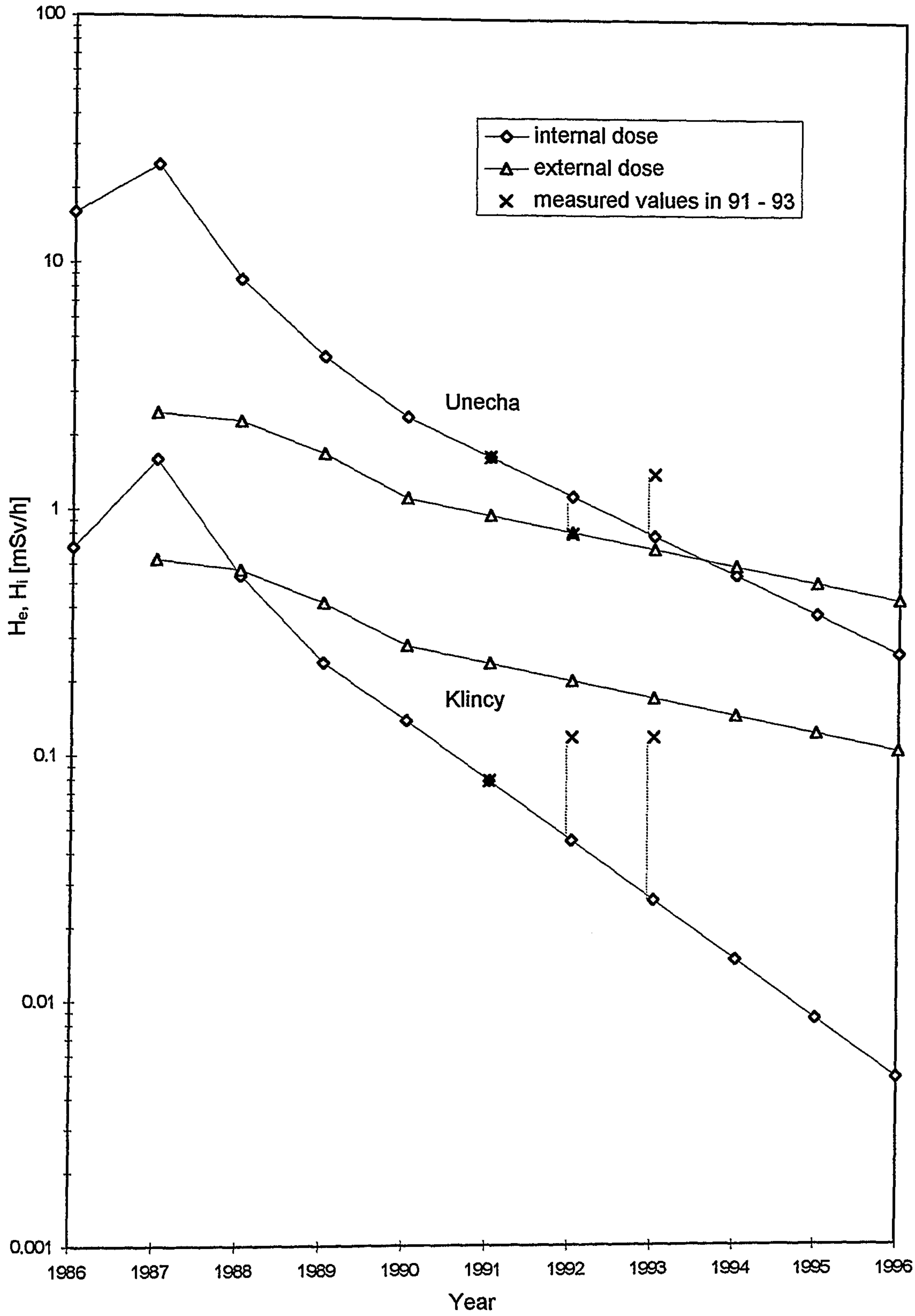


Fig. 5.2: Internal and external annual dose for the first ten years after the accident for the town of Klincy and the village of Unecha in Russia

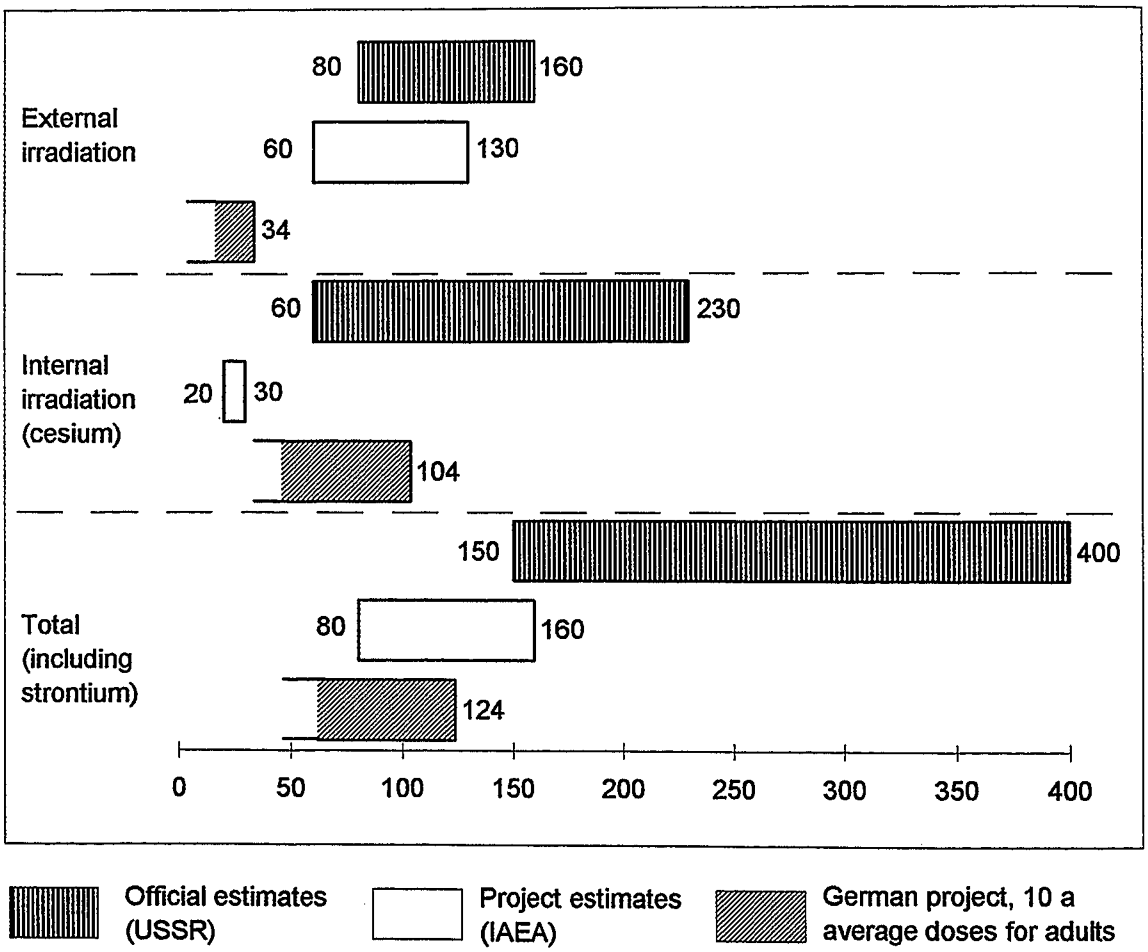


Fig.5.3: Comparison of project estimates and officially reported estimates for radiation dose to the population in selected settlements.

6. Concluding Remarks

The measuring campaign carried out by the Federal Republic of Germany in 1991 to 1993 has achieved its aim since it earned widespread recognition and acceptance both amongst the population and also from experts. The hope that this measuring campaign succeeded in providing objective information for the public on their actual radiation risk and, in this way, reduced unfounded fears has been confirmed by the information we have received. Indication is given that in many regions a climate of trust has been recreated and the psychological wellbeing of many people has been improved. It is hoped that this will be associated with a consolidation of the social structure and an improvement in public health.

Measurements carried out in the territories of the three republics contaminated by the reactor accident at Chernobyl of the radionuclides incorporated by the public provided data indicating that about 98 % of cases are within a completely safe range. There is no enhanced risk to the health of this proportion of the population from their food consumption or their behaviour. Whole-body measurements of more than 25 kBq Cs-137/134 were only found for about 2 % of the measurements. The elevated values were mainly found in the western Bryansk district in Russia, in the region of Gomel in Belarus and in the Rovno district in the Ukraine. In all cases the pollution was smaller than that permitted for persons professionally exposed to radiation by ICRP. These values therefore do not represent an acute health risk. However, the more heavily contaminated group should be monitored in the same way as persons professionally exposed to radiation. This means, in particular, the implementation of regular repeated measurements and accompanying medical check-ups.

In agreement with environmental measurements, apart from Cs-137, Cs-134 and nuclides of natural origin, no further gamma-emitting incorporations were determined in the population. Due to experience with environmental measurements, it was not necessary to determine Sr-90 in humans. It is clear that short-lived radionuclides like I-131 no longer existed during our campaign. The investigation of thyroid doses by investigating the I-129 concentration in soil could not be completed in the frame of this project.

A dose reconstruction in order to determine the total burden since the accident is hardly possible from the data gathered in single measuring campaigns alone and cannot go beyond rough estimates of averages. It is therefore indispensable to evaluate and collect into a data bank all data gathered in the affected areas in connection with the Chernobyl reactor accident. An appropriate method would be to incorporate our measured data into general data banks.

Not least with respect to the problems of dose reconstruction, more emphasis should be placed on long-term scientific aspects, in particular improved coordination between environmental and whole body measurements. In selected cases and/or at selected locations, medical methods of examining the radiation exposure (e.g. blood tests) should be expanded.

All our assistance can only be regarded as a support towards self-help. In the long term our goal was to integrate our Russian, Belarussian and Ukrainian colleagues on site to achieve the same confidence as expressed by the population in our work.

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