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Exposures from External Radiation and from Inhalation of Resuspended Material

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Abstract. In the modelling of external exposures due to cesium released during the reactor accident of Chernobyl, gamma dose rates in air over open undisturbed sites are considered to be different according to the unsoluble fraction in the deposit. This is taken into account by forming different classes according to the distance from the Chernobyl NPP. The effect of the different migration behaviour in these distance classes on the gamma dose rate in air is found to increase with time. Predictions of gamma dose rates in air are based on measurements of the nuclear weapons tests fallout. Various population groups in the CIS countries are defined according to their place of residence (rural or urban), their occupation or age (indoor resp. outdoor workers, pensioners, school-children, or preschool-children), and their kind of residence (wooden, brick, or multistorey house). Model results for various population groups are compared with the results of TLD-measurements of individual external exposures. For the calculation of inhalation doses, the new ICRP model for the respiratory tract was used. The dose assessments were conducted for measured size resolved activity distributions of resuspended material, obtained at different locations and for several kinds of agricultural operations. Inhalation doses vary considerably with respect to different kinds of work. Tractor drivers receive much higher doses than other agriculturual workers, especially when the cabin window of the tractor is open. Effective doses due to the inhaltion of resuspended plutonium are assessed to be a few μSv per initial deposit of one kBq/m^2. Inhalation doses from ^{137}Cs are usually smaller by an order of magnitude than the doses from Pu, provided a high solubility is assumed for resuspended Cs.

1. Introduction

Several years after the reactor accident of Chernobyl, radiation exposures of the population in contaminated areas still concern the living conditions. Currently and in the next decades, main exposure pathways are due to the deposited cesium. For most of the contaminated areas, the external irradiation contributes most to the population exposures [1,2]. An improved modelling of this exposure pathway is the main scope of the second part of the present paper.

As a general rule, the internal exposure due to incorporation of cesium by ingestion dominates the population exposure in regions with a high transfer of cesium to grass and other foodstuffs. The areas with the highest population doses due to the Chernobyl accident belong to these high transfer regions. Recent developments in improved modelling of ingestion doses are described in [3]. Another pathway of concern was in the recent years the potential hazard due to resuspension of radioactive material. A critical group consists here of agricultural workers on fields contaminated with plutonium. Potential inhalation doses are studied in the third part of the paper.

2. External exposures

The purpose of the present study is to evaluate data and develop a model for the external doses distributions of the population in contaminated areas. The developed model is assumed to be applicable from the year 1990 to the end of the lifetime of people being born before the reactor accident of Chernobyl.

At the time t after a deposition of a cesium isotope, the effective dose rate $\dot{H}_i(t)$ of a member of a population group i may be calculated by

$$\dot{H}_i = A \cdot \dot{g} \cdot \exp(-\lambda \cdot t) \cdot r(t) \cdot \sum_j f_j(t) \cdot p_{ij}(t) \cdot k_{ij}(t), \qquad (1)$$

where A is the activity deposited per unit area on a reference site, $\dot{g}$ the gamma dose rate in air per activity per unit area with a reference distribution of the cesium in the ground, λ the decay constant, and r(t) the gamma dose rate in air at the reference site divided by the gamma dose rate in air for the reference distribution. The summation index j indicates types of locations, f_j the gamma dose rate in air at a location j relative to the gamma dose in air at the reference site, $p_{ij}(t)$ the relative frequency of stay for members of population group i at location j and $k_{ij}(t)$ the conversion factor from the gamma dose in air to effective dose.

In this report, reference sites are considered to be open fields with undisturbed soil, normally lawns or meadows. A plane source below a soil slab with a mass per unit area of 0.5 g cm^{-2} has been chosen as a reference distribution to approximate the energy and angular distributions of the radiation field in air over an undisturbed field during the first years after the deposition. For this geometry, a value for $\dot{g}$ of 1.7 nGy h^{-1}per kBq·m^{-2} has been obtained for ^{137}Cs and of 4.7 nGy h^{-1}per kBq·m^{-2} for ^{134}Cs [4].

2.1 Gamma dose rates in air over open fields

A data base on the attenuation of the gamma dose rate in air due to the migration of cesium into the soil has been established. Main sources of information were [5], [6] and new results obtained in the framework of the projects JSP 5* and ECP 5**. The data base has about 450 data sets, each containing besides other information the time of measurement, the value of r(t), and the distance from the Chernobyl nuclear power reactor plant.

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The data set indicates the factor r(t) to decrease with increasing distance from the release point. Two factors may be responsible for this observation. First, cesium bound to fuel and condensation particles is known to be less available for migration than cesium that has been evaporated and then attached to the background aerosol. Second, in the data base for short distances, observation points with dry deposition dominate, whereas for long distances observation points with wet deposition dominate. Data grouped in three distance groups are given in Table 1.

Prediction of the future behaviour of r(t) may be based on observations of the fallout from nuclear weapons tests. An analytical approximation of the data for measurement sites at large distances from the release point and for the nuclear weapons tests in the form

$$r(t) = a_1 \cdot \exp(-\ln 2 \cdot t/T_1) + a_2 \cdot \exp(-\ln 2 \cdot t/T_2), \qquad (2)$$

yielded half lifes to be $T_1 = 1.5$ years and $T_2 = 50$ years. Based on the observation that fuel particles tend to dissolve over a period of several years in the environment, the same half lifes were assumed for the other distance categories and obtained values for a_1 and a_2 are indicated in Table 1.

The weapons fallout was deposited over long periods and column experiments have shown that under these conditions the initial penetration of the cesium into the soil is negligeable [7]. However, the sites in the far field from Chernobyl studied here had different meteorological conditions and an initial migration of the cesium during the wet deposition. It may be assumed that this initial difference retains and that the Chernobyl cesium will have migrated more deeply than corresponding profiles for the weapons fallout at the same time after deposition. Nevertheless both data sets are pooled here, since no better information is available for long times after deposition.

2.2 Location factors

In settlements of urban and rural type, the characteristics of radiation field differ from those over an open plot of virgin land due to shielding and to varying source distributions on the surfaces as a result of different deposition velocities, run-off and weathering. Location

Time after deposition (years)	Distance category		
	$D \leq 100$ km a_1=0.48; a_2=0.81	$100 \text{ km} < D \leq 1000$ km a_1=0.60; a_2=0.63	D>1000 km a_1=0.53; a_2=0.51
0.75	1.17 (1.15)	0.98 (1.05)	0.87 (0.88)
2.75	0.88 (0.93)	0.84 (0.78)	0.67 (0.64)
4.75	0.87 (0.82)	0.68 (0.66)	0.55 (0.53)
6.75	0.79 (0.77)	0.60 (0.60)	0.46 (0.48)
24	— (0.59)	— (0.45)	0.39 (0.36)
30	— (0.54)	— (0.42)	0.33 (0.33)

Table 1. Average values for r(t) of entries in data base for the first seven years after Chernobyl and derived from measurements of the atomic weapons tests fallout. The values in parentheses give the results of Eq. (2) with the paramters indicated in Fig. 1.

factors have been measured with thermoluminescence detectors in the higher contaminated areas of Ukraine ($A^{Cs137} > 185$ kBq · m^{-2}) in the period 1987-1989 [8]. In Russia, first measurements were performed in summer 1989 [9] in three large villages of the Bryansk region with Cs-137 activities per unit area above 1000 kBq/m^2 (Nikolayevka, Yalovka and Svyatsk). During this campaign, several thousand gamma dose rate measurements were performed in different points in the settlements and in their vicinity. The analysis included only data obtained in villages before decontamination actions were taken. Similar measurements were performed in the period 1992 to 1994 in ten villages in the contaminated area of Russia. The results obtained in the two countries were found to be consistent. Location factors measured after 1987 in rural environments were found to be independent of time. Values are given in Table 2.

The presence of a snow cover during the time period November-March reduces according to experimental measurements performed in the Brjansk region in the average the value of the dose rate in air over virgin plots by a factor of 0.76. Since the average reduction over streets may be assumed to be less, location factors during winter time might be a little higher than during summer time. This is confirmed by TLD-measurements of individual doses performed in the same settlements in winter and in summer time. According to these measurements, the mean reduction of the annual external dose to snow cover of 0.94 was derived. This small effect is not considered in the following.

2.3 Relative frequencies of stay

Information about relative frequencies of stay of the adult rural population at various locations were obtained in Ukraine from 8984 responses of evacuees from the 30 km zone to a questionnaire [8]. In Russia, in 1989, 1992, and 1993 corresponding responses of the rural population to questionnaires were obtained and results are summarized in Table 2.

Location	f_j	p_{ij} (anual)				
		Indoor workers	Outdoor workers	Pensioners	School children	Preschool children
Living areas						
house (wooden)	0.13					
house (brick)	0.07	0.50	0.48	0.68	0.60	0.52
multi storey house	0.02					
outside of houses	0.55	0.21	0.15	0.30	0.24	0.14
Work areas						
buildings	0.07	0.25	0.07	0.0	0.0	0.0
multi storey house	0.02	0.0	0.0	0.0	0.15	0.25
work yard	0.30	0.03	0.07	0.0	0.0	0.09
ploughed field	0.50	0.0	0.18	0.0	0.0	0.0
virgin land	1	0.0	0.04	0.0	0.0	0.0
Rest areas						
forest, meadow	1	0.01	0.02	0.02	0.01	0.0
$\sum p_{ij} \cdot f_j$ *	—	0.22/0.19/ 0.17	0.31/0.28/ 0.26	0.27/0.23/ 0.20	0.22/0.19/ 0.16	0.18/0.14/ 0.12

* The first number is for residents in wooden houses, the second in brick houses and the third in multistorey houses

Table 2. Location factors f_j and relative frequencies of stay p_{ij} for rural population groups.

2.4 Effective dose per gamma dose in air

The migration of the radionuclides into the soil changes the spectral-angular distribution of the photon fluence exposing the human body. In principle, the value of k_{ij} will change as well. To estimate experimentally conversion factors for a real vertical radionuclide distribution in soil, in summer 1991 and 1992 a series of phantom experiments was carried out. In the experiments three antropomorphical phantoms were used, the Alderson Rando phantom presenting adults and two phantoms of children of 5 and 1 year of age (produced by ATOM Ltd, Riga, Latvian Republic). As experimental sites, two open sites .. virgin land, one open ploughed site, and a location inside a wooden house in contaminated areas were chosen.

The experimental results for the three outdoor locations agreed within the limits of only 10% for each of the phantoms. Results of Monte Carlo calculations [10] for the reference distribution (plane source at 0.5 g/cm^2) of radionuclides were found to be intermediate between the experimental results for outdoor and indoor locations. In the present model a value of 0.9 $Sv \cdot Gy^{-1}$ for preschool children (0-7 years), of 0.8 $Sv \cdot Gy^{-1}$ for school children (8-17 years) and of 0.75 $Sv \cdot Gy^{-1}$ for adults is assumed for all locations.

2.5 Model results

Annual effective doses of rural indoor workers living in woodframe houses are shown in Fig. 1. For comparison results of the UNSCEAR model [11] are given approximating the attenuation due to the cesium migration into the soil by a constant factor. Dose estimates for Russian settlements in 1991 [1] agree within 10 % with the current model. In the present model for the period 1990-2056, annual effective doses of different population groups differ by a constant factor. The factors for rural population groups are given in Table 3. Urban population groups have lower external exposures due to the Chernobyl accident [12]. According to Table 3 rural outdoor workers living in woodframe houses are the critical group for external exposures. Among them, forestry workers and herdsmen receive the highest dose [12].

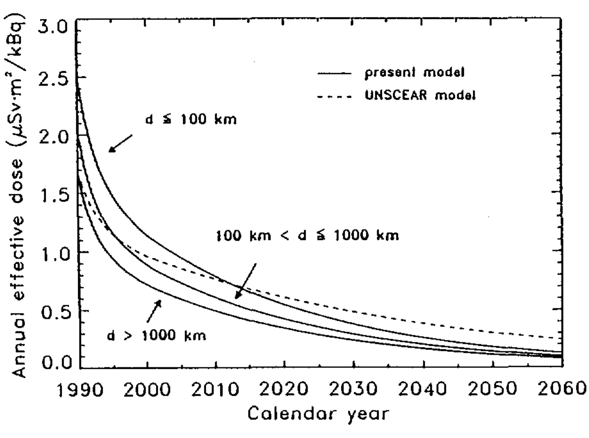

Fig. 1. Annual effective doses due to external exposures of rural indoor workers living in woodframe houses. Results are normalized to the ^{137}Cs activity initially deposited per unit area, a ratio of the ^{137}Cs activity to the ^{134}Cs activity of 1.8 has been assumed. The dotted line represents the UNSCEAR model [11].

Population group	Wooden houses	Brick houses	Multi storey houses
Indoor	1.00	0.86	0.77
Outdoor	1.36	1.23	1.09
Pensioners	1.23	1.05	0.91
Schoolchildren	1.07	0.92	0.78
Preschoolchildren	0.98	0.76	0.65

Table 3. Annual effective doses due to external exposures of rural population groups, normalized to the annual effective dose due to external exposures of rural indoor workers living in woodframe houses.

2.6 Comparison with measurements of individual external doses

Monthly external doses were measured with TLDs during the spring-summer periods of 1990-1993 in 21 settlements of the Bryansk region with average ^{137}Cs activities above 0.4 MBq/m^2 [13]. The conversion factor from readings of an individual dosimeter to the value of the effective dose was determined on the basis of the results of phantom experiments, and it was 0.9 Sv/Gy for adults, 0.95 Sv/Gy for schoolchildren and 1 Sv/Gy for preschoolchildren. Table 4 shows a good agreement between the average doses in the population groups obtained by the model and by measurements. A comparison with measured individual doses showed some of those to be larger than the largest of the average doses in the population groups. Doses to individuals are not the scope of the present model.

3. Inhalation doses

Resuspended material may be inhaled and deposited in the human respiratory system. Radionuclides attached to the inhaled particles lead to radiation doses to the lung and other tissues. Besides the radiation characteristics of a particular radionuclide, the main physico-chemical factors that affect inhalation doses are the size distribution of the inhaled particles and the solubility of the material. Consequently these factors require particular attention.

	Monthly effective dose (μSv)					
Year	Wood houses			Brick houses		
	Model	Measurement	Ratio	Model	Measurement	Ratio
1990	290	285(174)	1.02	271	230(100)	1.18
1991	246	235(414)	1.05	230	205(103)	1.12
1992	216	200 (95)	1.08	202	200 (49)	1.01
1993	195	150(195)	1.30	182	115 (84)	1.58
Average			1.11			1.22

Table 4. Monthly effective doses (μSv) of outdoor workers in the Bryansk region in the periods (April-November) of the years 1990 to 1993. Results of the present model and of TLD measurements [13] are given, the figures in parentheses give the number of measurements.

3.1 Method of inhalation dosimetry

The new ICRP lung model was used for the assessment of inhalation doses due to resuspended radioactivity from the Chernobyl accident. The dose calculations were performed applying the computer program LUDEP 1.0, developed at NRPB [14], which implements the model structure and parameter values approved by ICRP and described in ICRP Publication 66 [15].

Standard values for an adult male worker were assumed for the physiological and activity related parameters. This includes the assumption of a breathing rate of 1.2 m^3/h. Particle density was assumed to be 3.0 g/cm^3 The rate of particle dissolution and subsequent uptake to blood is described by the absorption parameters: type 'F' (fast), type 'M' (moderate) and type 'S' (slow), which are similar to the former lung retention classes D, W and Y, as specified in ICRP Publication 30 [16]. Radionuclide transformation data were taken from ICRP Publication 38 [17], and the biokinetic models were selected as given in ICRP Publication 30 [16]. Doses were calculated for various organs of the body and for the different regions of the respiratory tract, as specified in the new lung model. This includes explicitely the extrathoracic region of the respiratory tract, which was not considered in previous lung models.

3.2 Inhalation doses for agricultural activities

Dose calculations for inhalation of radionuclides were performed for actually measured size resolved aerosol concentrations of radionuclides and surface soil contaminations of the sampling campaigns of ECP1 in 1993 and 1994 [18, 19]. Due to the time-consuming analytical procedures, particularly for α-emitters, the particle size distribution was measured only for several experiments and also the complete radionuclide spectrum was measured only at a few places. As an example, Table 5 shows the airborne radionuclide composition and activity concentrations at one of the experimental sites.

Size resolved measurements of plutonium and ^{137}Cs inside a tractor cabin were made during several kinds of agricultural operation at Novozybkov and Bragin. Doses for the different radionuclides were calculated for each size fraction separately and summed up. The resulting lung doses and effective dose values for Novozybkov (surface soil contamination 1.6 x 10^2 Bq/m^2 for $^{239+240}Pu$ and 1.1 x 10^6 Bq/m^2 for ^{137}Cs) are shown in Table 6a, and for Bragin (surface soil contamination 6.4 x 10^2 Bq/m^2 for $^{239+240}Pu$ and 5.3 x 10^5 Bq/m^2 for ^{137}Cs) in Table 6b. It is obvious from this table that the doses may vary considerably with respect to different kinds of work, with highest values during cultivation at Novozybkov (with 18 μSv effective dose from $^{238+239+240}Pu$ and 5 μSv from ^{137}Cs). However, there is a remarkable increase in ^{137}Cs doses by an order of magnitude, when the

Site	Agricultural operation	Reference	Activity concentration (mBq/m^3) ^{137}Cs	^{90}Sr	^{238}Pu	^{239}Pu	^{240}Pu	^{241}Am
Zapolye	harrowing	ECP1 [18]	700	460	2.5	2.2	2.8	4.0

Table 5. Airborne activity concentrations at Zapolye during harrowing on 13 May 1993.

a) Novozybkov 1994

Kind of agricultural operation	Lung Dose (µSv) $^{238+239+240}Pu$	Lung Dose (µSv) ^{137}Cs	Effective Dose (µSv) $^{238+239+240}Pu$	Effective Dose (µSv) ^{137}Cs
ploughing	38	2.8	8.1	3.1
cultivation	74	4.4	18	4.9
fertilization	-	2.1	-	2.3
potato planting	77	1.8	16	1.9
potato planting (open cabin window)	-	23	-	26

b) Bragin, 1994

Kind of agricultural operation	Lung Dose (µSv) $^{238+239+240}Pu$	Lung Dose (µSv) ^{137}Cs	Effective Dose (µSv) $^{238+239+240}Pu$	Effective Dose (µSv) ^{137}Cs
ploughing	36	0.6	11	0.6
cultivation	5.1	1.4	1.3	1.5
rye harvesting	10	0.2	3.0	0.2
straw harvesting	6.8	0.1	1.5	0.1

Table 6. Inhalation life time doses per 832 h work (one working year) of tractor drivers for various agricultural activities at Novozybkov (a) and Bragin (b). Airborne activity concentrations and surface soil contaminations from ECP1 campaigns [18].

cabin window of the tractor is open (Table 6a). Unfortunately there are no data available for the plutonium activities under this condition. It is therefore difficult to assess the relevance of the plutonium doses for this situation.

Similar dose calculations for other sites and activities show that the internal doses for tractor drivers are between one and two orders of magnitude higher than for other agricultural workers.

The dose calculations of the above examples were performed using the actually measured size resolved aerosol concentrations. For the solubility of the material, the standard solubility classes (type 'F' for ^{137}Cs, and type 'S' for all plutonium isotopes) were used due to the lack of more precise information. Some experimental work on the solubility of inhaled dust particles was conducted at the UIAR in Kiev. In these investigations, simulated lung fluid (SLF) was used to evaluate the solubility of Cs, Sr, and Pu. In these measurements, all these nuclides exhibited a very low solubility. Although these *in vitro* experiments may be of limited significance for the real situation in the human lung, they strongly emphasize the need for better information of the physico-chemical characteristics of the resuspended material. If resuspended ^{137}Cs would be considered as a 'S'-type material, then the dose coefficients would change significantly. The longer retention of the material in the lung would increase the lung dose by a factor of 60 as compared to 'F'-type material, and also the effective dose would increase by a factor of 8. In this case, the dose contribution from ^{137}Cs would be quite significant.

As stated before, for an estimation of the total inhalation dose from resuspended material from the Chernobyl fallout, the full spectrum of the relevant radionuclides as shown in Table 5 must be taken into account. Table 5 shows that the activity concentrations in air are not very different for the three plutonium isotopes ^{238}Pu, ^{239}Pu and ^{240}Pu for this measuring site. If this finding can be generalized, then the radiation doses from the inhalation of these nuclides can be estimated, even where not all of these isotopes are determined separately, since also the inhalation dose coefficients are very similar for these isotopes. The concentration of ^{241}Pu was not measured here, but since the dose coefficients for this plutonium isotope are by several orders of magnitude lower than for the previous mentioned ones, the resulting doses seem to be not significant. The dose coefficients for ^{241}Am are again similar to those of the plutonium isotopes and dose contributions from this nuclide can be roughly estimated from the above activity ratios in the cases where no experimental data are available. It should be kept in mind, however, that americium is generally considered to be of considerably higher solubility than plutonium. As for ^{137}Cs, also for ^{90}Sr significant dose contributions must be expected if the solubility of the resuspended material is low.

With regard to the these uncertainties, it is obvious that it is difficult to derive an estimate for the total internal dose from the inhalation of resuspended radioactive material. With sufficient care it can be concluded, however, that even at sites inside the 30 km zone, life time doses per year of work generally will hardly exceed 200 μSv for lung dose and 35 μSv effective dose for agricultural workers.

4. Conclusions

A deterministic model has been developed to calculate annual external exposures to population groups in areas of Russia, Ukraine and Belarus that have been contaminated after the Chernobyl accident. Rural outdoor workers living in woodframe houses are the group with the highest average dose. In the period 1990 to 1993 annual external doses were obtained to be a few μSv per initial ^{137}Cs deposit of one kBq per m^2.

Effective doses due to the inhalation of resuspended plutonium were assessed to be a few μSv per initial ^{238}Pu + $^{239/240}$Pu deposit of one kBq per m^2, if 800 hours of agricultural work under dusty conditions are assumed. Therefore, effective doses due to the inhalation of resuspended material are under these conditions of the same order of magnitude as external exposures, if the ^{238}Pu + $^{239/240}$Pu activity initially deposited is of the same order of magnitude as the ^{137}Cs activity. For lung doses this would be the case if the ^{238}Pu + $^{239/240}$Pu activity was one fifth of the ^{137}Cs activity. The probability to find areas contaminated by the Chernobyl accident with such high relative plutonium activities is low even within the 30 km zone.

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