ASNR Autorité de sûreté nucléaire et de radioprotection

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STUDY REPORT

UPDATE TO THE ASSESSMENT OF RADIOLOGICAL CONSEQUENCES IN FRANCE OF THE CHERNOBYL ACCIDENT

Environmental contamination and exposure of the population

ABSTRACT

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At the beginning of May 1986, air masses contaminated by the Chernobyl accident passed over France. Some of the radionuclides present in these air masses were deposited on the surface of the soil, and then spread to all parts of the environment, resulting in radiological exposure of the population that continues to this day due to the persistence of caesium-137. This exposure is a component of the radiological exposome, which is the study topic of the CORALE project (Radiological component of the exposome, multiple exposures, risks of cancer and other chronic diseases in the Constances cohort; Sauce J. et al. 2024), conducted by ASNR in collaboration with UMS011 Inserm/Université Paris Cité/UVSQ/Université Paris-Saclay which manages the Constances cohort.

Exposure of the population to fallout from the Chernobyl accident was assessed for the first time in 1997 (Renaud et al., 1999), then reassessed in 2007 (Renaud et al. 2009). However, these assessments do not fully meet the needs of the CORALE project, which requires to reconstruct the annual equivalent doses to various organs, for all age groups and by municipality, from 1986 to the present day. The purpose of this study is to supplement the assessments performed in 2007 in this respect. It also provides an opportunity to consolidate all the assessments of radiological consequences of the Chernobyl accident in France, by using the observations made in Japan after the Fukushima accident and recent studies conducted by ASNR. In addition, these doses can be put into perspective with those resulting from fallout due to atmospheric testing of nuclear weapons in mainland France, recently estimated by ASNR (Renaud and Vray, 2024). This document provides a detailed account of all the methodological elements used to obtain these new estimates and discuss them

These dose calculations are essentially based on the activity concentrations of the main radionuclides of the fallout, measured in the air, soil, and foodstuffs by ASNR and its predecessors (the SCPRI[1], OPRI[2], IPSN[3] and then IRSN), with the use of modelling (via the Symbiose model or empirical models) to supplement the time series of measurement results, or interpolation to supplement the time series of doses. Estimates of doses received by inhalation are based on caesium-137 activity concentrations measured daily in May and June 1986 in atmospheric aerosols sampled at 36 stations across the country, and on isotopic activity ratios for around fifteen other radionuclides also measured in the air at some of these stations. Similarly, the doses received by ingestion were calculated on the basis of the activity concentration of the five main radionuclides that contributed to them, measured in foodstuffs produced from 1986 to 1989, supplemented by modelling results adjusted to these measurements, and then on the basis of activity concentrations measured from 2008 to 2018. Between these two periods, doses were estimated by interpolation. Finally, the doses associated with external exposure to radiation from the 15 radionuclides measured in the air and deposited on soil and surfaces were estimated for 1986 and then for 2008-2018, based on surface activity measurements or direct radiation measurements (equivalent dose rates). The kinetics of the reduction in equivalent dose rates over the first 8 years following radioactive deposits in rural areas, and the first 15 years in urban areas, were adjusted to those observed in Japan after the Fukushima accident.

In 1986, the effective doses due to fallout from the Chernobyl accident were estimated at between 10 µSv in the Bretagne region and a few hundred microsieverts in the areas of eastern France where radioactive deposits were highest. They were as high as 1,000 µSv (1 mSv) in the eight municipalities located on the east coast of Corsica and in the inland region of Nice, where caesium-137 deposits were around 50,000 Bq/m^2 following very heavy rainfall between May 1 and 5, 1986. That year, for most of the country, these doses resulted mainly from the ingestion of contaminated foodstuffs. However, in municipalities where radioactive deposits exceeded 20,000 Bq/m^2, external exposure was the main contributor to the total dose, particularly for adults who spent a lot of time outdoors. In all cases, the contribution from inhalation exposure was low and did not exceed 15% of the total dose in north-east France, where the highest airborne activities were measured.

[1] Central service for the protection against ionizing radiation
[2] Ionizing radiation protection office
[3] Nuclear protection and safety institute

In 1987, effective doses were 2 to 3 times lower due to the nearly disappearance of inhalation exposure and, above all, the significant reduction in the dose due to consumption of foodstuffs. Since 1988, annual effective doses have fallen steadily, with varying contributions from external exposure and ingestion depending on the place of residence and age of the person exposed. In 2000, annual effective doses were estimated between a few microsieverts in Bretagne and a few tens of microsieverts in the most affected areas in the eastern part of the country.

In 2020, the average effective dose due to fallout from the Chernobyl accident for an adult living in an urban environment[4], working indoors and not eating wild mushrooms or game meat, was around 1 µSv/year (from less than 1 microsieverts per year up to several microsieverts per year). This scenario undoubtedly represents the majority of the population. Generally speaking, the estimated effective doses for children are at the same level. For people working outdoors in rural areas most affected by fallout from the Chernobyl accident, doses could reach up to 10 µSv, or even 20 µSv in municipalities in the eastern part of the country where radioactive deposits in May 1986 were highest. However, these higher values can only be achieved if the person had spent several hours a day on undisturbed surfaces (which have never been ploughed or covered over, etc.) since 1986. Such areas are now often limited to natural or wooded areas.

Unlike foodstuffs produced through agriculture or livestock farming, with activity concentrations, and therefore with consumption-related doses that have been steadily decreasing since 1986, caesium-137 contamination in forest foods, mushrooms, and game meat has remained at a high level to this day. This contamination is also much more variable than for other foodstuffs, even on a municipality scale. As a result, even occasional consumption of these foods can lead to widely different and significant doses. For people who eat them regularly, the associated effective dose may have exceeded those due to other exposure pathways as early as the beginning of the 1990s. In the worst-affected municipalities of eastern France, the level may still be several dozen microsieverts.

In 1986, the equivalent doses to the thyroid were much higher than the effective doses and they were age-dependent. The highest estimates, around 7 mSv, were for children aged 2 to 7. They resulted almost exclusively from uptake of iodine-131 via the ingestion of foodstuffs, with contributions from inhalation or other radionuclides being very low. Equivalent doses to other organs are very close and often at the same level as effective doses, with the exception of equivalent doses to the colon which, for children aged 1 to 12, can be up to twice as high as effective doses.

[4] People are considered to live in an urban environment if their surroundings are mainly made up of artificial surfaces (roads, buildings, etc.); in addition to cities, they may live in villages or hamlets. People are considered to live in a rural environment if they have spent several hours a day on land that has been ploughed or on undisturbed land (neither ploughed nor covered) since 1986.

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1. Introduction

At the beginning of May 1986, air masses contaminated by the Chernobyl accident passed over France. Some of the radionuclides present in these air masses were deposited on the surface of the soil and then spread to all parts of the environment, resulting in radiological exposure of the population that continues to this day due to the persistence of caesium-137. This radiological exposure is a component of the exposome, which is defined as all environmental exposures from conception to death, and which are likely to have an effect on our health (Wild CP, 2005). The exposome includes all exposures to ionising radiation from all sources, which may then called the “radiological component of the exposome” (Sauce J. *et al*. 2024).

The "Constances" cohort is a group of 220,000 people residing in France who agreed to participate in an epidemiological study to investigate the potential causes of chronic diseases, including those of environmental origin (Goldberg M. et al, 2017). This cohort is managed by the UMS011 unit-Inserm/Université Paris Cité/UVSQ/Université Paris-Saclay. The presence of radioactive substances in our environment, as well as medical and professional exposure to ionising radiation, are among the potential causes of diseases to be studied. This is the subject of the CORALE project (Radiological component of the exposome, multiple exposures, risks of cancer and other chronic pathologies in the Constances cohort; Sauce J. *et al*. 2024) conducted by ASNR (previously IRSN) in collaboration with UMS011. The aim of CORALE is to reconstruct the radiation doses of environmental, medical, and occupational origin received by 76,000 participants of the Constances cohort since their birth, and then to estimate the risks of cancer and other chronic diseases potentially related to these doses, while taking into account the influence of other risk factors.

In order to reconstruct the individual dosimetric history, since birth, of each of the 76,000 people of the Constances cohort for whom residential histories are available, the doses must be estimated by year, age group, and municipality. This will provide an estimate of the doses received by each of these people at the age they had in their successive places of residence. Also, in addition to the effective doses, which account for the overall exposure of people, the assessment of associations with the incidence of specific diseases (e.g.: various cancer locations) requires access to equivalent doses to organs potentially affected by the diseases. To cover the most common diseases (notably, cancers), the first organs selected for dose assessment have been lungs, colon, prostate, breasts, thyroid, and brain.

Exposure of the population to fallout from the Chernobyl accident was assessed for the first time in 1997 (Renaud et al., 1999), then reassessed in 2007 (Renaud et al. 2009). The 2007 reassessment mainly concerned the mapping of radioactive deposits throughout France, based on specifically dedicated studies (Renaud et al, 2003; Renaud et al, 2004) and the doses due to external exposure to these deposits. The doses received from inhalation and ingestion of foodstuffs, estimated in 1997 and based mainly on the results of measurements in the air and in foodstuffs, were used in the 2007 assessment.

However, the 1997 and 2007 assessments do not fully meet the needs of the CORALE project. In these two studies, dose estimates were limited to effective doses for adults and thyroid doses for children of specific ages. While a range of spatial variability in doses was proposed in the 2007 study, the CORALE project requires a more precise and complete spatialization of doses throughout the country. Also, previous dosimetric estimates did not cover periods beyond 2006. However, a recent IRSN study (IRSN, 2022), based on numerous measurement results acquired between 2008 and 2018, provides an indication of the doses currently resulting from fallout due to the Chernobyl accident. Finally, the present study provides an opportunity to take a fresh look at all the assessments of the radiological consequences of the Chernobyl accident, particularly in the light of observations made in Japan after the Fukushima accident.

The present report document sets out to provide a detailed description of all the methodological elements used to complete and improve the dose assessments performed in 2007. The time series of dose measurements are then discussed in terms of their development over time, the main radionuclides that contribute to them, and their relative significance depending on organs affected and the age of individuals.

2. Reconstruction of doses received by inhalation

2.1. Available data and estimates from 2007

Around 15 radionuclides were measured in the air in France following the Chernobyl accident. These measurements were taken on atmospheric aerosol filters recovered from 36 CEA[5] and SCPRI sampling stations spread across the country (see Figure 1). However, the number of available measurement results varies greatly depending on the station, the day of sampling, and the radionuclide considered.

At most stations, only 137-Cs was measured, more or less regularly, throughout the month of May 1986. Figure 1 shows the 248 activity concentrations of this radionuclide measured throughout May 1986 at these 36 stations, which are located on the map. It shows that the contaminated air masses reached the eastern part of the country on April 30, 1986 and that the highest activities were most often measured on May 1st throughout the country, with a maximum value of 5.5 Bq/m^3 measured at the Verdun station. Activity concentrations then fell rapidly to less than 0.005 Bq/m^3 (i.e. around 1,000 times lower) starting May 7th, with the exception of the extreme south-east (Nice) where higher activities persisted until May 8th.

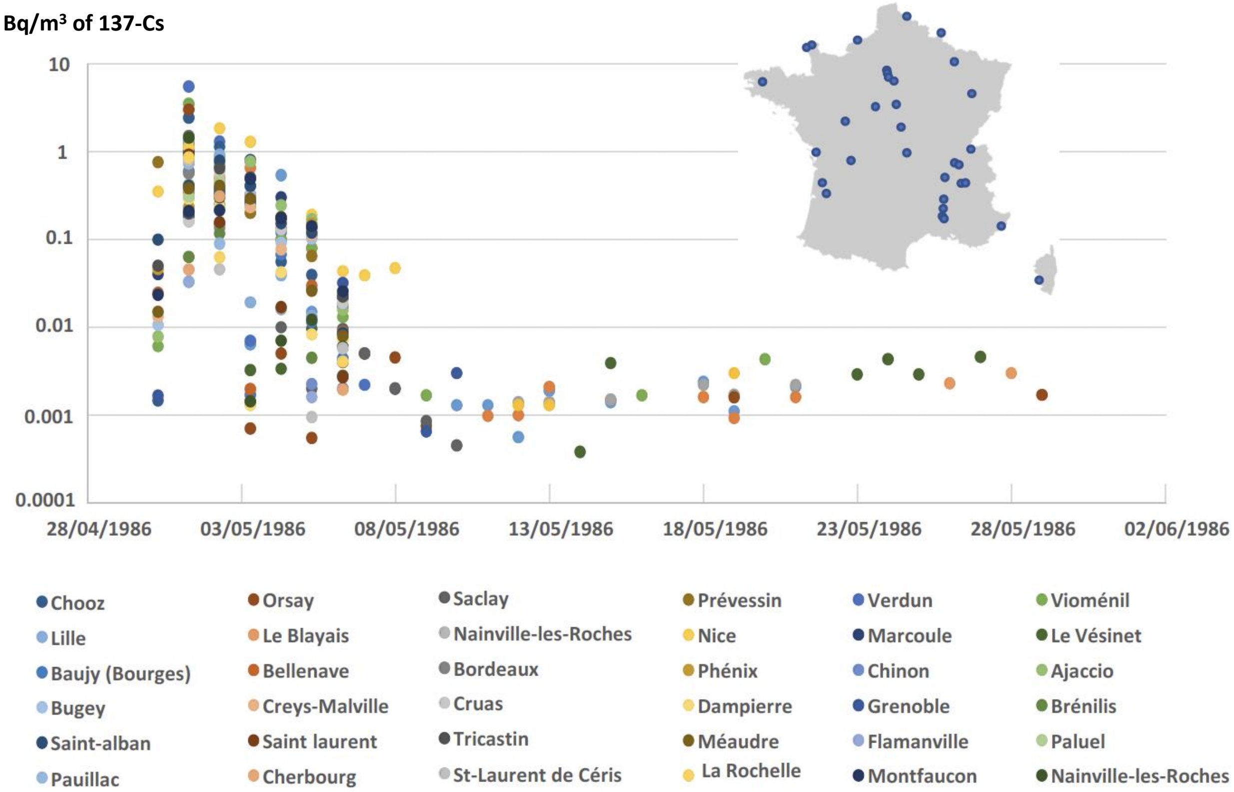


Figure 1: Activity concentrations of 137-Cs in the air measured on the daily filters of 36 aerosol sampling stations throughout France (Bq/m^3).

Because of this rapid decrease, only results of filters from a single day of sampling are usable; results from filters used for 3, 7, or even 10 days of sampling, which only provide averages of activities in the air over these longer periods, were therefore discarded.

Under these conditions, radionuclides other than 137-Cs were measured at only a dozen stations, mainly on May 1st and 2nd, 1986. As an illustration, Figure 2 shows the results obtained at the 2 stations where

[5] (French) Atomic energy commission

the most radionuclides were measured. Generally speaking, the number of available measurement results varies greatly depending on the radionuclide (see the last line of Table II). The radionuclides for which the largest number of measurement results were available over these 2 days, for all stations combined, were 106-Ru (36 results), 95-Zr (33 results), 103-Ru (18 results) and 134-Cs, 131-I, and 132-Te (17 results each). For some radionuclides, such as 136-Cs, 110m-Ag, 125-Sb, 99-Mo, or 141-Ce, fewer than 5 measurement results are available.

In the 2007 study, in order to provide an idea of the range of variability across the country, the doses received by inhalation were calculated for people who lived in Verdun (Meuse department), where activities measured in the air were the highest in France, and in Cléville (Calvados department), where they were among the lowest. The radionuclides used to estimate these doses were the main radionuclides measured in the air in France: 132-Te, 103-Ru, 106-Ru, 131-I, 134-Cs, and 137-Cs. For Verdun, the activities used to estimate doses were the activities actually measured. As only caesium-137 was measured at Cléville, the activities of the other radionuclides were deduced from the average isotopic activity ratios between each radionuclide and caesium-137, obtained for all the French sampling stations. Only effective doses for adults and thyroid equivalent doses for children were estimated. The inhalation dose factors used were those proposed by the International Commission on Radiological Protection (ICRP, 1995).

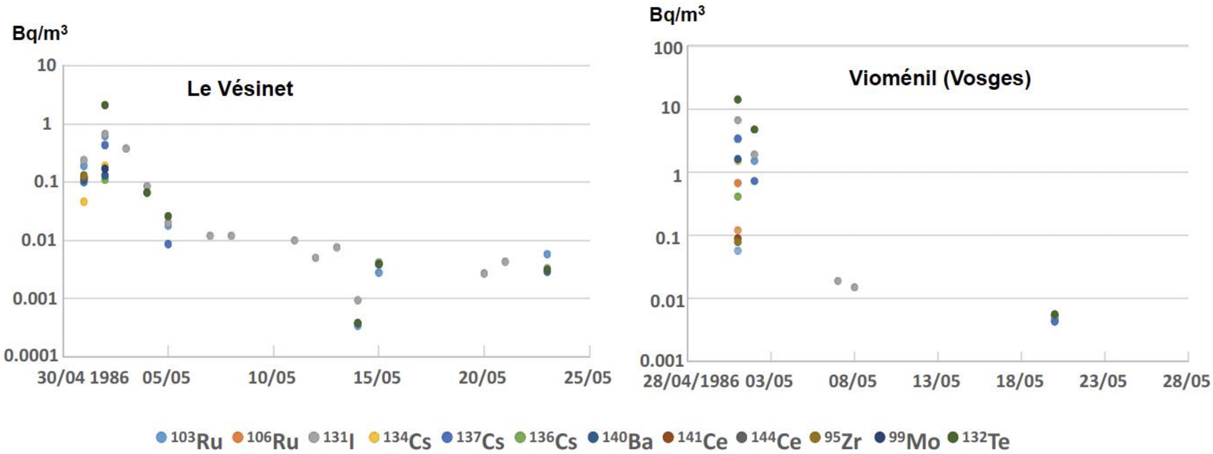


Figure 2: Activity concentrations of 12 radionuclides measured on May 1 and 2, 1986 on atmospheric aerosol filters recovered from the Le Vésinet and Vioménil monitoring stations (Bq/m^3)

The doses received by external irradiation and inhalation following the passage of contaminated air masses over France and estimated in 2007 are shown in Table I. Calculations were based on the assumption that the activities in the air inside buildings were equal to those measured outside, which is a slight over-estimation. These calculations indicate that the effective dose due to the radioactive plume did not reach 50 µSv for an adult. In the far western part of the country, it was around a few microsieverts. The highest thyroid doses were reached in children aged 1 to 7: between 400 and 500 µSv. External doses due to immersion in the radioactive plume were negligible compared with the doses received by inhalation: around 1/200th.

Table I: External and inhalation doses due to the passage of contaminated air masses in May 1986

		Effective dose µSv	Thyroid equivalent dose µSv			
		Adult	Less than 1 year	1-2 years	3-7 years	8-12 years
Inhalation	Verdun	46	260	470	410	360
	Cléville	4.1	23	41	36	32
External irradiation	Verdun	0.20	-	-	-	-
	Cléville	0.02	-	-	-	-

2.2. 2025 update

2.2.1. Objectives and methodology

The aim of this study is to estimate the committed doses received by inhalation for all age groups, for the 6 organs selected (thyroid, lungs, breasts, prostate, colon and brain), and by municipality. The doses received from external irradiation due to immersion in the radioactive plume are considered negligible (see Table I) and are therefore not estimated. The list of radionuclides taken into account, initially limited to the 6 main radionuclides in the 2007 study, was extended to include the 15 radionuclides (RN) measured in the air.

Figure 3 provides a schematic representation of the method used to assess inhalation doses based on available measurement results. The first step was to obtain a daily map of the activity concentrations of 137-Cs in the air **(1)** using the radionuclide measurement results shown in Figure 1. The second step was to deduce the daily changes in the airborne activities of the 14 other radionuclides **(2)** based on a study of the ratios between the activity concentrations of each of them and those of 137-Cs, and taking into account their radioactive decay. The effective daily doses, for all age groups, resulting from the inhalation of all the radionuclides, are then calculated for a unit activity concentration of 137-Cs **(3)** using the exposure parameters; respiratory flow rates and dose per unit intake coefficients (DPUI). Using **(1)** and **(3)**, the effective doses and equivalent doses to organs are estimated by age group and by municipality **(4).**

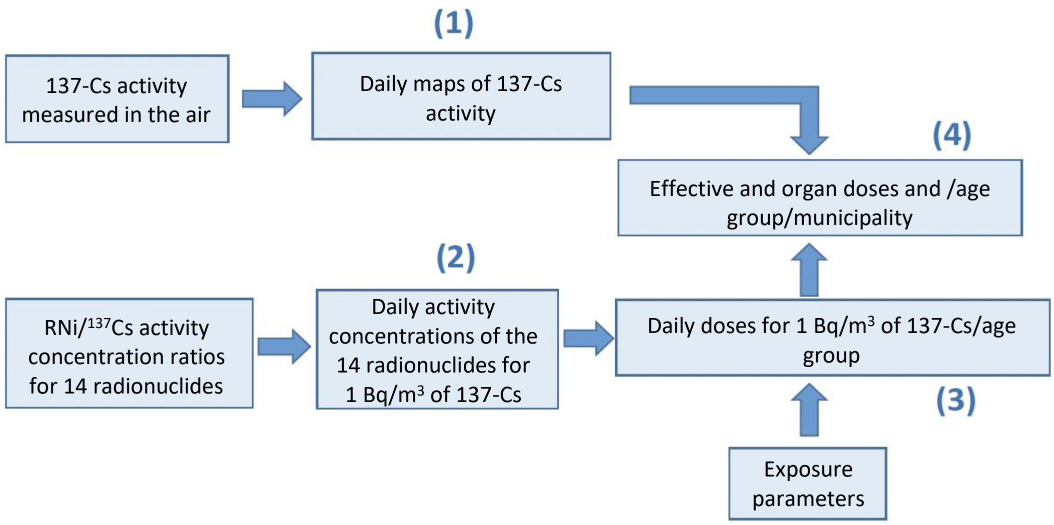


Figure 3: Schematic representation of the inhalation dose assessment method

2.2.1.1. Map of 137-Cs activity in the air

Figure 4 shows the daily maps of 137-Cs activity concentrations in air from April 30 to May 6, 1986. This is a spatial representation of the same measurement results as those presented in the form of a time series in Figure 1. Two interpolation methods were applied: a probabilistic method (ordinary kriging) within the equivalent convex polygon (interpolation), and a deterministic method (nearest neighbour) beyond this same polygon (extrapolation).

These maps confirm that the contaminated air masses entered from the east starting April 30th, within a high-pressure flow. Maximum activity concentration was reached on May 1st in the north-east of the country. Starting May 3rd, the westerly flow, characteristic of the most common atmospheric circulation over France, resumed, causing these contaminated air masses to flow back eastward. On May 5th and 6th, 137-Cs activity concentrations were lower and more uniform across the country. After May 6th, they became stable over time (see Figure 1). From May 7th to 31st, they ranged from 6×10^{-4} to 3×10^{-3} Bq/m^3, with an average of 2×10^{-3} Bq/m^3 retained for this study; in June, they ranged from 10^{-4} to 5×10^{-4} Bq/m^3, with an average of 2×10^{-4} Bq/m^3. After June, the activity concentrations were generally too low to be measured; the rare activities measured were lower than 10^{-4} Bq/m^3. Figure 4 shows the daily maps of airborne 137-Cs activity concentrations from April 30th to May 6th.

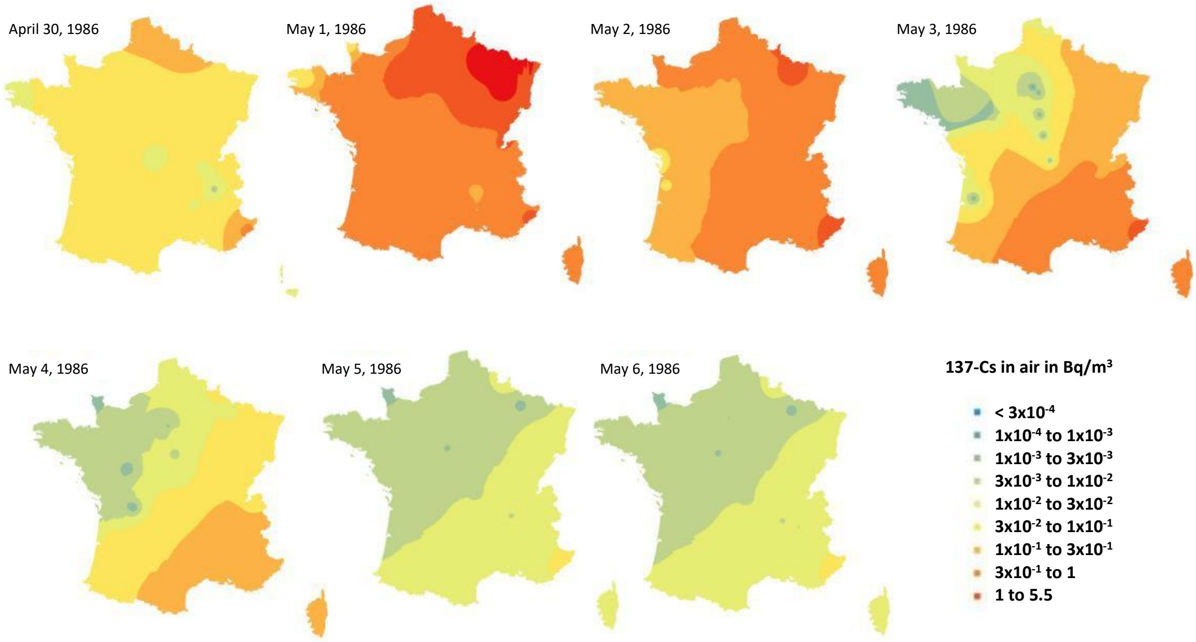


Figure 4: Maps of daily airborne 137-Cs activity from April 30th to May 6th (Bq/m^3)

2.2.1.2. Ratios of activity concentrations of 137-Cs to activity concentrations of other radionuclides

Figure 5 and Table II show the ratios of activity concentrations of various radionuclides measured in the air on May 1st and 2nd, 1986, to those of 137-Cs. Because of their longer half-life and the fact that measurements were available throughout the month of May, the average activity ratios 106-Ru/137-Cs and 144-Ce/137-Cs were obtained from all the data for May 1986.

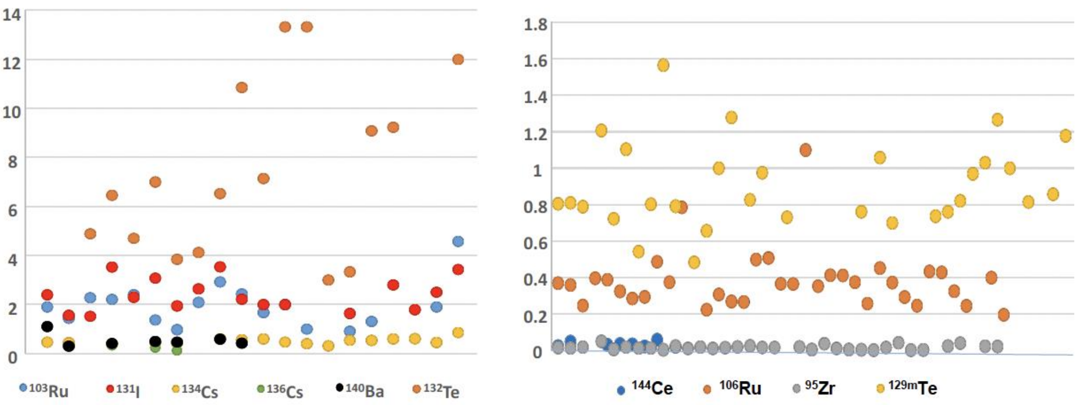


Figure 5: Activity concentrations in air of 10 radionuclides compared with those of 137-Cs

The isotopic activity ratios illustrate the relative abundance of radionuclides in the air at the beginning of May 1986. The four most abundant radionuclides in the air were 132-Te (activity concentrations almost 8 times those of 137-Cs), 131-I (2.4 times those of 137-Cs in aerosol form alone), 99-Mo and 103-Ru (around twice those of 137-Cs). The activity concentrations of all the other radionuclides were lower than those of 137-Cs: from 2 times lower for 134-Cs and 140-Ba, to more than 10 times lower for 95-Zr, 110m-Ag and 125-Sb.

The variability of activity ratios is generally moderate; most coefficients of variation are between 0.24 and 0.5. However, the coefficient of variation is 1 for 99-Mo and 1.6 for 141-Ce, and the number of ratios calculated (linked to the number of measurements results available) is low (5 and 3, respectively). The airborne activities of these two radionuclides are therefore not very well known.

Following the Chernobyl and Fukushima accidents, it was found that iodine, initially emitted in gaseous form, gradually fixed onto particles resuspended in air. The 131-I/137-Cs activity ratio of 2.4 mentioned in the table only reflects this particulate form measured on aerosol filters. For this study, it was assumed that three quarters (75%) of the iodine present in the air in May 1986 was in gaseous form (Maqua M. et al, 1987; Köhler H. et al, 1991); the 131-I/137-Cs activity ratio of 2.4 observed for particulate iodine was therefore multiplied by 4 to obtain the value of this ratio for total iodine (gaseous + particulate): i.e. a 131-I/137-Cs activity ratio of 9.6.

Table II: Activity concentrations (Bq/m^3) in air of 13 radionuclides compared with 137-Cs in May 1986

	103-Ru	106-Ru	^{131}I*	134-Cs	136-Cs	140-Ba	141-Ce	144-Ce	95-Zr	99-Mo	132-Te	110m-Ag	125-Sb
Average	1.90	0.38	2.40	0.51	0.22	0.54	0.43	0.04	0.02	2.10	7.70	0.02	0.04
Min.	1.00	0.20	1.50	0.32	0.12	0.30	0.03	0.02	0.003	0.40	3.00	0.02	0.04
Max.	4.60	1.10	3.50	0.85	0.32	1.10	1.20	0.06	0.05	5.00	13.00	0.02	0.04
Standard deviation	0.93	0.17	0.68	0.12	0.10	0.26	0.67	0.014	0.01	2.06	3.64	0.002	0.01
Coeff. of Variation	0.50	0.43	0.28	0.24	0.45	0.48	1.55	0.40	0.41	0.99	0.47	0.11	0.14
Nbr. of ratios	18	36	17	17	3	7	3	8	33	5	17	3	2

*iodine fixed to aerosols only

2.2.1.3. Daily committed doses for a unit activity concentration of 137-Cs

The application of respiratory flow rates (m^3/h) for different occupations (sleep, light or intense physical activities) and specific to each age group, provides estimates of the activity of radionuclide intake by inhalation per month (Bq/month). Table III provides the parameter values used in this study; they are derived from the time budgets and respiratory flow rates by type of occupation proposed in ICRP publication 66 (ICRP, 1994). The average daily respiratory flows used in this study are shown in the right-hand column.

Table III: Parameter values for inhalation exposure (men/women averages)

	Sleep		Rest		Light exercise		Intense exercise		Daily average
	m³/h	h	m³/h	h	m³/h	h	m³/h	h	m³/d
Infant	0.1	17	-	-	0.19	7.0	-	-	2.9
1-2 years	0.2	14.0	0.2	3.3	0.4	6.7	-	-	5.2
3-7 years	0.2	12.0	0.3	4.0	0.6	8.0	2.0	-	8.7
8-12 years	0.3	10.0	0.4	4.7	1.1	9.3	2.7	-	15.3
13-17 years	0.4	10.0	0.5	5.5	1.3	7.5	2.8	1.0	20.0
Outside worker	0.4	8.5	0.5	3.2	1.4	6.3	2.9	6.0	30.6
Sedentary worker	0.4	8.5	0.5	5.2	1.4	10.3	-	-	19.9

Use of dose per unit intake coefficients (DPUI) is then used to obtain estimates of the doses received by inhalation (effective doses and equivalent doses to the 6 selected organs). The DPUIs used in this study to calculate the effective and committed equivalent doses to the thyroid, brain, colon, lungs, breasts, and prostate, for the different age groups, are taken from the order of September 1st, 2003 published in the Official Journal of the French Republic (JO 2003) and publication 72 of the ICRP (ICRP, 1995).

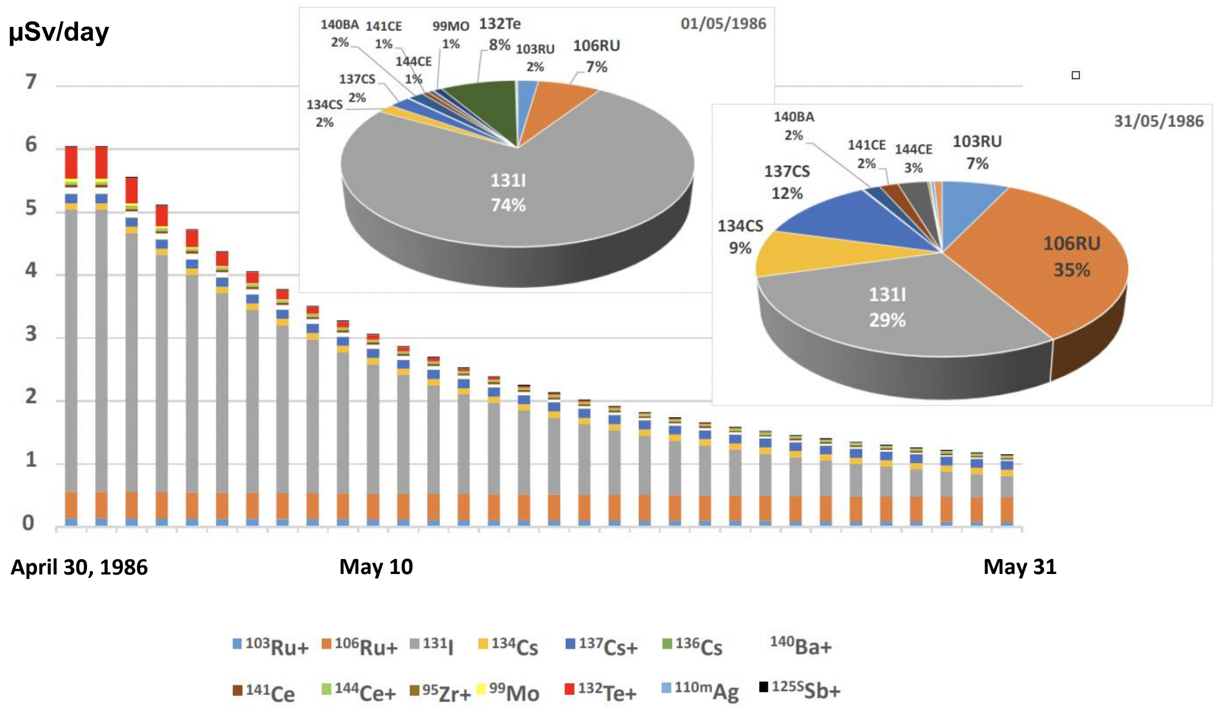


Figure 6: Trend in daily effective doses received by inhalation by an adult during the month of May 1986 for a unit activity concentration of 1 Bq/m³ of 137-Cs and contribution of the various radionuclides measured in the air (µSv/day)

Figure 6 shows the evolution, throughout the month of May 1986, of the daily committed effective doses[6] by inhalation for an adult and calculated for a unit activity in air of 1 Bq/m³ of 137-Cs+[7]. This graph shows the contributions of the various radionuclides to the effective dose, even if these doses do not correspond

[6] Once radionuclide intake has occurred, the radiation it releases within the body can last for several days, months, or years, depending on its half-life and how quickly the body eliminates it. But once intake has occurred, it is certain that the dose will be received. This is why we talk about committed dose. For short-lived radionuclides, this committed dose is equal to the effective dose in the year of intake.
[7] The daughters of radionuclides at equilibrium are taken into account in dose estimates. This is the case for all the radionuclide pairs mentioned in this study: 137-Cs and 137m-Ba, 132-Te and 132-I, 106-Ru and 106m-Rh, 103-Ru and 103-Rh, etc. To simplify writing, these pairs are designated by the "+" sign added to the parent radionuclide.

to any real exposure. In fact, as indicated above, the activity concentrations of 137-Cs changed considerably over time and spatially during the first few days of May 1986; it is only by multiplying these doses calculated for a unit activity of 137-Cs with the activity concentrations of this radionuclide in the air, mapped previously, that it is possible to obtain the doses received by inhalation in each municipality in mainland France.

Figure 6 also shows that 131-I is the main contributor to effective doses received by inhalation, with a contribution of 74% on May 1st, followed by 132-Te+ (8%) and 106-Ru+ (7%). At the end of May, 131-I still accounted for 29% of the effective dose, behind 106-Ru+, which accounted for 35%. It should be noted that the contributions of radionuclides for which the activities in the air are not precisely known, such as 99-Mo and 141-Ce, result in very low doses (with contributions of less than 1%). This study also validates the simplified approach used in the 2007 study, in which only 6 radionuclides were considered (132-Te, 103-Ru, 106-Ru, 131-I, 134-Cs and 137-Cs), which appear to have been the main contributors to doses received by inhalation. In the case of thyroid doses, particularly for young children, 131-I provides a contribution between 84% (on May 1st) and 98% (on May 31st), with the 132-Te and 132-I pair providing the additional contribution.

Doses to the brain, breasts, colon, and prostate are mainly due to three gamma-emitting radionuclides present throughout the body, in decreasing order of contribution: 137-Cs+, 134-Cs and 106-Ru+. 132-Te+ also made a significant contribution in the first few days, especially for equivalent doses to the colon. Doses to the lungs are mainly due to 106-Ru+[8] followed by 103-Ru+ and 129m-Te.

2.2.2. Effective and organ doses received by inhalation

Figure 7 shows maps of doses received by inhalation (effective doses for adults and thyroid doses for children aged 1 to 2 years). Effective doses for adults ranged from 1.5 µSv in Ouessant to 45 µSv in Belleray, a municipality near Verdun; thyroid doses for children aged 1 to 2 years ranged from 23 µSv to 754 µSv in the same two towns. In both cases, there is a very marked east-west gradient. This spatial trend is consistent with the maps of airborne activity concentrations presented in Figure 3 and with the decreases in 137-Cs and 131-I activity following longitude, presented in the 2007 study (Renaud et al. 2009).

While the maximum doses at Verdun (2007 study) and Belleray (present study) are virtually identical (46 µSv and 44 µSv, respectively), this study shows that in Bretagne and part of Normandie, effective doses may have been lower than the value of 4 µSv estimated for Cléville in 2007, with a minimum value of 1.4 µSv estimated for Ouessant in the present study.

Equivalent doses to the brain, breasts, and prostate are 10 to 30 times lower than effective doses. Equivalent doses to the colon are 3 times lower than effective doses and equivalent doses to the lungs are at the same level as effective doses.

In terms of time trends, the inhalation doses received by the French population between May 1 and 5, 1986 represent more than 95% of the total inhalation doses received from the Chernobyl accident. From June onwards, doses were negligible compared with those for May 1986 alone. At less than 0.004 µSv, the effective dose for adults committed in June was between 1,000 and 10,000 times lower than the dose in May 1986. Doses for the following months were even lower.

[8] The daughters of radionuclides at equilibrium are taken into account in dose estimates. This is the case for all the radionuclide pairs mentioned in this study: 137-Cs and 137m-Ba, 132-Te and 132-I, 106-Ru and 106m-Rh, 103-Ru and 103-Rh, etc. To simplify writing, these pairs are designated by the "+" sign added to the parent radionuclide.

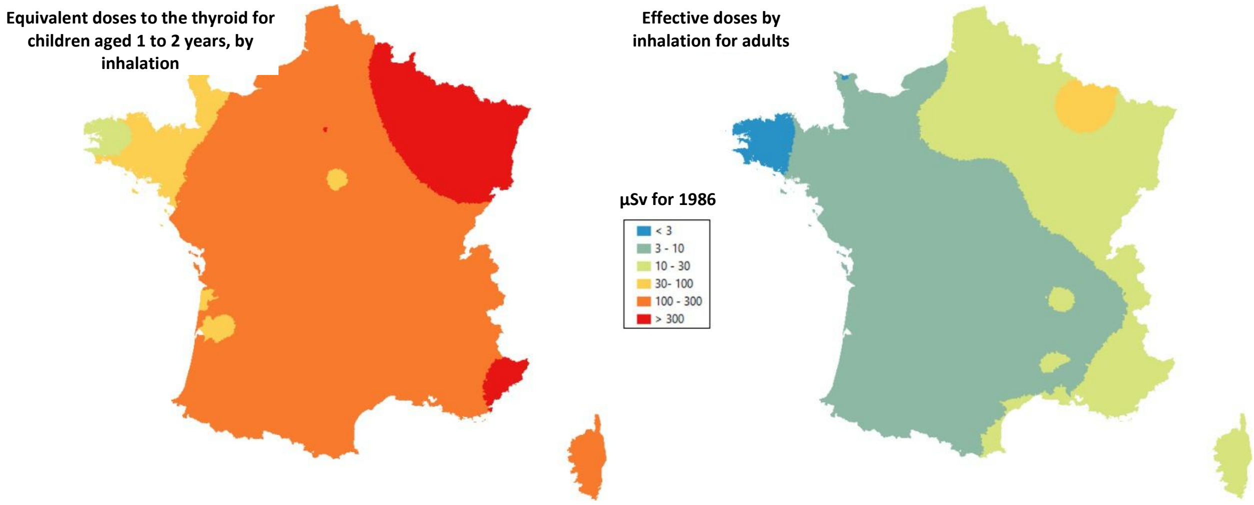


Figure 7: Effective doses received by inhalation in 1986 for adults (right-hand map) and equivalent doses to the thyroid for children (left-hand map) aged 1 to 2 years (µSv).

3. Reconstruction of external doses

3.1. General information

External doses from the Chernobyl accident fallout result from external irradiation due to radiation (gamma equivalent dose rate in air) emitted by the persistence, on various types of surfaces, of the radioactive deposits that built up at the beginning of May 1986. External doses due to immersion in the radioactive plume were negligible, even compared with exposure by inhalation alone, of which they represented less than one hundredth (Renaud et al., 2009).

Radionuclides deposited on the surface of the soil penetrate rather quickly to a few millimetres due to rain. The deeper migration is then slower and practically only concerns radionuclides with a half-life longer than one year. In addition, ploughing an agricultural area leads to the penetration and homogenisation of the activity of a radionuclide in 10 to 20 cm of soil, depending on the type of crop.

The deposited radionuclides emit radiation that induces a gamma equivalent dose rate in the air. As soon as the radionuclide penetrates the soil, part of its radiation is absorbed by the thickness of the soil above it. As a result, the radionuclides likely to generate significant external exposure to people are gamma radiation emitters with sufficient energy to cross a thickness of soil, including: 106-Ru+, 103-Ru+, 137-Cs+, and 134-Cs. For the same deposited surface activity, the intensity of radiation in the air above the soil, and therefore the dose rate, decreases when the depth reached by the radionuclide increases; this varies according to the type and energy of the radiation and the density of the soil.

Some of the radionuclides deposited on artificial surfaces are temporarily fixed by adsorption, while the rest run off and end up in urban sewer systems. Wear related to rain, the passage of pedestrians and vehicles, as well as the cleaning of roadways, remobilises the radionuclides initially adsorbed on the surface of the materials; they are then carried away by runoff. The kinetics of migration to the depths of the soil or of fixation/remobilisation of radionuclides on artificial surfaces are extremely variable. They depend on the characteristics of the soil or materials, the radionuclide, as well as many other parameters, including the characteristics of precipitation, the use and maintenance of surfaces, etc. Moreover, except for 137Cs+, for which there have been some experimental data and observations following the Chernobyl and Fukushima accidents, all the phenomena mentioned above are poorly understood.

Finally, the dose rate inside buildings is lower than it is outdoors. This is mainly due to the fact that the indoor spaces are less contaminated than the outdoors, in particular because of regular cleaning, the fact that the external radiation source is further away, and, to a lesser extent, the fact that building materials shield radiation that comes from outside.

This reduction in the dose rate indoors compared to outdoors is defined by a protection factor as well as by daily time spent inside buildings, assigned for two categories of adults (one working outdoors and the other working indoors), and three age groups of children. The protection factor varies greatly depending on the type of dwelling, from 0.4 (the indoor dose rate represents 40% of the outdoor dose rate) for a single-storey wooden house, to less than 0.01 on the upper storeys of a concrete building.

So, unlike exposures related to inhalation and ingestion of radionuclides, external exposure can give rise to a very wide variety of exposure situations and conditions. For this study, the choice was made to consider a limited number of exposure scenarios. Their suitability with respect to the main foreseeable exposure situations will be discussed.

3.2. Available data and review of assessments from 2007

In the absence of usable direct dose rate measurements[9] until the early 2000s, doses due to external exposure must be calculated based on activities measured or estimated in the soil, using mainly dose factors expressed in Sv/h per Bq/m2 or Sv/h per Bq/kg of soil.

The only reliable data available for estimating radioactive deposits following the Chernobyl accident are those acquired between 2000 and 2002 for this purpose, which led to the mapping of 137-Cs deposits throughout France (Renaud et al, 2004), validated by the IRSN Scientific Council in 2005 (Figure 8). This map shows that deposition was greater and more heterogeneous in the eastern part of the country, ranging from less than 5,000 Bq/m^2 to sometimes more than 40,000 Bq/m^3 just a few dozen kilometres away. This heterogeneity of deposits is related to the heterogeneity of rainfall that occurred during the passage of the contaminated air masses between May 1st and 5th, 1986. In fact, it was the study of the "rainfall-deposition" relationship that led to the mapping of the eastern part of the country (Renaud et al., 2003). Further west, deposition was lower and more homogeneous due to less precipitation (in the central longitudes) and lower activity concentrations of air masses from east to west (Renaud et al, 2004).

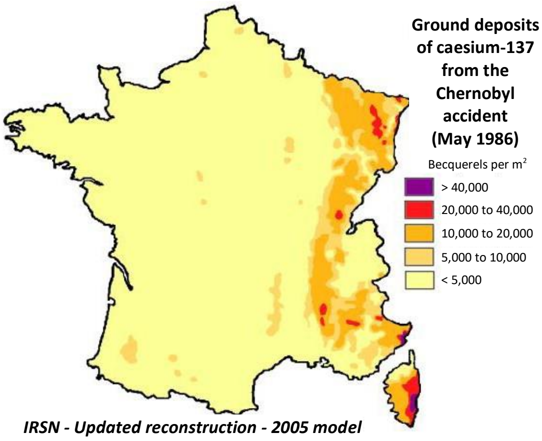


Figure 8: Map of surface activity of caesium-137 deposited on French soil in May 1986 after fallout from the Chernobyl accident (Renaud *et al*, 2007)

In the 2007 study, the external doses induced by radioactive deposits were estimated for a unit 137-Cs deposit of 1 kBq/m^2. The deposited activities of 4 other radionuclides assumed to be the other main contributors (134-Cs, 131-I, 103-Ru+ and 106-Ru+), were estimated based on the isotopic activity ratios established in the air between each of them and 137-Cs (see paragraph 2.2.1.2). External doses were estimated for a conservative scenario, that of an adult living in a rural environment and spending 14 hours a day inside buildings, 4 hours a day outdoors on ploughed land, and 6 hours a day outdoors on undisturbed land (neither ploughed nor covered since the accident). This scenario will hereafter be referred to as "rural". As early as 1987, ploughing is assumed to have homogenised radionuclides in cultivated soils to a depth of 20 cm. Successive ploughings and the deeper migration of radionuclides under the effect of water (rain and irrigation) increased this depth by 0.5 cm/year, leading to a total depth of almost 30 cm in 2006. In unploughed soils, after a migration of 1 cm during the first year (from 1986 to 1987), radionuclides migrated downwards at a rate of 0.5 mm/year, reaching a depth of almost 15 cm in 2006. These assumptions on radionuclide burial are based on a study published in 2007 (Roussel-Debet *et al*, 2007) and on activities measured in French soil between 1991 and 2004. Dose factors from the US Environmental Protection Agency (EPA, 2019) were interpolated to obtain the corresponding annual values for the depths of interest between 1986 and 2006.

[9] In situ gamma spectrometry measurement, which can distinguish radiation emitted by each of the radionuclides deposited following the Chernobyl accident from the natural background radiation, was not developed until the early 2000s, and few measurements were carried out in the following years. The number of measurements has only recently become sufficient to estimate external doses (IRSN, 2022).

In terms of spatialization, the publication (Renaud *et al.*, 2007) limits itself to indicating that, throughout the country, the external effective doses for an adult can be deduced from those estimated for a 137-Cs unit deposit of 1 kBq/m^2 using the mapping of deposits of this radionuclide, and that they were therefore as high as 400 µSv in 1986 in the worst-affected areas of the country (40,000 Bq/m^2).

The external effective doses estimated in 2007 for the period 1986-2007 are compared with those estimated in chapter4.3.2.2 of this study.

4. Knowledge gained from observations made after the Fukushima accident

4.1.1. Trend in the equivalent dose rate in the first few years following the radioactive deposits

Following the Fukushima accident in 2011, numerous direct measurements of airborne equivalent dose rates were taken in Japan. Figure 8 summarises the results of these measurements in the 5 years following the accident (Siato K. et al, 2019). It shows the decrease in the equivalent dose rate observed through measurements taken using different devices/techniques, by comparing all the measurement results to an initial unit value in June 2011 (i.e. after the disappearance of the short-lived radionuclides mentioned above).

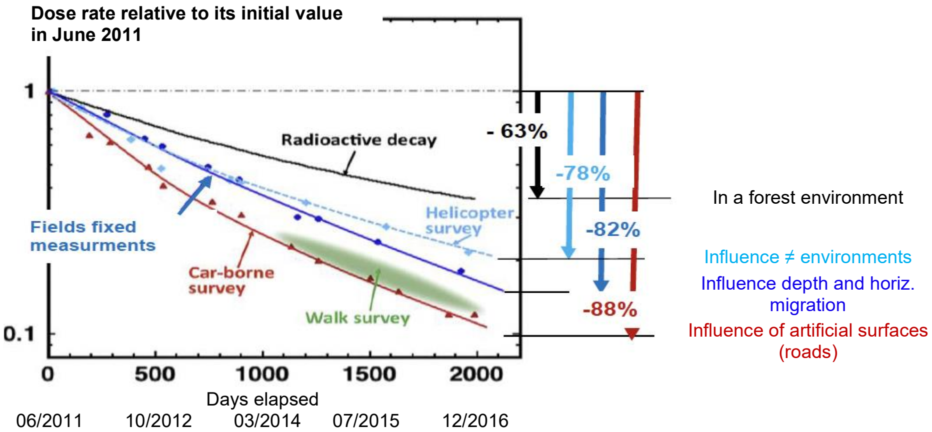


Figure 9: Decrease in dose rate observed in different types of environments by different types of measurement over the five years following the Fukushima accident (from Siato K. et al, 2019)

The black curve shows the decrease due solely to radioactive decay of caesiums, particularly 134-Cs (half-life of 2.1 years). In 5 years, radioactive decay led to a reduction of -63% in the equivalent dose rate (for a deposited activity of 134-Cs equal to that of 137-Cs, as was the case during the fallout from the Fukushima accident). The curve in dark blue represents the decrease observed on natural, undisturbed surfaces (field measurements) through dose rate measurements taken using fixed devices; it shows a reduction of -82%[10] over the first 5 years. The light blue dotted line represents the decrease observed by devices on board a helicopter: -79% in 5 years. The position of this slope, an intermediate between the slope due to radioactive decay and that observed by fixed field measurements, is explained by the fact that several types of surfaces contribute to the dose rate "seen" by the helicopter: cultivated and uncultivated fields, some artificial (urban) surfaces and forested areas. In a forest environment, the rate of dose reduction is slow, barely faster than that due to radioactive decay. Finally, measurements taken by on-board equipment in road vehicles (red curve) are, for reasons of geometry, heavily influenced by the road itself. These road measurements confirm what was already observed after the Chernobyl accident, namely that artificial surfaces decontaminate faster than non-artificial soils. This is because only some of the atmospherically deposited radionuclides adsorb onto the surface of the materials, and the wear caused by the use of these materials (traffic), rainfall, and cleaning results in the radionuclides being desorbed or "pulled away" into the rainwater collection system. It should be noted that measurements taken while walking (using a measuring device in a backpack, for example) are intermediate between road measurements and “field” measurements.

[10] Still for an initial 134-Cs/137-Cs ratio equal to 1. In the case of fallout from the Chernobyl accident, where 134-Cs/137-Cs = 0.5, the decrease would be -78%.

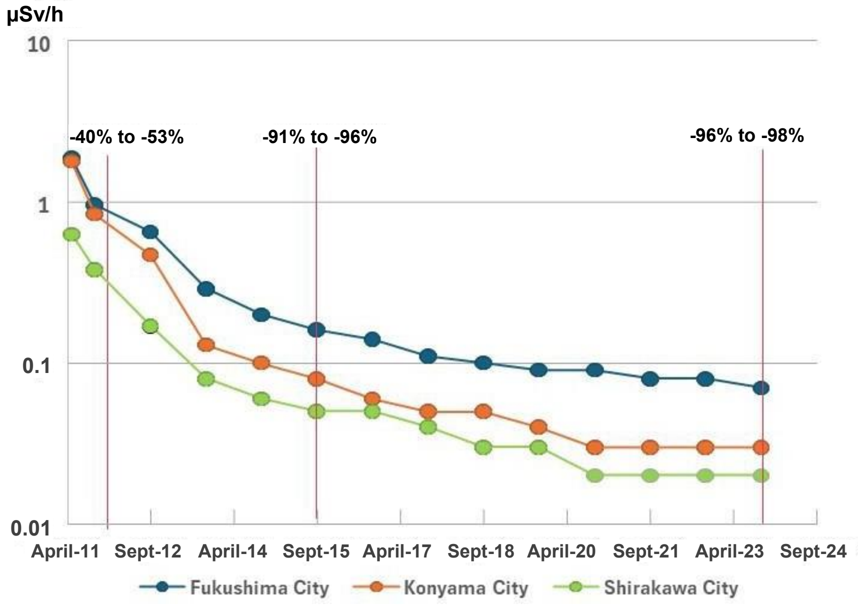


Figure 10: Decrease in dose rate observed in 3 major Japanese cities (urban areas) in the 12.5 years following the Fukushima accident

Figure 10 shows the decrease in dose rate observed in 3 major Japanese cities representing urban areas, in the 12.5 years following the Fukushima accident. This figure shows that between April and September 2011, the dose rate fell by half (-43 to -53%). This reduction is mainly due to the elimination of radionuclides adsorbed onto artificial surfaces during rainfall, cleaning operations, and traffic on roads. The decrease linked to radioactive decay of short-lived radionuclides, which contribute most to the dose rate (^{132}Te-I, ^{131}I, ^{140}Ba-La), is most noticeable during the first month following deposits (see Figure 11), i.e. between March and April 2011, before the first measurements presented in Figure 10. After 4.5 years, in September 2015, the reduction was between -91 and -93%, fairly close to the -88% observed on roads and presented in Figure 9. After the first five years, the decline is slower, reaching -96% to -98%, 12.5 years after the deposits.

In the case of non-artificial soils, whether disturbed or not, the decrease in dose rate over the years is studied in section 4.3.2, based on measurement results acquired in France and presented in section 4.2.

4.1.2. Protection factor inside buildings

The protection factor (see section 3.1) was well documented after the Fukushima accident. It varies greatly depending on the type of dwelling, from 0.4 (the indoor dose rate represents 40% of the outdoor dose rate) for a single-storey wooden house, to less than 0.01 on the upper storeys of a concrete building (Yoshida-Ohuchi et al. 2019).

The protection factor values adopted for this study are 0.28 in rural areas (the indoor dose rate represents 28% of the outdoor dose rate) and 0.025 in urban areas (the indoor dose rate represents 2.5% of the outdoor dose rate). The values come from an American bibliographical study (Lawrence Livermore National Laboratory, 2016), which integrates the many data points acquired after the Fukushima accident. The value of 0.28 corresponds to a single-storey family home built of brick. This is lower than 0.4, from the same study, which corresponds to a single-storey house built of light materials such as wood. But it is higher than the value of 0.2 recommended by the IAEA (IAEA, 1979 and IAEA, 2000) on the basis of previous data. The value of 0.025 chosen for the urban environment corresponds to the lower storeys of an apartment building. It is conservative (i.e. it tends to overestimate the dose) compared with the average value of 0.01 mentioned in the same study for upper storeys.

For information purposes, Table IV presents the ratio of external doses calculated with protection factors of 0.28 to those calculated with other protection factors, taking into account the various periods of time spent indoors as presented in Table XI. For adults working outdoors, the ratios are close to 1, regardless of the type of dwelling. The same is true for single-storey dwellings, regardless of age group. On the other hand, ratios are significant between single-storey dwellings and multi-family apartment buildings (3rd and 4th lines of the table), like those found in urban areas, especially for young children.

Table IV: Corrective factors of external doses for an indoor protection factor of 0.28

	Adult working outdoors	0-1 year	1-2 years	Other age groups and sedentary adults
0.28/0.4	0.9	0.7	0.7	0.8
0.28/0.2	1.1	1.4	1.3	1.2
0.28/0.025	1.5	16	6.5	3.2
0.28/0.01	1.5	40	6.0	2.8

4.2. External doses estimated by IRSN for France from 2008-2018

As part of an assessment carried out by IRSN (now ASNR) on the background noise of artificial radionuclides in the environment of mainland France (IRSN, 2022), external effective doses from caesium-137 in soil were estimated for the period 2008-2018 for adults living in rural areas and working outdoors, using two approaches: one based on direct measurements of the equivalent dose rate and the other based on activity concentrations measured in soils. These two approaches led to concordant results. Table V shows the estimated doses for different regions and areas of France based on 137-Cs activities measured in grassland soils undisturbed since 1986 (a similar and concordant table exists for ploughed soils).

Table V: External effective doses due to caesium-137 in French soils, estimated for an adult living in different regions of mainland France and working outdoors in a rural environment, based on measurement results acquired over the period 2008-2018 (IRSN, 2022)

	Western 2/3 of the country			Eastern 1/3 of the country	
	Occitanie Nouvelle-Aquitaine	Bretagne, Normandie, Hauts-de-France	Centre-Val de Loire, Pays de la Loire	Grand Est	Rhône-Alpes, Provence-Alpes-Côte d'Azur
	Estimated annual external effective doses (µSv/year)				
min.	1	0	1	2	4
max.	8	7	6	19	18
Inter-regional averages	2	3	3	6	8
Area averages	3			7	

	Areas of high persistence (ZRE)			
	ZRE Rhone valley	ZRE Corsica	ZRE Jura and Doubs	ZRE Vosges
	Estimated annual external effective doses (µSv/year)			
min.	10	18	17	15
max.	35	22	39	79
Inter-regional averages	19	20	27	36
Area averages	23			

This table shows that, in the western two-thirds of the country, the average effective dose for adults was estimated at 3 µSv/year for the period 2008-2018, ranging from 1 µSv/year (lowest regional average in Occitanie and Nouvelle-Aquitaine) to 8 µSv/year (highest regional average in these same regions). In this vast area, 137-Cs deposits following the Chernobyl accident ranged from less than 1,000 to 5,000 Bq/m^2. These deposits are added to the persistence of those from atmospheric testing of nuclear weapons worldwide, estimated between 1,500 and 4,500[11] Bq/m^2.

In the eastern part of the country, outside the areas most affected by fallout from the Chernobyl accident (known as ZRE for the French “area of high persistence”), the average dose is estimated at 7 µSv/year (regional averages ranging from 2 to 19 µSv/year). In this area, 137-Cs deposits following the Chernobyl accident were between 5,000 and 10,000 Bq/m^2. Finally, in areas of high persistence in the eastern part of the country, where 137-Cs deposits ranged from 10,000 Bq/m^2 and locally over 40,000 Bq/m^2, the average dose was estimated at 23 µSv/year within a regional range of 10 to 40 µSv/year (apart from an isolated maximum value of 79 µSv/year estimated in the Vosges).

As a reminder, these estimates correspond to the worst-case scenario of an adult living in a rural environment and working outdoors on land that has either been ploughed or undisturbed (neither ploughed nor covered) since the radioactive deposits of May 1986.

[11] In areas with high annual precipitation, deposits linked to fallout from atmospheric testing of nuclear weapons have been as high as 7,500 Bq/m^2 very locally.

4.3. 2025 update

4.3.1. External doses in 1986

As in the 2007 study, external doses were estimated for a unit deposit of 1 kBq/m^2 of 137-Cs. However, the number of radionuclides considered is 15 (compared with 6 in the 2007 study) using the isotopic activity ratios observed in the air in France on May 1st and 2nd, 1986, between each of these radionuclides and 137-Cs (see Table II). For 1986, radioactive deposits are assumed to have penetrated the soil by 3 mm (compared with 1 cm in the 2007 study); the dose factors used are those proposed by the International Commission on Radiological Protection in its publication 144 (ICRP, 2020). External doses were calculated for two categories of adults, one working outdoors and the other indoors (referred to here as “sedentary workers”), as well as for the 5 age groups of children in both rural and urban environments. People are considered to live in an urban environment if their surroundings are mainly made up of artificial surfaces (roads, buildings, etc.); in addition to cities, they may live in villages or hamlets. People are considered to live in a rural environment if they have spent several hours a day on land that has been ploughed or on undisturbed land (neither ploughed nor covered) since 1986. Table VI shows the corresponding outdoor and indoor times. They are taken from ICRP Publication 66 (ICRP, 1994).

Table VI: Daily time spent outside and inside buildings (ICRP, 1994)

	Inside	Outside	Inside	Outside
	h/d	h/d	Daily fraction	
Infant	24	0	1.000	0.000
1-2 years	23	1	0.958	0.042
3-7 years	21	3	0.875	0.125
8-12 years	21	3	0.875	0.125
13-17 years	21	3	0.875	0.125
Outside worker	16	8	0.667	0.333
Sedentary worker	21	3	0.875	0.125

Table VII shows the corresponding estimated effective doses, for all radionuclides combined, for a unit deposit of 1 kBq/m^2 of caesium-137.

The external effective dose for an adult working outdoors in 1986 (between May 1st and December 31st, 1986) and for a 137-Cs unit deposit of 1 kBq/m^2, is estimated in this study at 14.8 µSv. This dose is slightly higher than the 12 µSv estimated in the 2007 study, mainly because more radionuclides were taken into consideration (15 radionuclides instead of 6 in 2007). It is very close to estimates made in other countries, again for a unit deposit of 1 kBq/m^2 of 137-Cs: 15 µSv for Belarus and Russia and 17 µSv for Ukraine (IAEA, 1999).

Table VII: External effective doses in µSv, all radionuclides combined, estimated for 1986 and for a unit deposit of 1 kBq/m^2 of caesium-137

Age group	Rural environment	Urban environment
0-1 year	13.0	1.16
1-2 years	12.9	2.72
3-7 years	14.2	5.63
8-12 years	13.0	5.15
13-17 years	12.2	4.82
Adult working indoors	11.8	4.67
Adult working outdoors	14.8	10.0

The estimated doses for city dwellers living in apartment buildings are lower than those for people living in single-storey houses in rural areas, particularly for young children who spend most of their time indoors.

Furthermore, although the dose factors are higher for children, their effect on the estimated doses is offset by the fact that adults spend more time outdoors.

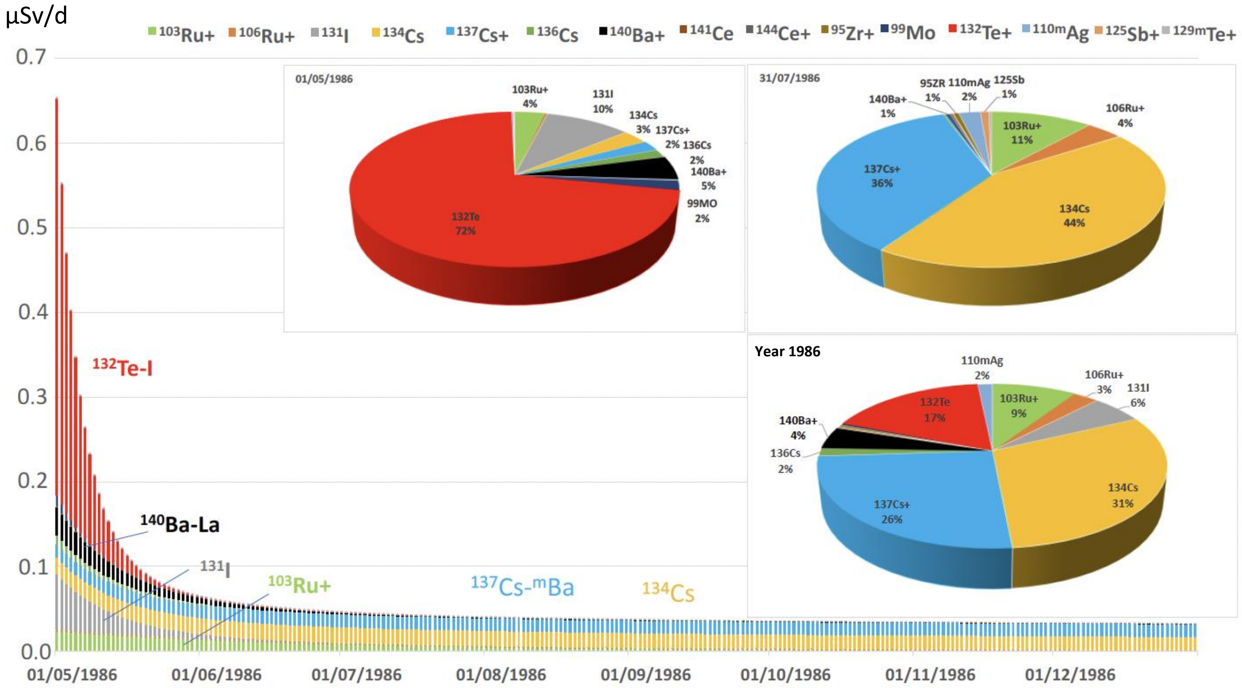


Figure 11: Trend in daily external effective dose (µSv/d) for an adult working outdoors in a rural environment in 1986 and for a unit deposit of 1 kBq/m^2 of 137-Cs, with contributions from the various radionuclides

Figure 11 shows the daily change in external effective dose for an adult between May 1st and December 31, 1986, again for a unit 137-Cs deposit of 1 kBq/m^2. Although the highest levels of airborne activity were measured on May 1st, most of the deposits were formed during the rains of May 2, 3 and 4, 1986. However, it is not possible to reconstruct deposits day by day; in this study, they are allocated in full to May 1st alone, the consequences of this simplification being negligible. Deposits of each radionuclide were estimated on the basis of isotope activity ratios measured in the air on May 1st and 2nd, particularly for short-lived radionuclides. Because of their longer half-life and the fact that measurements were available throughout the month of May, the average activity ratios 106-Ru+/137-Cs + and 144-Ce+/137-Cs+ were obtained from all the data available for May 1986.

The rapid reduction in this external dose in May 1986 alone is due to the radioactive decay of short-lived radionuclides, particularly 132-Te+, 140-Ba+ and 131-I, which were the main contributors to this dose in the first few days. The pie chart at the top left of Figure 11, for May 1st, 1986, shows that these three radionuclides accounted for around 87% of the dose just after the radioactive deposits. A very similar trend was observed through dose rate measurements in Japan after the Fukushima accident, and IRSN demonstrated its link with radioactive decay of short-lived radionuclides (IRSN, 2012).

Figure 11 shows that by July 31, 1986, 137-Cs+ and 134-Cs were already contributing 80% of the external effective dose (top right pie chart). This contribution exceeded 95% on December 31, 1986, for an overall contribution of 57% for the whole year (bottom pie chart).

4.3.2. External dose trends from 1986 to 2020

4.3.2.1. General formulation

A different approach to that used in the 2007 study is proposed here for changes in external dose between 1986 and 2020. It consists in using observations made in Japan on the decrease in dose rate in the years following the Fukushima accident (see Chapter 4.1), and external doses estimated from measurements taken in France from 2008 to 2018 (see Chapter 4.2). This makes it possible to distinguish between people living in urban and rural areas. In both cases, to account for the rapid decrease in dose rate in the first few years after deposition, and the slower decrease thereafter, the proposed equation is of biexponential form:

$$Dext_i = (t_{ext} + t_{int} \cdot FP) \cdot D_j \cdot Fd_{i,j} \cdot [\, a \cdot e^{-((i-1986)/T_{eff1})} + (1-a) \cdot e^{-((i-1986)/T_{eff2})} \,]$$

$$1/T_{eff} = 1/T_{rj} + 1/T_m$$

Equation [1] Where:

$Dext_i$:	External dose in year i	Sv/year
t_{ext}, t_{int}:	Time spent outdoors and indoors	h/year
FP:	Protection factor inside buildings	Dimensionless
D_j:	Deposit of radionuclide j in 1986	Bq/m^2
$Fd_{i,j}$:	Dose factor of radionuclide j for year i	Sv/h per Bq/m^2
a:	Distribution coefficient of half-lives T_{eff1} and T_{eff2}	Dimensionless
T_{eff1} and T_{eff2}:	Effective short and long decay half-lives of fractions a and 1-a of the deposit	year
T_{rj}, T_{m1} and T_{m2}:	Radioactive and mechanical decay half-lives	year

The values of the parameters a, Tm and FP vary depending on whether the decrease is simulated in an urban or a rural environment.

4.3.2.2. Case of rural environments

People are considered to live in a rural environment if they have spent several hours a day on land that has been ploughed or on undisturbed land (neither ploughed nor covered) since 1986. In this environment, the radionuclides contained in the 3 mm of surface soil in 1986 gradually migrated and were homogeneously distributed at a depth of 15 cm[12] in 2020. This gradual migration is taken into account by linear interpolation of the dose factors between the values corresponding to 3 mm in 1986 and 15 cm in 2020. In addition, a further reduction in the dose rate, which could be attributable to losses due to horizontal or vertical migration of part of the 137-Cs to greater depths, is defined by a "mechanical" activity decay half-life. The values of the relevant fraction "a" of the activity and its "mechanical" half-life were determined by adjusting the external dose predicted by equation [1] with, on the one hand, the -82% decrease observed in Japan 5 years after the Fukushima accident ("field" measurements in figure 9) for a 134-Cs/137-Cs ratio of 1, which corresponds to a decrease of -78% for the 134-Cs/137-Cs ratio of 0.5, characteristic of the Chernobyl accident; and on the other hand, the dosimetric estimates made for the 2008-2018 time period[13]. This adjustment is shown in Figure 12 for an adult working outdoors in a rural environment. The value of the parameter "a" is 0.55 (dimensionless) and there is only a "mechanical" half-life, Tm1, which is 1.5 years. The mechanical decay therefore only concerns the first years following deposits. Beyond that, the progressive burial of caesium (integrated through the interpolation of dose factors as indicated above) and radioactive decay (mainly of 134-Cs) are sufficient to explain the range of doses estimated for the period 2008-2018. This seems to confirm that the rapid decay observed in Japan in the first few years after the deposits resulted from losses (horizontal and vertical) related to the high mobility of caesium during this period. It should be noted that a recent Japanese publication (Andoh et al, 2020) proposes a model of the same type to simulate changes in dose rate over the 7 years following the Fukushima accident, and compares them with those obtained in Russia and Ukraine after the Chernobyl accident.

To illustrate the consistency between the estimate in equation [1] and that in the 2022 study, we note that the range of doses obtained for a deposit of 40,000 Bq/m^2 of caesium-137 (blue curve on Figure 12) over the period 2008-2018, using the present modelling, is between 35 µSv in 2008 and 17 µSv in 2018 with a value of 25 µSv in 2013 (central year of the period); whereas the 2022 study provided a range between 40 µSv[14] and 10 µSv for an average of 23 µSv (blue rectangle). Similarly, the range of doses obtained for a deposit of 5,000 Bq/m^2 of caesium-137 (black curve) over the period 2008-2018 is estimated between

[12] Measurements taken by IRSN in different types of undisturbed soil have shown that, 30 years after the Chernobyl accident, 137-Cs was essentially found in this 15 cm layer (IRSN, 2022 and Observation of areas of persistence).
[13] The estimated doses for this period were applied to the central year of 2013.
[14] A value of 79 µSv was only obtained locally in the Vosges area.

4 µSv (in 2008) and 2 µSv (in 2018), with a value of 3 µSv in 2013 (the central year of the period), while the 2022 study provided a range between 8 µSv and 1 µSv, with an average of 3 µSv (grey rectangle).

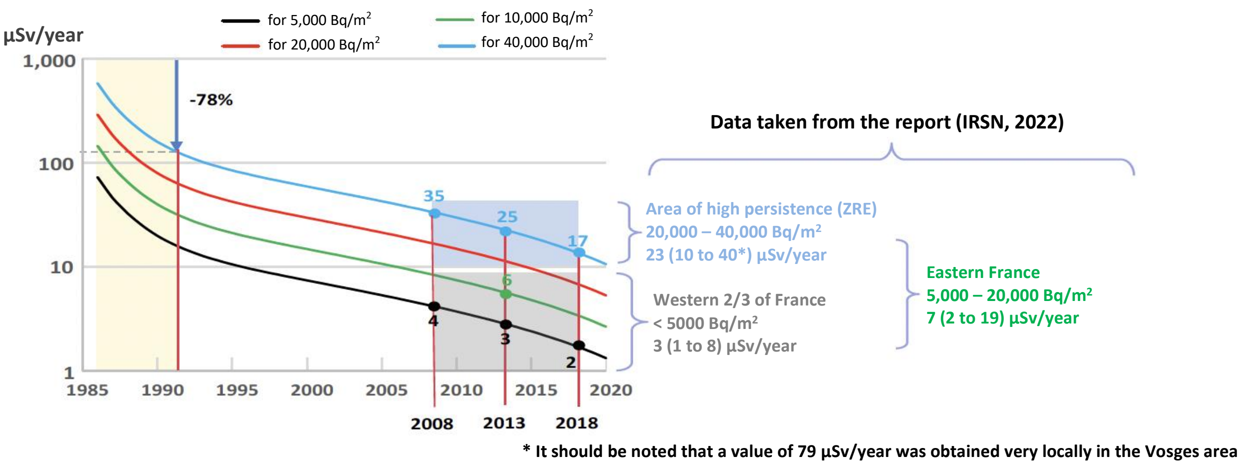


Figure 12: Trend in the external effective dose (µSv/year) estimated here for an adult working outdoors in a rural environment according to the surface activity of deposited Cs-137 (Bq/m²) and comparison with dose estimates from the report (IRSN, 2022) for the period 2008-2018

It is also interesting to compare the estimates made using the 2007 approach (Renaud et al. 2009) and those of the present study. This is the purpose of Figure 13, which compares them between 1986 and 2006 for a unit deposit of 1 kBq/m² of 137-Cs. The differences between the two estimates are small, despite the major differences in the assumptions of the two approaches (number of radionuclides considered, value of the protection factor, dose coefficient values used[15], and other scenario elements). For 1986, the estimate in the present study (14.8 µSv) is higher than the estimate in the 2007 study (12.1 µSv). This is mainly due to the fact that radionuclides not considered in the 2007 study have been included in the current study (140-Ba+, 136-Cs, 99-Mo, 129m-Te, 141-Ce). It should also be noted that, in the 2007 study, the estimated dose for 1986 (12.1 µSv) is lower than that estimated for 1987 (15 µSv); the neglect of certain short-lived radionuclides highlighted the fact that, unlike 1987, 1986 was not a complete year (only 8 months of exposure from May to December 1986). For the other years, the doses estimated here are 60% to 75% lower than those estimated in 2007. This difference is mainly due to a more rapid decrease in the dose rate estimated in this study, particularly in the first few years following the deposits, as a result of the adjustment following observations made in Japan.

[15] Those proposed by the US Environmental Protection Agency (Eckermann et al, 1993) for the 2007 study and those recommended by the International Commission on Radiological Protection (ICRP, 2020) for the present study.

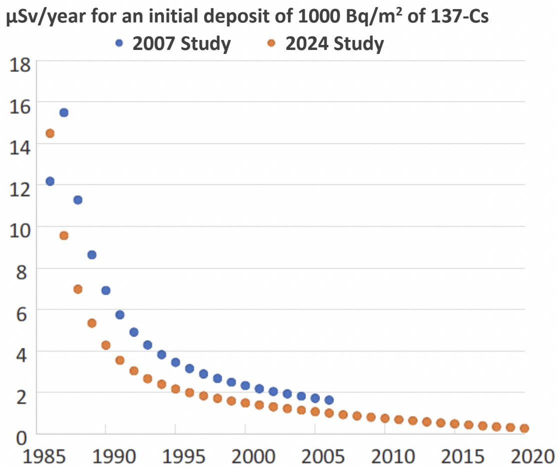


Figure 13: Comparison of changes in external doses between the 2007 study and this update, all radionuclides combined, and for an initial deposit of 137-Cs of 1000 Bq/m^2, estimated for an adult working outdoors in a rural environment.

4.3.2.3. Case of urban environments

As a reminder, people are considered to live in an urban environment if their surroundings are mainly made up of artificial surfaces (roads, buildings, etc.); in addition to cities, they may live in villages or hamlets. In this urban environment, in order to simulate trends in the dose rate in France between 1987 and 2020, the values of the parameters "a" and "Tm" in equation [1] were determined in order to adjust these trends to those observed in Japanese cities over the 12.5 years following the radioactive deposits from the Fukushima accident. The value of "a" thus determined is 0.85 (dimensionless), and the values of Tm1 and Tm2 are 1.2 years and 20 years, respectively.

4.3.2.4. Rural vs. urban and outdoor vs. indoor worker overview

Figure 14a (left) compares the estimated external doses for adults working outdoors and indoors in the two types of environment (rural and urban), for all radionuclides combined and for an initial deposit of 1,000 Bq/m^2 of caesium-137. In 1986, as expected (see Table IV), the differences were small, with the exception of the estimated dose for a city dweller working indoors, which was twice as low as the others (blue curve in the graph on the left). In subsequent years, the higher value of the "a" coefficient for the urban environment than for the rural environment, together with a shorter "Tm1" value (1.2 instead of 1.5 years), led to a much greater reduction in doses in the urban environment during the first few years. A Tm2 value of 20 years for the urban environment, while radioactive decay was sufficient for long-term adjustment in the rural environment, also results in a faster long-term decrease. As a result, in 2020, the estimated dose for a person working indoors in an urban environment was 10 times lower than that estimated for an adult working outdoors in a rural environment.

In the absence of sufficient hindsight on the observations made in Japan (only 12.5 years since the deposits), and in the absence of dose rate measurements in an urban environment in France, it is questionable whether the dose rate in this environment will fall over the long term, and in particular whether the renewal of surfaces (road repairs, etc.) could lead to even lower dose rates. In this case, is it possible to determine a lower limit for the external dose received by a person living in an urban environment? We can imagine city dwellers spending part of their time in a natural environment, either during holidays in rural areas or by visiting an urban park. Figure 14b (right) shows the evolution of the dose associated with a thirty-day stay per year in a rural environment, which could also correspond to daily visits to an urban park lasting 1.5 h/d (the stay including time spent indoors, whereas the time spent in the park is exclusively outdoors). It appears that, from 1996, i.e. 10 years after the deposits were made, the dose related to this temporary exposure to a natural environment became equivalent to the dose related to exposure during the rest of the year for a city dweller working indoors (the black and red dotted curve joins the blue curve). Thus, in the event of a slump in the dose rate in urban areas due to surface renewal, the lower limit dose likely to be received in 2020 by a city dweller visiting natural areas would remain extremely low, with a maximum value of 1.2 µSv/year if the natural area is located in one of the areas of high persistence (initial caesium-137 deposit of 40,000 Bq/m²).

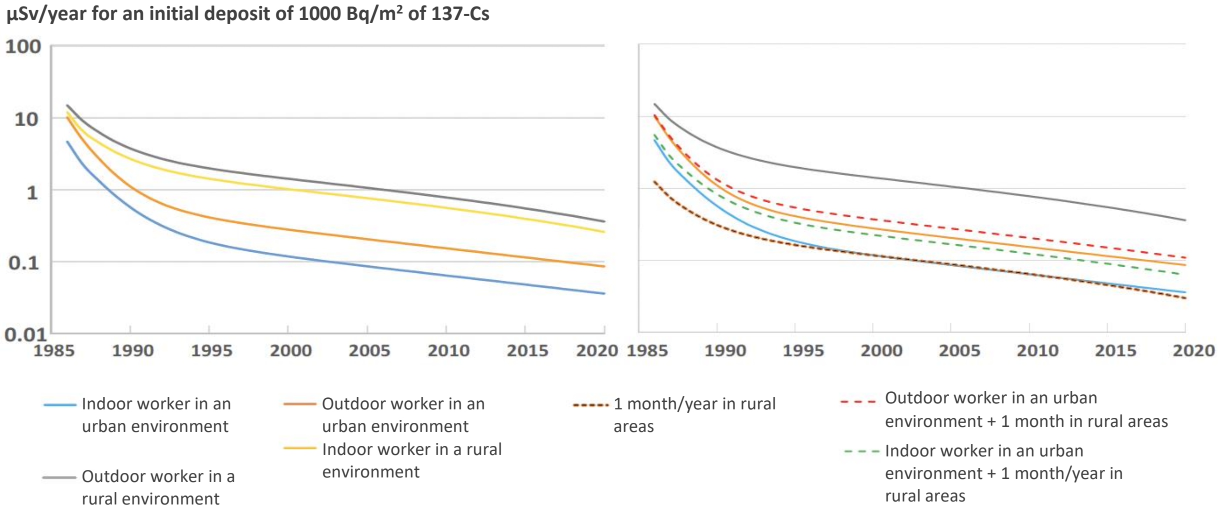


Figure 14: Comparison of external effective doses (µSv/year) estimated for adults working outdoors or indoors in rural or urban environments (left graph); taking into account the exposure time (stay) spent by a city dweller on natural surfaces (right graph).

Figure 15 shows the maps of effective doses due to external exposure estimated for an adult working outdoors in a rural environment and for an adult working indoors in an urban environment. As expected, the general appearance of these maps is directly related to radioactive deposits. In 1986, for adults working outdoors in rural environments, estimates ranged from less than 3 µSv in the Côtes-d'Armor to more than 300 µSv in some areas of eastern France, and up to almost 750 µSv in some municipalities where deposits reached 50,000 Bq/m². In 2020, they ranged from 0.1 µSv to 10 µSv, potentially reaching 20 µSv in these same municipalities.

Since 1986, the estimated external effective doses for indoor workers in an urban environment have been much lower than those for outdoor workers in a rural environment. This difference is due partly to the greater protection afforded by the home and partly to the longer time spent indoors. In the western part of the country, they did not reach 10 µSv and in the eastern part, they ranged from 10 to 100 µSv, exceeding this value only in a limited number of municipalities. In 2000, estimates of external effective doses fell sharply and are now 10 times lower than those estimated for an adult in a rural environment, due to the faster elimination of contamination from artificial surfaces than from non-artificial soils. In 2020, the estimated effective doses from external exposure for an adult working outdoors in an urban environment were below 1 µSv almost everywhere.

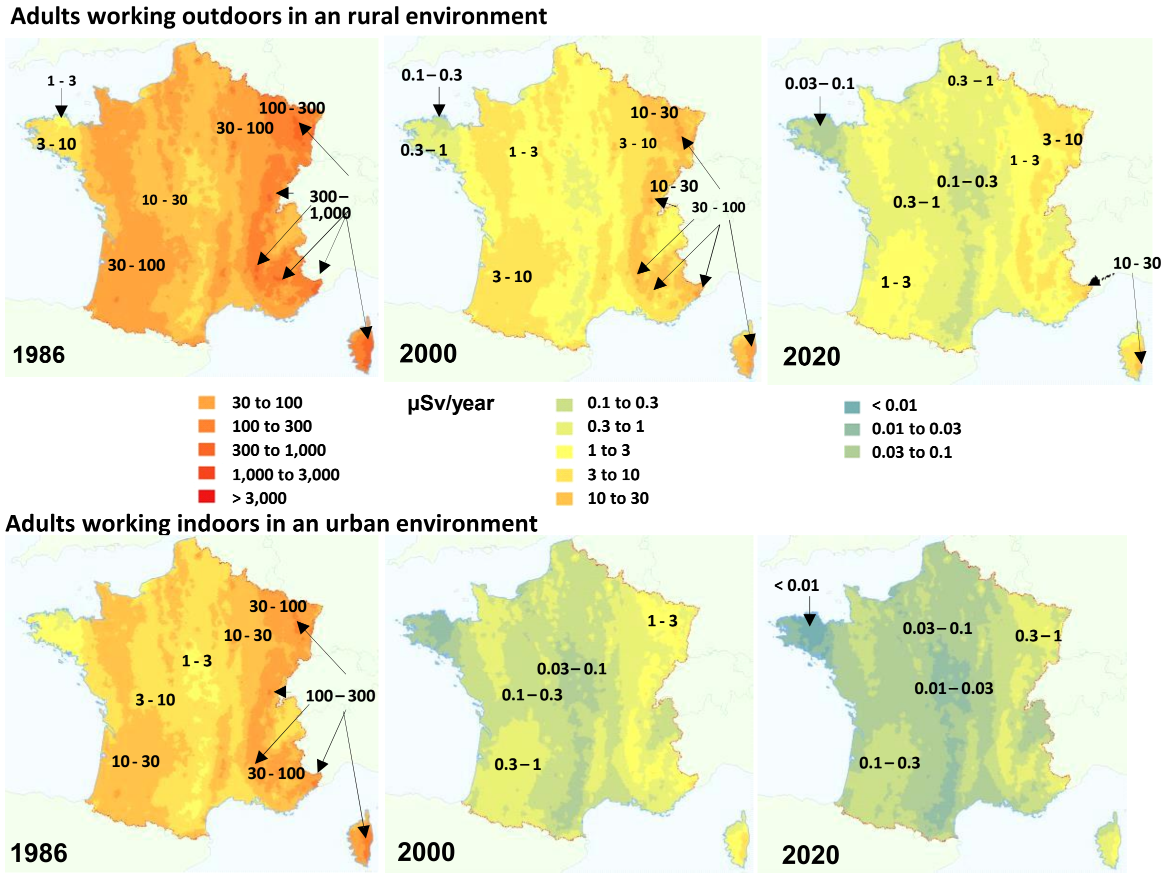


Figure 15: Mapping of effective doses (µSv/year) due to external exposure for adults working outdoors in a rural environment or indoors in an urban environment for the years 1986, 2000 and 2020.

4.3.2.5. External effective doses as a function of age and organ doses

Figure 16 presents the trend in effective doses as a function of the exposed person's age, for all radionuclides combined, and for an initial unit deposit of 1 kBq/m^2 of caesium-137. The discrepancies between the estimated doses for the different age groups previously indicated for 1986 (see table VII) are maintained over time. The estimated doses for city dwellers living in apartment buildings are lower than those for people living in single-storey houses in rural areas, particularly for young children (grey and yellow curves) who spend most of their time indoors. Furthermore, although the dose factors are higher for children, their effect on the estimated doses is offset by the fact that adults spend more time outdoors. The effective external doses for children are very similar to those for adults working indoors, as shown in Figure 16

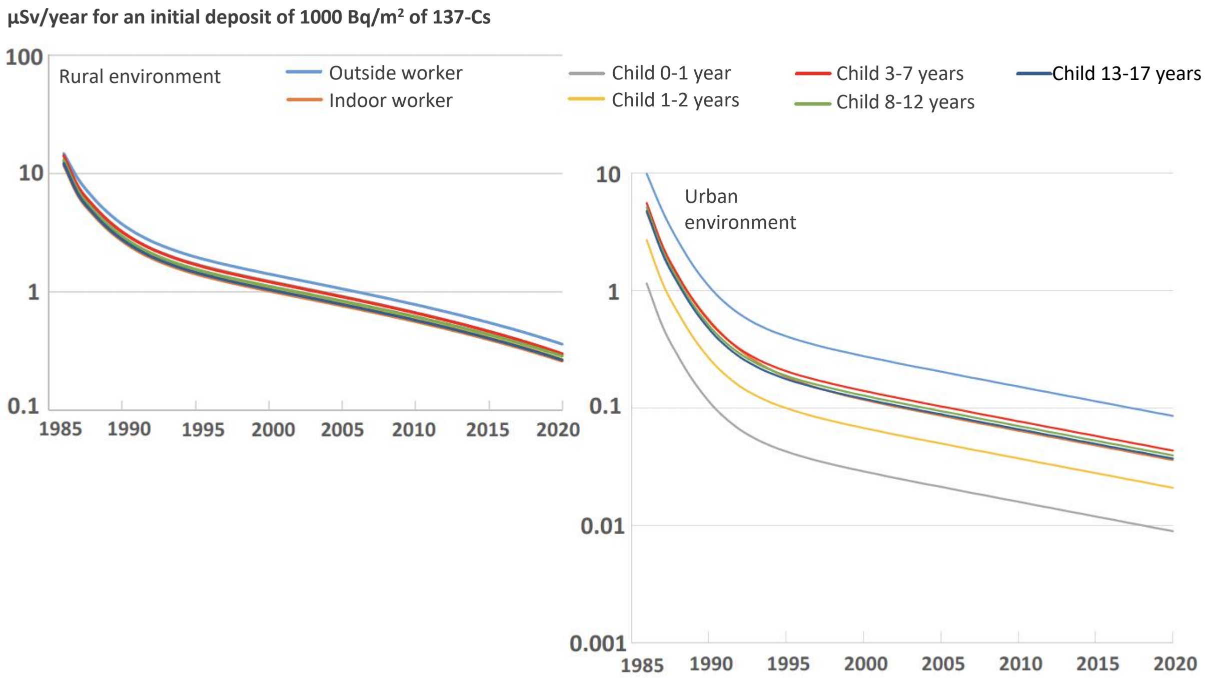


Figure 16: Trends in annual external effective doses as a function of time since the Chernobyl accident and by age group (µSv/year), all radionuclides combined, and for an initial unit deposit of 1 kBq/m² of caesium-137

The equivalent doses to organs due to external exposure to deposits were not calculated in detail in this study since the external dose factors for organs, very close to those for effective doses, do not induce significant differences between organs.

5. Reconstruction of ingested doses

5.1. Available data and review of 2007 assessments

The doses received by ingestion are directly linked to the contamination of foodstuffs. Thousands of activity concentration measurements in all types of foodstuffs produced in France were taken after the Chernobyl accident by SCPRI, CEA/IPSN, and the CNEVA (National Centre for Veterinary and Nutritional Studies). The foodstuffs that were the subject of the largest number of analyses were the most relevant to analyse, both from a dosimetric point of view and in terms of the overall knowledge of contamination levels of all foodstuffs that could be deduced from them; these were cow's milk, leafy vegetables and grains, with complete coverage of the entire country. It is regrettable, however, that very few meat analyses were carried out, mainly by the CNEVA. The many other analyses carried out on foodstuffs that are consumed less, or very little, or on processed foods, provide indications but are difficult to use for dosimetric assessments. The radionuclides regularly measured are 131-I, 137-Cs, and 134-Cs. The fact that there are far fewer analysis results (or none at all) for the other radionuclides is explained by their relative abundance in the air and in deposits, and by the relative intensities of their transfer in the food chain. This point will be discussed in section 4.2.

There were a sufficient number of analyses on milk, vegetable, and grain samples to study the spatial variability of the activity concentrations and activity concentrations of 131-I, 137-Cs, and 134-Cs throughout the country. The 2007 study showed that these activity concentrations were fairly homogeneous within 4 vast longitudinal zones, with activities decreasing from east to west in line with the general pattern of the spatial distribution of radioactive deposits (Figure 17).

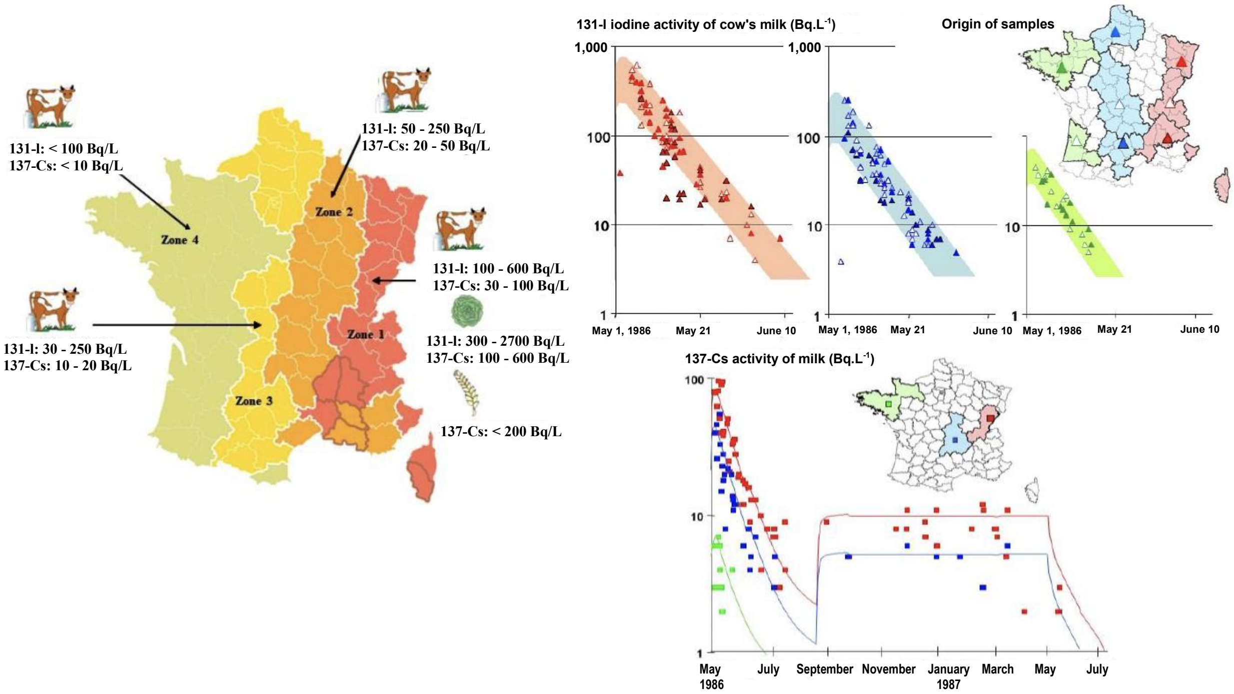


Figure 17: Zones of homogeneous activity concentrations of iodine-131 and caesium-137 observed in 1986 in foodstuffs.

As a result, contamination of foodstuffs was spatially much more homogeneous than radioactive deposit contamination. This is due to the fact that when radioactive deposits are greater because of rainfall (which was the case, see section 3.2), contamination of the food chain does not increase proportionally to these deposits. The leaves of plants drip in the rain and can therefore retain only an increasingly small proportion of the contamination contained in rainwater. Overall, when deposits are ten times greater due to rainfall, contamination of the food chain is only two to three times higher.

IRSN's ASTRAL model (now part of ASNR's SYMBIOSE modelling platform) was adjusted to the available results of milk and leafy vegetable measurements[16] for the four zones of homogeneous contamination of foodstuffs shown in Figure 17. Figure 18 illustrates these adjustments, which were then applied to reconstruct trends over time for the contamination of the 5 radionuclides considered in the 2007 study, i.e. 131-I, 137-Cs+, 134-Cs, 106-Ru+, and 103-Ru+, in different types of food for which few or no measurement results were available. Figure 18 shows examples of reconstructions for 137-Cs+, 131-I, and 106-Ru+.

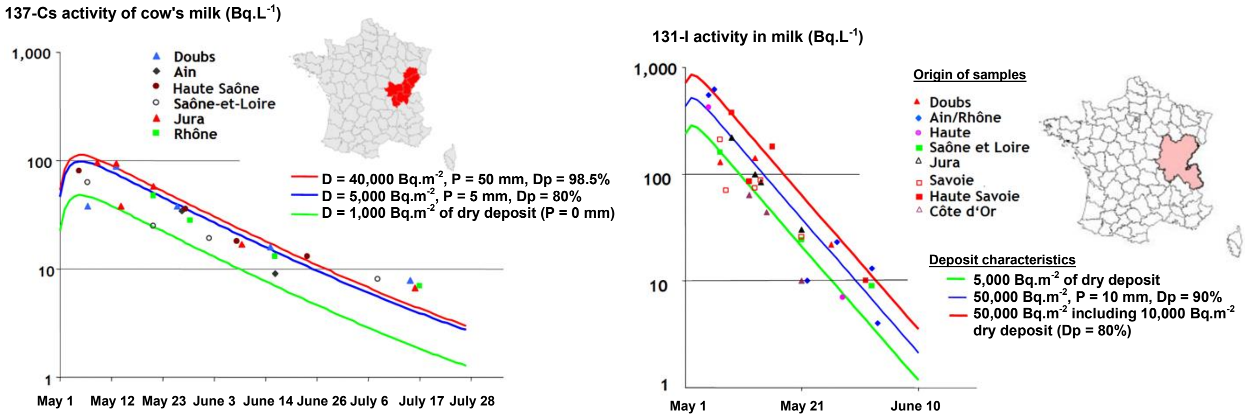


Figure 18: Examples of adjustment of simulations from the SYMBIOSE model (curves) to the131-I and 137-Cs+ activities measured in 1986 in milk (points) according to the initial deposit and the département of origin of the samples

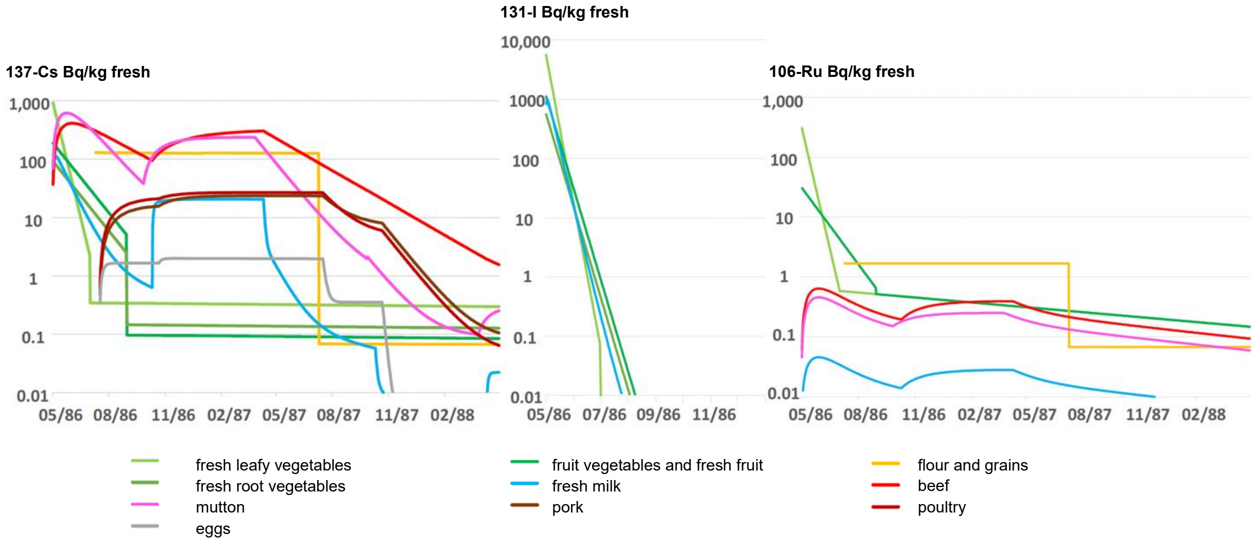


Figure 19: Examples of reconstruction by the SYMBIOSE model of the temporal trend of activity concentrations for radionuclides and foodstuffs for which few measurement results are available.

Using a food ration specific to each age group, it was then possible to estimate the activity intake on a daily basis between 1986 and 1988. For adults, these activity intakes were compared with the 137-Cs body burdens and activity concentrations of this radionuclide in urine, measured in workers at CEA and COGEMA

[16] This adjustment mainly involved determining the value of three parameters: the "effective" deposit retained by the leaves of plants (leafy vegetables and pasture grass), the contribution of the rainfall deposit to the total deposit, and the rainfall height (Renaud et al, 2007 pp 95-104).

(now ORANO) nuclear sites as part of monitoring for occupational exposure, to check consistency[17]. Figure 20, relating to the 137-Cs body burdens measured and estimated for the Tricastin site, illustrates this consistency, which constitutes an element of validation of the doses received by ingestion of foodstuffs following the Chernobyl accident$.

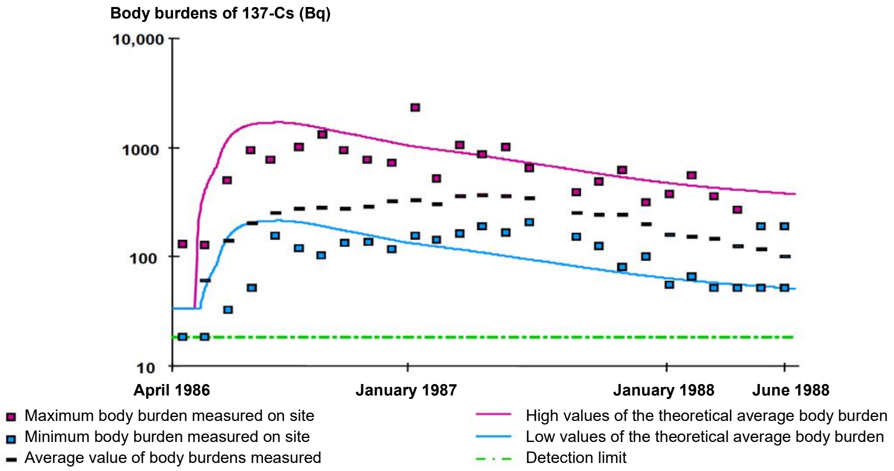


Figure 20: Comparison between estimated and measured body burdens (in Bq/person) for workers at the Tricastin site between 1986 and 1988.

Trends in the annual effective doses received by adults through ingestion between 1989 and 2006 were then estimated based on the effective half-lives of 137-Cs activity reduction, established on the basis of a very limited number of activity concentration time series for lettuces and milk produced in France, and over an equally limited period from 1993 to 2004 (Roussel-Debet, 2006). The overall effective half-life for the decrease of doses received by ingestion was 6 years.

5.2. 2025 update

5.2.1. Radionuclides to be considered

The first element of this update was to determine whether some radionuclides, other than the 5 selected in the 2007 study, may have made a significant contribution to the doses received by ingestion. Table VIII shows the main characteristics of the radionuclides deposited in May 1986 compared with those of 137-Cs+: abundance ratios in the deposits (1st line), ratios of transfer factors to agricultural foodstuffs (translocation[18] to fruit vegetables and root vegetables, 2nd line), to foodstuffs of animal origin (cow's milk and beef, 3rd and 4th lines), ratios of effective dose per unit intake coefficients (DPUI) by ingestion, as well as their radioactive half-lives. This table shows that, apart from 137-Cs+, the main potential contributors to doses by ingestion are those considered in the 2007 study. These are 134-Cs, 131-I, 103-Ru+ and 106-Ru+, either because of their abundance in deposits (in the case of 131-I, and to a lesser extent 103-Ru+) or because of a higher DPUI by ingestion (in the case of 134-Cs). While they are very abundant in deposits, 132-Te+ and 99-Mo are transferred to a lesser extent to foodstuffs and have lower DPUIs than 137-Cs+, mainly because of their short half-lives. The other radionuclides can only make a negligible contribution compared to the above radionuclides, given their characteristics of abundance, transfer to foodstuffs, or radiotoxicity.

[17] These workers, who were not occupationally exposed, are representative of the local population around the sites. The causes of certain discrepancies were sometimes able to be identified. For example, in this graph, the point above the others in February 1987 corresponds to a person who spent a holiday in the former Yugoslavia, eating food that was more contaminated than food produced in France. The whole-body radiation measurement performed on his return demonstrates this.

[18] Translocation refers to the transfer between leaves, which intercept the radioactive deposits, and the edible part of the plant, in this case a fruit or root

Table VIII: Relationship between the main characteristics of the various radionuclides and those of 137-Cs+: abundance in deposits (1st line), transfer factor to agricultural products (translocation[19] to fruit vegetables and root vegetables, 2nd line), to animal products (cow's milk and beef, 3rd and 4th lines) and dose per unit intake coefficient by ingestion (DPUI); radioactive half-life values

Rn i/137-Cs ratios	103-Ru	106-Ru	131-I	134-Cs	136-Cs	137-Cs	140-Ba	141-Ce	144-Ce	95-Zr	99-Mo	132-Te	110m-Ag	125-Sb
Air/deposits	1.9	0.38	7.2	0.51	0.22	1	0.54	0.43	0.04	0.02	2.1	7.7	0.02	0.04
Vegetable transloc.	0.5	0.5	1	1	1	1	0.5	0.5	0.5	1	1	1	0.5	1
Milk transfer	0.03	0.03	1.00	1.00	1.00	1	0.12	0.01	0.01	0.01	0.5	0.1	6.7	0.03
Meat transfer	0.03	0.03	0.03	1.00	1.00	1	0.004	0.02	0.02	0.1	0.2	0.4	0.8	0.03
DPUI	0.06	0.54	1.69	1.46	0.23	1	0.35	0.05	0.40	0.12	0.05	0.29	0.22	0.09
Radioactive half-life	39 d	1.0 year	8.0 d	2.1 years	13 d	30 years	13 d	32.5 d	0.78 year	64 d	2.7 d	3.2 d	0.68 year	2.76 years

5.2.2. Trend over time of doses received by ingestion

For the years 1986 and 1987 and in the case of adults, considering the whole-body validation elements presented above, this study uses the estimates made in 2007 with, in particular, the food rations used at the time. In 1986, the highest dose was estimated for zone 1 in the eastern part of the country: 231 µSv. This dose, which is an average for the zone, is 25% lower than the maximum value of 308 µSv indicated in the 2007 study. The average dose in zone 4 (15 µSv) is practically equal to the minimum dose mentioned in the 2007 study.

For the period 1989-2020, a different approach than that used in 2007 is proposed here. A recent IRSN study calculated the effective doses due to 137-Cs+ for adults over the period 2008-2018 using numerous results from measurements carried out on different types of food sampled during this period (IRSN, 2022).

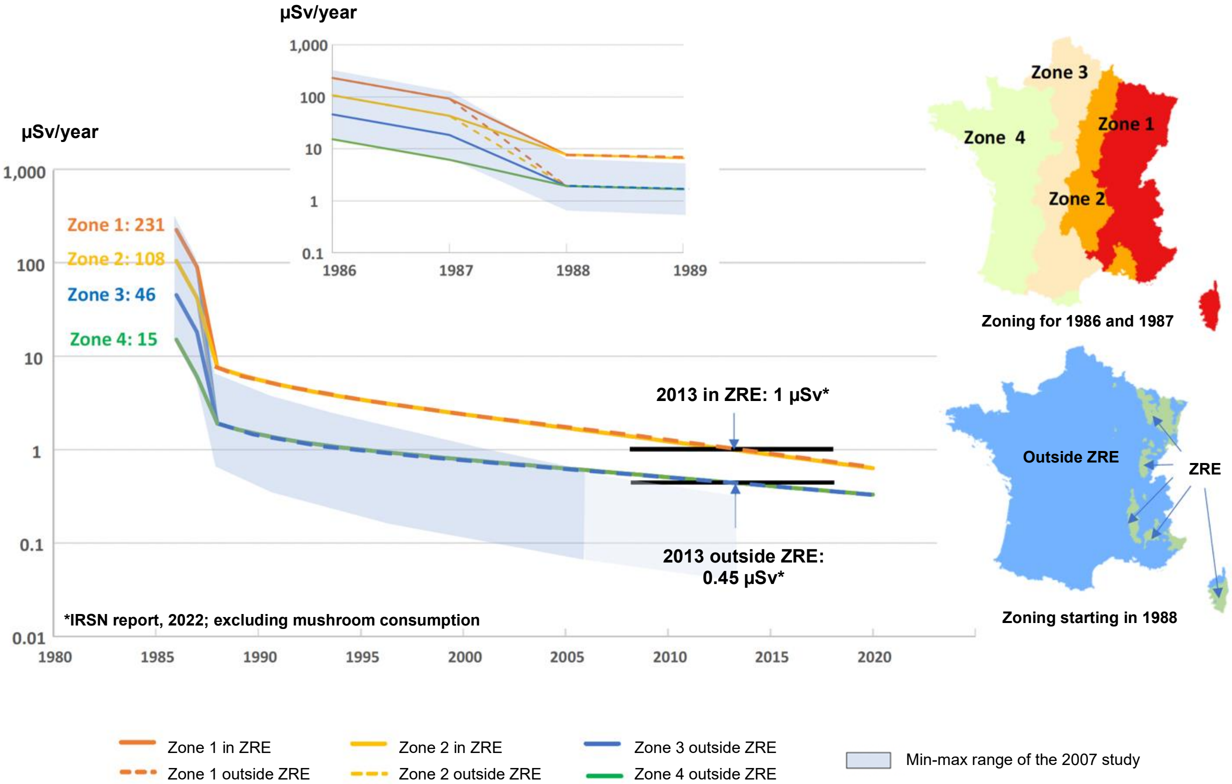


Figure 21: Trends in estimates of effective doses received by ingestion of foodstuffs between 1986 and 2020 in µSv/year and corresponding zones. ZRE: Area of high persistence.

[19] Translocation refers to the transfer between leaves, which intercept the radioactive deposits, and the edible part of the plant, in this case a fruit or root.

We therefore have reference dosimetric estimates based on measurement results for two periods: 1986-1987 and 2008-2018. For adults, the estimated effective doses received by ingestion fell from a range of 2 to 7 µSv in 1988 to a range of 0.5 to 1 µSv in 2013 (the central year of the 2008-2018 period). This decrease is related to the radioactive decay of 134-Cs and 137-Cs+, and to the decrease in root transfer, which is the source of contamination of all foodstuffs. This slow decline has probably been constant, as shown by the time series of activity concentration measurements available and used in the 2007 study.

In the present study, the doses received by ingestion during the 1989-2012 period were estimated by interpolation of the estimated doses for 1988 and 2013 (see Figure 21). This interpolation leads to the application of an effective half-life of dose reduction of 16 years outside the ZREs, and 10.5 years inside the ZREs, which corresponds to "biological" half-lives (excluding radioactive decay) of 35 years and 16 years, respectively, which combines with the 30-year radioactive half-life of 137-Cs+. These same "biological" half-lives were applied to 134-Cs.

As previously indicated, since 1988, 137-Cs+ contamination of foodstuffs has mainly come from root transfer. However, this transfer decreases over time as the bioavailability of caesium in the soil decreases and it is "diluted" by the potassium contained in fertilisers. The chemical properties of fertilisers are similar to those of caesium, and it therefore competes with the isotope for root uptake. Livestock products (milk and meat), whose higher activity concentrations are responsible for the higher doses in ZREs, are less affected by these fertiliser inputs, which could explain the slower decline in doses received by ingestion in ZREs (biological half-life of 16 years) than outside ZREs (biological half-life of 10.5 years).

The half-lives previously determined by interpolation over the period 1989-2013 were then used to estimate doses for the period 2014-2020 by extrapolation. Figure 22 shows that the estimated effective doses from 137-Cs+ ingestion for adults in 2020 are extremely low, less than 1 µSv/year.

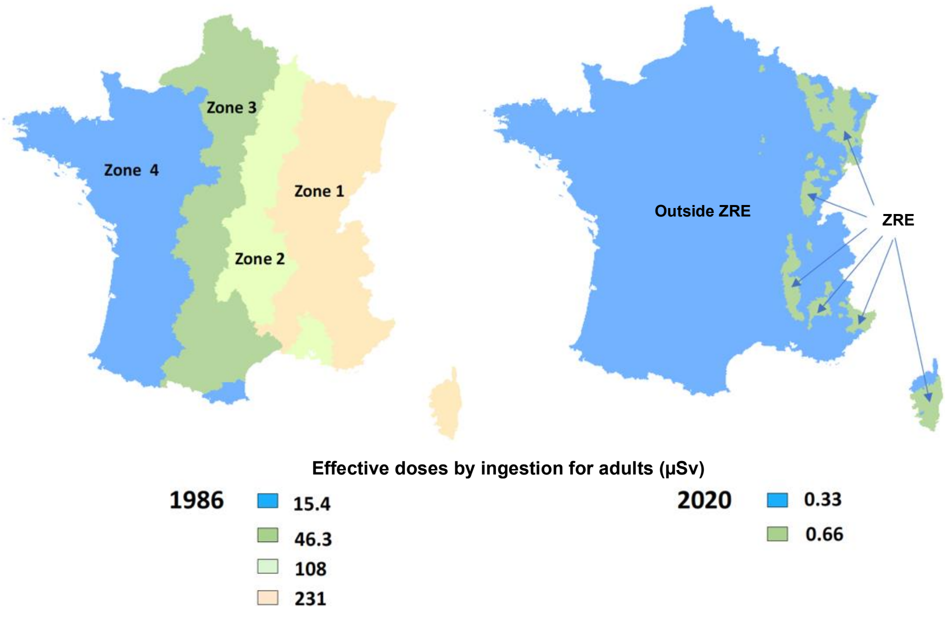


Figure 22: Effective doses (µSv/year) from ingestion of 137-Cs+, estimated for adults in 1986 and 2020

Figure 21 also shows how these dose trends compare with those in the 2007 study. The effective 16-year half-life used for most of the country in this study is 2.7 times longer than the 6-year period used in the 2007 study. As a result, the doses estimated by the 2007 model (blue background in Figure 21) decreased faster than those estimated by interpolation in this study. In contrast, the 10.5-year half-life in the ZRE is only 1.7 times longer than that of 2007. As a result, the dose of 0.3 µSv predicted for 2013 by the 2007 model is only 3 times lower than the dose of 1 µSv used in this study. These discrepancies, both within and outside the ZREs, can be explained in part by the fact that the 2007 model was based on a very limited number of time series, which were also fairly short (from 1993 to 2004). In addition, the measurements of 137-Cs activity concentrations used to estimate the doses received by ingestion of this radionuclide in the study (IRSN, 2022) do not allow 137-Cs from fallout due to nuclear testing to be distinguished from 137-Cs from fallout due to the Chernobyl accident. The estimated doses are therefore due to the intake of caesium from both sources, whereas the doses predicted by the 2007 model result exclusively from the intake of 137-Cs+ from the Chernobyl accident.

For this reason, doses related to the intake of 137-Cs since 1986 are processed, by convention, as part of this study, whatever the origin of this radionuclide. For the sake of consistency, the doses estimated for the study of consequences of nuclear weapons testing in France (Renaud, 2024) are set to zero starting from 1986.

5.2.3. Spatial variability of doses received by ingestion

The 2007 study provides only two 'bounding' estimates of doses received by ingestion in the country: a maximum value for zone 1 and a minimum value for zone 4. To meet the objectives of the CORALE project, it was necessary to assign each person a dose according to his/her place of residence. An average dose received by ingestion was therefore estimated for each of the 4 zones where contamination measured in foodstuffs was homogeneous in 1986 (see section 4.1 and Figure 17). This zoning was used as long as foodstuffs contaminated directly (plant foodstuffs) or indirectly (foodstuffs of animal origin or processed foods) by leaf interception of radioactive deposits were consumed, i.e. until the end of 1987. The persistence of leaf transfer in foodstuffs consumed is related in particular to the time between production of the foodstuff and its consumption. For example, grains harvested in the summer of 1986 were consumed at least until the following harvest in the summer of 1987, or even longer because of the preserved products derived from them (if grains are used to feed pigs, for example).

The recent assessment of the doses received from 137-Cs+ ingestion, based on measurements of the activity concentrations of all types of foodstuffs taken between 2008 and 2018 (IRSN, 2022), also shows that today, across most of the country, the variability of 137-Cs+ activities in foodstuffs is no longer solely related to the initial heterogeneity of radioactive deposits. The activity concentrations measured show close regional averages and very significant variability within the same geographical area, related to the characteristics of the soils, as well as farming and livestock practices. However, the measurements also show that some foodstuffs produced in areas where radioactive deposits were the highest (above 10,000 Bq/m2) can still be significantly different from those produced elsewhere. This is the case for milk, cheese, beef, and mutton, as well as mushrooms and game meat from areas described as "zones of high persistence" (ZRE) for fallout from the Chernobyl accident. People living in these ZREs and who consume some of the food produced there would receive doses twice as high as the national average if we excluded the consumption of wild mushrooms. The 2022 study also showed that the consumption of wild mushrooms, even in very moderate quantities (2.5 kg per year), could account for 50% of the total annual dose from the ingestion of food outside the ZRE and up to 75% within the ZRE. As the consumption of these natural foods varies greatly from one person to another, and is probably very low, or even exceptional for most people, the choice was made for this study to exclude mushrooms from the standard diet and to make personalised estimates using a specific questionnaire (see chapter 6).

The spatial variability of contamination of foodstuffs produced is likely to have changed between the periods 1986-1988 and 2008-2018: At the end of the 1980s, it probably reflected the variability of the soils at the production sites, and the radioactive deposits, before evolving towards the situation observed through measurements taken from 2008 to 2018 (IRSN 2022 report), where variability related to the initial deposits is less significant than the variability of soil characteristics and farming/livestock practices. This variability in the contamination of foodstuffs produced cannot be defined in dosimetric assessments. Even if it could be, the problem of variability in the origin of foods consumed would still remain. The origins of these products also vary greatly depending on the type of food, the region in which they are consumed and, above all, people's eating habits. Given the impossibility of integrating these causes of variability, both from the perspective of radioecological knowledge and from the perspective of knowledge of individual practices, the choice was made for this study to distinguish only two populations for the dosimetric assessments from 1989 to 2020: people living in ZREs and those living in the rest of the country.

5.2.4. Estimation of doses received by children and equivalent doses to organs

In addition to effective doses, only thyroid doses for children in various age groups were estimated in the 2007 study. For the purposes of the CORALE project, effective doses and equivalent doses to the 5 other organs selected for the first stage of this project (brain, lungs, colon, breasts, and prostate) were estimated in the present study. For the sake of consistency with the study on fallout from nuclear weapons testing (Renaud and Vray, 2024), the ratios of quantities of food consumed by "children/adults" recommended by the IAEA (IAEA, 1999) for the different age groups were used for this new study. The dose per unit intake coefficients used are taken from the Journal Officiel de la République (JO, 2003) for effective doses, and proposed by the ICRP (ICRP, 1995) for equivalent doses to the organs.

Figure 23 compares the estimated effective doses for different age groups for people living in zone 1 in the ZRE. In 1986, the highest effective doses were estimated for children aged 1 to 2 (448 µSv) and 3 to 7 years (334 µSv). For these age groups, the effective dose is strongly influenced by high thyroid doses (see Figure 24) linked to the DPUI of 131-I, which is much higher for these age groups than for adults. Because of the very small quantities ingested, the estimated doses for children under 1 year of age remain low. Those of teenagers aged 13 to 17 are almost indistinguishable from those of adults.

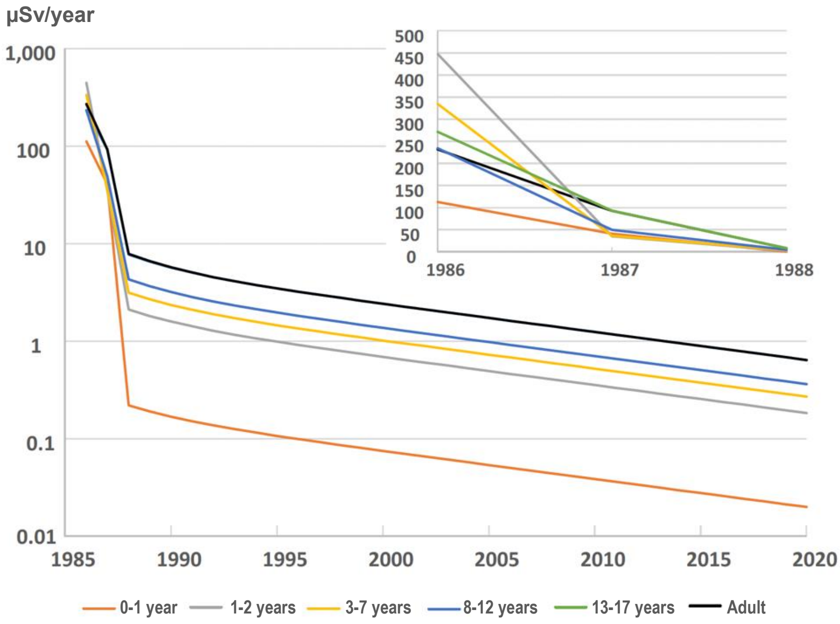


Figure 23: Comparison of trends in estimated effective doses received by ingestion for various age groups (µSv/year)

From 1987 onwards, the effective doses estimated for children are lower than those estimated for adults and adolescents (the doses received by these two age groups being practically equal). As the contribution of doses to the thyroid is lower due to the disappearance of 131-I, the fact that adults and adolescents consume larger quantities of food takes precedence over the decrease in DPUI with age.

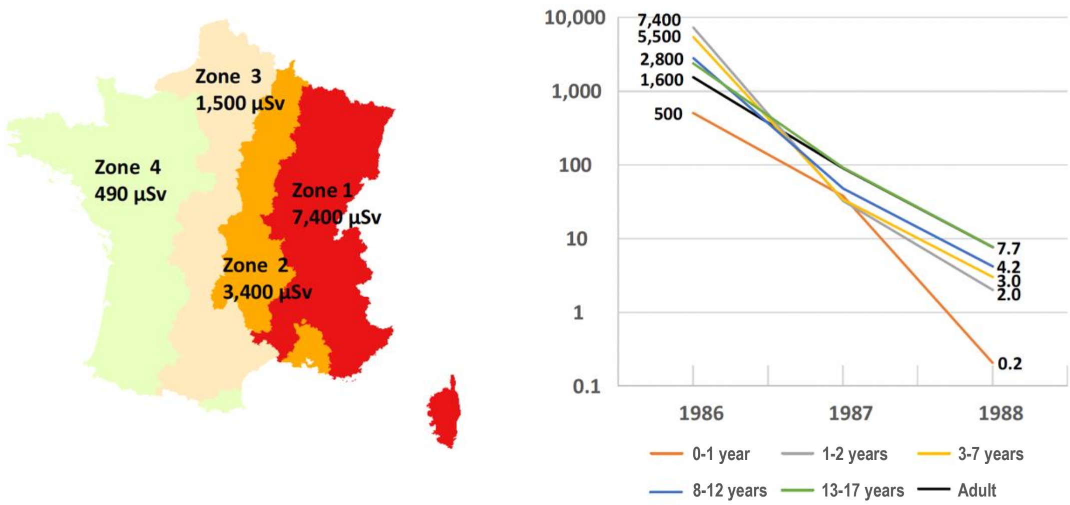


Figure 24: Map of estimated equivalent doses to the thyroid by ingestion (µSv/year) for children aged 1 to 2 in 1986, and comparison of trends between 1986 and 1988 for the various age groups

In 1986, regardless of age group, equivalent doses to the thyroid were always significantly higher than effective doses, often by more than an order of magnitude (compare the maps in Figures 22 and 24 for 1986). These estimates of equivalent doses to the thyroid are close to those estimated in the 2007 study, as shown in Table IX. The differences are mainly due to the use, for this study, of the "child/adult" food ration ratios recommended by the IAEA.

Table IX: Comparison between equivalent doses to the thyroid estimated in this update and those estimated in the 2007 study.

Equivalent doses to the thyroid in µSv	0-1 years	1-2 years	3-7 years	8-12 years	13-17 years	Adult
2007 Study	1,700	9,500	5,700	2,700	-	-
2024 Study	500	7,400	5,500	2,800	2,400	1,600

Since 1987, equivalent doses to the colon have been the highest, particularly for children aged 1 to 2, followed by doses to the prostate for adolescents and adults. But generally speaking, from 1987 onwards, all organ doses are very close to one another, as are effective doses. In 1988, for example, organ doses ranged from 7 µSv for the breasts to 10 µSv for the colon.

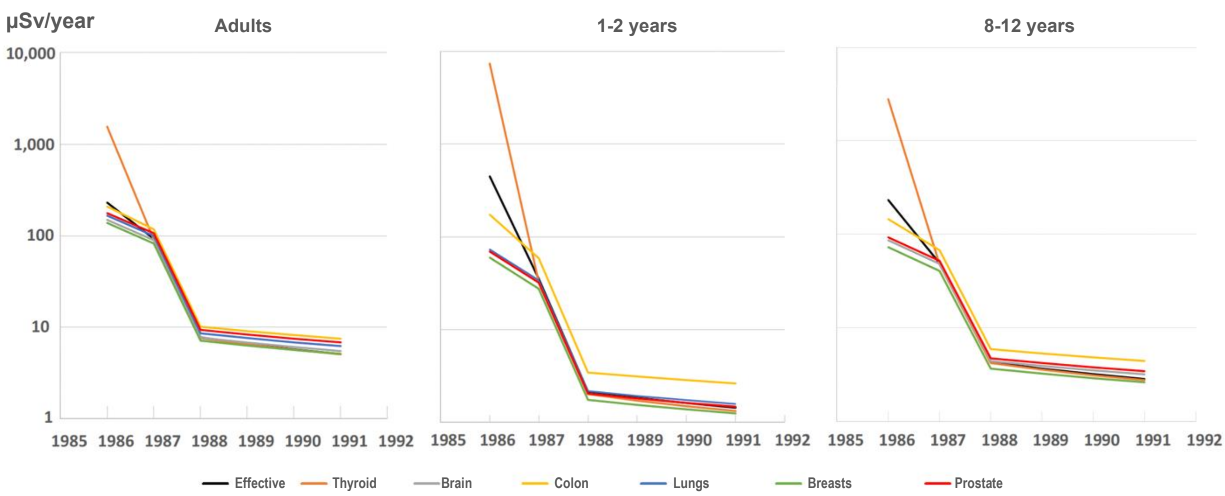


Figure 25: Comparison of trends in equivalent doses to the various organs received by ingestion and estimated for adults, children aged 1 to 2, and children aged 8 to 12 (µSv/year)

6. Trend of total doses, all exposure pathways combined

Figure 26 shows the evolution over time of the total effective dose, all exposure pathways combined, estimated for two representative adults, one working outdoors in a rural environment (graphs on the left), the other working indoors in an urban environment (graphs on the right). Both are in the vicinity of Verdun, where the air contamination measured was highest between May 1 and 5, 1986, but where the deposition of caesium-137 was moderate (6,900 Bq/m^2) due to moderate rainfall. It shows that, even in this part of the country, the contribution from inhalation exposure in 1986 was very small compared with the other two exposure pathways (12% to 15% of the total dose).

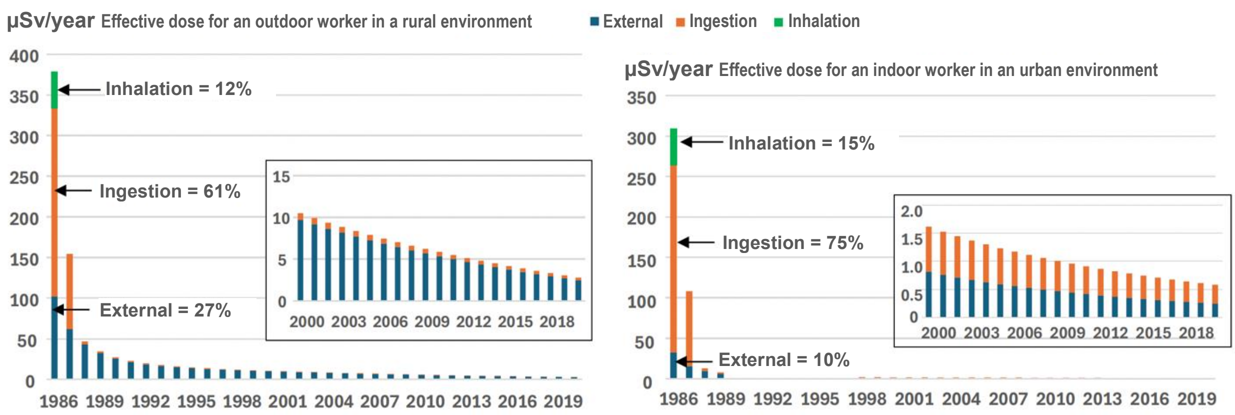


Figure 26: Trend in total effective dose, all exposure pathways combined, estimated for adults, one working outdoors in a rural environment (left graph), the other working indoors in an urban environment (right graph), in the vicinity of Verdun

Total doses and the relative contributions of external and internal exposure by ingestion then vary according to the year, whether the exposed person lives in a rural or urban area, and according to the time spent outdoors. For an adult working outdoors in a rural environment, the external dose became predominant in 1988, accounting for 92% of the total dose of 47 µSv. In 2020, it accounts for 88% of the total dose of 2.8 µSv. In the case of adults working indoors in an urban environment, the external dose also predominated from 1988 onwards (72%), but the situation gradually reversed and, from the early 2000s onwards, it was the ingestion of foodstuffs that became the main contributor due to the more rapid decrease in the external dose. In 2020, the ingestion of foodstuffs accounted for 57% of the total dose, estimated at just 0.6 µSv, i.e. 4.5 times lower than that of adults working outdoors in a rural environment.

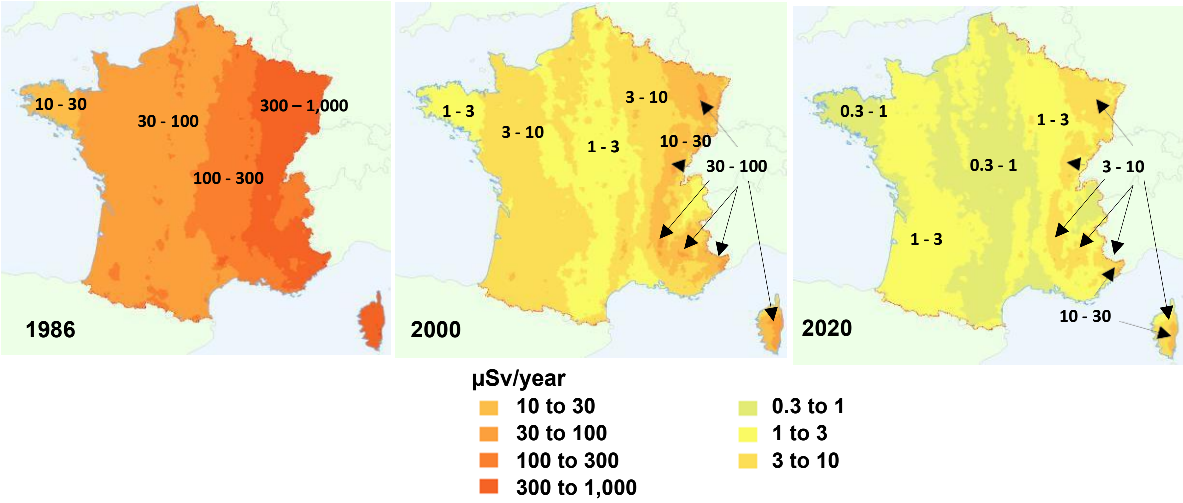
It follows from the above that the highest effective doses received in France after the Chernobyl accident, excluding the consumption of wild mushrooms and game meat (see chapter 6), were received by adults working outdoors in rural areas, where radioactive deposits were greatest. In May 1986, only eight municipalities in France, all on the east coast of Corsica or in the inland region of Nice, received deposits of caesium-137 around 50,000 Bq/m^2. In these municipalities, the total effective dose, estimated at around 1 mSv in 1986, was mainly (75%) due to external exposure to radioactive deposits. This estimate of 1 mSv, representative of the highest effective doses likely to have been reached in France in 1986 as a result of fallout from the Chernobyl accident, is higher than the estimate of 0.75 mSv made in the 2009 study. This difference is mainly due to the fact that, in the 2009 study, the deposit used to represent maximum values was 40,000 Bq/m^2, compared with 50,000 Bq/m^2 in the current study. To a lesser extent, the inclusion of a greater number of radionuclides also contributes to this discrepancy. Since 1986, outside the urban environment, external exposure has remained predominant in the most affected municipalities, accounting for 97% of the total dose in 2020, estimated at 20 µSv. This maximum dose, excluding the consumption of wild mushrooms and game, likely to be reached in France almost 40 years after the Chernobyl accident, is probably overestimated because the Corsican soils and the mountain soils (inland region of Nice) where the radioactive deposits were greatest are known to retain very little caesium-137 due to low clay content or a low level of organic matter.

In 1986, the total effective doses estimated for children aged 1 to 2 years were up to twice those received by adults, due to the extent of exposure by ingestion of foodstuffs, and to a lesser extent by inhalation. Since 1987, effective doses received by children have been at the same level as those received by adults. This is related to the predominance of external exposure in total effective doses. There is, however, one exception: the estimated effective doses for children in an urban environment are 2 to 3 times lower than those for adults working outdoors. In 1986, the relationships between effective doses and equivalent doses to the various organs were dictated by the heterogeneous nature of exposure by ingestion of foodstuffs and, to a lesser extent, exposure by inhalation (mainly related to the affinity of radioactive iodine for the thyroid gland). Since 1988, equivalent doses to the organs have been at the same level as effective doses, due to the predominance of external exposure.

Figure 27 shows maps of estimated effective doses for two adults, one working outdoors in a rural environment (top maps) and the other working indoors in an urban environment. The general appearance of the 2 maps for 1986 is the result of the combined effect of zoning related to exposure by ingestion (see figure 22) and the rain-deposition relationship characteristic of the external dose. For the same year, the areas of eastern France corresponding to the highest doses are more extensive in the case of outdoor workers in rural environments (top), due to the greater contribution of the external dose. In 2000, external exposure determines the general appearance of the two maps, and doses for indoor workers in urban environments (bottom) are much lower because the ambient equivalent dose rate decreases more rapidly in urban areas and due to the protection provided by dwellings. In 2020, the dose due to external exposure in urban environments dropped (bottom map) and the zoning related to exposure by ingestion was predominant, differentiating the areas of high persistence from the rest of the country.

Figure 28 shows maps of estimated equivalent doses to the thyroid for children aged 1 to 2 living in rural environments. In 1986, the general shape of the map was determined by exposure through ingestion, which clearly predominated. In the eastern part of the country, estimates are in excess of 3,000 µSv, with some reaching 7,400 µSv. After 1986, the equivalent doses to the thyroid slumped as a result of the disappearance of iodine-131; they are now only due to the radiation emitted by the other radionuclides distributed throughout the body (after intake) or deposited outside the body (external exposure).

Adults working outdoors in a rural environment



Adults working indoors in an urban environment

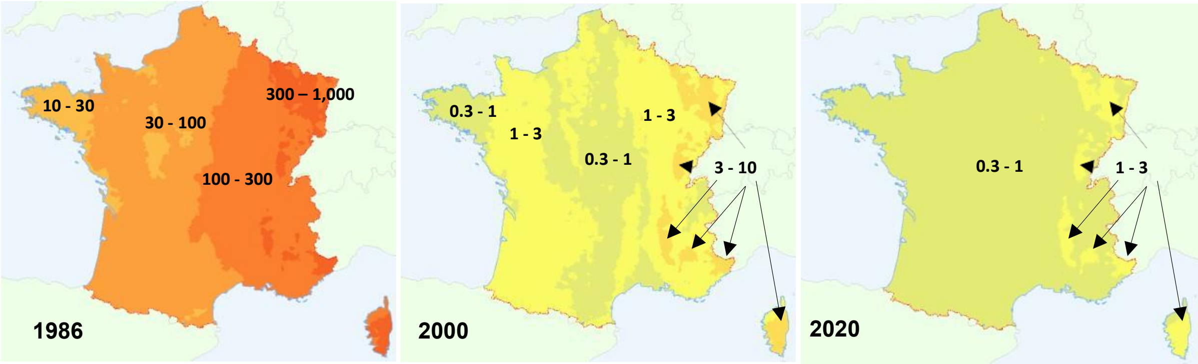


Figure 27: Estimated effective doses for adults working outdoors in a rural environment (top maps) or indoors in an urban environment (bottom maps) in 1986, 2000, and 2020 (in µSv/year)

Equivalent doses to the thyroid for children aged 1 to 2 in a rural environment

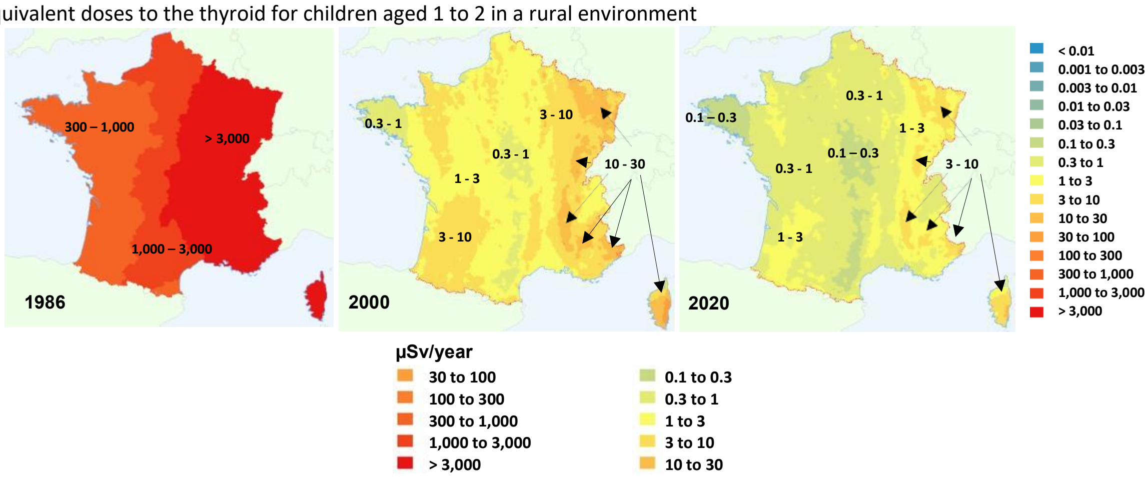


Figure 28: Equivalent doses to the thyroid estimated for children aged 1 to 2 living in a rural environment in 1986, 2000 and 2020 (in µSv/year)

7. Case of consumption of wild mushrooms and game meat

7.1. General information

Measurements carried out in all the countries affected by the fallout from the Chernobyl accident, as well as those carried out in Japan after the Fukushima accident, have shown that forest foods (mushrooms, game meat, and, to a lesser extent, berries) have the particularity of retaining high levels of contamination, particularly in caesium-137, for years or even decades following the radioactive deposits.

In the first few days after the fallout, contamination levels of mushrooms were no higher than those of leafy vegetables or beef, for example. But while caesium-137 levels in foodstuffs from agriculture and livestock farming fell sharply in the months following the deposits, then more slowly but steadily in the following years, those in mushrooms and game remained high (see Figure 28). This is due to the fact that, in a forest environment, caesium-137, which is fixed to organic matter, remains highly bioavailable to plants and therefore to the animals that feed on those plants (Renaud, 2019).

As a result, over the decades, the gap between the caesium-137 contents of forest foods and agricultural and livestock foods has become very wide.

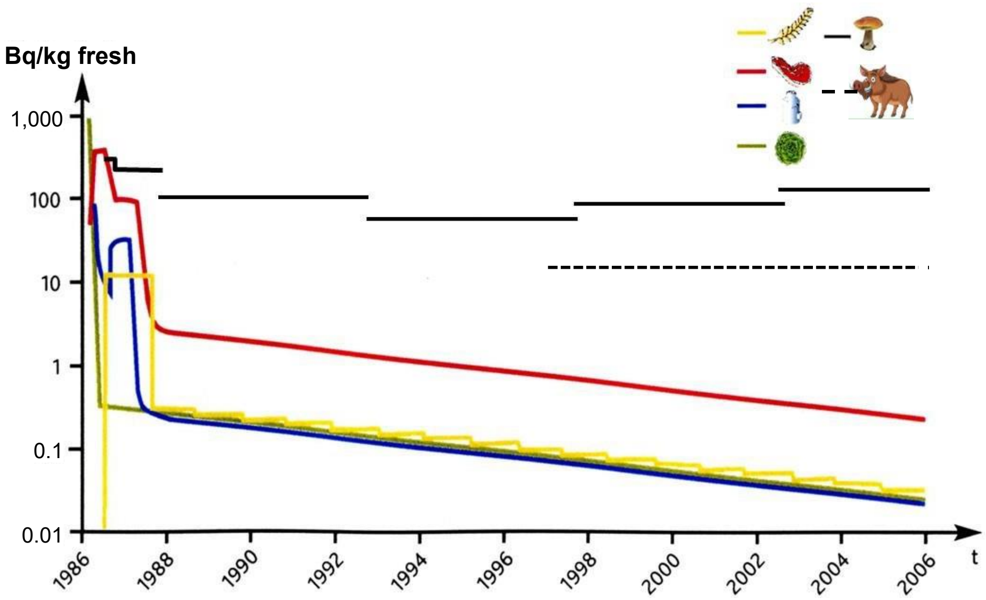


Figure 28: Overall trends in caesium-137 activity concentrations in certain types of French foodstuffs (Bq/kg fresh; Renaud et al., 2007)

7.2. Available data and dosimetric estimates for 2007 and 2022

On the basis of measurements taken during the period of 2008-2018, IRSN estimated the average activity concentrations of agricultural and livestock foodstuffs, produced over most of France, as being between 0.01 and 0.7 Bq/kg fresh, and up to 1.7 Bq/kg fresh for beef produced in ZREs (areas of high persistence; IRSN, 2022). Over the same period, the activity concentrations of mushrooms were estimated at 94 Bq/kg in the ZRE and 16 Bq/kg for the rest of the country. In the case of game meat, these averages were 55 Bq/kg in the ZRE and 2 Bq/kg for the rest of the country.

While these average values for agricultural and livestock foodstuffs hide a variability of up to 2 orders of magnitude, much of which is related to the spatial variability of initial radioactive deposition, those for forest foods are spread over 5 orders of magnitude and the relationship with initial deposits is much less clear, as shown in Figures 29 and 30. These figures show 297 results from measurements of caesium-137 activity concentration in mushrooms and 133 results from game meat (wild boar, roe deer, other deer) acquired by ASNR and its predecessors (SCPRI, OPRI and IRSN) between 1986 and 2021, and for which the precise origin is known (municipality or département, in some cases). Firstly, they confirm the absence of any observable decline in these activity concentrations over the 35 years covered by this period (26 years for game). They also show that while the highest levels of activity (in excess of 100 Bq/kg fresh for mushrooms and 1 Bq/kg for game) all come from eastern France, below these values, foodstuffs with high levels of contamination can come from anywhere in the country. It should also be noted that the highest levels of activity measured in game meat were in wild boar.

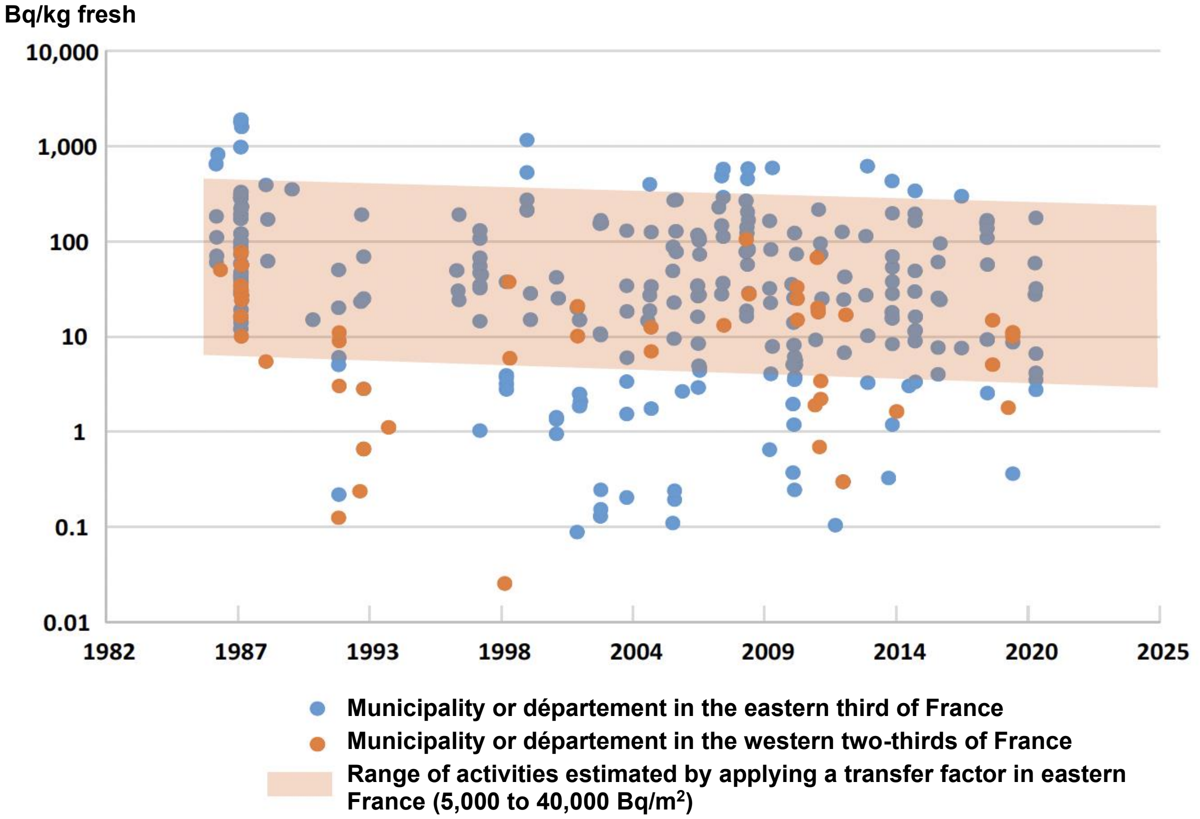


Figure 29: Activity concentrations of caesium-137 (Bq/kg fresh) measured since 1986 by ASNR (and its predecessors IPSN, OPRI and IRSN) in mushrooms collected in mainland France.

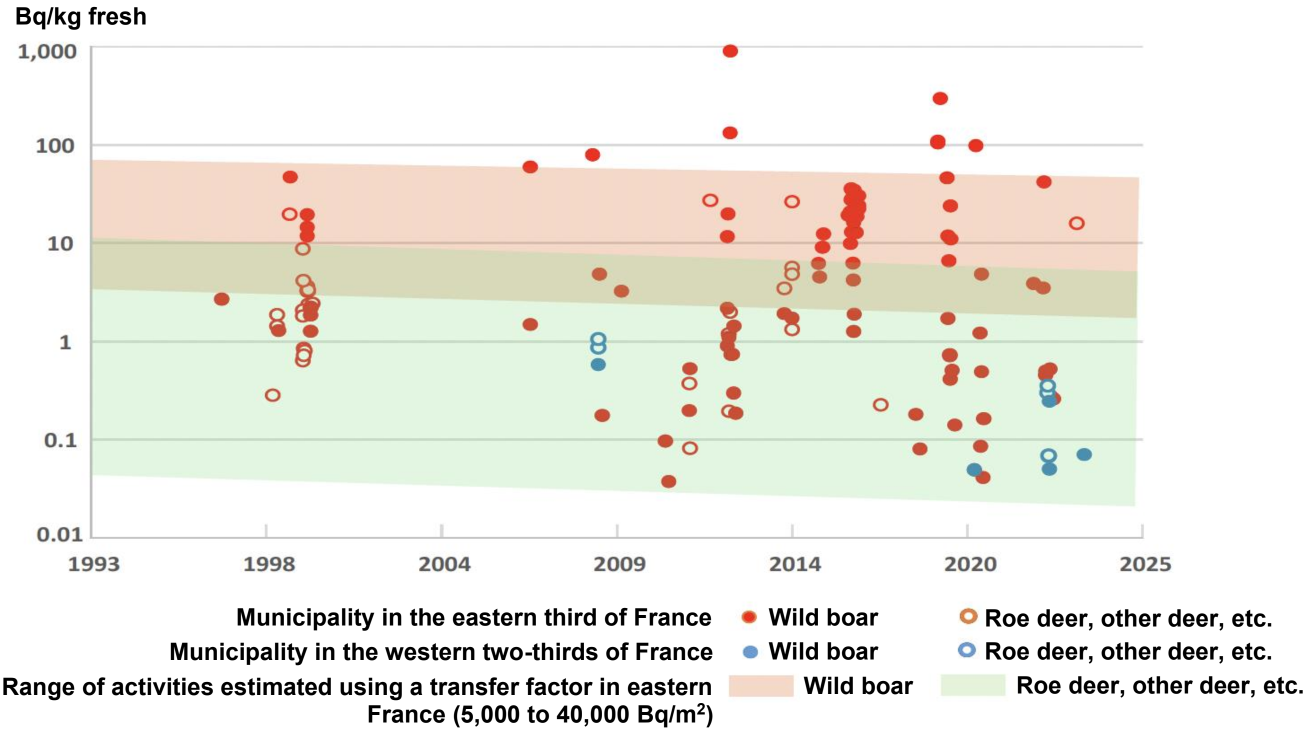


Figure 30: Activity concentrations of caesium-137 (Bq/kg fresh) measured since 1986 by ASNR (and its predecessors IPSN, OPRI and IRSN) in game meat from mainland France.

Some measurement results provide more specific information on these observations at the scale of a few municipalities in eastern France. Figure 31 shows the caesium-137 activity concentrations measured in mushrooms from municipalities, or groups of municipalities, in the Jura, Vosges, Haut-Rhin and Alpes-de-Haute-Provence. For each of these geographical areas, the results are spread over 2 orders of magnitude, including for samples of the same species picked on the same day. For example, three delicious milk cap mushrooms picked on October 5, 2015 in the municipality of Les Rousses in the Jura, exhibited activity concentrations of 9 Bq/kg fresh, 160 Bq/kg fresh and 194 Bq/kg fresh, respectively. In the same municipality, 2 samples of boletus mushrooms picked on October 12, 2020 exhibited activities of 3 and 176 Bq/kg. The same was true for samples of hedgehog mushrooms taken at Hauteroche and Conliège in 2008, exhibiting activities of 83 and 590 Bq/kg fresh, respectively.

Data on game meat comes mainly from the Alpes-Maritimes, Vosges, and Haute-Corse areas. They are presented in Figure 32. Intra-municipal variability is as great as for mushrooms. Two samples of wild boar meat collected on October 21, 2006 in the municipality of Saint-Dié in the Vosges area showed activity concentrations of 1.5 Bq/kg fresh and 59 Bq/kg fresh, respectively. The same was true for the two samples from Hurbache in December 2012 and January 2013: 19.8 and 903 Bq/kg fresh (the latter value standing out very clearly from all the others).

This intra-municipal variability, observed in both mushrooms and game meats, may 'overwhelm' the potential inter-municipal variability related to the initial deposits if it is low. For example, the measurements do not make it possible to distinguish between mushrooms from municipalities where deposits were around 10 to 15 kBq/m^2 (Banon, Redortiers, Les Rousses, Ribeauvillé) and those picked in municipalities that received 25 to 30 kBq/m^2 (Hurbache, Denipaire and Aubure). This intra-municipal variability is probably due to micro-local variations (on a metric or decametric scale) in initial deposits, related to run-off or dripping phenomena from the radioactive rain of the first week of May 1986, or to the subsequent re-concentration phenomena (horizontal migration of soil particles).

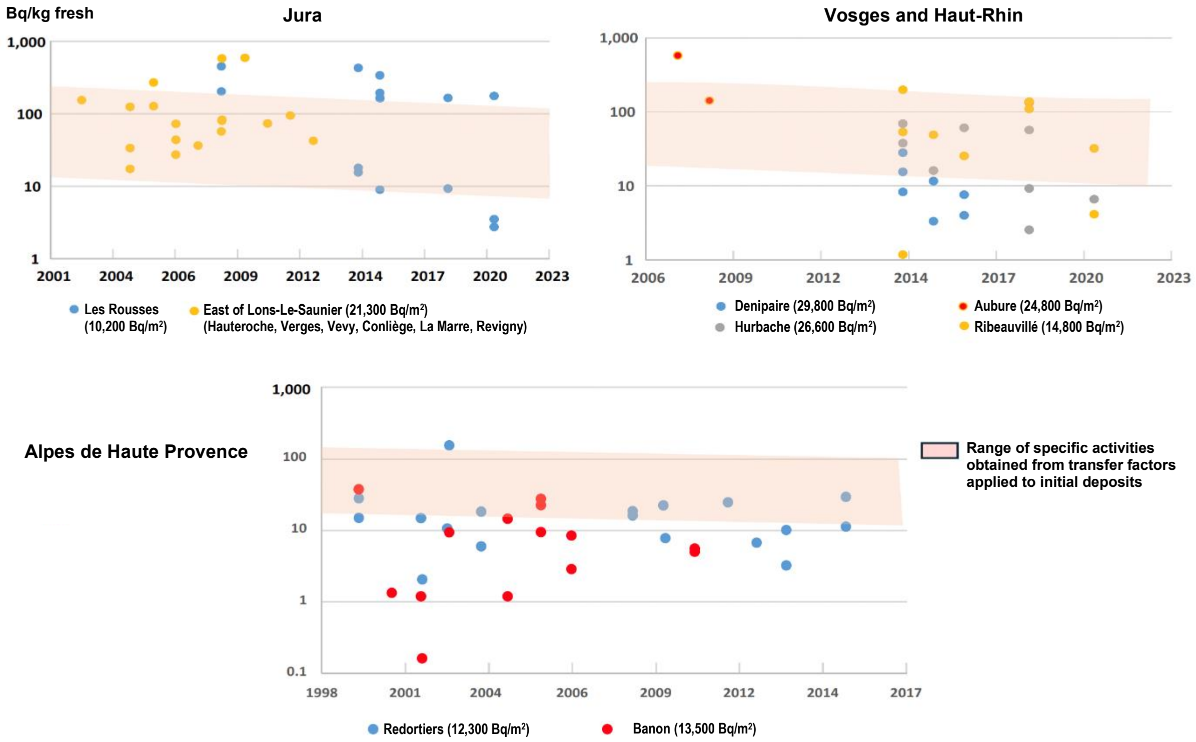


Figure 31: Caesium-137 activity concentrations (Bq/kg fresh) measured by ASNR (and its predecessors IPSN, OPRI and IRSN) in mushrooms collected in a number of municipalities or groups of municipalities in eastern France.

While the relationship between activity concentrations and initial deposits is generally respected for game meat, the case of Haute-Corse is a counter-example that is worth mentioning. The municipalities of Ventiseri and Solaro received the highest radioactive deposits in France after the Chernobyl accident, around 50 kBq/m^2, due to very heavy rainfall. Studies conducted in this area (IRSN, 2015) have shown that low clay content makes these soils unsuitable for retaining caesium-137, so they retained only a very small proportion of the fallout. Between 1986 and 2001, these soils therefore lost up to 85% of their initial radionuclide deposits. As a result, there is very little persistence from fallout due to the Chernobyl accident in foodstuffs. The results of measurements taken on wild boar from this area, ranging from 0.18 to 1.7 Bq/kg fresh, illustrate this point. The soil studies conducted by ASNR show that this is a special case compared with the rest of the country.

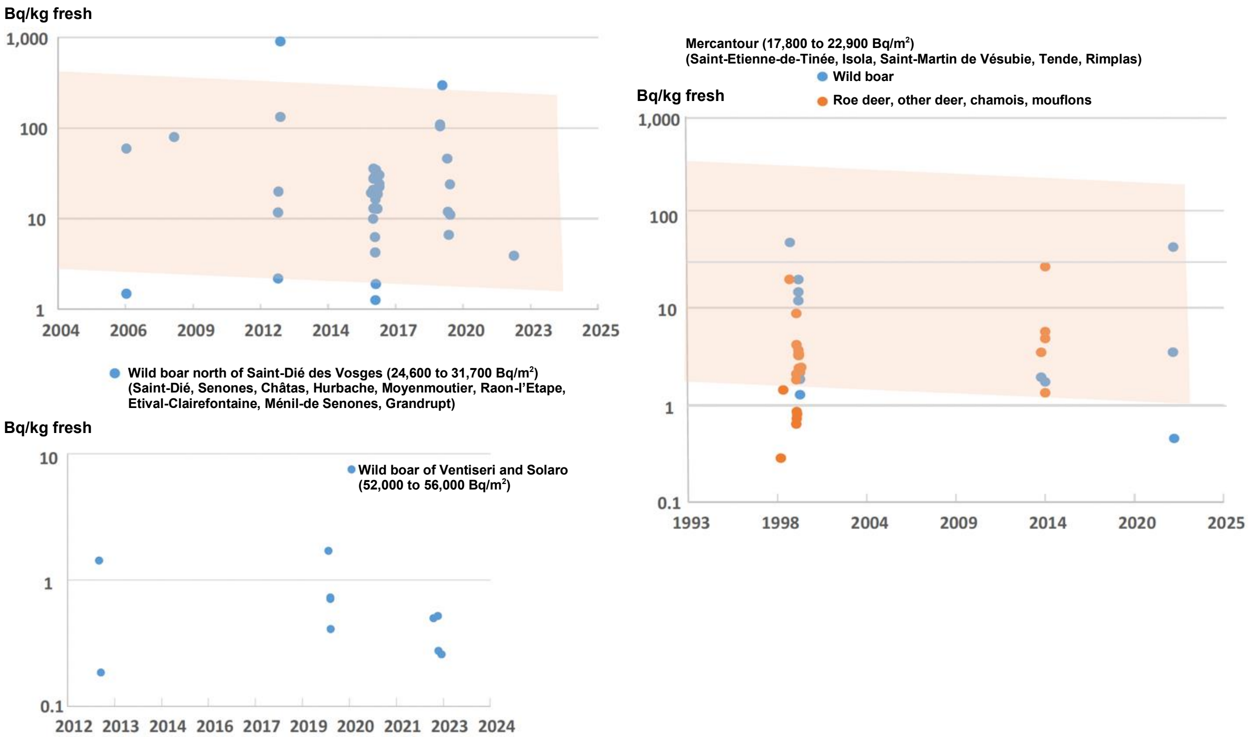


Figure 32: Caesium-137 activity concentrations (Bq/kg fresh) measured by ASNR (and its predecessors IPSN, OPRI and IRSN) in game meat collected in a number of municipalities or groups of municipalities in eastern France.

Thus, the use of a municipal average activity concentration for dosimetric assessment purposes, even based on measurement results, is to be considered with caution. In fact, two people living in the same municipality may be accustomed to procuring mushrooms in places where their activity concentrations are very different, and therefore receive very different doses as a result of their consumption. Moreover, consumption habits for these foods vary greatly. Most people consume very little of them, and a small proportion of the population may consume a lot of them.
Given this twofold observation (the impossibility of identifying an "average" activity concentration on any scale, and the wide variability of consumption practices), ASNR's previous estimates (Renaud *et al*, 2007 and IRSN, 2022) were limited to demonstrative and bounding examples: on the one hand, the demonstration that, 20 to 30 years after the Chernobyl accident, moderate consumption of these foods could be sufficient to double the total dose, all exposure pathways combined; and on the other hand, maximising estimates based on unrealistic scenarios likely to lead to doses of up to 1 mSv/year (2007 study) and 0.57 mSv/year (2022 study).

7.3. 2025 update: possibilities and limits

With regard to doses due to the ingestion of forest foodstuffs, the main objective of this update is to refine previous estimates: by taking into consideration more realistic consumption habits, by spatializing activity concentrations through their connection to initial radioactive deposits and the study of their evolution since 1986.

ASNR recently carried out a study on caesium transfer factors from soil to mushrooms and game meat, expressed in Bq/kg of fresh or dry mushroom per Bq/m^2 of soil (m2/kg) (IRSN, 2024-102). This study, based partly on a literature review of European publications and partly on the results of measurements taken in France, confirms the previous findings: transfer factors to mushrooms have been stable over time since 1986 and their variability, over 3 to 4 orders of magnitude, partly explains the variability of mushroom concentrations. Table X presents the distribution parameters (median, 25 and 75 percentiles) of the transfer factors obtained for symbiotic mushrooms (which represent the majority of mushroom species consumed in France) and meat from wild boar and deer. The difference between the values of the 25 and 75 percentiles is 1 to 2 orders of magnitude and reflects some of the variability previously observed in the measurements, discarding the extreme values.

Table X: Parameters for the distribution of soil-forest food transfer factors, obtained for France (IRSN, 2024)

m^2/kg fresh	Median	25th percentile	75th percentile
Symbiotic mushrooms	0.0045	0.0015	0.012
Wild boar	0.0007	0.00008	0.018
Deer	0.0002	0.0001	0.0003

The theoretical ranges of caesium-137 activity concentrations in mushrooms and game meat obtained by applying these transfer factors to the range of deposits in eastern France (from 5 kBq/m^2 to 40 kBq/m^2) are shown by a pink band in Figures 29 to 32. At this scale, the application of transfer factors provides a fairly satisfactory account of the range of variability in the measurement results.

In Figures 29 to 32, the pink band represents the application of the median transfer factors applied to the theoretical deposits on the municipalities or groups of municipalities provided in brackets, taken from Figure 8 of this report, corrected for radioactive decay. The results of this application fall within the range of variability of the measurement results, however, they only represent a small part of it (thickness of the pink band).

Thus, despite the variability of the estimates resulting from this modelling, it would appear that it is just as reliable, in the light of the observations made in the first part of this chapter, to base a dosimetric estimate on results obtained by applying a transfer factor to an estimate of deposition on a municipal scale, as on an average of measurement results acquired in this municipality. In addition, as there is a very limited number of municipalities for which data is available, the application of a transfer factor provides an estimate for the whole country and a relationship between the activity concentrations of foodstuffs and the initial deposits at this scale. Lastly, applying the transfer factors corresponding to the 25th and 75th percentiles, allows elimination of the extreme values from the highest and lowest activity concentrations measured on samples, of which recurrent or even continuous consumption is highly unlikely.

In the questionnaire sent to members of the "Constances" cohort eligible for the CORALE study, they were asked whether, at different times in their lives (before the age of 15, between the ages of 16 and 24, and after the age of 25), they had eaten wild mushrooms and/or game meat produced around their place of residence: (i) never or less than once a month, (ii) once or twice a month, (iii) once or twice a week or (iv) more than twice a week. It is considered that a meal consists of 200 g of meat and 200 g of mushrooms. One meal a month corresponds to an annual consumption of 2.4 kg of game and 2.4 kg of mushrooms; 2 meals a week corresponds to an annual consumption of 21 kg of each. The latter consumption is significant, representing almost a third of the average annual consumption of meat from livestock (all types of meat combined) or of grains.

Figures 33 and 34 present the dosimetric consequences of the consumption of forest foods[20] using the example of Figure 22 for the area around Verdun (see chapter 5), where caesium-137 deposits in May 1986 were moderate (6,900 Bq/m^2). Figure 33 (left graph) shows that for an adult working outdoors in a rural environment, the added dose resulting from just 2 meals a month of forest foods is low, even negligible, particularly in the first years, compared with doses from other exposure pathways (external exposure and ingestion of food from agriculture and livestock farming). On the other hand, the consumption of forest foods twice a week may induce a higher dose than those linked to other exposure pathways from 1991 onwards. In 2020, this practice even led to a 5-fold increase in the total dose due to fallout from the Chernobyl accident, increasing from 3 µSv/year to 15 µSv/year.

[20] These doses include caesium-137 and 134.

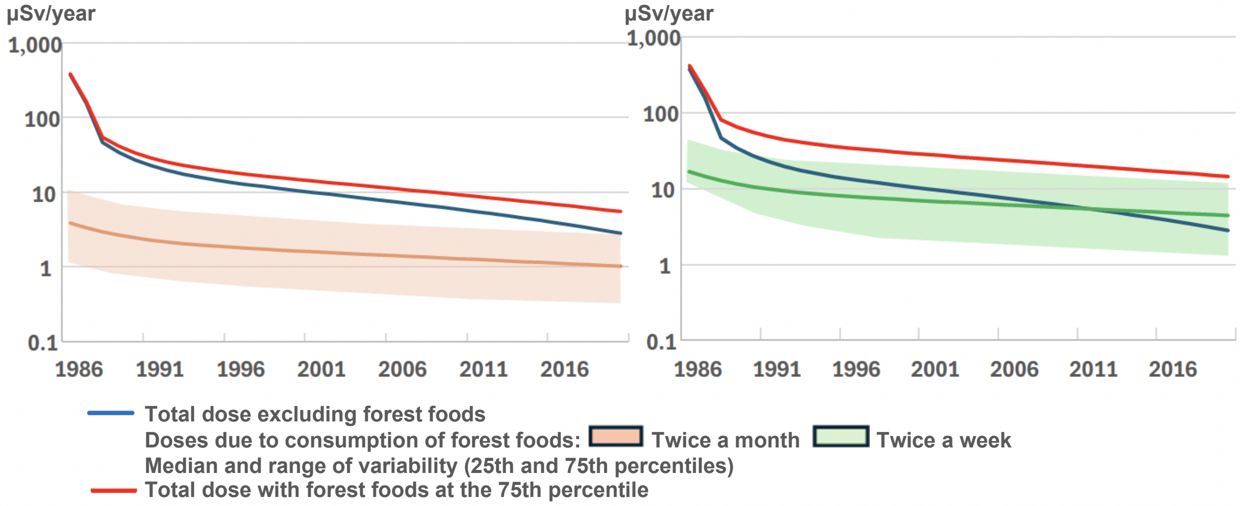


Figure 33: Comparison of trends in potential doses from the consumption of forest foods and doses from other exposure pathways, estimated for a moderate deposit of 6,900 Bq/m2 in the case of an adult working outdoors in a rural environment (µSv/year)

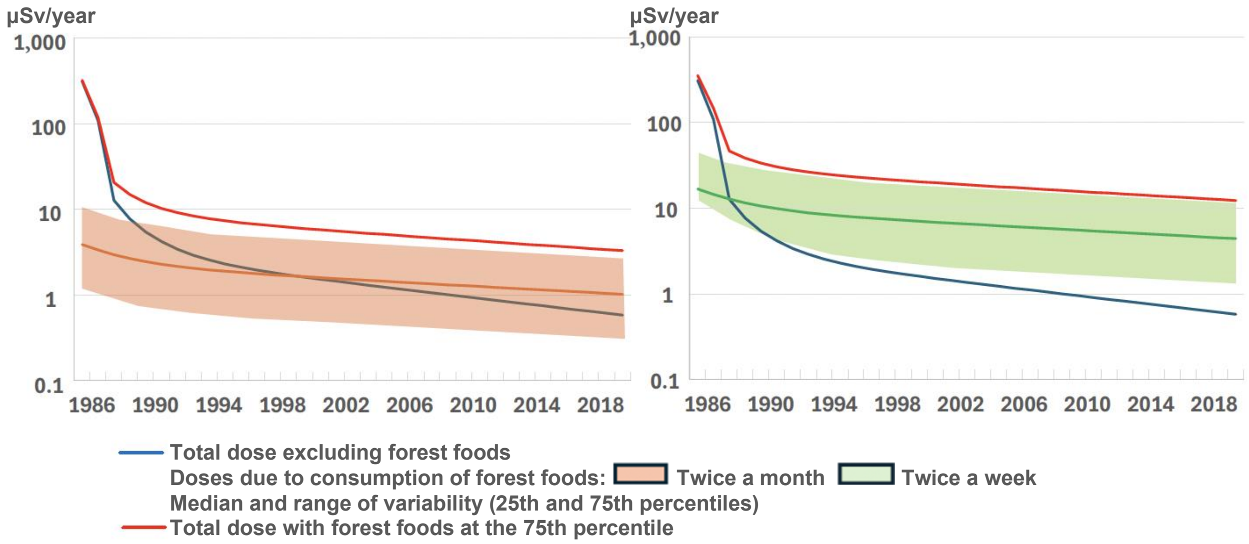


Figure 34: Comparison of trends in potential doses from the consumption of forest foods and doses from other exposure pathways, estimated for a moderate deposit of 6,900 Bq/m^2 in the case of an adult working indoors in an urban environment (µSv/year)

In the case of an adult working indoors in an urban environment (Figure 34), the dose due to frequent consumption of forest foods became clearly predominant from 1988 onwards. In 2020, it may be up to 20 times higher than that associated with other exposure pathways (12 µSv compared with 0.6 µSv).

This level of consumption of forest foods by a person living in one of the areas most affected by fallout from the Chernobyl accident (50 kBq/m^2) would lead to a dose of 85 µSv in 2020. This estimate is more realistic than the 580 µSv (6 times higher) estimated using a deliberately extreme but implausible scenario (IRSN, 2022), where it was assumed that all the game meat and mushrooms consumed throughout the year would have had the highest activity concentrations measured, in particular the value of 903 Bq/kg fresh measured in wild boar meat in January 2013, which stood out very clearly from the other measurement results (see Figure 30).

8. Conclusion

This document provides details of all the methodological elements used to obtain estimates of effective doses and equivalent doses for a selection of six organs, received by adults and children in the French population following fallout from the Chernobyl accident, by municipality and by year, between 1986 and 2020, to meet the objectives of the CORALE project (Sauce J. *et al*. 2024) conducted in collaboration between ASNR and the UMS011.

These dose calculations are essentially based on activity concentration values for the main radionuclides of the fallout, measured in the air, soil, and in foodstuffs by ASNR and its predecessors (SCPRI, OPRI, IPSN, and then IRSN), with minimal use of modelling. Estimates of doses received by inhalation are based on caesium-137 activity concentrations measured daily in May and June 1986 in atmospheric aerosols sampled at 36 stations across the country, and on isotopic activity ratios for around 15 other radionuclides, also measured in the air at some of these stations. Similarly, the doses received by ingestion were calculated on the basis of the activity concentrations of the 5 main radionuclides that contributed to them, measured in foodstuffs produced from 1986 to 1989, supplemented by modelling results that were adjusted using these same measurements, and then on the basis of activity concentrations measured from 2008 to 2018. Between these two periods, doses were estimated by interpolation. Finally, the doses associated with external exposure to radiation from the 15 radionuclides measured in the air and deposited on soil and surfaces, were estimated for 1986 and then for 2008-2018 using surface activity measurements or direct ambient radiation measurements (equivalent dose rates). The kinetics of the reduction in equivalent dose rates following radioactive deposits over the first 8 years in rural areas and the first 15 years in urban areas were adjusted to those observed in Japan after the Fukushima accident.

In 1986, the effective doses due to fallout from the Chernobyl accident were estimated at between 10 µSv in Bretagne and a few hundred µSv in the areas of eastern France where radioactive deposits were highest. They were as high as 1,000 µSv (1 mSv) in the eight municipalities on the east coast of Corsica and in the inland region of Nice, where caesium-137 deposits were around 50,000 Bq/m^2 following very heavy rainfall between May 1 and 5, 1986. That year, for most of the country, these doses resulted mainly from the ingestion of contaminated foodstuffs. However, in municipalities where radioactive deposits exceeded 20,000 Bq/m^2, external exposure was the main contributor to the total dose for that same year, particularly for adults who spent a lot of time outdoors. In all cases, the contribution from inhalation exposure was low and did not exceed 15% in north-east France, where the highest airborne activities were measured.

In 1987, doses were 2 to 3 times lower due to the almost total disappearance of the contribution of inhalation and, above all, the significant decrease in the dose due to consumption of foodstuffs. Since 1988, annual effective doses have fallen steadily, with varying contributions from external exposure and ingestion, depending on place of residence and age. In 2000, annual effective doses were estimated between a few µSv in Bretagne and a few tens of µSv in the most affected areas in the eastern part of the country.

In 2020, the average effective dose due to fallout from this accident for an adult living in an urban environment[21], working indoors, and not eating wild mushrooms or game meat, was around 1 µSv/year (from less than 1 µSv/year up to several µSv per year). This scenario undoubtedly represents the majority of the population. Generally speaking, the estimated effective doses for children are at the same level. For people working outdoors in rural areas most affected by fallout from the Chernobyl accident, doses could reach up to 10 µSv, or even 20 µSv in municipalities in the eastern part of the country where radioactive deposits in May 1986 were highest. However, these higher values can only be achieved if the person has spent several hours a day on undisturbed surfaces (which have never been ploughed or covered over, etc.) since 1986. However, such areas are now often limited to natural or wooded areas.

Unlike foodstuffs produced through agriculture or livestock farming, with activity concentrations, and therefore with consumption-related doses that have been steadily decreasing since 1986, caesium-137 contamination in forest foods, mushrooms, and game meat has remained at a high level to this day. This contamination is also much more variable than for other foodstuffs, even on a municipality scale. As a

[21] People are considered to live in an urban environment if their surroundings are mainly made up of artificial surfaces (roads, buildings, etc.); in addition to cities, they may live in villages or hamlets.

result, even occasional consumption of these foods can lead to widely different and significant doses. For people who consume them regularly, the associated effective dose may have exceeded those due to other exposure pathways as early as the beginning of the 1990s. In the worst-affected municipalities of eastern France, the level may still be several dozen µSv.

In 1986, the equivalent doses to the thyroid were much higher than the effective doses and they were age-dependent. The highest estimates, around 7 mSv, were for children aged 2 to 7. They were due almost exclusively to intake of iodine-131 via the ingestion of foodstuffs, with contributions from inhalation or other radionuclides being very low. Equivalent doses to other organs are very close and often at the same level as effective doses, with the exception of equivalent doses to the colon which, for children aged 1 to 12, can be up to twice as high as effective doses.

These doses can be compared with those resulting from fallout from atmospheric testing of nuclear weapons in mainland France, recently estimated by ASNR (Renaud and Vray, 2024).

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