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Introduction to the Radioecology of Forest Ecosystems and Survey of Radioactive Contamination in Food Products from Forests

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SUMMARY

The first part of this report consists in a review of the various factors determining the contamination of forest ecosystems after an accident such as the one that occurred in Chernobyl. The following factors are successively analysed: the contamination source term (Chernobyl accident and atmospheric nuclear tests), the disposal of the contamination by wind, the dry and wet deposition, the influence of the forest cover on this fallout (changes in spatial and temporal distribution, modified concentrations of radionuclides in pluvial washoff), the behaviour of radionuclides in soil and humus (absorption, percolation, biological availability) as well as their uptake in plants and animals (variability of the contamination according to the location in the forest stand and to the physiology of the considered species, e.g. accumulating species, ...) and, finally, the factors related to the distribution of radionuclides in the various tissues of plants and animals.

The second part encompasses a survey of the contamination levels observed in the main plants and animals growing in forest ecosystems and likely to be consumed as food by man. Three chapters deal with higher plants (mainly with eatable fruits such as blueberry, blackberry, wild strawberry, ...), game and wild fungi. For the main species, the following data are provided: latin name, common names in several European languages, natural area in Europe, main uses (of plants only), as well as the diet of game species. A survey of the observed contamination levels is proposed thereafter, based both on literature data and on unpublished reports. It is followed by some comments and conclusions.

The third part of this report presents a set of preventive measures aimed at reducing doses ingested by the public when consuming products from forest ecosystems after a radioactive contamination: harvest of species which do not accumulate contamination, harvest on types of soil in which the biological availability of radionuclides is low, preservation of food products before consumption, washing, peeling, boiling, steeping in water or in a salt solution, ...

The fourth part is a quick presentation of the evolution of the contamination levels in forest ecosystems and, among others, of the plants and animals living there. After a usually rapid phase of increase, starting from the deposition, the contamination level reaches a maximum and then starts to decrease rather markedly. This phase of quick evolution corresponds with the period during which the distribution of the radionuclides among the different compartments in the ecosystem evolves and then stabilises in a state of dynamic equilibrium (turnover). As soon as this equilibrium state is reached, the contamination decrease becomes very slow, probably not much faster than radiological decay.

A bibliographical list of 366 references is given at the end of the report.

I. CONTAMINATION FACTORS

Radioecologists often have difficulty in studying natural ecosystems because of their diversity and complexity. In order to obtain a correct idea of the phenomena occurring there, researchers must know the mechanisms determining the natural equilibria and thus regulating the behaviour of radionuclides.

In this chapter we shall analyse the influence of the various factors successively affecting radionuclides as they move from their point of release through the environment and then enter the tissues of biota living in natural ecosystems, in particular plants and animals living in a forest environment and which are likely to be consumed by humans.

Factor	Type of factor	Influence on ...
Source term	Technological	- Amount, type and chemical species of radioisotopes produced - Duration and kind of emission (altitude reached by emission, etc.)
Winds (speed and direction at different altitudes)	Meteorological	Dispersion of contaminated air masses
Air turbulence	Meteorological	Intensity of dry deposition
Rain (duration & intensity	Meteorological	Intensity of wet deposition
Type and density of plant cover	Biological (phytosociological)	Redistribution of deposition (in time and space) by the canopy
Type of soil (type of humus and profile, chemical composition and grain size of horizons)	Pedological	Rate of percolation and biological availability of radionuclides in the various horizons
Type of fungal species : - position of mycelium in soil - accumulative capacity	Biological (physiological)	 Amounts of radionuclides available Amounts of radionuclides absorbed

Table 1 : Overview of the various factors determining the contamination transferred to biota (here a wild fungus) following a nuclear accident such as the one at Chernobyl.

I.1. SOURCE TERM

The first factor to be taken into account is clearly the source term, i.e. the quantity and nature of radioisotopes emitted as well as their chemical species. The site and type of release (prolonged or brief emission, escape or explosion at a power plant; atmospheric, undersea or underground nuclear test, etc.) are also important. In our particular case, the radioisotopes came from two sources - the accident at Chernobyl and atmospheric nuclear tests.

I.1.1. The Chernobyl accident

On 26 April 1986 at 0123 hours reactor No 4 (of the RMBK-1000 type) exploded at the Chernobyl nuclear power station in the Ukraine. This accident was due to a "power excursion" which caused the instantaneous vaporisation of primary circuit water, which ruptured the containment, followed by a second explosion caused by exothermic reactions involving an explosive mix consisting mainly of hydrogen, carbon monoxide and air. The explosions and subsequent fire badly damaged the reactor itself and the building in which it was located. Several detailed descriptions of the accident have been published, as well as an analysis of the factors causing it (see inter alia : Ahearne 1987, Kress et al. 1987, Mathieu 1986, USSR State Committee on the Utilisation of Atomic Energy 1986).

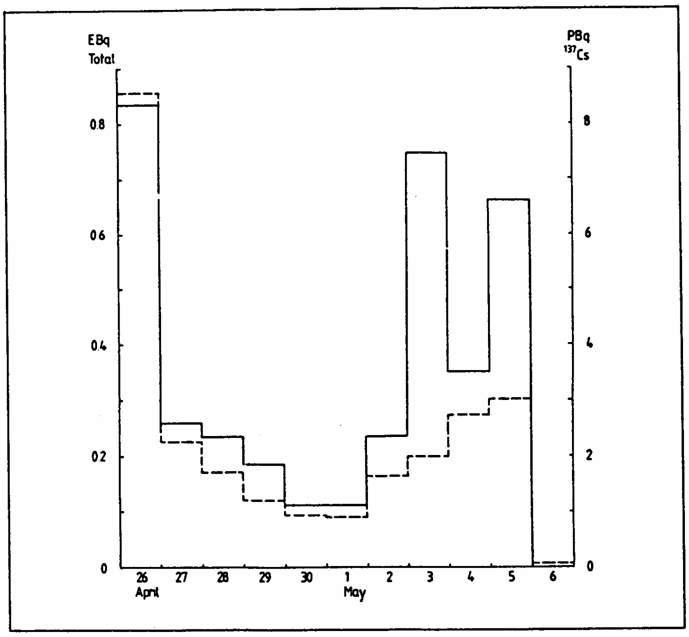

Figure 1 : Daily emissions from Chernobyl (after Persson et al. 1987); the broken line indicates the total activity excluding inert gases (left-hand scale) while the solid line indicates the emissions of ^{137}Cs (right-hand scale); 1 EBQ = 10^{18} Bq, 1 PBq = 10^{15} Bq.

Large amounts of radioactive substances were emitted into the atmosphere during the first 10 days after the accident, and this at levels which varied in time (Ahearne 1987, Norman & Dickson 1986, Persson et al. 1987), as Figure 1 shows. The total quantity released is estimated at 50 000 000 curies of noble gases (especially xenon, half-life = 9 hours) and 50 000 000 curies of other radionuclides; this is equivalent to a total of 3.7 x 10^{18} Bq, i.e. a quantity of radionuclides in which 3.7 x 10^{18} atoms undergo nuclear fission every second to produce radiation. The amounts of ^{137}Cs released are estimated differently by different authors and vary from 37 to 100 PBq (Gudiksen et al. 1989).

The radioactive isotopes emitted were of very different types. The literature studied (Ballestra et al. 1987, CEN-SCK [= Centre for Nuclear Energy Studies, Mol] 1986, Clarke 1986, Devell et al. 1986, Gudiksen et al. 1989, Horyna et al. 1989, Liljenzin et al. 1988, Osuch et al. 1989, Persson et al. 1987, Pienkowski et al. 1987, Rantavaara 1987, Reisch 1987, Schelenz & Abdel-Rassoul 1986, Zarnowiecki 1988) mentions 52 different radioisotopes. Those detected in the fallout are listed in Table 2. After Gudiksen et al. (1989) the following radionuclides (emitted but not observed in the fallout) should be added : ^{85}Kr ≥ 18 PBq, ^{91}Y = 7.2 PBq, ^{127m}Te = 3.7 PBq, ^{131m}Te = 6.7 PBq, ^{133}Xe ≥ 4 200 PBq, ^{133m}Xe ≥ 86 PBq and ^{147}Pm = 0.9 PBq.

Of course, not all these isotopes are dangerous to the same degree : Table 2 shows that most of the isotopes produced during the accident had totally or almost totally disappeared by the end of 1989 due to their fission. Although they might have been a cause of some concern during the weeks following the accident (^{131}I), they no longer play a role nowadays. Of the isotopes still existing today due to their slow rate of disintegration, most are present in only small quantities and can likewise be ignored. Only ^{134}Cs and ^{137}Cs still need to be monitored. And perhaps one should add the beta-emitter ^{90}Sr, of which only a fairly small amount was deposited but whose toxicity is high because - like Ca (of which it is the homologue) - it is incorporated into bone.

I.1.2. Atmospheric nuclear tests

The Chernobyl accident received broad media coverage (Otway et al. 1987) and greatly agitated public opinion (Ahearne 1987). Nevertheless, it appears that for most of the EEC's territory the resulting contamination was much lower than that caused by nuclear weapons testing in the early Sixties (Ward et al. 1989, Gudiksen et al. 1989). By way of example : these latter authors believe that the amount of ^{137}Cs emitted by the Chernobyl reactor is no more than some 6% of that produced by nuclear weapons testing, with the equivalent figures being only 0.1% for ^{90}Sr and ^{131}I. At that time the unbridled increase in the number of explosions and the explosive power of the devices tested caused very major contamination, which has not been talked about much. The Treaty signed in Moscow on 5 August 1963 put an end to such atmospheric testing (apart from weaker blasts carried out by China and France) and the contamination level began to fall. The measurements carried out in Belgium by the CEN/SCK on people not using radioisotopes in their work clearly shows this phenomenon (Figure 2). A similar curve was obtained by Kang (1989); Liljenzin et al. (1988) give comparable figures for ^{137}Cs activities in the human body and in milk.

I.1.3. Consequences for our study

It follows from the above that the two main radioisotopes likely to pose a radioprotection problem are ^{134}Cs and ^{137}Cs. During the Chernobyl accident these two isotopes were released with a concentration ratio (^{134}Cs/^{137}Cs) near to 0.52. The figures quoted by the authors vary somewhat. Cambray et al. (1987) measured minimum and maximum values of 0.40 and 0.71, Hoffmann et al. (1987) noted some fluctuation in this ratio, quoting values of 0.44 to 0.47 for their aerosol filters in Germany. Consiglio et al. (1990) observed a value of 0.48 in Italy, while Lowe & Horrill (1988) measured values of 0.50

Isotope	Period		‰ end 89	Emission Persson	Emission Gudiksen
^{54}Mn	312	d	51	.	.
^{58}Co	70.91	d	0	.	.
^{60}Co	5.272	y	617	.	.
^{89}Sr	50.52	d	0	94	22
^{90}Sr	29	y	916	8.1	1.3
^{95}Nb	34.98	d	0	.	.
^{95}Zr	64.03	d	0	160	8.5
^{99}Mo	65.94	h	0	160	17
^{99m}Tc	6.01	h	0	.	.
^{103}Ru	39.24	d	0	140	27
^{105}Ru	4.44	h	0	.	.
^{105}Rh	35.4	h	0	.	.
^{106}Rh	29.8	s	0	.	.
^{106}Ru	372.6	d	83	59	6.3
^{110m}Ag	249.8	d	24	.	.
^{111}Ag	7.47	d	0	.	.
^{124}Sb	60.2	d	0	.	.
^{125}Sb	2.76	y	398	.	.
^{126}Sb	12.4	d	0	.	.
^{127}Sb	3.84	d	0	.	18
^{129m}Te	33.4	d	0	.	13
^{130}I	12.36	h	0	.	.
^{131}I	8.04	d	0	670	1300
^{132}Te	78.2	h	0	450	200
^{132}I	83	m	0	.	.
^{133}I	20.8	h	0	.	300
^{134}I	52.5	m	0	.	.
^{134}Cs	2.065	y	292	19	48
^{136}Cs	13.1	d	0	.	20
^{137}Cs	30.17	y	919	37	89
^{140}Ba	12.76	d	0	280	37
^{140}La	40.28	h	0	.	10
^{141}Ce	32.5	d	0	130	8.5
^{143}Ce	33	h	0	.	.
^{144}Ce	284.4	d	38	88	5.2
^{147}Nd	10.99	d	0	.	.
^{237}Np	$214\ 10^4$	y	1000	.	.
^{238}Pu	87.74	y	971	0.03	.
^{239}Np	2.35	d	0	970	.
^{239}Pu	24110	y	1000	0.026	.
^{240}Pu	6537	y	1000	0.037	.
^{241}Pu	14.4	y	838	.	.
^{242}Cm	162.9	d	3	0.78	.
^{244}Cm	18.11	d	0	.	.

Table 2 : List of main isotopes observed in Chernobyl fallout, including corresponding half-lives (given as follows after Weast 1988 : y = year, d = day, h = hour, m = minute, s = second), the proportion of the quantity emitted still present at the end of 1989 (expressed in thousandths) and the main activities released (expressed in PBq = 10^{15}Bq, after Persson et al. 1987 and Gudiksen et al. 1989).

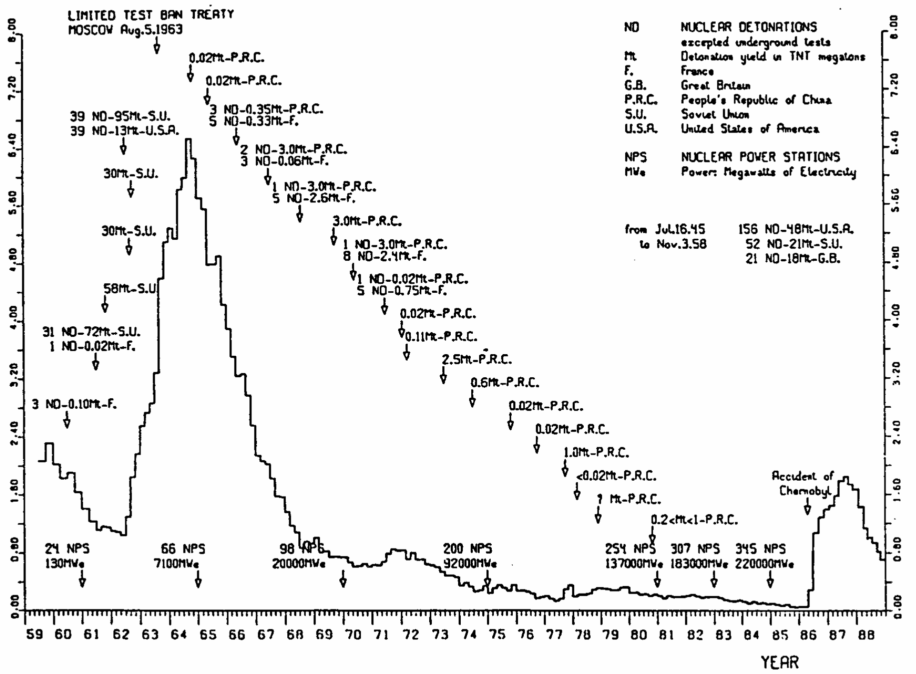

Figure 2 : Changes in internal contamination in the human body (^{137}Cs) measured systematically by the CEN/SCK since 1959 in a group of people in Belgium (after Deworm 1987, supplemented by recent data from the CEN/SCK in Mol). The graph clearly shows the strong contamination existing in the 1960s following atmospheric nuclear weapons testing, the fairly rapid decrease which occurred once the tests had stopped, and the contamination due to the Chernobyl accident.

to 0.53 in Great Britain and Gudiksen et al. (1989) give 0.54 for Scandinavia. The authors quoted by Jackson (1989) observed ratios of 0.53 to 0.62. Pienkowski et al. (1987) give the figure of 0.55; Tobler et al. (1988) that of 0.58; Hötzl et al. (1987) values of 0.56 to 0.59. Due to the huge difference in the length of their respective half-lives (2.065 years for ^{134}Cs and 30.17 years for ^{137}Cs) this ratio changes with time : whereas during the Chernobyl fallout period it was 0.52, in October 1986 it was near 0.44, in October 1987 0.33, in October 1988 0.24 and in October 1991 0.09. This trend is shown in Figure 3.

As for nuclear explosions, these produce only a small amount of ^{134}Cs or none at all. After Gudiksen et al. (1989) the $^{134}Cs/^{137}Cs$ concentration ratio in this fallout was under 0.001. Whatever the case, the decrease is such that at the moment no more than one 10 000th or one 20 000th of the quantity produced in the early Sixties is still around. However, less than half of the large amounts of ^{137}Cs which began to circulate at that period has disappeared by today. The contamination observable nowadays in natural ecosystems therefore consists of a mixture of ^{134}Cs from Chernobyl and of ^{137}Cs produced by the Chernobyl accident and by nuclear bombs.

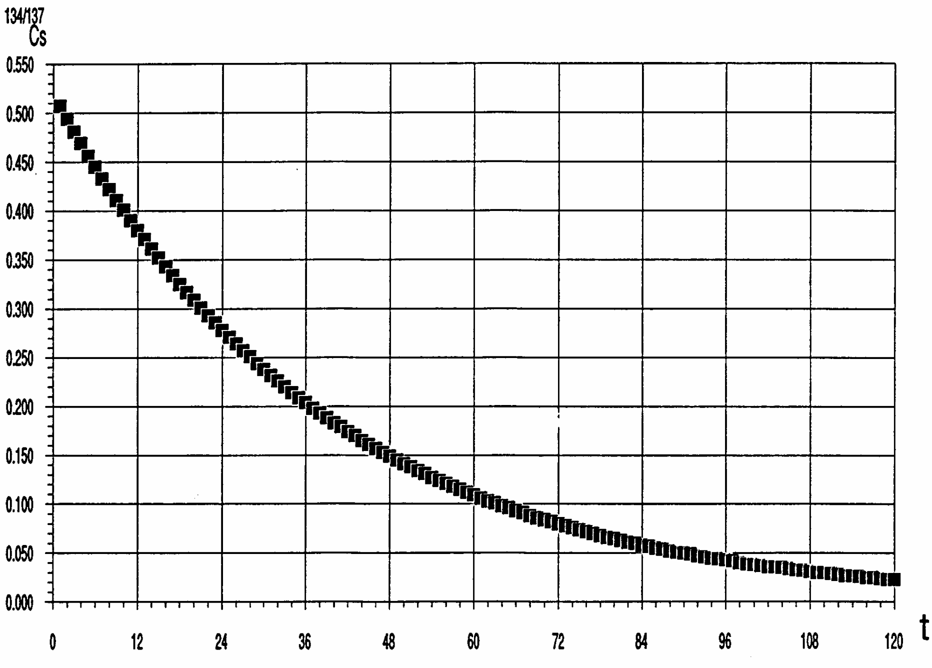

Figure 3 : Changes in $^{134}Cs/^{137}Cs$ concentration ratio over time (expressed in months since the date of the Chernobyl accident; initial concentration ratio = 0.52).

I.2. DISPERSAL OF CONTAMINATION

After being released into the atmosphere the contamination was dispersed by contaminated air masses. In the case of the Chernobyl accident, changes in wind direction during the 10 days over which the radioactive cloud was released gave rise to several plumes which followed very diverse trajectories (Figure 4; see, too, the figures given by Müller 1986, Duvernet 1989 and Gudiksen et al. 1989), all the more so since the air masses moved in different directions depending on their altitude. The contamination in the air masses therefore also varied with altitude (Jaworowski & Kownacka 1988).

Several mathematical models based on regular recording of wind speed and direction at a large number of European sites made it possible to reconstruct more or less satisfactorily the trajectories taken by the plumes. The main models are MESOS (Imperial College, United Kingdom) and GRID (RIVM/KNMI [= National Institute of Public Health and Environmental Protection/Royal Netherlands Meteorological Institute], Netherlands) and the model put forward by Van der Auwera & de Sadeleer (1982) and Van der Auwera & Vanlierde (1986). Other models, aimed at predicting the trajectory of a contaminated cloud, have also been developed (Duvernet 1989, Sinnaeve 1991a and 1991b). The SPADE model is intended to evaluate short-range atmospheric dispersion (20 km), model MC 31 covers distances of 100 to 200 km and the 3-DRAW model covers the area of a continent, and even of a hemisphere. The PATRIC Model (Lange 1978), developed by the U.S. Airforce Global Weather Centre, covers the northern hemisphere.

Use of such mathematical models currently allows us to obtain a fairly good picture of the movement of radioactive clouds as well as of contamination via dry deposition. On the other hand, contamination via wet deposition (rain, etc.) is often not taken into account enough or even left out of this type of calculation despite the large role they play in radioactive fallout (see I.3). Such meteorological phenomena vary too much in time and space for them to be modelled.

As for radioactive contamination caused by atmospheric testing of nuclear bombs, the large number of explosions (229, totalling 87 Mt, from 1945 to 1958; and 116, totalling 298 Mt, from 1959 to 1962, figures quoted by Deworm 1987), the explosive power of the devices used, as well as the spreading of these tests over time and space gave rise to a very large number of clouds which spread major contamination throughout the whole world.

I.3. RADIOACTIVE FALLOUT

During their journey radioactive clouds deposit some of their activity on the countryside they are crossing. This process takes place in two main ways : dry deposition and wet deposition.

Dry deposition is the descent of radioactive particles suspended in the air. It is governed by numerous factors, including air speed and turbulence, particle shape and density, etc. (Sinnaeve & Olast 1991). Deposition occurs more quickly the denser the particles and the calmer the air. In the Chernobyl accident case most of the fallout occurred on Soviet territory.

Wet deposition is the name given to contamination carried to the ground by various types of precipitation (rain, snow, hail, etc.). Given that such precipitation "washes" the contaminated air as it passes through it, wet deposition often contains more contamination than dry deposition. As a guideline, it should be noted that the "washout ratio", i.e. the ratio between the contamination in rain and that in air, varied between 300 000 and 5 000 000 in Sweden during deposition of Chernobyl fallout (Persson et al. 1987); it increases with increasing particle diameter. As a result, when

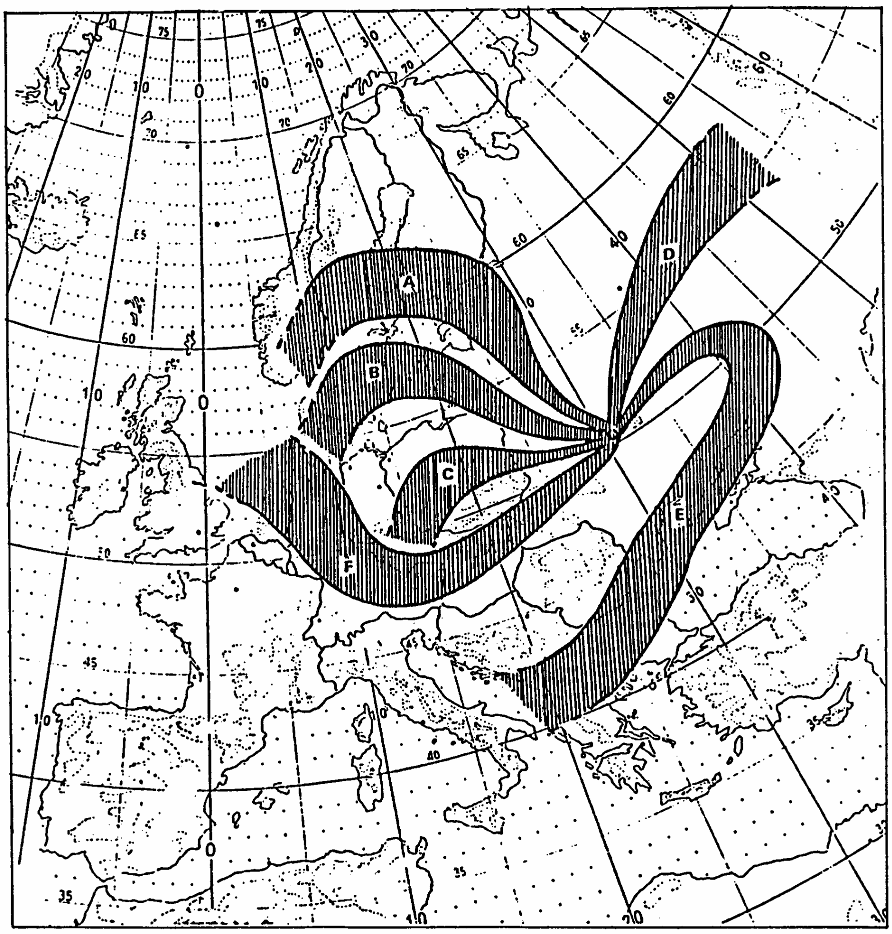

Figure 4 : Trajectories of the various radioactive plumes across Europe (after Martens 1986); the striped areas indicate only the approximate dispersion of the radioactivity with distance from the source. The letters stand for the following dates : A = 26-28/4; B = 26-29/4; C = 27-30/4; D = 28-30/4; E = 29/4-2/5; F = 29/4-3/5.

precipitation occurs during the passage of a radioactive plume, the resulting degree of contamination often strongly correlates with the amount of rainfall recorded (see for example Clark & Smith 1988, Frissel et al. 1987, Guillitte et al. 1987, Liljenzin et al. 1988, Livens et al. 1992, Menzel et al. 1963, Persson et al. 1987, Puhakka et al. 1988, Tataruch et al. 1989). Bunzl & Kracke (1988b) observed that the deposition which accumulated before 1985 significantly correlated with the average annual rainfall recorded at the various sites studied. The positive correlation observed in Austria between altitude and the degree of contamination of game (Tataruch et al. 1989) is probably no more than a consequence of variations in rainfall with altitude.

Nevertheless, while this link between deposition and rainfall is very striking for ^{137}Cs, which was mainly present in particulate form in the Chernobyl cloud, it is less striking for ^{131}I, which was in both particulate and gaseous form and fell to the ground as dry deposition and wet deposition (Clark & Smith 1988). Livens et al. (1992) put at 20% the fraction of ^{131}I from wet deposition, the remainder coming from dry deposition.

Consequently, this stage in the spread of radioactivity is strongly linked to meteorological conditions. A number of analyses of the impact of these conditions on deposition have been carried out (see, inter alia, Persson et al. 1987, Puhakka et al. 1988).

This contaminated deposition is usually measured in Bq/m^2. Many maps of the deposition recorded following the Chernobyl accident have been published for various countries. We consulted the following :

Austria : Clarke (1986), Henrich et al. (1988), Tataruch et al. (1989, game contamination map);
Belgium : Cottens (1986);
Denmark (incl. Faroes and Greenland) : Aarkrog (1987);
Finland : Rantavaara et al. (1987);
France : Fourré (1988), Laylavoix et al. (1988);
Germany : Dörr & Münnich (1987), Der Spiegel (1988);
Great Britain : Clark & Smith (1988);
Italy : Battiston et al. (1989), Belli et al. (1988);
Netherlands : Frissel et al. (1987);
Poland : Piasecki (1987), Zarnowiecki (1988);
Sweden : maps drawn up by the Swedish Geological Co. and reproduced in particular by Hammar et al. (1988), Liljenzin et al. (1988), Mascanzoni (1987, 1989) and Persson et al. (1987).

In the case of Europe, in addition to the theoretical maps drawn up on the basis of the mathematical models referred to above (OMS-Europe 1986), we also consulted a very simplified map (BEUC 1988).

I.4. INFLUENCE OF FOREST CANOPY

Before reaching the ground the fallout comes into contact with the forest canopy. Passage through this changes the fallout in various ways - by changing its distribution in space, its chemical composition and by delaying its arrival on the ground. These phenomena are analysed in greater detail below.

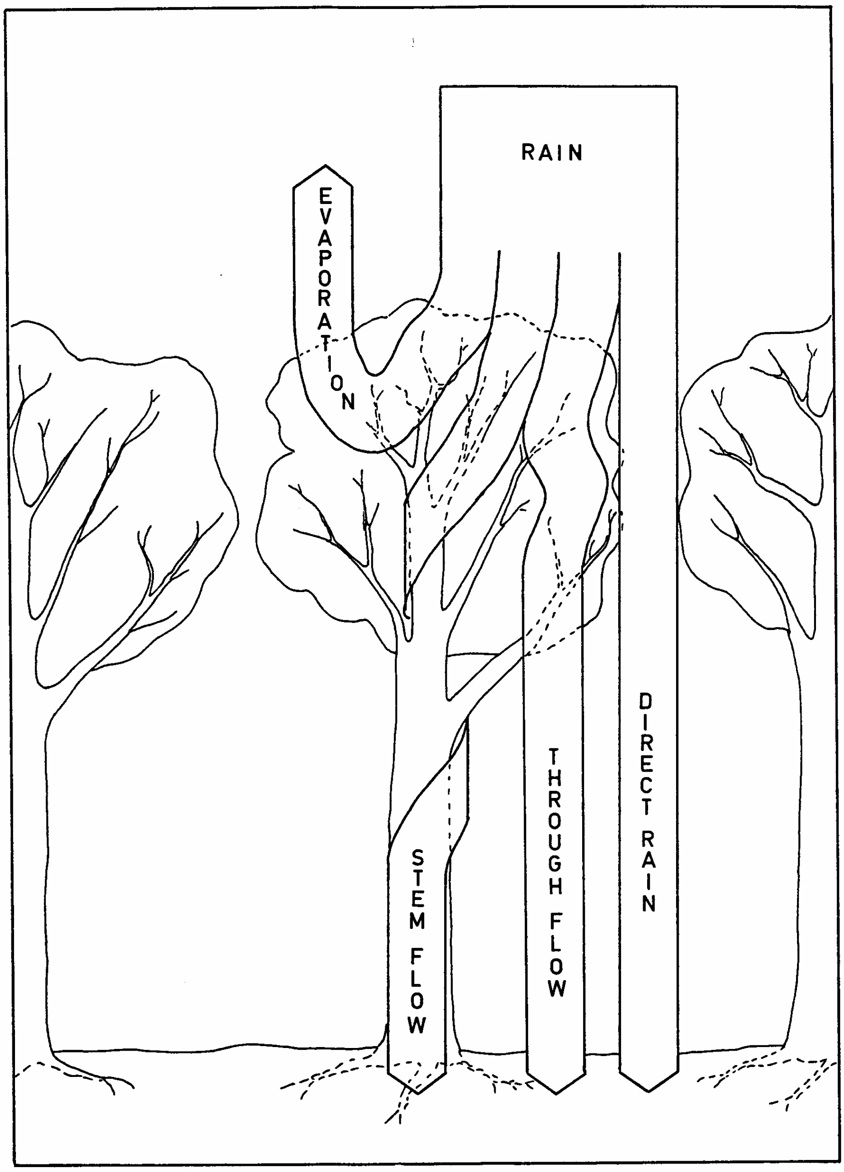

Figure 5 : Redistribution of precipitation by the forest canopy (see text).

I.4.1. Changes in spatial distribution of fallout

When raindrops fall onto the forest canopy various things can happen to them (Figure 5) :

a. They can pass through small gaps between the branches and reach the ground directly without touching the forest canopy. Thus, this fraction of the wet deposition is not changed in any way. The amount involved will be larger the sparser the forest canopy : in large openings it can account for 100% of the total volume of precipitation and can be practically 0% in very dense stands.

Several factors can be involved in determining this percentage :

- The nature of the forest canopy plays some role. Roughly speaking, the various species can be placed in the following order of increasing canopy density : *Larix, Pinus, Betula, Quercus, Carpinus, Fagus, Abies, Pseudotsuga, Picea.*

- The age of the trees is also very important, at least for forests planted by man, where stands are generally even-aged and are usually denser the younger they are.

- The influence of forest management methods can also be substantial : coppice system, high forest system or coppice with standards, heavy or light thinning, etc.

- Finally, the season in which fallout occurs is also very important, at least in the case of deciduous forests, because a tree without leaves (or without needles : *Larix*) intercepts considerably less precipitation than a tree in leaf. In Belgium, the leaves of most of the woody species had already started to come out at the time Chernobyl fallout came down, but the leaves had not, as a rule, reached their final size.

b. They can be deposited on the branches and leaves and then evaporate before reaching the ground. The fraction of precipitation in this category is determined in particular by the type of precipitation : during very short downpours it can be very close to 100%, although it is usually very small in the case of strong rainfall lasting several hours. This fraction is also larger for trees in leaf than for those without leaves. Very uneven bark and abundant epiphytic cover (lichens) also tend to increase this fraction.

c. They can fall to the ground after touching one or several leaves/branches. This part of the precipitation is called through flow. In the case of spruce it was observed to be greater with increasing distance from the trunk towards the edge of the crown (Guillitte et al. 1989a, Henrich et al. 1989).

Schnock & Galoux (1967) measured under mixed oak a through flow of 74% of total precipitation. This figure was 82.5% under *Quercus pedunculata*, 80.3% under *Fagus sylvatica* and 65.5% under *Carpinus betulus*. It depends on the amount of rain falling during the downpour : the same authors observed that under mixed oak and for light rain equivalent to one mm of rainfall, through flow accounted for no more than 60% of the total amount, whereas the corresponding figure was 90% for 15 mm of rainfall. This difference stems from the amount of rain wetting the leaves and branches and evaporating without reaching the ground. It is obvious that in deciduous species through flow is much lower during periods when the trees are not in leaf.

d. They can run along the branches and then the tree stem, finally reaching the ground at the base of the trunk. This fraction is called stem flow, and its size compared to that of through flow is determined by tree shape (Franklin et al. 1967). The crown of the spruce (*Picea abies*), for

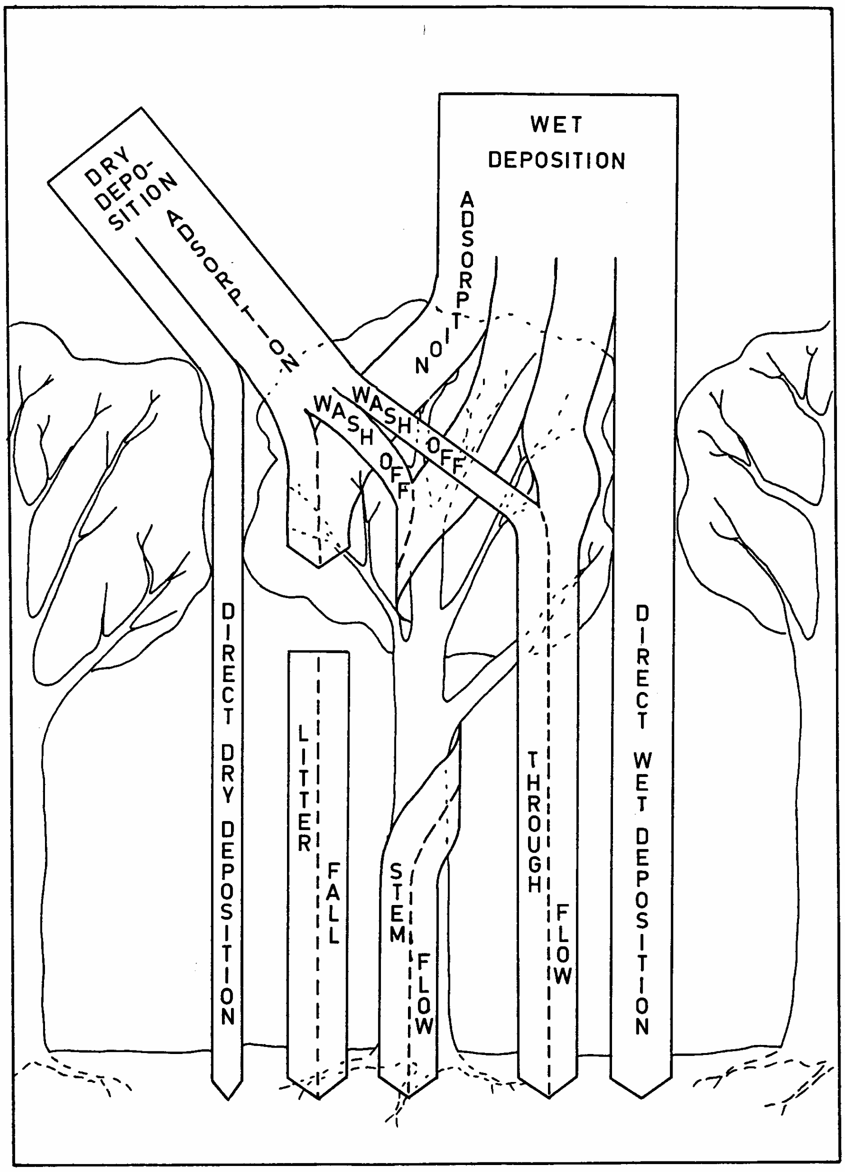

Figure 6 : Redistribution of dry and wet deposition by the forest canopy (see text).

example, takes the form of a cone standing on its base; their branches are generally oblique and pointing downwards; this type of cone has an umbrella effect and through flow therefore clearly predominates in this species. On the other hand, the beech (*Fagus sylvatica*) has a crown in the shape of a cone standing on its apex, with oblique branches pointing upwards; this type of crown has a funnel effect, and stem flow is therefore very large. Most other species come somewhere between these two types. Oak trees (*Quercus robur*, *Q. petraea*, etc.) are different again : their sinuous branches and rough bark clearly act as an obstacle to stem flow. Figure 7 shows these phenomena.

The season can also be very important in this context. Schnock (1967) carried out measurements for different species, in the leaved and leafless phases. Compared to the total volume of rain, stem flow in these two phases accounted for 4.60 and 8.01% respectively for *Carpinus betulus*, 2.27 and 5.23% for *Fagus sylvatica*, 1.39 and 5.07% for *Acer campestre*, and 0.39 and 2.19% for *Quercus*. On average, stem flow accounts for 8% in the mixed oak grove studied by Schnock; it is 4.4% during the leaved phase and 11.3% during the leafless phase.

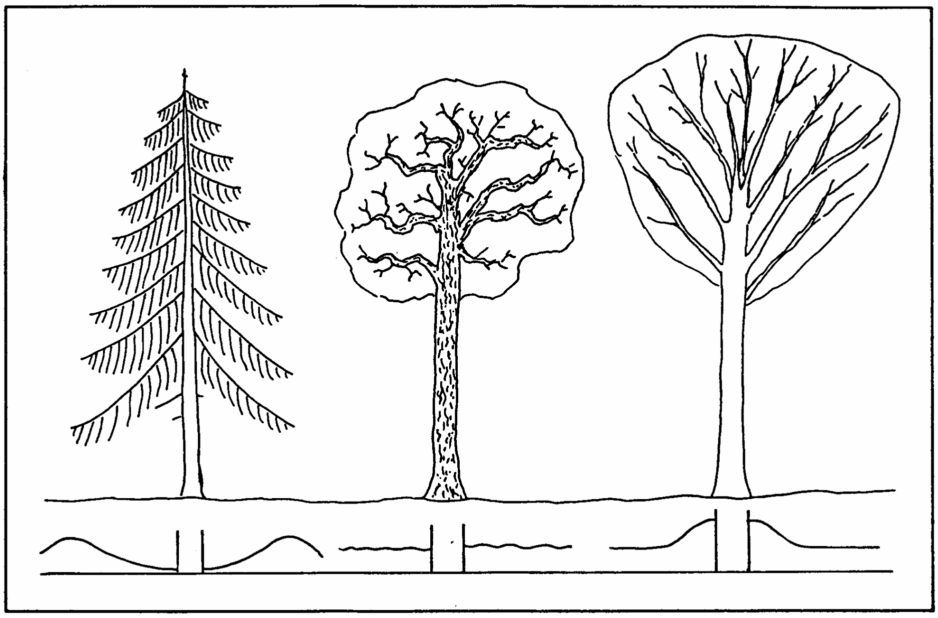

Figure 7 : Crown structure of three forest trees and the changes they cause in deposition of radioactivity on the ground : left = spruce (*Picea*), which causes the deposition to move towards the edge of the crown; centre = oak (*Quercus*), which has hardly any influence on the distribution of deposition; right = beech (*Fagus*), which collects the deposition at the base of the trunk.

The measurements carried out by Guillitte et al. (1989a) are a good illustration of this phenomenon and also show that for broad-leaved trees (*Fagus*) deposition can vary greatly around the base of one and the same tree. On average, the contamination observed by these authors on the least contaminated side of the *Fagus* trunk is comparable to the average contamination recorded at the base of *Picea*, whereas that observed on the most contaminated side of the *Fagus* trunk is twice as high. These results can possibly be explained by the fact that stem flow does not take place in a homogeneous manner over all the trunk surface but follows particular routes. The direction of prevailing winds and that of winds blowing during fallout can also be of some importance.

Furthermore, the configuration of some crowns (*Picea* for example) is also likely to produce a fairly strong accumulation of rain at the centre of small gaps in a stand ("impluvium" effect). On several occasions we have observed very major contamination in fungi taken from such sites. Finally, the samples taken from the edge of stands often contain above-average contamination - due to this phenomenon and the fact that trees situated on the edge of stands collect larger amounts of dry and wet deposition than trees situated within the main body. For further information on the influence of the forest canopy on fallout redistribution see Guillitte et al. (1989a).

I.4.2. Changes in the concentration of radionuclides in pluvial washoff

The radionuclide concentrations in contaminated rainwater undergo changes as the latter passes through the forest canopy. These changes can have three different origins (Figure 6) :

a. Washoff of dry deposition from leaves and branches. This process leads to a greater increase in the concentrations in the rainwater the larger the dry deposition is; this normally happens when a long period of dry weather precedes the downpour. It is known that forests are very efficient filters in collecting dry deposition. Bunzl & Kracke (1988b) observed that the average rate of dry deposition over a long period (before 1985) was approximately nine times greater in forest than in grassland, and Adriano et al. (1981) give similar results. However, Bunzl et al. (1989a) and Bunzl & Kracke (1988b) observed differences of only 20 to 30% between the total radiocaesium deposits on a spruce stand and neighbouring grassland. The reason for this is that the differences observed between these two types of biotopes are much larger for dry deposition than for wet deposition, which makes up the major part of fallout (with the possible exception of fog and mist).

This ability to capture dry deposition goes a long way, no doubt, towards explaining why contamination of forest ecosystems is very often much greater than that of nearby open areas. Gerzabek et al. (1988) observed that contamination of fungi varied with environment : conifers > broad-leaved trees > grassland.

Sombre et al. (1989) observed washoff by rain of Cs deposited on a spruce (*Picea abies*). They concluded that there were two fractions of Cs, the first (approximately 50% of the total) having an ecological half-life of five days and the second an ecological half-life of 50 days. They also noted that two years after initial contamination, the rain under *Picea abies* (spruce) was some four times more contaminated than under *Quercus* (oak). This state of affairs is very probably due to the fact that the leaves on the oak at the time of measurement had formed in the preceding spring and therefore had not received direct contamination, whereas most of the spruce needles had already grown at the time of fallout.

Slinn (1977) studied the mechanisms involved in dry and wet deposition on the forest canopy. The amount of dry deposition depends on the height and biomass of the canopy, the plant type (evergreen or deciduous) and average wind. The finest particles are adsorbed the most lastingly; the largest particles become detached more easily (Kovar 1990). Wedding et al. (1975) also showed that pubescent leaves were clearly more effective than smooth leaves in collecting aerosols. Chamberlain & Chadwick (1972) observed that particles attached much more easily to moist leaves. It was also observed that the deposits were larger along the leaf veins and in the hollows of leaves; in contrast, such deposits were smaller on the underside of leaves and at the edges. The size, shape and orientation of leaves also play a role (Kovar 1990).

b. An increase in radionuclide concentration in the rainwater deposited on the crowns, due to evaporation of some of the water.

c. A decrease in such concentrations due to adsorption of part of the radionuclides on the leaf and branch surfaces. This can lead to different changes depending upon the radionuclides involved, due to a marked difference in their adsorptive ability. Bunzl et al. (1989a) observed that 70% of the radiocaesium fallout on an old spruce stand was retained by the crowns. Milbourn & Taylor (1965) measured an approximately 50% retention rate for ^{89}Sr applied in spray form to pastureland. According to Ronneau et al. (1987) and Sombre et al. (1989), retention of radiocaesiums is high (around 80%) while radioiodines are hardly ever retained. The result of this process is that the rain flowing off the foliage contains less Cs and more I than rain reaching the ground without touching the forest canopy.

On the other hand, the percentage thus retained by crowns and trunks probably varies greatly, depending upon tree species. In the Höglwald (Germany) Schimmack et al. (1991) observed a 20% interception rate for a beech forest (*Fagus sylvatica*) and 70% for a spruce stand (*Picea abies*). The difference between these two figures is probably due to the fact that the foliage of the beech trees was only slightly developed during the fallout period.

I.4.3. Changes in distribution of fallout over time

As noted above, some of the contamination contained in the rainwater is adsorbed on the leaf and branch surface while passing through the forest canopy. This adsorbed fraction nevertheless ends up by reaching the ground, including by being returned into solution or suspension during later downpours. This pluvial washoff is quite slow, however : Vallejo et al. (1989) noted that 3-year-old *Pinus* needles still contained average activity of 9 Bq ^{137}Cs/kg dry matter (DM), whereas 2- and 1-year-old needles, which had not formed when the fallout occurred, contained only 3 and 2 Bq ^{137}Cs/kg DM respectively.

Another way in which contamination transfers from foliage to the ground is through falling leaves or needles. In the case of the latter (with the exception of resinous deciduous trees such as *Larix*), this can be spread over several years, since needles normally stay on the tree for between 3 and 6 years.

Transfer of contamination to the ground from crowns of *Picea* via rain, wind and falling needles was studied by Bunzl et al. (1989a). The half-life of Cs in the crown was 90 days during the first 130 days and 230 days between the 131st and 600th day following deposition. During this period of 600 days falling needles accounted for no more than 7% of the quantity of Cs transferred from the forest canopy to the ground.

Interception of contamination by the forest canopy has the following effects in particular :

- it slows down the rate of deposition on plants and the ground beneath the canopy (Block & Pimpl 1989, Vallejo et al. 1989), and
- reduces the total contamination received at this level because the radionuclides undergo radioactive decay during residence in the tree crown; this holds particularly true for isotopes with a short half-life such as ^{131}I.

I.5. INFLUENCE OF TYPE OF SOIL AND HUMUS

I.5.1. Holo- and hemiorganic horizons

When they reach the ground the radionuclides are adsorbed on the different reception "sites", the nature and effectiveness of which vary according to the type of soil and humus. These are in turn determined by :

- the geological substrate;

- the nature of the forest canopy, which determines the acidity and the decomposability of the deposited litter; certain species are well known to generate litter which decomposes less readily (*Picea, Larix, Pinus, Fagus*, etc.), especially due to its acidity, its high C/N ratio and the antibiotic substances with which it is sometimes impregnated; certain species, in particular *Picea abies*, can also have a negative impact on the pH and the richness of the soil itself (Bonneau et al. 1976, Herbauts 1987, Manil 1971, Noirfalise 1964);

- the agricultural, silvicultural and grazing practices occurring there in the past (over-exploitation having led to temporary disappearance of the forest and washout - and even podzolisation - of the soil; fertilisation or clearing of land, etc.).

Because it is located on the top part of the soil it is the slightly decomposed litter (OL horizon) which is the first to come into contact with the radionuclides, a large part of which become fixed at this level (Andolina & Guillitte 1989a, Belli et al. 1989, Römmelt et al. 1989, Sombre et al. 1989, Vallejo et al. 1989, Witkamp & Frank 1967), with most of the remainder being adsorbed by the immediately underlying layers consisting of more or less decomposed organic matter (holorganic horizons OF, OH and OAH). One can therefore conclude that during the weeks following the fallout events, the radiocaesium isotopes deposited on the forest floor basically remained confined to the holorganic horizons, and that only a minute part of the contamination percolated through into the AH hemiorganic horizons (with the exception of the best kinds of humus [mull] in which the holorganic horizons are very thin or practically non-existent).

Schimmack et al. (1991) recorded a residence half-time in the organic horizons of 1 050 days in a spruce stand and 700 days in a beech stand for Chernobyl fallout. The difference is probably due to the fact that the organic horizons of spruce-stand soils are thicker and decompose more slowly than those in beech stands.

Compared to clay minerals, organic matter is a relatively poor fixer of Cs and the biological availability of this element is fairly high in organic matter. Barber (1964), Sandalls et al. (1989) and Frissel et al. (1989) observed a clear correlation between Cs availability in soil and the percentage of organic matter. However, this relationship holds good only for soils with over 15% organic matter. In contrast to this, the correlation between the percentage of organic matter and Sr availability is negative. Thus, it comes as no surprise that the biological availability of Cs is particularly high in peaty soil, which consists for the most part, if not totally, of organic matter (Cremers et al. 1989, Sanchez et al. 1988).

As for radiocaesium fixation sites in the organic matter in soil, Andolina & Guillitte (1989a) showed that the Cs retention capacity of lignin is much higher than that of the two other major components of organic matter - humic and fulvic acids. The fact that many fungi are able to decompose lignin provides one explanation for the high contamination levels of certain species of fungus.

Fixation of ^{239}Pu, ^{240}Pu and ^{241}Am by the humic and fulvic acids making up organic matter has also been shown (Livens et al. 1987, Livens & Singleton 1991).

During the months and years which followed contamination, the radiocaesium isotopes migrated very slowly into the soil profile. Adsorption on the organic matter is usually fairly effective, but in forest soils such migration continues even if the radiocaesium isotopes are fixed, because the matter making up the litter breaks down in the course of time, either simply due to mechanical or chemical degradation or action by microfauna, fungi and bacteria. After being broken down in this manner such matter is even more likely to percolate into the cracks and small interstitial spaces in the soil. Furthermore, it is covered over each year by a new layer of (less contaminated) leaves.

What is more, the radiocaesium isotopes can be removed from their organic substrate, mainly through the action of microfauna and microflora, and thus regain the ability to move within the soil profile. Higher plants also have a certain impact on desorption of radionuclides (Muramatsu et al. 1991b). The mycelia of certain fungi selectively accumulate radiocaesium isotopes and transfer them to their fruit bodies; this constitutes transport and horizontal concentration as well as redeposition on the soil surface (see I.7.1 for details). Likewise, earthworms also have an impact on the vertical distribution of Cs, as observed in permanent grassland (Caput et al. 1989). This could also be the case in certain types of forest where such organisms are relatively abundant.

I.5.2. Mineral horizons

a) Clay content

In the mineral part of the soil the clay minerals are more or less the only specific fixation sites. Their adsorptive capacity with regard to Cs has been known for several decades (see, inter alia, Squire & Middleton 1966, Witherspoon 1964). This has also been shown by more recent experiments (Bunzl & Schultz 1985, Cremers et al. 1988, Kerpen 1988, Ocker 1987, Sweeck et al. 1989) as well as in the measurements taken after the Chernobyl accident (Papanicolaou et al. 1989, Römmelt et al. 1989). Consequently, radiocaesium isotopes are usually more available in sandy soils.

It has also been shown that the capacity to retain Cs varies from clay to clay (Graham & Killion 1962, Schulz et al. 1960, Tamura 1964), and the following classification can be established as a function of Cs fixation capacity : vermiculite > chlorite and illite > kaolinite, montmorillonite and hydrobiotite. More recently, Sinnaeve & Olast (1991) restudied the cation exchange capacity of various types of clay with regard to Cs, and put it at 0.01-0.09 mmol/g for kaolinite, 0.20-0.29 mmol/g for illite and 0.50-0.70 mmol/g for beidellite-montmorillonite.

This classification is not necessary the same for other radionuclides. It also varies depending on the amounts of Cs used : above 2.10^{-3} meq per unit of exchange capacity the adsorptive capacity of illite drops below that of kaolinite and montmorillonite (Tamura 1964). Schulz (1965) confirms that two Cs fixation mechanisms can be present in soil, depending upon whether this element is present in large or very small amounts. Sheppard et al. (1987) also observed that Kd values varied depending on the concentration of the elements studied. Gerzabek et al. (1989) indicate that the ^{137}Cs transfer factors from soil to selected crops are smaller the more contaminated the soil is.

It should also be noted that laboratory experiment results have sometimes been very different depending upon the solvent used to extract Cs (Schulz et al. 1960).

Cs fixation in clay minerals takes place at only a small number of sites, situated in the interlayer-edge zones, which demonstrate a strong preference for the ions of lightly hydrated alkali metals such as K, Rb and Cs. Such sites are called frayed edge sites (FES).

More precisely, Cremers et al. (1988) discovered the existence of three types of sites for adsorption of Cs : the first one not being very selective but very abundant (95-97% of FES), the second being moderately selective and not very abundant (2-5% of FES) and the third being super-selective but rare (0.5% of FES).

Not all elements are fixed by a particular type of soil with the same degree of effectiveness. Schimmack et al. (1987) give the following classification for Kd in a forest soil : Tc < I < Ru, Co, Zn, Sr < Cd < Ce < Cs. The mobility of elements in a sandy brown soil is classified as follows : Tc ≥ I >> Mo ≥ Cr >>> V ≥ Np, Cs, Th (Sheppard et al. 1987). Schulz (1965) classifies ions as follows as a function of their adsorptive energy : $Cs^+ > Rb^+ > K^+$ and NH_4^+.

Bunzl et al. (1989b) determined the rate of progression of ^{134}Cs from Chernobyl in podzolic soil in a spruce stand : it ranges from 4 ± 2 cm/year in the Of_1 horizon, 3 ± 1 cm/year in the Of_2 and 2 ± 1 cm/year in the Oh. ^{106}Ru migrates at a quicker rate.

b) Homologous cations

To move from one compartment to another, Cs^+ enters into competition with other monovalent cations - with Rb^+ and K^+, which are alkali metal homologues of Cs (same column in the Mendeleev Table), and ammonium ions (NH_4^+).

Thus, when a certain amount of one or several of these ions is added to a nutritive solution containing Cs, plant uptake of Cs is reduced (Handley & Overstreet 1961, Jackson et al. 1966, Schulz 1965). Likewise, when the proportion of exchangeable K is high in soil, Cs availability is low (Andolina & Guillitte 1989a, Schulz et al. 1960, Squire & Middleton 1966). This is direct competition, with these different ions being more or less absorbed by the plants in proportion to their concentration in the solution.

In soils, the presence of fixation sites (FES) capable of immobilising such ions in a form unavailable to plants makes the problem more complicated (Figure 8). In soils with a high specific radiocaesium interception potential (SRIP), the distribution of Cs between the liquid and solid phases is basically governed by the FES (clay minerals!) and by the concentrations of K^+ and NH_4^+ in the liquid phase.

In contrast to this, in peaty soils, where clay minerals are not very abundant and whose SRIP is approximately 10 times lower than in soils containing more clay, the FES play hardly any role and the concentrations of free NH_4^+ and K^+ are the main factor in determining the availability of Cs (Cremers et al. 1989, 1991; Heaton et al. 1989; Sweeck et al. 1989). These authors also provide equations to determine the Kd (Cs) in soils.

The existence of such fixation sites explains why the addition of NH_4^+, K^+, Rb^+ or Cs^+ ions to soil contaminated with radiocaesium usually produces increased absorption of this radionuclide by plants (Evans & Dekker 1969, Schulz 1965). What happens is that some of these ions substitute for radiocaesium at the fixation sites and make it more available for plants. Other cations (Ca^{++}, Mg^{++}, Ba^{++}, Na^+ and H^+) have an identical, but much less marked, impact (Schulz 1965). According to Memom et al. (1983), the first three of these are homologues of Sr. According to other studies (Desmet 1991, Grauby et al. 1991), K is the only element which can reduce transfer of Cs to plants (and of Sr to a lesser degree). This effect is only obtained in soils with a high cation exchange capacity. When cation exchange capacity is low the addition of large amounts of K induces Cs desorption and thus increases uptake of Cs by plants.

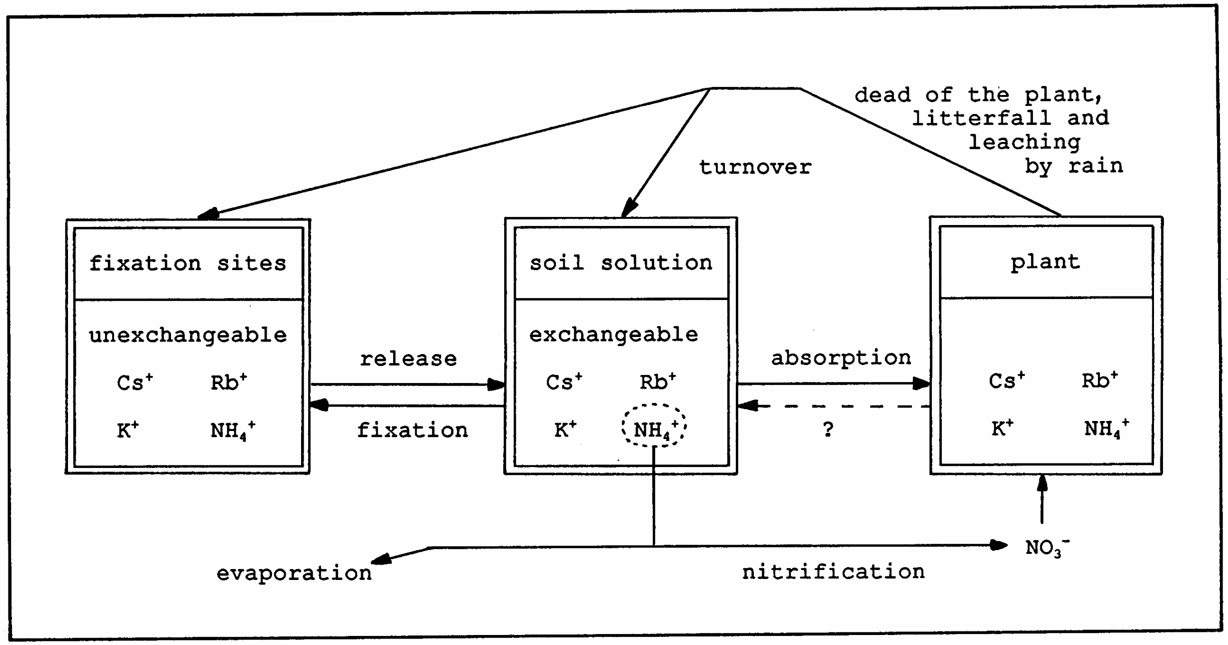

Figure 8 : Movement of Cs and its homologues among the various soil compartments.

It has also been observed that NH_4, although the homologue of K, appears to be much more effective than the latter in lowering the retention capacity (Kd) of soil with regard to Cs (Schulz 1965, Sweeck et al. 1989). This could be explained in two ways. First of all, a certain amount of exchangeable NH_4 is perhaps lost through volatilisation. Secondly, the microorganisms responsible for nitrification metabolise some of the exchangeable NH_4^+, changing it into NO_3^-, which does not compete with Cs. The concentration in competing ions in the soil solution would therefore be lowered, thus making Cs more available. By contrast, since they are unaffected by nitrification, the immobilised NH_4 ions continue to occupy the fixation sites which would thus remain inaccessible for Cs.

The water balance of soils could therefore be of some importance in this respect. In moist and poorly drained soils nitrification is easier and the nitrogen is mainly found in nitric form. This state of affairs considerably increases the availability of radiocaesium for plants (Cremers et al. 1989).

c) pH

The influence of soil acidity on the availability of radionuclides is well known. In a laboratory comparison of 17 types of soil Kerpen (1988) even says that this is the major factor : the most acidic soils are those in which the Cs is the most available. Frissel et al. (1989) observed that higher plants take up more Cs, Sr, Pu, Np and Co the more acidic the soil (with the inverse effect for Am).

This feature is perhaps another way of explaining the relative ease with which fungi seem to extract numerous substances in the soil (heavy metals, radionuclides, etc.) in comparison to the absorptive capacities of higher plants. It is known that the fungus mycelium generally acidifies the area immediately next to the hyphae, which could increase the biological availability of numerous substances. Fisher (1972) observed that the mycelia of *Hydnellum scleropodium* speed up the development of the soil profile and increase the availability of nutrients, especially K.

Progressive acidification of soils, due to the action of atmospheric pollutants ("acid rain"), will therefore have a negative impact.

d) Practical conclusions

It follows from the above that even if contaminated to the same degree, two types of soil can have very different Cs adsorptive capacities. The two following extreme examples give some idea of the phenomena involved.

1) In a very acidic sandy soil (podzol), the organic horizons are thick. Initially, the Cs remains fixed here but - due to the high acidity - this adsorption is not totally effective and some of the Cs remains free or is re-released. This fraction moves towards deeper horizons where the acidity and the absence of effective adsorption sites (clay minerals) also prevent it from being irreversibly fixed. The relative weak adsorptive capacity of this type of soil and humus has another consequence : that of the radiocaesiums (since they are not strongly fixed) also being more available for green plants and fungi. Given identical soil contamination levels, it is on this type of substrate that the activity observed in these organisms is at its highest. Furthermore, relatively large quantities of Cs are extracted in this manner by plants and redeposited on the soil surface when the leaves are shed and fruit bodies rot, etc. Therefore, Cs turnover is rapid in this type of soil.

2) In the case of a rich soil, lying on limestone or calciferous rock, the humus is of the mull type, and the holorganic horizons, reduced to the litter and very quickly decomposed, are thin or non-existent. Therefore, retention of Cs will be low here and the radioisotopes rapidly enter into contact with the mineral fraction of the soil, where fixation sites abound (clay minerals) and the Cs radioisotopes are strongly adsorbed. Such fixation is more effective the higher the pH. Thus, the

contamination migrates extremely slowly in the soil profile. In contrast, the effectiveness of the adsorption to which they are subject means that the Cs radioisotopes are not readily available for green plants and fungi. Given equal soil contamination levels, it is on this type of substrate that the concentrations observed in these organisms are at their lowest. Because of this, Cs recycling via the dead organic matter is fairly low and turnover very slow.

It is interesting to note that since "young" Cs (from Chernobyl) and "old" Cs (from atomic bombs) are distributed differently in the soil horizons, their availability likewise differs. For example, in 1989 most of the Cs from Chernobyl was still located in the upper part of the holorganic horizons of the soil where it was relatively available for plants, while most of the Cs from bombs had already migrated to the AH_1 and AH_2 soil horizons, where it was strongly adsorbed on clay minerals.

Therefore, it seems probable to us that this difference in location accounts for the differences in availability observed between the two types of Cs, rather than the factors put forward in hypotheses by certain authors, i.e. differences in the chemical form of the "old" and the "young" Cs, or the ability of fungi to discriminate between the two isotopes of Cs. This perhaps also explains why the Chernobyl Cs seems to have migrated more quickly in the soil than the Cs from bombs. In reality, the latter type has probably migrated just as quickly in the holorganic layers of the soil during the years following its deposition, but it has subsequently been more or less immobilised in the mineral horizons. Therefore, its average migration rate since deposition is slower than that of "young" Cs, which is still present in considerable amounts in the holorganic horizons.

I.6. FACTORS RELATED TO SPECIES

If one analyses various species collected at the same time in one and the same forest it will be noted that large differences exist between the contamination levels of the various samples (Gerzabek et al. 1989, Horrill and al. 1989, Memom et al. 1983). These differences are to some extent due to factors related to the species, and in particular to the type of nutrition, which determines the contamination ingested, as well as to the physiological behaviour of the species with regard to radionuclides.

I.6.1. Contamination via nutrition

a. Higher plants and fungi

The food supply of these plants is clearly determined by the substrate on which they grow and thus, indirectly, by the ecological requirements of each species, these having a sometimes major impact on the degree of contamination of such organisms :

1. Spatial location of species within a given territory

Each living species has its own requirements in terms of the physical factors determining its environment (mainly the pH and the chemical composition of the soil, temperature and moisture content of the air and soil, as well as the amount of light). In practice, for each of these factors each plant possesses a minimum and maximum threshold beyond which it cannot subsist, as well as an optimum at which it flourishes.

Because of their differing requirements two plants living in the same region may, for example, grow on different substrates, one on limy soil and the other on acid soil, and thus develop very different contamination levels (due to large differences in the biological availability of radionuclides in these two types of soil) despite the fact that they are situated in areas of identical contamination. The same phenomenon can exist, for example, among plants requiring a lot of light (heliophiles) and growing outside forests, and others which prefer the shade (sciaphiles) and which develop under tree cover, where radionuclides are more abundant and more available.

2. Type of substrate utilised

The higher plants covered by this study are all terricolous. In contrast, the edible fungi growing in a forest ecosystem can be divided into humo-terricolous (mycorrhizal or saprophytic) and lignicolous. This latter type, which grow on a substrate (wood) generally formed prior to the Chernobyl accident and not contaminated to any great extent (with the exception, perhaps, of very rotten large stumps), have very low contamination levels only.

3. Depth of roots or mycelium in soil

This factor is also important because, as we mentioned in I.5 above, contamination is clearly greater in the top layers of the soil than in deeper layers (Jackson 1989, Guillitte et al. 1989b). Furthermore, the concentration of organic matter is higher in the top layers, while the pH and the clay mineral concentration are lower there, thus making Cs more available (Nimis et al. 1989).

Furthermore, the Chernobyl accident provided researchers with an elegant and unexpected method for determining the depth at which plants in general, and the mycelium of fungi in particular, exploit the soil for nourishment (Guillitte et al. 1989b, Lambinon et al. 1988). As we saw earlier, radioactive contamination from atomic bombs contains ^{137}Cs but no ^{134}Cs, while that from the Chernobyl accident consists of the two isotopes in a well-defined concentration ratio ($^{134}Cs/^{137}Cs$ = approximately 0.52 at the time of the accident). However, the radiocaesium from bombs is much older (dating from before 1963) than that of Chernobyl. Therefore, it is generally located in deeper layers of the soil. Because of this, the various soil horizons making up the litter and the soil have different $^{134}Cs/^{137}Cs$ concentration ratios. In 1986 this ratio was higher the closer the horizons were to the surface. Since 1987 the addition of only slightly contaminated dead leaves has greatly decreased the radiocaesium concentrations in the upper layers of the litter.

Since there is no reason to believe that fungi are able to discriminate between these two isotopes, it can be stated that the $^{134}Cs/^{137}Cs$ ratio observed in a fungus should be identical to that in the layer of soil exploited by its mycelium. By comparing the values of this ratio in the various soil horizons and the various species of fungus collected at the same time from the same soil, it is possible to determine both the depth and the nature of the soil horizon in which the mycelium of the various species is located (Figure 9).

b. Game

The diet of game species is determined by the following factors :

1. The species of animal involved

Each animal species has its own dietary habits. The presence of highly contaminated species in diet (lichens, mosses, *Ericaceae* and in particular certain species of fungus) clearly contributes to increasing the contamination of the game animals consuming them.

As we shall see further on, many species of game eat fungi. Often, the amounts ingested are only a very small percentage of the total food ingested but, given the very high contamination levels of certain species of fungus, the dose absorbed is not by any means negligible. Livestock have also been

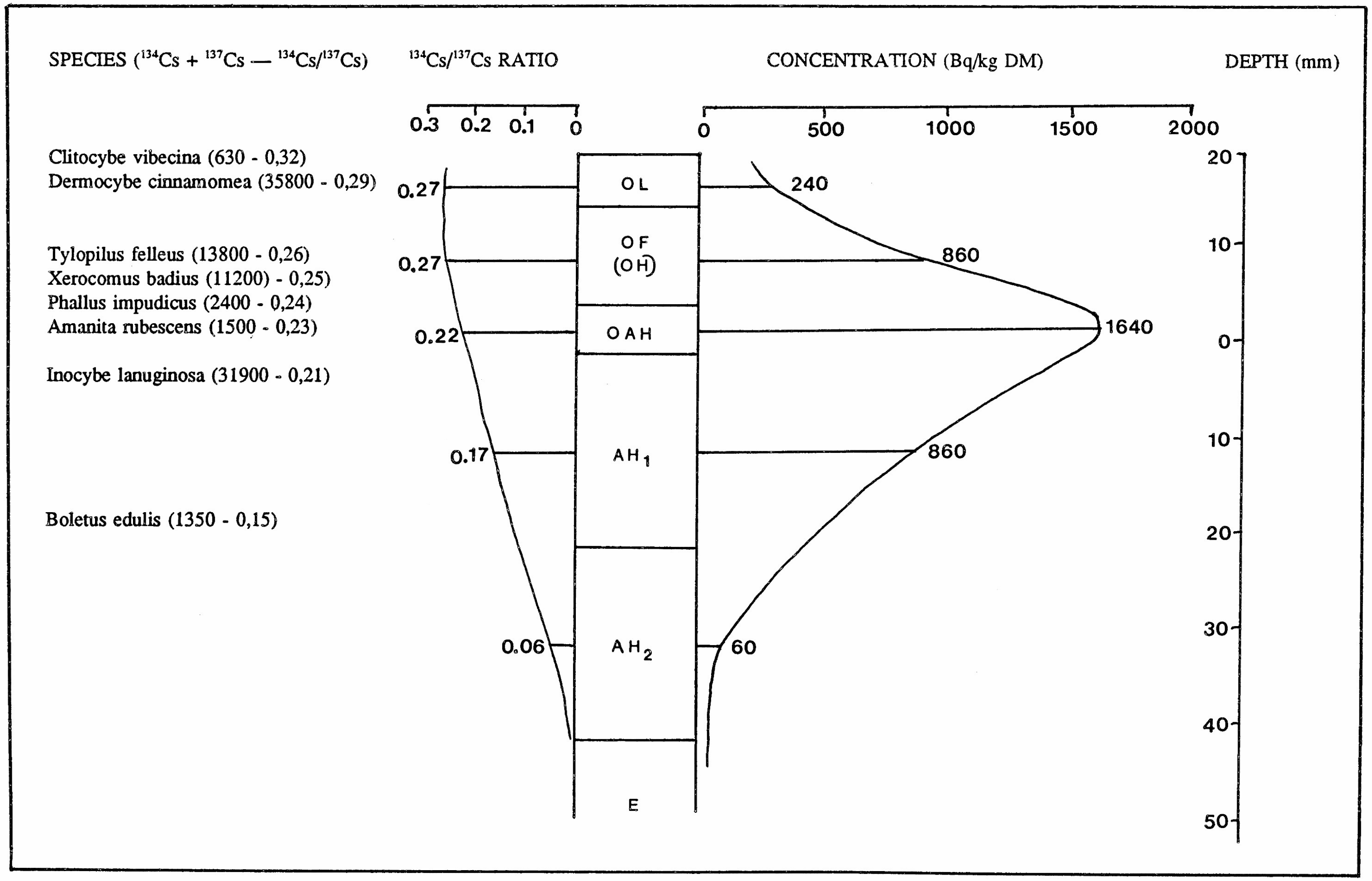

Figure 9 : Vertical distribution of ^{134}Cs and ^{137}Cs and of their concentration ratio in moder-type humus; the comparison with the values observed for this concentration ratio in the fruit bodies of various species of fungi collected from the same soil make it possible to determine the depth of the mycelium of these species. (After Guillitte et al. 1989b)

observed to eat fungi (Fauvel 1951), as have small rodents (Durrieu et al. 1984, Fogel & Trappe 1978, Heim de Balsac 1951, Maser et al. 1978, Mihok et al. 1989, Palo et al. 1989, Polaco et al. 1982, Ure & Maser 1982).

Many studies have been devoted to the diet of wild animal species consumed by man. The references relating to the species studied in this section are given in III.2.

The methods used in these studies are very diverse : recording of tracks left during grazing, direct observation through binoculars, observation of the food eaten by domesticated animals, noting down the species a captive animal takes when given a choice, study of fistulated animals, study of stomach contents, analysis of faeces and even (Silvennoinen et al. 1991) spectroradiometric measurements. These methods provide results which sometimes vary a great deal. Comparative analyses of the value of some of them were carried out by Gaare et al. (1977), Holišová et al. (1986) and Wallmo et al. (1973). A review of these techniques has also been presented by Susmel et al. (1989).

In general, ruminants have a higher contamination level than monogastric herbivores (Tataruch et al. 1989). This is due to their having a larger absorption area in their digestive system, and to greater digestion efficiency. It is often found that carnivores are more contaminated (Johanson et al. 1989, Palo et al. 1989, Pendleton et al. 1965); this is because they are further up the food chain than herbivores. Finally, small mammals are often more contaminated because of their rapid metabolic rate.

2. Region inhabited by game

The geographical spread of game species is usually quite large. Depending on latitude, altitude and the nature of the soil, the range of vegetation available differs and, as a result, diet varies greatly : it goes without saying that a hare living in Italy does not eat the same food as a Swedish hare. Furthermore, as we saw in I.5 above, given identical contamination of the soil, plants absorb many more radionuclides on acid substrates than on alkaline substrates. Finally, it has been observed that certain game species are less contaminated the higher the percentage of their territory given over to agriculture. This is due to the richness of cultivated soils, which decreases the biological availability of the radionuclides found there.

Establishing the diet of a species and the resultant contamination becomes an extremely complicated matter when dealing with migratory animals, such as certain mammals (reindeer, elk/moose) and numerous birds. In this case, calculating the transfer factors is sometimes nigh impossible because the place where the animals are killed is not usually the place where they became contaminated.

3. Season

It has been shown that the radiocaesium content of plants can vary a lot during the year (Bunzl & Kracke 1989). Furthermore, in Europe alternating hot and cold seasons engender an annual rhythm in the development of vegetation.

The most notable case is that of fungi, the most contaminated species of which normally grow in August-October (at least in temperate western Europe). Consumption of the fruit bodies of these species is without doubt a major factor in contamination of game and explains the sharp increase in radioactivity observed in autumn in the tissues of certain animals (Johanson et al. 1989).

1.6.2. Physiological behaviour of species with regard to radionuclides

When it ingests nutritive substances contaminated by a radionuclide a species may either accumulate the nuclide, hinder its absorption or react indifferently to it. A species of the first type mentioned above is known as an accumulator species.

This term is often found in the literature and we deem it important to point out that the terms "accumulator species" and "highly contaminated species" are not necessarily synonymous. A species should only be considered as accumulating an isotope when it absorbs the latter selectively and, as a consequence, the concentrations of this isotope in its tissues are higher than in the substrate on which it feeds.

As shown in Figure 9, *Amanita rubescens* has a level at this particular site of 1 500 Bq/kg DM and *Boletus edulis* 1 350 Bq/kg DM. However, the corresponding soil horizons have readings of some 1 500 and 500 Bq/kg DM; calculation of the transfer factors thus gives 1.0 for *Amanita rubescens* and 2.7 for *Boletus edulis*. This means that it is the latter species which accumulates radiocaesium isotopes, although its contamination level is under that of *Amanita rubescens*.

Other conclusions can be drawn from this figure : *Xerocomus badius*, *Phallus impudicus* and *Amanita rubescens* have a very similar $^{134}Cs/^{137}Cs$ ratio, which means that their mycelium feeds on more or less the same soil layer (OH and OAH horizons); however, their contamination levels are very different, being 11 200, 2 400 and 1 500 Bq/kg DM respectively. Given that the corresponding soil contamination values are around 1 000-1 500 Bq/kg DM, this is scientific proof of selective accumulation of radiocaesium by *Xerocomus badius*, low accumulation by *Phallus impudicus* and indifference on the part of *Amanita rubescens* towards such isotopes.

Among the higher plants, *Ericaceae* family species often have high activities (Bunzl & Kracke 1986, Colgan et al. 1989, Horrill et al. 1989, Johanson et al. 1989, Römmelt et al. 1989). The most contaminated of all is often *Calluna vulgaris*; the others (*Vaccinium myrtillus*, *V. uliginosum*, *V. vitis-idaea*, *V. oxycoccus*, *Erica tetralix*, *E. cinerea*, etc.) accumulate radiocaesium isotopes to a slightly lesser degree. This state of affairs is due first and foremost to the fact that the roots of these plants basically grow in the lower part of the organic horizons, where Cs availability is high. Furthermore, *Ericaceae* have a particular type of mycorrhizae - ericoid mycorrhizae (Strullu 1985) - which no doubt influence their accumulative abilities. Finally, the fact that *Ericaceae* usually grow on poor and acid, even peaty, soils in which Cs availability for plants is high, also increases their contamination level.

I.7. FACTORS LINKED TO THE INDIVIDUAL TAKEN AS A SAMPLE

When carrying out measurements on various samples of the same species collected at the same time and at the same place, further differences are noted in contamination levels. These are due to factors associated with the individual (higher plant, fungus, animal) and, more particularly, the exact location of this individual in the stand as well as its particular physiological characteristics, which are related to seasonal physiological variations, age, etc.

I.7.1. Location of individual (plant) in forest stand

As we saw in I.4.1 above, the forest canopy redistributes precipitation spatially. Depending on its location in the stand, a fungus or plant can therefore have a low contamination level (if it benefits from the "umbrella effect" of a particularly dense canopy) or - on the contrary - become highly contaminated (if it is situated within small gaps engendering the "impluvium effect" or at the foot of a tree subjected to major stem flow).

Another factor which could be a source of variation is the decomposition, on the soil surface, of large fruit bodies from accumulator species. Take, for example, *Xerocomus badius*, whose fructifications can weigh up to 200 g and have a contamination level of 10 000 Bq/kg fresh matter in the most contaminated regions (see in particular Henrich et al. 1989 and IV.2 below); fruit bodies of this species can contain up to 2 000 Bq of radiocaesium isotopes. If these fungi rot on an area of 1 to 2 dm^2, redeposition on this area would correspond to a concentration of some 100 000 to 200 000 Bq/m^2.

These figures are, of course, extreme values and the averages obtained for Belgium, for example, are 20 times lower. Nevertheless, they correspond to deposition of 5 000 to 10 000 Bq/m^2 (on 1-2 dm^2), which is not negligible. Thus, fungi play some part in the migration of radiocaesium isotopes in the soil, as they do in recycling radioactive elements, which they thus return to the surface.

I.7.2. Physiological characteristics of the individual

1. Fungi

Significant differences in radiocaesium concentrations have been observed between samples of the same species collected at the same time from the same biotope. During analyses carried out by us in collaboration with the Faculté des Sciences Agronomiques [= Faculty of Agronomic Sciences] in Gembloux (Fraiture et al. 1989), we observed that young fruit bodies were less contaminated on average than ripe fructifications but the latter, by contrast, were generally more contaminated than old fruit bodies. The explanation for this is probably to be found in the two following phenomena.

1) During formation of the fruit body a continual flux of contaminated cytoplasm enters the hyphae of the young growing fructification. Parallel to this, a certain amount of water is lost via evapo-transpiration while the Cs is retained. Thus, this leads to progressive concentration of Cs in the flesh of the fungus, a process which can continue for some time after the fruit body has reached its final size.

2) Just like K, to which it is chemically close (Bunzl & Kracke 1986, 1989, Nimis et al. 1988a), Cs is mainly found in the cytoplasm and not in the walls. During senescence of the fruit bodies, the impermeability of the cell walls progressively decreases and a certain quantity of cytoplasm (and thus of Cs) can exude from the hyphae and be washed away by rain.

2. Higher plants

The physiology and "lifestyle" of plants also influence radionuclide content. It has been observed in particular (Bunzl and Kracke 1986, 1989, Eriksson 1989, Salt & Mayes 1989) that there is a decrease over the year in radiocaesium and K levels in the aerial parts of *Gramineae*, *Cyperaceae* and *Chamaenerion angustifolium*, this being greater in May-July, i.e. during the tissue growth and maturation period. During this process, the proportion of walls in the overall weight of the plant increases considerably, which contributes to lowering the Cs concentration because this element remains mainly in the free state in the cytoplasm. Given this, the suggestion made by Nimis et al. (1988a) - to express Cs content in relation to water content (= weight of fresh matter minus weight of dry matter) rather than in relation to dry weight - is an interesting one.

The decrease in the K and radiocaesium levels continues during the summer and autumn, probably as a result of the gradual return of nutritive substances towards the roots, which continue to live during the winter while the leaves die. Henrich et al. (1989) observed a similar phenomenon in the fern *Dryopteris filix-mas* : the fronds of the current year were clearly more contaminated than the old fronds on the same plant. This return of nutritive substances towards the underground organs during the winter would be especially important in plants growing on poor soils.

The situation is quite different for species whose aerial parts, including the leaves, persist during winter. Heather (*Calluna vulgaris*), for example, displays hardly any seasonal variations in the K and radiocaesium levels measured in its leaves and stems (Bunzl & Kracke 1989).

3. Game

Significant differences in contamination levels have been observed in animals. These result from a number of factors.

The age of individuals can be of some importance (Danell et al. 1989, Johanson & Bergström 1989, Johanson et al. 1989, Lowe & Horrill 1988, Rantavaara 1987, Tataruch et al. 1989). The main difference is found between young animals (still in the growing phase) and adult individuals, the young being generally more contaminated (Table 3). This difference does not appear to be attributable to different diets, but rather to the physiology of the growing organism (faster metabolism, greater incorporation of Cs into growing tissues).

Certain differences in contamination levels have also been noted between the sexes (Danell et al. 1989). Females are usually more contaminated than males. This is also shown by the data in Table 3 (roe deer) and Table 4 (chamois). Since males are very often larger than females, this phenomenon is related to observations showing that contamination is generally inversely proportional to animal size.

Finally, unmistakeable fluctuations in contamination levels due to season have been measured (Danell et al. 1989, Eriksson 1989, Johanson et al. 1989, von Bothmer et al. 1989). These are partly due to changes in diet, but also to seasonal changes in activity and metabolism; for example, although the metabolism slows down and the quantity of nourishment absorbed decreases in winter, they both pick up again in spring.

I.8 FACTORS LINKED TO DISTRIBUTION OF RADIONUCLIDES IN THE VARIOUS PARTS OF THE INDIVIDUAL (fungus, higher plant, animal)

I.8.1. Fungi

When studying the distribution of radiocaesium isotopes in fruit bodies, a very clear and constant phenomenon is observed : radiocaesium concentrations are always higher in the cap than in the stem. The analyses carried out by us in collaboration with the Faculté des Sciences Agronomiques in Gembloux (Fraiture et al. 1989) show that the latter is usually no more than 48% of the former on average. Comparable results were obtained by Rückert & Diehl (1987) and by Heinrich (1987); those presented by Heinrich et al. (1989) vary more.

When boletus tubes were analysed separately, they displayed activity some 50% to 100% higher than those of caps minus the tubes (Rückert & Diehl 1987, Bakken & Olsen 1989); the same phenomenon is found in the gills (Heinrich et al. 1989). By contrast, the gills appear to contain less Sr than the rest of the fungus (Seeger et al. 1982).

		C	N	^{137}Cs min	^{137}Cs mean	^{137}Cs max	^{134}Cs min	^{134}Cs mean	^{134}Cs max	R
flesh	buck	A	18	74	606	1961	52	294	814	0.49
"	cow	A	14	104	632	1665	44	317	814	0.50
"	fawn	A	4	322	793	1443	141	370	666	0.47
"	unknown	A	73	0	735	3293	0	354	1480	0.48
viscera	buck	A	5	155	263	370	67	132	196	0.50
"	cow	A	4	255	280	340	130	141	148	0.50
"	fawn	A	3	481	678	962	263	338	481	0.50
"	unknown	A	14	52	469	1221	26	235	666	0.50
flesh	unknown	I	81	0	635	2744	0	239	1345	0.38
liver	"	I	33	0	412	1832	0	136	586	0.33
heart	"	I	10	174	560	1177	48	145	285	0.26
kidney	"	I	12	63	799	2171	52	361	1055	0.45
thyroid	"	I	3	37	1362	3021	0	272	792	0.20

Table 3 : Radiocaesium contamination of the flesh and viscera of roe deer (*Capreolus capreolus*) killed in Austria in June-July (-August) 1986 and in Italy in (May-) October (-December) 1986; C = country, N = number of samples, R = $^{134}Cs/^{137}Cs$ concentration ratio; the figures are given in Bq/kg FM and are based on data from the CEC Joint Research Centre in Ispra.

		C	N	^{137}Cs min	^{137}Cs mean	^{137}Cs max	^{134}Cs min	^{134}Cs mean	^{134}Cs max	R
flesh	buck	A	13	126	859	4810	48	399	2183	0.46
"	cow	A	8	74	1702	5180	41	796	2442	0.47
"	unknown	A	21	0	885	4440	0	411	1924	0.46
liver	buck	A	2	126	396	666	78	224	370	0.57
"	cow	A	2	333	629	925	185	315	444	0.50
"	unknown	A	3	222	513	1073	111	258	518	0.50
kidney	buck	A	3	352	820	1739	170	410	851	0.50
"	cow	A	2	629	1332	2035	296	629	962	0.47
"	unknown	A	2	592	703	814	296	370	444	0.53
flesh	unknown	I	8	83	199	384	26	90	198	0.45
kidney	"	I	1		196			133		0.68
thyroid	"	I	1		11211			4403		0.39

Table 4 : Radiocaesium contamination of the flesh and viscera of chamois (*Rupicapra rupicapra*) killed in Austria and Italy in May-December 1986; C = country, N = number of samples, R = $^{134}Cs/^{137}Cs$ concentration ratio; the figures are given in Bq/kg FM and are based on data from the CEC Joint Research Centre in Ispra.

Finally, the cuticle does not appear to have higher ^{137}Cs contamination than the flesh (Bakken & Olsen 1989), with the exception of cases of direct contamination and in the particular case of *Xerocomus badius*, whose pigments (norbadione) complex Cs (Aumann et al. 1989).

I.8.2. Higher plants

Radionuclide distribution in plant tissues is often uneven, in particular in perennial species with lignified parts. Henrich et al. (1989) observed radiocaesium concentrations approximately two times lower in the old parts of *Vaccinium myrtillus* than in the recently formed parts (berries, leaves and stems formed that year). Bunzl & Kracke (1986, 1989) observed similar ratios for *Calluna vulgaris*. Brown (1989) observed that Cs and Sr activity in the leaves of this species was respectively 7.2 and 5.7 times higher than that in the stems. Variations of this nature have also been observed in trees (Cooper & Mattie 1989). In 1986, Tobler et al. (1988) analysed branches of *Picea abies* and observed that 58% of the total activity was in the branch itself, 17% in the needles and 25% was adsorbed on the surface of the latter.

This is probably due to the fact that the old parts of these plants (roots, stems), are highly lignified, i.e. the cell walls make up a large proportion of their weight. Since radiocaesium isotopes mainly remain in the cytoplasm, it is normal that their content in these tissues is lower. It should be recalled that these features led Nimis et al. (1988a) to propose that Cs content in the various parts of plants no longer be expressed in terms of dry matter weight but in terms of water content, i.e. the difference between the weight of fresh matter and that of dry matter. Using this approach, Nimis et al. (1989) observed some accumulation of radiocaesium in the root system of plants growing in *Fagus* forest openings.

By contrast, Jackson (1989) found the exact opposite when studying Cs content in several *Ericaceae*, i.e. young shoots are less contaminated than stems from previous years. The figures given are all the more surprising since they indicate a decrease in the soil-plant transfer factor in 1987 as compared with 1986, although the authors are generally agreed that contamination of *Ericaceae* berries remained stable or even increased from year to year at this time (Mascanzoni 1989a, Rantavaara 1989). This disagreement may be explained by strong adsorption of radiocaesium on the plants during fallout (the environment in question having no cover), a fair amount of which was still present after the winter. The period in which the measurements were taken may also be important (see I.7.2).

In May 1987 Nylen & Ericsson (1989) measured, for *Pinus sylvestris*, clear differences in contamination in needles which grew in 1986 (500 Bq ^{137}Cs/kg DM) and older needles (4 000 Bq ^{137}Cs/kg DM). This is due to the fact that the latter received Chernobyl fallout, whereas the others were still only in the process of forming at the time the fallout occurred. A similar phenomenon was observed by Baldini et al. (1987) in *Pinus sylvestris* and *Picea abies* and by Heinrich (1987) in *Picea abies*, *Pinus nigra* and *Pinus mugo*.

Heinrich et al. (1989) studied the phenomenon in more detail in *Picea abies*. The needles formed in 1985 are more contaminated (892 Bq ^{137}Cs/kg DM) than those formed in 1986 (403 Bq/kg), which were in turn more contaminated than those formed in 1987 (185 Bq/kg). This principle holds true for the five radionuclides of man-made origin analysed by these authors, but not for ^{40}K, a natural radionuclide, which is more abundant the younger the needles (respectively 207, 222 and 485 Bq ^{40}K/kg). It is normal for K to be more abundant in the young needles than in the old ones, because the latter have thicker walls and lower metabolic activity. If the radiocaesium isotopes do not have the same distribution, although plants use them as the homologue of K, this is because a large quantity of these radioisotopes remains adsorbed on the surface of the needles following fallout and, in old needles, add to the much lower quantities contained in the cytoplasm.

Finally, Hugon et al. (1991) measured two gradients in the activities of needles of a spruce subjected to contaminated rain. The first of these is vertical : the activity in the needles is higher the further one moves away from the base of the tree up towards the crown; the second is horizontal : the contamination of the needles is greater the further one moves away from the trunk out towards the edge of the crown. This phenomenon is due to the conical form of the spruce, with the outside layer of the crown intercepting most of the precipitation. The same authors observed, however, that spruce trees are contaminated in a uniform manner by dry deposition under non-turbulent conditions, contamination per unit weight of needles being a function of the surface area of the latter.

When studying contamination of several species of plants producing edible berries, Heinrich (1987) observed that the leaves of these plants were much more contaminated than the fruit. These analyses were carried out in 1986 and this phenomenon is probably attributable to the fact that the leaves, in contrast to the fruit, received a large amount of direct contamination. This author also observed that the skin of the fruit is more contaminated than the pulp. This latter observation was also made by Schelenz & Abdel-Rassoul (1986).

I.8.3. Game

Johanson & Bergström (1989) did not observe any difference in ^{137}Cs contamination in the different muscles of elk (neck, thigh, leg). An analysis of data from the CEC Joint Research Centre in Ispra covering roe deer (Table 3) and chamois (Table 4) contamination in Austria and in Italy shows that on average offal is less contaminated than muscle. Nevertheless, the kidneys generally have Cs contamination close to that for flesh, while the thyroid has Cs contamination which is much greater still. Moreover, the thyroid accumulates radioiodine in large amounts.

II. EDIBLE WILD PLANTS IN FOREST ECOSYSTEMS

II.1. INTRODUCTION

This chapter contains data collected on higher plants growing in a forest ecosystem and likely to be consumed by man. In practice, we concentrated on species used as foods. Nevertheless, we have also included plants used to make beverages such as teas, herbal teas, "wines" and alcoholic drinks. On the other hand, we have ignored the innumerable species likely to be gathered for medicinal purposes or consumed only occasionally.

Here we deal mainly with plants with edible fruit (berries of *Rosales* and of *Ericales*, seeds of various *Fagales*). For the main plants we give below their names in Latin, English, French, German, Italian, Spanish and Dutch, an overview of their geographical distribution and main uses in human diet, together with a summary of data collected by us on the contamination levels found.

We consulted the following :

- Latin names : Clapham et al. (1981), De Langhe et al. (1983), Fiori (1969), Polunin (1959) and Rothmaler et al. (1976);
- English names : Clapham et al. (1981), Gove (1986), Mansion (1961) and Polunin (1959);
- French names : De Langhe et al. (1983);
- German names : De Langhe et al. (1983), Rothmaler et al. (1976) and Sachs-Villate (1964);
- Italian names : Fiori (1969);
- Dutch names : De Langhe et al. (1983) and Gallas (1985);
- for the distribution maps : Hulten & Fries (1986), Jalas & Suominen (1976) and Meusel et al. (1965, 1978);
- for the culinary uses : Couplan (1983);
- for berry production figures : Belonogova (1988), Cherkasov (1988), Kujala (1988) and Raatikainen (1988);
- references covering radioactive contamination of edible higher plants are given in the relevant section.

II.2. REVIEW OF MAIN EDIBLE SPECIES, PRESENTATION AND CONTAMINATION LEVELS

VACCINIUM MYRTILLUS L.

Order : *Ericales*
Family : *Ericaceae*

English : Whortleberry, Huckleberry, Whinberry, Bilberry, Blueberry
French : Myrtille
German : Heidelbeere, Blaubeere
Italian : Mirtillo, Bagolo, Baggiole
Spanish : Arandano, Raspanera, Arandanera, Mirtila
Dutch : Blauwe bosbes

Geographical distribution : temperate and cold zones in Europe and north-east Asia (see map in Meusel et al. 1978). In Europe it is found everywhere, with the exception of the Mediterranean and southern Iceland.

Main uses : The berries are used to make jams, tarts/pies and cakes ("blueberry muffins"); they can also be eaten raw.

Blueberries are one of the most collected berries in Europe and the quantities used are relatively large. By way of example, we can quote Raatikainen (1988) who collected data on the yield of this species in Finland : according to estimates, the wild areas there produce 42 000 to 200 000 t of blueberries per year. Kujala (1988) reports that only 700 to 5 000 t of these berries found their way onto the market each year from 1977 to 1986. However, these statistics do not take into account the quantities harvested privately and non-commercially. Finally, Belonogova (1988) notes that 100 000 t of blueberries are produced each year in Karelia (USSR) but that only 1% of this quantity is harvested.

Contamination levels observed :

Rantavaara (1987) gives the analysis results for 67 samples collected in Finland in 1986. The levels (in Bq/kg FM) vary from 0 to 170 for ^{134}Cs and from 5 to 341 for ^{137}Cs. Other radionuclides have been detected in a small number of samples from the worst-affected areas of the country : the levels (in Bq/kg FM) go up to a maximum of 0.8 for ^{95}Nb, 4.9 for ^{95}Zr, 3.1 for ^{103}Ru, 2.7 for ^{141}Ce and 2.8 for ^{144}Ce.

Mascanzoni (1987, 1989a) reports the result of analyses carried out in Sweden. The ^{137}Cs concentrations observed ranged from < 2 to 1 130 Bq/kg FM (average = 150, for 367 samples) in 1986 and from 12 to 1 920 Bq/kg FM (average = 248, for 46 samples) in 1987.

Liljenzin et al. (1988) give the following figures : for 216 samples collected in Sweden in 1986, an average of 1.67 Bq/kg (DM?) for ^{134}Cs and 1.99 Bq/kg (DM?) for ^{137}Cs (figures corrected for accident date; the unit should be kBq, no doubt).

Nelin & Palo (1989) measured the contamination of stems of *Vaccinium myrtillus* collected in Sweden; they contained approximately 800 Bq of ^{137}Cs/kg DM in 1986, approximately 400 Bq/kg in 1987 and approximately 330 Bq/kg in 1988. For the same type of material Palo et al. (1989) give figures of 1 480, 810 and 521 Bq ^{137}Cs/kg DM for 1986, 1987 and 1988 respectively.

Henrich et al. (1989) note harvested blueberries containing from 2 400 to 8 200 Bq/kg DM in Austria in 1988. These researchers decided to take *Vaccinium myrtillus* as the indicator plant because it is very abundant in the type of forest they study, because it grows everywhere in Austria up to an altitude of 2 000 m, because it is easy to find, is eaten by man (berries) and by game, and because it is a perennial shrub and thus constitutes a good indicator for monitoring contamination in the long term.

A communication from the Ludwig-Maximilians-Universität in Munich (1986) contains the results of an analysis of 13 samples in 1986. The levels, expressed in Bq/kg (FM?), range from 50 to 1 265 for Cs and from 0 to 21 for Ru.

It is interesting to note that from 1984 onwards Bunzl & Kracke (1986) found high ^{137}Cs contamination in this plant in Bavaria : 560 Bq/kg DM in the berries, 740 in the leaves, 590 in the flowers, 370 in the stems and 430 in the roots. These were collected from a very acid peat bog (pH = 2.8), which no doubt explains such high levels. The same authors also note that four samples collected during this time in various types of German forest gave readings of only 47 + 21 Bq/kg DM.

In Belgium, we received two sets of results from the IHE [= Hygiene and Epidemiology Institute] covering harvests in 1987. They correspond to < 5.1 and 13 Bq/kg FM for ^{134}Cs and 26.7 and 40 Bq/kg FM for ^{137}Cs.

In July 1989 we collected several samples of blueberries in Belgium. The result of the analyses carried out by the Cyclotron Laboratory at Liège University is given below (in Bq/kg DM, the FM contamination being 7 times lower on average).

No	Place	^{134}Cs	^{137}Cs	$^{134}Cs/^{137}Cs$
2	Chiny	11.5	59	0.19
3	Rossignol	34	179	0.19
6	Jalhay	24	150	0.16
8	Samrée	73	583	0.13
14	Couvin	15	44	0.34

These results show strong concordance with the level of fallout in Belgium's various regions, apart from No 8, whose very high level is attributable to the plants having grown in a peat bog (Fange aux Mochettes). The $^{134}Cs/^{137}Cs$ concentration ratio indicates that the roots of this species are not located very deep in the soil and tend to flourish at the bottom of the organic horizons.

VACCINIUM ULIGINOSUM L.

Order : *Ericales*
Family : *Ericaceae*

English : Bog whortleberry, Bog blueberry, Bog bilberry
French : Myrtille de loup
German : Rauschbeere, Trunkelbeere, Moor-Heidelbeere
Italian : Bagolo, Baggiole
Spanish : Arandano negro
Dutch : Rijsbes

Geographical distribution : temperate and cold zones of the northern hemisphere (see map in Meusel et al. 1978). In Europe this species is practically absent from the Mediterranean, Ireland and most of France, Belgium, the Netherlands and Britain.

Main uses : the berries are edible, being somewhat less tasty than blueberries. They are reputed to be slightly toxic when consumed in large quantities.

Contamination levels observed :

Very little data were found on this species.

Rantavaara (1987) gives the results of an analysis of one sample of this species collected in 1986 : 5.6 Bq/kg FM for ^{137}Cs and practically no measurable ^{134}Cs. This sample came from northern Finland, a region not much affected by fallout.

Furthermore, we should note that Bunzl & Kracke (1986) report fairly high levels detected in 1984 in a peat bog in southern Bavaria : 380 Bq/kg DM for the berries, 510 for the leaves, 740 for the flowers, 200 for the stems and 270 for the roots. These figures are slightly lower than those for the blueberries collected at the same spot (see above).

VACCINIUM VITIS-IDAEA L.

Order : *Ericales*
Family : *Ericaceae*

English : Red whortleberry, Lingberry, Lingonberry, Foxberry, Mountain cranberry, Cowberry
French : Airelle
German : Preiselbeere, Kronsbeere
Italian : Vite d'orso, Vite idea
Dutch : Rode bosbes

Geographical distribution : temperate and cold zones in the northern hemisphere (see map in Meusel et al. 1978). In Europe it is to all intents and purposes absent from the Mediterranean and Iceland, from most of France, Belgium and the Netherlands, as well as from southern England and Ireland.

Main uses : the berries are used to make jams and compots.

This species is widespread and very much a favourite in certain regions. Raatikainen (1988) gives several estimates of the quantities produced in natural habitats in Finland. These range from 70 000 to 500 000 t per year. Kujala (1988) notes that from 1977 to 1986 1 800 to 10 000 t of these berries were sold in Finland. These figures do not include those harvested for personal use. Belonogova (1988) gives a figure of 80 000 t for the yield of this type of berry in Karelia (USSR), with the proportion harvested by local populations being no more than 1%.

Contamination levels observed :

Rantavaara (1987) reports the result of analysis of 41 samples harvested in 1986 in Finland. The levels observed (in Bq/kg FM) range from 0 to 330 for ^{134}Cs and from 4 to 630 for ^{137}Cs. Other radionuclides were also detected at the same time. These come for the most part from the worst contaminated regions of Finland : up to 10 for ^{95}Nb, up to 8.0 for ^{95}Zr, up to 14 for ^{103}Ru, up to 11 for ^{141}Ce, up to 8.9 for ^{144}Ce, up to 41 for ^{106}Ru and up to 1.3 for ^{110m}Ag (figures in Bq/kg FM).

Mascanzoni (1987, 1989a) gives the levels observed in Sweden for ^{137}Cs (in Bq/kg FM) : in 1986 they ranged from < 2 to 904 (average = 187, for 343 samples) and from 11 to 1 010 (average = 214, for 51 samples).

A communication from the Ludwig-Maximilians-Universität in Munich (1986) gives the results of measuring 2 samples collected in 1986. The first consisted of berries and contained 132 Bq Cs/kg (FM?) and 14 Bq Ru/kg. The second was marmalade and contained 75 Bq Cs/kg and 1 Bq Ru/kg.

The CEC Joint Research Centre in Ispra provided us with data on 19 samples collected in Austria in 1986. The figures, expressed in Bq/kg (FM?), range from 7.4 to 174 for ^{134}Cs, from 7.4 to 363 for ^{137}Cs and from 11 to 26 for ^{103}Ru.

Bunzl & Kracke (1986) give the following concentrations for the various parts of this plant collected in 1984 in a peat bog in southern Bavaria (in Bq ^{137}Cs/kg DM) : 420 for the fruit, 840 for the flowers, 310 for the leaves, 240 for the stems and 260 for the roots.

Finally, for Belgium we have two results provided by the IHE and expressed in Bq/kg FM : < 1.9 and 14.4 for ^{134}Cs and 16.8 and 36.2 for ^{137}Cs in samples collected in 1986.

VACCINIUM OXYCOCCOS L.

Syn : *Oxycoccus palustris* Pers.; *O. quadripetalus* Br.-Bl.

Order : *Ericales*
Family : *Ericaceae*

English : Cranberry, Swamp cranberry
French : Canneberge
German : Gemeine Moosbeere
Italian : Ossicocco palustro
Spanish : Canaheja, Arandano agrio
Dutch : Veenbes

Geographical distribution : temperate and cold zones of the northern hemisphere (see map in Meusel et al. 1978). In Europe the species is practically absent from the Mediterranean, Iceland and from most of France, Belgium and the Netherlands.

Main uses : the berries are edible and can be used to make compot.

Raatikainen (1988) reports several estimates regarding the yield of this species and of *V. microcarpum* in Finland : these vary between < 25 000 and 50 000 t per year. It is probable that only a small fraction (1% ?) of these quantities are harvested by the population.

Cherkasov (1988) states that this species yields on average 200 kg of berries per hectare in the USSR but that, at the best sites, yield may be as high as 2 600 kg per ha. The most suitable areas for this species are in the north-western part of the USSR

Contamination levels observed :

Rantavaara (1987) gives the results of analysing 8 samples harvested in Finland in 1986. The levels, expressed in Bq/kg FM, range from 0.8 to 274 for ^{134}Cs and from 4.1 to 530 for ^{137}Cs. One sample contains 7.7 Bq/kg FM of ^{103}Ru.

Mascanzoni (1987, 1989a) gives the concentrations of ^{137}Cs observed in two samples of berries harvested in Sweden in 1986. These are 281 and 327 Bq/kg FM.

Liljenzin et al. (1988) reproduce statistics based on analysis results for 309 samples collected in Sweden in 1986. The average levels, expressed in Bq/kg (DM?; this is probably an oversight for kBq) are 1.85 for ^{134}Cs and 2.12 for ^{137}Cs.

Finally, it is interesting to note that Bunzl & Kracke (1986) took samples of this species from a peat bog in southern Bavaria in 1984. The levels observed at that time for ^{137}Cs were (in Bq/kg DM) 330 in the berries, 440 in the flowers, 380 in the leaves and 210 in the stems.

EMPETRUM NIGRUM L.

Order : *Ericales*
Family : *Ericaceae*

English : Crowberry, Common crowberry, Black crowberry
French : Camarine noire
German : Gemeine Krähenbeere
Dutch : Kraaiheide

Geographical distribution : temperate and cold zones of the northern hemisphere (see map in Meusel et al. 1978). In Europe this species is practically limited to the northern half of the temperate zone, as well as to Finland, Iceland and the north-western part of the USSR.

Main uses : the berries are edible; Raatikainen (1988) estimates yield in the wild in Finland at 350 000 t of berries per year. We have no indication of how much of this is harvested by the public.

Contamination levels observed :

We have only one analysis result (Rantavaara 1987). This is 70 Bq/kg FM for ^{137}Cs, with no ^{134}Cs being detected. The sample was collected in 1986 in northern Finland, a region which received very little fallout, which explains the latter finding.

RUBUS IDAEUS L.

Order : *Ericales*
Family : *Rosaceae*

English : Raspberry bush, Raspberry cane (fruit : raspberry)
French : Framboisier (fruit : framboise)
German : Himbeere (fruit : id.)
Italian : Lampone, Framboè
Spanish : Frambueso
Dutch : Framboos (fruit : id.)

Geographical distribution : temperate and cold zones of the northern hemisphere (see map in Meusel et al. 1978). This species is found in almost all of Europe, with the exception of Iceland, southern Spain and certain Mediterranean areas.

Main uses : the fruit are edible and very tasty. They can be eaten raw or turned into jams (jellies), tarts/pies, etc. This species is widely planted and numerous cultivars exist.

Contamination levels observed :

Rantavaara (1987) gives contamination levels observed for 8 raspberry samples collected in Finland in 1986 (in Bq/kg FM) : these range from 3.7 to 47 for ^{134}Cs and from 6.2 to 91 for ^{137}Cs. Figures are also given for cultivated raspberries (10 samples) : they range from 1.9 to 37 for ^{134}Cs and from 3.5 to 71 for ^{137}Cs.

Mascanzoni (1987, 1989a) provides statistics on ^{137}Cs concentrations observed in Sweden (in Bq/kg FM) : these vary from < 2 to 945 in 1986 (average = 115, for 198 samples) and from < 2 to 613 in 1987 (average = 213, for 11 samples).

Liljenzin et al. (1988) give a summary of analysis results for 166 samples harvested in Sweden in 1986. On average the figures are 1.61 Bq/kg (DM? - probably an oversight for kBq) for ^{134}Cs and 1.91 Bq/kg (DM?) for ^{137}Cs.

The CEC Joint Research Centre in Ispra has passed on to us the results of an analysis of 170 samples collected in Austria in 1986. The results give levels, in Bq/kg (FM?), which vary between 4 and 185 for ^{134}Cs (except for one sample which gave a reading of 1 739), between 4 and 370 for ^{137}Cs (except for one result of 2 997), between 0 and 41 for ^{131}I and between 11 and 30 for ^{103}Ru.

The Radioprotection Division in the Grand-Duchy of Luxembourg has sent us a number of analysis results. In 1986 contamination of 6 samples, expressed in Bq/kg (FM?), varied from 4.5 to 28 for ^{134}Cs and from 9.5 to 67 for ^{137}Cs. In 1987 the corresponding figures were <1 and <3 for three other samples.

A communication from the Ludwig-Maximilians-Universität in Munich (1986) covers analysis of 11 samples. The levels vary between 54 and 773 Bq/kg (FM?) for Cs and between 0 and 13 Bq/kg (FM?) for Ru. Measurements of jam gave, respectively, 170 and 0 Bq for these two elements, while measurements of raspberry brandy gave 89 to 110 and 1 to 2 Bq/kg.

We collected several samples of raspberry in Belgium in July 1989. The results are given below (in Bq/kg DM, measured by the Cyclotron Laboratory at Liège University).

No	Place	^{134}Cs	^{137}Cs
1	Chiny	0.8	4.5
4	Chiny	11	54
5	Strainchamps	8.5	25
7	Polleur	3.3	6

The difference in concentration observed in the two samples from Chiny is clear proof of the influence of the soil : sample No 1 was taken from an area without cover on the edge of pastureland on fairly rich soil, whereas sample No 4 was taken from a felling site at the edge of a spruce stand on fairly acid soil.

RUBUS FRUTICOSUS L. s.l.

Order : *Rosales*
Family : *Rosaceae*

Observation : Collective species, nowadays divided into numerous species which are difficult to distinguish.

English : Blackberry bush, Bramble (fruit : Blackberry, Brambleberry)
French : Ronce (fruit : mûre, not to be confused with the fruit of the *Morus*, Asiatic shrubs introduced to Europe)
German : Brombeere (fruit : id.)
Italian : Rovo, Rogo (fruit : More)
Spanish : Zarza
Dutch : Braambes (fruit : id.)

Geographical distribution : taken as a whole, the various species grouped under the name *R. fruticosus* cover the whole of EEC territory.

Main uses : the fruit of several species are edible and tasty. They can be eaten raw, in jams (jelly) or in tarts/pies.

Contamination levels observed :

Mascanzoni (1987, 1989a) gives an analysis result obtained in Sweden in 1986 for ^{137}Cs of 64 Bq/kg FM in the fruit.

Juzňič (1987) recorded Sr levels in samples harvested on 6 July (1986?) : 15 Bq/kg (FM?) for ^{89}Sr + ^{90}Sr and 7.6 Bq/kg (FM?) for ^{90}Sr.

The Radioprotection Division in the Grand Duchy of Luxembourg has communicated to us the following analysis results (in Bq/kg FM?) : in 1986 (4 samples) the levels varied from 5 to 19 for ^{134}Cs and from 9 to 37 for ^{137}Cs; in 1987 (3 samples, other collection areas) from < 0.1 to 0.31 and from 0.17 to 0.74 respectively.

The listing from the Ludwig-Maximilians-Universität in Munich (1986) contains the results of analysis of 3 samples collected in 1986 : the Cs content varies from 42 to 233 Bq/kg (FM?) and Ru content from 0 to 14.

The CEC Joint Research Centre in Ispra provided us with 17 readings for samples collected and analysed in 1986 in Austria. These are 7 to 89 Bq/kg (FM?) for ^{134}Cs, 7 to 237 for ^{137}Cs and 11 to 26 for ^{103}Ru.

The Cyclotron Laboratory at Liège University analysed two samples collected in Belgium in August 1989 by us. The results (in Bq/kg DM) are as follows :

No	Place	^{134}Cs	^{137}Cs
11	Sart-lez-Spa	4.5	7.1
12	Francorchamps	5.6	16

RUBUS CHAMAEMORUS L.

Order : *Rosales*
Family : *Rosaceae*

English : Cloudberry, Baked-apple (fruit : id.)
German : Moltebeere (fruit : id.)

Geographical distribution : cold parts of the northern hemisphere (see map in Meusel et al. 1978). In Europe, the species is practically confined to Fennoscandia and the northern half of USSR and Britain.

Main uses : the fruit are edible and used in Scandinavia to make tarts/pies and other desserts.

Raatikainen (1988) has collected various estimates, putting at between 25 000 and 30 000 t the annual yield of this species in Finland. According to Kujala (1988), the amounts sold there are generally below 1 000 t, but this does not include consumption of berries not sold commercially.

Contamination levels observed :

Rantavaara (1987) gives the results of analysis of 24 samples collected in Finland in 1986. The levels recorded (in Bq/kg FM) range from 0 to 118 for ^{134}Cs and from 12 to 280 for ^{137}Cs. A number of other radioisotopes were occasionally detected : the maximum levels recorded were 14 for ^{95}Nb, 5.5 for ^{95}Zr, 2.3 for ^{103}Ru, 2.1 for ^{141}Ce and 22 for ^{144}Ce (figures in Bq/kg FM).

Mascanzoni (1987, 1989a) summarises data stemming from numerous analyses of ^{137}Cs in Sweden. The levels recorded (in Bq/kg FM) vary from < 2 to 2 750 (average = 362, for 446 samples) in 1986 and from 26 to 1 780 (average = 437, for 45 samples) in 1987.

Liljenzin et al. (1988) give the following averages in Bq/kg (DM?; probably an oversight for kBq) for 218 samples harvested in Sweden in 1986 : 1.94 for ^{134}Cs and 2.30 for ^{137}Cs.

FRAGARIA VESCA L.

Order : *Rosales*
Family : *Rosaceae*

English : Wild strawberry, (fruit : id.)
French : Fraisier sauvage (fruit : fraise des bois)
German : Wald-Erdbeere (fruit : id.)
Italian : Fragola
Spanish : Fresera, Viruébano, Viruégano, Miruéndano (fruit : Fresa silvestre)
Dutch : Bosaardbei (fruit : id.)

Geographical distribution : Europe, western and central Asia, Macronesia (see map in Meusel et al. 1978). In Europe the species is found everywhere, except for northern Fennoscandia, southern Spain and southern Greece.

Main uses : the fruit can be eaten raw, in jams or in tarts/pies. The species is sometimes planted and should not be confused with the cultivated variety (*F.* x *ananassa*), whose fruit are usually much larger.

Contamination levels observed :

Mascanzoni (1987, 1989a) summarises observations made in Sweden. Content in ^{137}Cs (given in Bq/kg FM) varies from < 2 to 708 (average = 141, for 37 samples) in 1986 and is 60 and 123 in 1987 (2 samples). It is interesting to note that the figures for cultivated strawberries (*Fragaria* x *ananassa*) range from < 2 to 140 (average = 32, for 90 samples) in 1986 and are < 2 and 3 (2 samples) in 1987. Thus, contamination of cultivated strawberries is much lower than that of wild strawberries, largely due to the richness of the soils in which the former are grown.

Liljenzin et al. (1988) reproduce the results of analyses done in Sweden in 1986 on 3 samples : the average levels observed are 2.06 Bq/kg DM (?) for ^{134}Cs and 2.33 Bq/kg (DM?) for ^{137}Cs. Measurements of 6 samples of cultivated strawberries gave readings of 1.29 and 1.51 Bq/kg (DM?; perhaps an oversight for kBq) respectively.

The results from the Radioprotection Division of the Grand Duchy of Luxembourg give 3.7 to 12.8 Bq/kg (FM?) for ^{134}Cs and 7.2 to 25 for ^{137}Cs in 3 samples in 1986, and < 0.1 for ^{134}Cs and < 0.1 to 0.13 for ^{137}Cs in 3 other samples in 1987.

A listing from the Ludwig-Maximilians-Universität in Munich (1986) gives levels (in Bq/kg FM?) of 98 to 560 for Cs and of 0 to 1 for Ru in 3 samples of wild strawberries in 1986. Thirteen samples of cultivated strawberries gave levels of 0 to 150 for Cs and 0 to 15 for Ru.

The data from the CEC Joint Research Centre in Ispra contain some 60 readings concerning strawberries (wild?) measured in Austria 1986. The levels vary from 0 to 122 Bq/kg (FM?) for ^{134}Cs, from 0 to 263 for ^{137}Cs, from 10 to 30 for ^{103}Ru and from 0 to 26 for ^{131}I.

MISCELLANEOUS

Fruit of other species of the order Rosales are likely to be consumed : *Rubus* div. sp., *Rosa* div. sp., *Prunus* div. sp., *Sorbus* div. sp., etc. We have some analysis results concerning :

***Rubus arcticus* (arctic bramble)** : Mascanzoni (1987) summarises the results of analysing 9 samples collected in Sweden in 1986 : the ^{137}Cs levels vary from 29 to 320 Bq/kg FM (average = 130).

***Rosa* sp. (rose)** : The Radioprotection Division of the Grand Duchy of Luxembourg provided us with an analysis of fruit from the dog rose (*Rosa canina* L. s.l.) collected in 1987 : 2.6 Bq/kg (FM?) for ^{134}Cs and 7.9 Bq/kg (FM?) for ^{137}Cs.

***Prunus spinosa* (blackthorn, sloe)** : The same source gives figures for blackthorn harvested in 1987 : 1.5 Bq/kg (FM?) for ^{134}Cs and 3.9 Bq/kg (FM?) for ^{137}Cs.

Sorbus intermedius : Mascanzoni (1987) reproduces the results of analyses carried out on material collected in Sweden in 1986. The ^{137}Cs levels vary from < 2 to 190 Bq/kg FM (average = 69 for 3 samples).

Other forest species have edible fruits. We have the following figures :

***Sambucus* sp. (elder)** : The Radioprotection Division of the Grand Duchy of Luxembourg analysed elderberries. In 1986 a sample gave a reading of 5 Bq/kg (FM?) for ^{134}Cs and 11.1 Bq/kg (FM?) for ^{137}Cs. Two samples analysed in 1987 contained 0.25 and 1 Bq ^{134}Cs/kg (FM?) and 0.45 and 2.8 Bq ^{137}Cs/kg (FM?).

The CEC Joint Research Centre in Ispra passed on to us the results of 38 analyses carried out in Austria in 1986. The levels observed (in Bq/kg FM?) vary from 7 to 229 for ^{134}Cs, from 11 to 481 for ^{137}Cs, from 11 to 137 for ^{131}I and from 11 to 56 for ^{103}Ru.

***Corylus avellana* (hazel, cob-nut)** : In 1987 the Radioprotection Division of the Grand Duchy of Luxembourg measured 5.9 Bq/kg (FM?) of ^{134}Cs and 16.4 Bq/kg (FM?) of ^{137}Cs in a sample of such nuts.

Roca et al. (1989) studied the contamination of *Corylus avellana* in Campania (southern Italy), a region where this bush is much grown : in 1986 the leaves contained 300-400 Bq/kg, the green nuts (in May) 60-80 Bq/kg and ripe nuts (October) 300-400 Bq/kg. In 1987 all these levels had dropped to 20-40 Bq/kg.

It should be pointed out that the levels were much higher in nuts from Turkey and greatly exceeded the threshold of 600 Bq/kg (Agence Belga 1987). This country is the largest producer of nuts in the world (annual yield : 300 000 t, i.e. 70% of world production) and the Chernobyl accident led to serious economic problems there.

Other wild plants can be used as vegetables. The species used most frequently in this way is ***Urtica dioica*** **(nettle)**. Rantavaara (1987) gives results of analysing 4 nettle samples; one of them, collected on 5 May 1986, had received a large amount of direct fallout and totalled 12 565 Bq/kg FM, including 4 880 Bq of ^{131}I. However, contamination of the three other samples did not exceed 180 Bq for the total of man-made radionuclides.

Mascanzoni (1987, 1989a) gives a summary of the results of analysing nettles collected in Sweden in 1986. The ^{137}Cs levels measured range from 13 to 4 600 Bq/kg FM (average = 420 for 15 samples). Liljenzin et al. (1988) give an average of 1.86 Bq/kg DM (oversight for kBq?) for 15 nettle samples harvested in Sweden in 1986.

Thymus vulgaris **(thyme)**, used as seasoning or medicinal plant (infusions), can have fairly high levels of radioactivity. Grauby and Foulquier (1987) give the following values (in Bq/kg FM) for the total activity of man-made radioisotopes in thyme in the Cadarache region in France : prior to the Chernobyl accident : 200; 13 May 1986 : 4 000; but only 2 000 on 3 June and 1 600 on 1 July. These figures are averages, there being wide variations between the various harvests. These authors add that in order to reach the limit set down in Directive 86/836 EEC, an individual would have to eat 150 kg of thyme or drink 6 200 litres of infusion per year.

These high levels are not too worrying, though, because this plant is consumed in minute amounts only. It should be recalled here that prolonged consumption of large quantities of thyme could seriously damage health.

Tea, when harvested in a highly contaminated region, can contain radioactivity exceeding EEC limits. In 1986, a stockpile of 75 000 t of contaminated tea (including 30 000 t with a very high level of radioactivity) was built up in Turkey pending a decision on what to do with it. The value of this stockpile of tea was put at 190 million dollars (Agence Belga 1987).

On the other hand, Yule and Taylor (1989) studied the transfer of radioactivity into the infusion. They used tea from Turkey, which had an activity of 14 400 Bq/kg. The infusion they made contained 45% of the Cs contained in the tea. This Cs was mainly present in cationic form, i.e. in a form easily absorbable by the digestive tract. On the basis of this data they estimate that someone drinking 1/2 litre of this tea every day would, at the end of a year, have received an effective dose equivalent of 0.3 mSv. Further details concerning contamination of Turkish tea and the consequences for human contamination are contained in Gedikoğlu & Sipahai (1989) and Hayball et al. (1989).

Honey produced by bees gathering nectar in a natural habitat might be contaminated to some degree. A listing from the Radioprotection Division of the Grand Duchy of Luxembourg gives activities of 5.5 to 115 Bq/kg for Cs there in 1986 (33 samples), from <1 to 65 in 1987 (7 samples) and <1.5 in 1988 (2 samples). Analyses carried out in 1986 in Rhineland-Palatinate give 11 to 118 Bq/l for ^{137}Cs and 4 to 241 Bq/l for ^{131}I.

The Ludwig-Maximilians-Universität in Munich (1986) analysed 30 samples of honey in 1986. The activities recorded vary from 2 to 865 Bq/kg for Cs and from 0 to 127 Bq/kg for Ru.

Radioactivity in honey had already been studied by Bunzl & Kracke in 1981. These authors observed the presence of ^{40}K, ^{137}Cs, ^{90}Sr and $^{239/240}Pu$. The activities measured in honey from heather and expressed in Bq/kg were respectively 55.5, 52.2, 0.17 and 0.0018 for these four radionuclides. They also noted that the type of honey greatly influenced the contamination and that if honey is to be used as a bioindicator, its composition has to be known. The transfer of various radionuclides from flowers to honey has also been studied by Bunzl & Kracke (1988a).

II.3. SUMMING-UP AND DISCUSSION

Of the forest plants likely to be consumed by man, the most common are the edible fruit species, in particular wild berries. Among these, those mostly represented are plants belonging to the genera *Vaccinium* (*Ericaceae*) and *Rubus* (*Rosaceae*). These species are widely used in some regions (northern Europe, for example). How they grow and the factors determining their yield are being studied there (Vanninen & Raatikainen 1988), and efforts are being undertaken by the public authorities in certain countries to encourage people to harvest the edible wild products (Härkönen 1988). The data in this chapter therefore mainly cover these plants, although some figures are given for other types of plants, such as those used as a vegetable or to make various teas or infusions.

The radionuclide concentrations observed in edible plants in general vary as follows (Rantavaara 1987) : the lowest concentrations are found in plants whose fruit, leaves or roots are used as vegetables, as well as in potatoes (0.3 to 8 Bq/kg FM on average). Higher concentrations (10 to 30 Bq/kg FM) have been observed in cultivated berries and activities of 80 to 120 Bq/kg FM in wild berries.

A number of authors note greater contamination in *Ericaceae* (cf. I.6.2.) than in the other wild plant species (Bunzl & Kracke 1986, 1989, Horrill et al. 1989, von Bothmer et al. 1989). It is possible that the contamination levels in the different species depend more on the depth of their roots in the soil than on differences in the physiological mechanisms of absorption (Belli et al. 1989). It is, however, interesting to note that *Ericales* have a very special type of mycorrhizae, viz. ericoid mycorrhizae (Strullu 1985). It is possible that these structures are a factor promoting absorption of radionuclides by the roots. Rogers & Williams (1986) have demonstrated the favourable influence of another type of vesicular-arbuscular mycorrhizae (VAM) on uptake of ^{137}Cs by *Melilotus officinalis* and of ^{60}Co by *Sorghum sudanense.*

Furthermore, a relatively high contamination level has also been detected in elderberries (*Sambucus*), but this species is consumed less extensively.

Edible berries are as a rule produced by perennial species with lignified stems. The activity measured in the berries is often of the same order as that observed in the leaves or flowers, but is, in contrast, higher (often two times higher) than that in the lignified parts of the plant (stems, roots) (Bunzl & Kracke 1986). This phenomenon is probably due to the extensive development of cell walls in these organs, which reduces - in the overall weight of the plant - the proportion of cytoplasm and thus of Cs (since this element is found in the free state in cytoplasm) (Nimis et al. 1988a).

Contamination of wild edible plants, even in contaminated regions, usually remains far below the European limit of 600 Bq/kg, at least in terms of **average** figures. Nevertheless, this limit has been greatly exceeded in certain places, with levels of several thousand Bq/kg being frequently observed.

Some 15 radionuclides were detected, but only ^{131}I, ^{134}Cs and ^{137}Cs were found in important concentrations. Since August 1986 radiocaesium isotopes have been practically the only ones still observed.

Contamination trends in wild berries from 1986 to 1989 show that contamination remained stable or even increased from year to year (Eriksson 1989, Mascanzoni 1989a, Rantavaara 1989, von Bothmer et al. 1989). This phenomenon is no doubt linked to the fact that the contamination moved downwards in the soil and thus increased at the plant root level. Furthermore, we are dealing with perennial species which can retain a certain amount of contamination in their tissues from one year to the next.

Another interesting fact, reported by Rantavaara (1989), is that the contribution of products growing in the wild to the dose absorbed annually by the population has increased from year to year, because the contamination levels in agricultural products have dropped sharply since 1986.

Apart from the amount deposited, the factor most influencing the level of contamination is the type of soil :

- Cultivated soils are usually very rich by nature and as a result of the fertiliser added to them. The availability of radionuclides in them is thus low, and wild species, when grown therein, produce fruit which is much less contaminated than that harvested in the wild (Rantavaara 1987).

- Soil richness also plays an important role in radionuclide availability in other soils, apart from cultivated ones. For example, many species cited by us above also grow outside the forest. Berries collected in such areas are generally less contaminated than those harvested in the same region but under forest canopy.

Another example showing the important role of soil richness is that of peat. With the exception of eutrophic peat bogs, peaty areas are generally very acid. Furthermore, the soil there almost exclusively consists of organic matter, and no clay minerals are present. Consequently, the availability of radiocaesium isotopes is at its highest there, and it comes as no surprise that plants growing occasionally or always (*Rubus chamaemorus*) in such places develop high contamination levels (Bunzl & Kracke 1986, Heaton et al. 1989, Johanson et al. 1989, Mascanzoni 1989a).

III. ANIMAL SPECIES CONSUMED AS GAME

III.1. INTRODUCTION

The reference works used are as follows :

- for the Latin and vernacular names of mammals : van den Brink & Barruel (1971);
- for the Latin and vernacular names of birds : Peterson et al. (1972);
- bibliographic references covering the diet of game species and the radioactive contamination observed in them are given in the text for each species.

III.2 REVIEW OF MAIN GAME SPECIES, PRESENTATION AND CONTAMINATION LEVELS OBSERVED

CAPREOLUS CAPREOLUS (L.)

Order : *Artiodactyles*
Family : *Cervidae*

English : Roe deer (male : Roe buck)
French : Chevreuil (female : Chevrette; male : Brocard; young : Faon)
German : Reh (female : Rehgeiss; male : Rehbock; young : Rehkitz)
Italian : Capriolo
Spanish : Corzo
Dutch : Ree

Geographical distribution : occurring in most of Europe but absent from Iceland, Ireland, northern Fennoscandia and most of the Mediterranean and Britain (after a map by van den Brink & Barruel 1971).

Diet :

Data concerning qualitative analysis of roe deer diet have been published by Degrez (1989), Fichant (1974), Helle (1980), Henry (1978), Johanson et al. (1989), Kałuziński (1982), Siuda et al. (1969) and Szmidt (1975).

This animal has a diversified diet, with as many as 178 different species being counted. Trees and shrubs (*Quercus*, *Pinus*, *Betula*, *Populus*, *Sorbus*) play an important role, especially at the end of autumn and in winter. Large amounts of *Ericaceae* (*Vaccinium* spp. and *Calluna*) are also consumed at this time of the year. In winter, hunters sometimes provide these animals with fodder, which tends to reduce significantly the dose ingested.

The leaves of *Rubus* also play an important role in roe deer diet. Various herbaceous plants are also consumed, as well as lichens and fungi.

Herbaceous plants are eaten mainly in spring and summer, as are certain cultivated plants : Kałuziński (1982) observed 85 plant species in the stomach contents of roe deer, but 6 cultivated plants (rye, etc.) accounted for 66% of the volume.

Various authors mention intake of fungi by the roe deer (Cederlund et al. 1980, Fichant 1974, Gębczyńska 1980, Heim de Balsac 1951, Henry 1978, Holišová et al. 1983 and 1986, Johanson et al. 1989, Kałuziński 1982, Maizeret & Tran Mahn Sung 1984, Siuda et al. 1969). The total amounts of fungi eaten account, according to the authors, for 0.5 to 6.2% of the total weight eaten annually and 1.4 to 15% of the weight of food eaten in autumn, the season in which consumption of fungi is at its highest. Cederlund et al. (1980) indicate another maximum in April, suggesting that this involves fungi having spent the winter under snow (Sweden). Finally, Fichant (1974) notes that bucks consume a lot more fungi than does, and that high proportions in stomach contents (up to 35%) can be observed now and again. Degrez (1989) observed that fungi can make up as much as 82% of the large fraction of the stomach contents of roe deer bucks in October-November.

Contamination levels observed :

Rantavaara et al. (1987) give the result of an analysis of 4 samples of roe deer meat collected in southern Finland in 1986. The activities (in Bq/kg FM) range from 11 to 59 for ^{134}Cs and from 24 to 159 for ^{137}Cs.

Mascanzoni (1987, 1989a) reproduces the summary of an analysis of 439 samples collected in 1986 on Swedish territory. The ^{137}Cs levels, expressed in Bq/kg FM, range from < 2 to 11 000 (average = 770). Müller (1986) notes a Swedish sample with a reading of 8 273 Bq/kg in July 1986.

Johanson et al. (1989) studied roe deer contamination in central Sweden in a zone where deposition had been very high (30 000 to 40 000 Bq of ^{137}Cs/m^2). A marked increase in ^{137}Cs activity was measured in roe deer at the beginning of the hunting season. In September 1988 average contamination was 9 000 Bq/kg FM, the highest level observed for this species at this location. It is probable that the contamination level of certain individuals rose from 1 000 to 10 000 Bq/kg FM from the end of July to the beginning of August, probably as a result of eating highly contaminated fungi.

Johanson et al. (1989) report that 135 000 roe deer were killed during the hunting season (16 August-31 December 1988) in Sweden. These authors note that the seasonal variations in radioactive contamination of this species are much larger than in elk. Peaks are usually observed in August and September. A marked increase in contamination was observed in August-September 1988; this probably parallels that measured at the same time in elk (Nelin & Palo 1989, von Bothmer et al. 1989), and is probably due to a temporary change in diet (proliferation of fungi, especially during August).

Johanson et al. (1989) also studied the daily intake of roe deer in Sweden. Fungi seem to be the main factor in the large seasonal variation in contamination recorded in this animal. Intake of ^{137}Cs via fungi is at its highest in August and September (3 000 Bq/day) and in October (2 500 Bq/day), while *Ericaceae* make the greatest contribution during the remainder of autumn.

Müller (1986) reviews the contamination observed in roe deer meat in Germany during the first few weeks following the Chernobyl accident. In general, the contamination remained below the 600 Bq/kg limit. Nevertheless, certain record values greatly exceeded this threshold : 3 287 Bq/kg in Lower

Saxony and 3 700 in Baden-Württemberg. The levels observed in Bavaria were the only ones to have exceeded the limit on a wide scale : 810 to 2 700 on average, with certain record values even exceeding 5 000 Bq/kg.

An article in Der Spiegel magazine (1988) surveyed the contamination situation in Germany. Roe deer continued to exhibit high levels at this time and some spectacular figures were given (15 000 Bq/kg for an animal killed in Bavaria in October 1988).

A listing from the Rhineland-Palatinate gives results of measuring 68 samples of roe deer harvested in May-June 1986. The values observed for ^{131}I in the flesh range from < 0.3 to 260 Bq/kg (FM?), except for three values between 2 000 and 4 000 and dating from the first two weeks in May. The concentrations of this same isotope in the entrails vary from 56 to 301 Bq/kg (FM?). As for ^{137}Cs, the values observed in the entrails range from 99 to 294 Bq/kg (FM?) and from 36 to 600 Bq/kg (FM?) in the flesh, with the exception of five samples with activity of 742 to 902 Bq.

The CEC Joint Research Centre in Ispra provided us with the findings from analysing roe deer killed in Austria and Italy in 1986. A summary of the data for ^{134}Cs and ^{137}Cs is given in Table 3. Other radionuclides were observed in these samples, in particular in Italy. The main one is ^{131}I, detected in 87 samples. This isotope has a very short half-life and high levels were observed only in May and June 1986. The contamination is never above 500 Bq ^{131}I/kg FM, except for in samples of the thyroid and trachea (probably together with the thyroid) : 1 747 to 9 883 Bq/kg FM for four trachea samples and 4 662 to 536 560 Bq/kg for six thyroid samples! Finally, other isotopes were sometimes detected in the Italian samples, these being radionuclides with a short or very short half-life : ^{103}Ru (39 samples, maximum level observed = 201 Bq/kg FM); ^{140}Ba (17 samples, max. = 382 Bq/kg); ^{140}La (5 samples, max. = 56 Bq/kg); ^{132}Te (6 samples, max. = 287 Bq/kg); and ^{136}Cs (7 samples, max. = 56 Bq/kg).

Tataruch et al. (1989) looked at roe deer contamination in Austria since 1986. They observed that this species had, on average, contamination exceeding that of the other species they had studied. The record value of 11 500 Bq ^{137}Cs/kg (FM?) was observed in 1986, although most of the samples were below 5 000 Bq/kg. Still in Austria, Henrich et al. (1989) note that in 1988 contamination of roe deer meat rose to 11 800 Bq/kg.

The Radioprotection Division in the Grand Duchy of Luxembourg sent us a listing giving radiocaesium activities measured in over 400 roe deer samples. The levels observed are between 1.0 and 277 Bq/kg FM for ^{134}Cs and between 4.8 and 612 Bq/kg FM for ^{137}Cs. They show a clear decrease from May to September 1986. On the basis of these figures we calculated that the ecological half-life of ^{134}Cs and ^{137}Cs was 51 days during the June-September 1986 period.

Sépulchre-De Bie et al. (1988) studied radiocaesium contamination of roe deer kidneys in Belgium in 1986. The average values measured range between 9.77 and 246 Bq/kg for ^{137}Cs and between 9.87 and 104 Bq/kg for ^{134}Cs. Average figures for 1985 are below 35 Bq Cs/kg. Furthermore, these authors observed a good correlation between this contamination and that in blackberry leaves (*Rubus* div. sp.), a species playing an important role in roe deer diet. The lowest figures (Ciergnon) are attributable to the richness of the soil in this area (and thus to the low availability of Cs) and not to the "greater protection against air pollutants" which this region enjoys, the explanation put forward by the authors.

Lowe & Horrill (1988) analysed samples of meat from 7 bucks killed in England in May-July 1986. The activities (in Bq/kg FM) vary between 72.2 and 215.7 for ^{137}Cs and between 17.4 and 108.4 for ^{134}Cs. On the basis of their figures these authors calculate an ecological (not biological) half-life of 28 days for ^{134}Cs in roe deer.

CERVUS ELAPHUS L.

Order : *Artiodactyles*
Family : *Cervidae*

English : Red deer (female : Hind, Doe; male : Stag, Hart)
French : Cerf rouge, Cerf elaphe (female : Biche; young : Faon)
German : Rothirsch, Edelhirsch
Italian : Cervo
Spanish : Ciervo comun, Venado
Dutch : Edelhert

Geographical distribution : occurring in most of European countries (always in big forests, except in Scotland) but not in Iceland and Finland; very rare in Scandinavia (after a map by van den Brink & Barruel 1971).

Diet :

According to Borowski & Kossak (1975), Dzięciołowski (1970a and 1970b) and Jamrozy (1980), the red deer has a varied diet; the first of these authors cite 137 species. Trees and shrubs play an important role here : *Quercus*, *Salix*, *Carpinus*, *Corylus* and *Sorbus* are among the favourites; *Pinus* and *Juniperus* are consumed during periods of food scarcity.

Ericaceae such as *Calluna vulgaris* and *Vaccinium* spp. are also frequently consumed and are a source of contamination. The leaves of the *Rubus* and of herbaceous plants are also eaten in considerable amounts.

Fungi are consumed on a wide scale, especially in August-November, with a maximum in September-October during the period of fructification of most of the higher fungi. However, they make up only a small proportion of the total annual weight of food ingested (in the order of 1% or less). In autumn they account for some 3% of diet, but much higher quantities are sometimes found : up to 30% of the contents of a rumen analysed by Jensen 1968. According to Gębczyńska (1980), the females of the species eat more fungi than the males (almost twice as much). Dzięciołowski (1967) cites 13 species of fungus eaten by the red deer, including *Armillaria mellea*, *Russula alutacea*, *Lactarius volemus* and *Paxillus involutus*. Some of these species, in particular the last one mentioned, are capable of accumulating radiocaesiums and thus constitute major sources of contamination.

Finally, seasonal variations exist in the quantity of food taken in, with a maximum in summer (July-August) and a minimum in winter (Bobek et al. 1972, Borowski & Kossak 1975). These latter authors also indicate a second maximum in December and a minimum in May. Qualitative seasonal variations have been observed, in particular by Dzięciołowski (1967).

Contamination levels observed :

Mascanzoni (1987, 1989a) gives ^{137}Cs contamination levels measured in two samples in Sweden in 1986 : 51 and 100 Bq/kg FM.

The CEC Joint Research Centre in Ispra sent us the results of an analysis of meat from *Cervus elaphus* killed in Austria and Italy in 1986. In Austria the contamination among 39 samples varies from 0 to 1 110 Bq ^{134}Cs/kg FM (average : 218) and from 0 to 2 405 Bq ^{137}Cs/kg FM (average : 492). In Italy, the contamination among 7 samples varies from 0 to 59 Bq ^{134}Cs/kg FM and from 0 to 188 Bq ^{137}Cs/kg FM.

The IHE sent us the results of analysing two samples of muscle collected in Belgium in October and November 1986 : 29.3 and 111.1 Bq/kg FM for ^{134}Cs and 91.7 and 274.1 Bq/kg FM for ^{137}Cs.

Tataruch et al. (1989) indicate that red deer occupy third place among ruminants, with the roe deer and chamois being more contaminated. The record figure of 1 000 Bq 137 Cs/kg (FM?) was recorded for an animal killed in Austria in 1986.

RANGIFER TARANDUS (L.)

Syn. : *Rangifer fennicus*

Order : *Artiodactyles*
Family : *Cervidae*

English : Reindeer, Caribou
French : Renne, Caribou
German : Ren, Rentier
Italian : Renno
Spanish : Rengifero, Reno
Dutch : Rendier

Geographical distribution : in Europe, occurring only in Finland and northern USSR; smaller populations in Iceland and southern Norway (after a map by van den Brink & Barruel 1971).

Diet :

See Eriksson 1989 and Holleman et al. (1979). The diet of the reindeer is very varied : some 450 species have been counted. However, lichens (especially species of *Cladina*, *Bryoria* and *Alectoria*) form the staple food (50%) of the reindeer from September to April. These cryptogams are radiocaesium accumulators and thus constitute a major source of contamination. According to Holleman et al. (1971), 20 to 30% of the radiocaesium in lichens can be assimilated by reindeer.

Ericales (*Vaccinium myrtillus*, *Calluna vulgaris* and *Empetrum hermaphroditum*) are the second most important group in reindeer diet (17 to 68%). These species also accumulate caesium.

Contamination levels observed :

A sample of reindeer taken in Finland in 1986 (Rantavaara et al. 1987) contained 281 Bq/kg FM of ^{134}Cs and 529 Bq/kg FM of ^{137}Cs. Information on contamination of this species in Finland in 1979 is given by Rantavaara (1982). The average ^{137}Cs concentrations observed for this period in 30 samples varied between 457 and 529 Bq/kg (FM?).

Mascanzoni (1987, 1989a) reproduces a summary of data collected in 1986 for Sweden. The ^{137}Cs levels vary between 12 and 16 000 Bq/kg FM, with an average of 2 200, for 180 samples.

MacKenzie (1986) reports that, of 21 000 reindeer killed in Sweden in October 1986, only 5 000 were below the Swedish limit of 300 Bq/kg. Average contamination was 4 000 Bq/kg in September 1986 and 8 000 Bq/kg in October 1986, with a record value of 30 000 Bq/kg.

Liljenzin et al. (1988) give averages for 95 samples of reindeer meat harvested in 1986 in Sweden : 2.64 Bq/kg (FM?) for ^{134}Cs and 2.95 for ^{137}Cs (oversight for kBq?), with standard deviations of 0.66 and 0.58 respectively.

Eriksson (1989) studied contamination of reindeer in northern Sweden, where some 300 000 reindeer live. Their contamination levels vary considerably during the year, being between 100 and 80 000 Bq/kg in winter and between 30 and 3 000 Bq/kg in July. No more than a slight reduction in contamination was observed between 1986 and 1989 in reindeer meat and in the plants constituting the staple diet of this species (lichens, blueberry, etc.), although contamination in the latter plant increased slightly.

By way of comparison, it should be noted that 296 samples of caribou meat analysed in Canada (Meyerhof & Marshall 1989) produced readings varying from 40 to 750 Bq ^{137}Cs/kg FM.

Reindeer are a special case because the Laplanders live more or less in symbiosis with these animals and follow their annual transhumances (MacKenzie 1986). For them the reindeer is a means of transport and the main source of food and of raw materials for making clothes, tools, etc. All parts of the animal are used : flesh, milk, blood, antlers, hide, sinews, etc. If, as is feared, reindeer contamination remains high during many years to come, the Laplanders' culture and way of life are in danger of disappearing, in the same way that the culture of some Indian tribes in North America did not survive the settlers' massacre of the bison.

ALCES ALCES (L.)

Order : *Artiodactyles*
Family : *Cervidae*

English : Moose, Elk
French : Elan, Orignal
German : Elch
Italian : Alce
Spanish : Alce
Dutch : Eland

Geographical distribution : in Europe, only occurring in Fennoscandia and USSR (after a map by van den Brink & Barruel 1971).

Other data on the distribution and seasonal migrations of this animal are given by Bédard (1974) and in particular by Pulliainen (1974) and Markgren (1974). The latter notes that 200 000 elk live in Fennoscandia, mainly in southeast Norway, central Sweden and southern Finland.

Diet :

According to von Bothmer et al. (1989), trees and shrubs constitute these animals' staple diet throughout the year. The shoots of *Pinus sylvestris* predominate in January-February and the leaves of various trees (*Betula*, etc.) are frequently consumed from June to September, whereas low-growing shrubs such as *Vaccinium myrtillus* and *Calluna vulgaris* are mainly eaten in April, September and October. *Epilobium angustifolium* is also a favourite.

According to Heikkilä (1991), the rowan (*Sorbus aucuparia*) and the aspen (*Populus tremula*) are preferred to the Scots pine (*Pinus sylvestris*) and to birch (*Betula pendula* and *B. pubescens*) when elk have the choice. *B. pendula* is preferred to *B. pubescens* and planted birch are grazed more than naturally regenerating birch.

According to Morow (1976), 90% of food eaten in autumn comes from trees and shrubs and some 10% from *Ericaceae*. Fungi are consumed, especially *Armillaria mellea* and *Xerocomus subtomentosus*, but make up no more than 0.1% of the weight annually consumed. Fungi are also cited by Cederlund et al. (1980) as accounting for 0.5% of the mass consumed.

Further information on the diet of the elk is given by Johanson et al. (1989), Heikkilä (1991), Silvennoinen et al. (1991) and Bédard (1974), and in particular by Peek (1974).

Contamination levels observed :

Rantavaara et al. (1987) reproduce the results of analysing 276 samples collected in Finland in 1986. The activities measured (in Bq/kg FM) vary from 0 to 882 for ^{134}Cs and from 10 to 1 610 for ^{137}Cs. The correlation between the contamination level of elk meat and the deposition recorded is quite clear. Data on contamination of elk meat in Finland in 1979 can be found in Rantavaara (1982); the figures range from 4 to 400 Bq/kg, with an average of 34.4 for 176 samples. Some other radionuclides are mentioned by Rantavaara et al. (1987), mainly ^{129m}Te (max. 118 Bq/kg FM), ^{131}I (max. 231 Bq/kg), ^{132}Te (max. 67 Bq/kg) and ^{136}Cs (max. 66 Bq/kg). These latter analyses all date from before the end of August 1986.

Mascanzoni (1987, 1989a) reproduces data collected on this species in Sweden in 1986. Analysis of ^{137}Cs in 561 samples gives levels ranging between < 2 and 4 800 Bq/kg FM, with an average of 290 Bq.

Johanson & Bergström (1989) studied radiocaesiums in the elk in Sweden. Prior to Chernobyl, ^{137}Cs activities were 23 Bq/kg on average. In October 1986 they were 760 and in October 1987 664 Bq/kg. There was no significant difference in contamination between the various types of muscle. The influence of diet (amount of crop farming in the area) and of the amount of deposition recorded is very clear. Johanson et al. (1989) give corresponding figures for 1988.

Nelin & Palo (1989) and Palo et al. (1989) studied contamination of the elk in northern Sweden. From May 1986 to December 1988 some 5 500 analyses of elk muscle were performed. The contamination observed correlates closely with deposition. A sudden increase in contamination took place in September 1988 : the adults contained an average activity of 640 Bq ^{137}Cs/kg whereas the corresponding figures were only 300 in September 1986. This phenomenon was also in evidence in elk calves : 1 300 Bq ^{137}Cs/kg in 1988 as against 500 in 1986. From October 1988 onwards the situation reverted to what it was in 1987, however. This irregularity was also noted by von Bothmer et al. (1989). It was probably caused by a temporary change in diet due to exceptional growth of fungi in August 1988 (Johanson et al. 1989).

Johanson et al. (1989) indicate that 135 000 elk were killed in Sweden during the hunting season (October-November 1988). These authors collected some 250 samples of elk meat per year during the 1987 and 1988 hunting seasons in central Sweden. The contamination levels, which vary greatly from sample to sample, are inversely proportional to the percentage of arable land in the animals' territory. The levels measured vary from 300 to 3 000 Bq/kg FM, most being between 500 and 1 000 Bq. The same authors noticed hardly any trend over time : the average level in 1987 was some 10% lower than the average level in 1986, but the average level in 1988 was 10% higher than the 1986 level. Over the three years average contamination of calves was slightly higher (in the order of 10-15%) than that of adults.

Johanson et al. (1989) also measured the contamination of various plants making up the diet of the elk in Sweden and calculated the daily intake. This was some 25 000 Bq ^{137}Cs in October 1988.

Bergman & Johansson (1989) provide information on contamination trends in elk meat in Sweden. Average activities, expressed in Bq ^{137}Cs/kg, were 27 in 1985, 220 in 1986 and 1987 and 450 in 1988.

Von Bothmer et al. (1989) studied elk diet and ^{137}Cs accumulation in elk tissues. Contamination of the elk clearly varies according to an annual cycle, with a maximum in autumn and a minimum in spring, more or less marked depending upon the year. The authors analysed the contamination of the staple plants making up the diet of this species and used the data to explain the changes in contamination during the year. The role played by fungi was not studied, however.

The elk is very popular among hunters in certain regions : Mascanzoni (1987) and von Bothmer et al. (1989) report that some 130 000 elk are killed each year in Sweden, and Johanson & Bergström (1989) note that 50 000 animals are killed every year in Finland.

Danell et al. (1989) published a study on elk contamination in central Sweden, where Chernobyl fallout amounted to between 2 000 and 60 000 Bq/m^2. Some 3 661 animals were looked at in the course of this major study. The calves were on average more contaminated (470 Bq/kg FM) than adults (300 Bq/kg FM). Contamination was slightly higher in females than in males. Prior to the Chernobyl accident contamination of elk meat was 33 Bq/kg FM.

SUS SCROFA L.

Order : *Artiodactyles*
Family : *Suidae*

English : Wild boar
French : Sanglier (female : Laie; young : Marcassin)
German : Wildschwein (female : Bache; male : Eber, Keiler; young : Frischling)
Italian : Cinghiale
Spanish : Jabali
Dutch : Wild zwijn

Geographical distribution : in Europe, the species is absent from Iceland, the British Isles, Denmark and Fennoscandia as well as from many regions of Italy, Austria, Czechoslovakia, Hungary and Yugoslavia (after a map by van den Brink & Barruel 1971).

Contamination levels observed :

Mascanzoni (1987, 1989a) notes that an analysis of 3 samples of this species collected in Sweden in 1986 gave ^{137}Cs levels ranging from 46 to 450 Bq/kg FM.

A listing from the Rhineland-Palatinate provided data on four samples of wild boar collected in May-June 1986. The ^{131}I levels vary from < 0.3 to < 1.0 Bq/kg FM (?) and those for ^{137}Cs from 25.9 to 362 Bq/kg.

Tataruch et al. (1989) studied contamination trends since 1986 among wild boar in the highly contaminated zones of Austria. They noted that this species was not contaminated very much in 1986, with the activities measured normally being below 37 Bq ^{137}Cs/kg (FM?) and the highest value being 600 Bq/kg. However, very high ^{137}Cs activities were observed in the subsequent years, which was attributed to contamination of the staple plants found during analysis of stomach contents. In 1988 all the wild boar samples from the Kobernausser Wald were above 2 000 Bq ^{137}Cs/kg (FM?) and a record level of 17 600 Bq/kg was recorded there. This was the highest contamination level observed in game in Austria. Henrich et al. (1989) note that contamination of this species in Austria in 1988 reached a maximum of 11 800 Bq/kg.

The CEC Joint Research Centre in Ispra sent us the following results after analysing wild boar killed in 1986. In Austria, the analysis of 29 samples showed contamination ranging from 0 to 407 Bq ^{134}Cs/kg FM (average : 58) and from 0 to 814 Bq ^{137}Cs/kg FM (average : 129), while in Italy 21 samples of this species had contamination levels of from only 0 to 73 Bq ^{134}Cs/kg FM (average : 23) and from 0 to 180 Bq ^{137}Cs/kg FM (average : 53).

The listing received from the Radioprotection Division of the Grand Duchy of Luxembourg contains results of analyses of 28 samples of wild boar for 1986-1988. The ^{134}Cs levels vary from 0.5 to 53.9 Bq/kg FM (?), while those of ^{137}Cs vary from 2.1 to 140.6 Bq/kg FM (?). Contrary to what was observed in Austria by Tataruch et al. (1989), the levels observed in 1987 and 1988 are lower than those in 1986.

Finally, the listing sent to us by the IHE (Belgium) contains data on contamination of 2 samples of this species. The levels (in Bq/kg FM) are 11.4 and 121.7 for ^{134}Cs and 33.1 and 403.9 for ^{137}Cs.

ORYCTOLAGUS CUNICULUS (L.)

Order : *Lagomorphes*
Family : *Leporidae*

English : Rabbit
French : Lapin de garenne
German : Wildkaninchen, Kaninchen
Italian : Coniglio selvatico
Spanish : Conejo de monte
Dutch : Wild konijn, Konijn

Geographical distribution : in Europe, the species is absent from Iceland, Denmark, Fennoscandia (except southern Sweden), USSR and the Balkan, as well as from most of Italy and Austria (after a map by van den Brink & Barruel 1971).

Contamination levels observed :

Mascanzoni (1987, 1989a) reproduces the results of an analysis of two samples collected in Sweden in 1986. The activities observed for ^{137}Cs were 82 and 1 300 Bq/kg FM.

The listing sent by the Radioprotection Division in the Grand Duchy of Luxembourg contains data on 6 samples of rabbit collected in the 10 months following the Chernobyl accident, the levels for ^{134}Cs varying between 7.6 and 26.0 Bq/kg FM and those for ^{137}Cs between 17.7 and 56.6 Bq/kg FM.

The listing from the IHE contains data on two samples collected in Belgium in 1986. The activities, expressed in Bq/kg FM, are < 1.3 and 32.0 for ^{134}Cs and 2.1 and 65 for ^{137}Cs.

The CEC Joint Research Centre in Ispra sent us the results of analysing 24 samples of rabbit killed in Austria in 1986. Contamination varied from 0 to 407 Bq ^{134}Cs/kg FM (average : 120) and from 0 to 777 Bq ^{137}Cs/kg FM (average : 245).

LEPUS CAPENSIS L.

Syn. : *Lepus europaeus*

Order : *Lagomorphes*
Family : *Leporidae*

English : Brown hare
French : Lièvre brun (female : Hase)
German : Feldhase, Hase
Italian : Lepre comune
Spanish : Liebre europea, Liebre común
Dutch : Haas

Geographical distribution : most of Europe, except Iceland, northern Fennoscandia as well as most of Ireland and the Alps (after a map by van den Brink & Barruel 1971).

Contamination levels observed :

Rantavaara et al. (1987) present activities recorded in 7 samples of brown hare collected in Finland in 1986. The levels for ^{134}Cs range from 0 to 671 Bq/kg FM and those for ^{137}Cs from 3 to 1 408 Bq/kg FM.

Johanson et al. (1989) studied ^{137}Cs contamination in 11 brown hares killed in central Sweden in 1986-1989. The activities vary from 7 to 921 Bq/kg FM, with an average of 280.

Tataruch et al. (1989) noted the record value of 2 200 Bq ^{137}Cs/kg (FM?) in 1986 in Austria.

A listing from the Radioprotection Division in the Grand Duchy of Luxembourg gives data for 6 samples collected there in 1986-1987. The levels for ^{134}Cs fluctuate between 2.7 and 20.5 Bq/kg FM and those for ^{137}Cs between 4.3 and 46.9 Bq/kg FM, with the exception of a sample collected on 12.6.1986 containing 207.7 Bq ^{134}Cs and 355.3 Bq ^{137}Cs/kg FM.

A sample analysed by the IHE and collected in December 1986 in Belgium contained 1.3 Bq ^{134}Cs/kg FM and 4.6 Bq 137 Cs/kg FM.

The CEC Joint Research Centre in Ispra sent us the results of analyses of meat from brown hare killed in 1986. In Austria, the readings ranged from 22 to 666 Bq ^{134}Cs/kg FM (average : 149 for 12 samples) and from 37 to 1 406 Bq ^{137}Cs/kg FM (average : 313), while in Italy 8 samples contained from 0 to 240 Bq ^{134}Cs/kg FM (average : 72) and from 0 to 481 Bq ^{137}Cs/kg FM (average : 163).

LEPUS TIMIDUS L.

Order : *Lagomorphes*
Family : *Leporidae*

English : Blue hare, Mountain hare
French : Lièvre variable, Lièvre changeant
German : Schneehase
Italian : Lepre bianca

Geographical distribution : occurring in Iceland, Ireland, Fennoscandia, USSR as well as in the Alps and in parts of Britain (after a map by van den Brink & Barruel 1971).

Contamination levels observed :

Rantavaara et al. (1987) give the results of analysing 6 samples taken in 1986 in southern Sweden. The levels vary from 313 to 926 Bq/kg FM for ^{134}Cs and from 608 to 1 888 Bq/kg FM for ^{137}Cs.

Johanson et al. (1989) measured the contamination in 8 blue hares killed in central Sweden in 1986-1989. The readings obtained vary from 988 to 5 313 Bq/kg FM, with an average of 3 300. These activities are over 10 times greater than those obtained for the brown hare. The authors note that these differences are due to the fact that the brown hare lives mainly in areas where there are plenty of cultivated crops.

Mascanzoni (1987, 1989a) summarises the data collected in 1986 on this species in Sweden. The activities for ^{137}Cs vary from < 2 to 13 000 Bq/kg FM, with an average of 1 200, for 225 samples.

OTHER MAMMALS

Rantavaara et al. (1987) give the results of analyses of 27 samples of *Odocoileus virginianus* (white-tailed deer, *Cervidae*) collected in Finland in July to September 1986. The levels measured (in Bq/kg FM) range from 23 to 500 (except for one sample which goes as high as 979) for ^{134}Cs and from 52 to 775 (apart from two samples with 1 025 and 1 954) for ^{137}Cs.

A sample of *Dama dama* (fallow deer, *Cervidae*), collected in Finland in August 1986 had an activity of 41 Bq/kg FM for ^{134}Cs and of 125 Bq/kg FM for ^{137}Cs (Rantavaara et al. 1987). Mascanzoni (1987, 1989a) reports that analysis of 3 samples of fallow deer collected in Sweden in 1986 gave ^{137}Cs levels of between 21 and 280 Bq/kg FM. Finally, a listing from the Rhineland-Palatinate gives the contamination observed in two fallow deer killed in 1986. The activities for ^{131}I were 5.2 and 42 Bq/kg (FM?) and 97 and 457.1 Bq/kg (FM?) for ^{137}Cs.

The CEC Joint Research Centre in Ispra sent us the results of analyses of 8 samples of fallow deer killed in 1986 in Austria. The readings observed ranged from 0 to 777 Bq ^{134}Cs/kg FM (average : 335) and from 0 to 1 591 Bq ^{137}Cs/kg FM (average : 682).

Data on contamination of chamois (*Rupicapra rupicapra*, *Bovidae*) in Austria and Italy were sent to us by the above-mentioned centre; a summary of these is given in Table 4.

Mascanzoni (1987, 1989a) gives data on 3 other mammals sampled in Sweden in 1986. Three samples of *Castor fiber* (beaver, *Castoridae*) had activities for ^{137}Cs ranging from 140 to 920 Bq/kg FM. Seven samples of *Ursus arctos* (brown bear, *Ursidae*) had ^{137}Cs levels ranging from 62 to 420 Bq/kg FM. Three samples of *Meles meles* (badger, *Mustelidae*) had contamination ranging from 38 to 3 600 Bq/kg FM for ^{137}Cs.

BIRDS

Table 5 contains data on radiocaesium activities observed in various species of bird.

The CEC Joint Research Centre in Ispra sent us the results of analysing 133 samples covering 27 species of bird killed in Italy in 1986. The radiocaesium levels (^{134}Cs + ^{137}Cs) are, as a rule, below 300 Bq/kg FM and never exceed 600 Bq/kg, with the exception of a sample of wood pigeon which had a level of 1 106 Bq/kg FM, and a sample of quail with 1 344 Bq/kg FM. Some of these data are included in Table 5.

^{110m}Ag was observed by Rantavaara et al. (1987) in various waterfowl killed in Finland in 1986 : 30 to 86 Bq/kg FM for the 3 samples of *Fulica atra* (coot, *Rallidae*); 6 to 175 Bq/kg FM for 15 of 22 samples of *Bucephala clangula* (goldeneye, *Anatidae*), the other samples of this species not retaining any radioactive silver; 7 to 79 Bq/kg FM for 11 out of 27 samples of *Anas platyrhynchos* (mallard, *Anatidae*) and 6 to 30 Bq/kg FM for 9 out of 25 samples of *Anas crecca* (teal, *Anatidae*).

Davis (1986), dealing with the problem of migratory birds, indicates that over 20 million birds are killed and eaten every year in Italy.

Species	English name	Auth.	N	137Cs min	137Cs mean	137Cs max	134Cs min	134Cs mean	134Cs max	R
ANATIDAE										
Anas crecca	*Teal*	Rant.	25	39	875	4349	19	455	2317	0.52
		Masc.	2	43	59	75				
Anas penelope	Wigeon	Rant.	2	373	2617	4861	210	1365	2520	0.52
Anas platyrhynchos	Mallard	Rant.	2	0	983	4673	0	521	2529	0.53
		Masc.	20	18	450	1600				
		Ispra	4	0	50	122	0	22	70	0.43
Anser anser	Graylag	Masc.	1	-	64	-				
Branta canadensis	Canada goose	Masc.	20	12	580	3800				
Bucephala clangula	Goldeneye	Rant.	22	0	1196	6861	0	624	3608	0.52
		Masc.	2	120	345	570				
Mergus merganser	Goosander	Masc.	2	52	76	99				
		Rant.	1	-	77	-	-	30	-	0.39
Somateria mollissima	Eider	Masc.	20	<2	48	250				
RALLIDAE										
Fulica atra	Coot	Rant.	3	43	118	178	13	57	87	0.48
		Ispra	1	-	29	-	-	15	-	0.52
TETRAONIDAE										
Lagopus lagopus	Willow grouse	Masc.	5	25	340	600				
		Rant.	1	-	12	-	-	3	-	0.25
Lagopus mutus	Ptarmigan	Masc.	22	<2	250	980				
Lyrurus tetrix	Black grouse	Masc.	36	39	520	2000				
		Rant.	3	218	366	662	115	187	330	0.51
Tetrao urogallus	Capercaillie	Masc.	38	<2	830	3200				
		Rant.	1	-	94	-	-	15	-	0.16
Tetrastes bonasia	Hazel hen	Masc.	18	<2	510	1800				
		Rant.	1	-	764	-	-	390	-	0.51
SCOLOPACIDAE										
Scolopax rusticola	Woodcock	Masc.	45	19	1500	17000				
		Reis.	?	?	?	16000				
PHASIANIDAE										
Coturnix coturnix	Quail	Ispra	4	0	230	918	0	107	426	0.46
Phasianus colchicus	Pheasant	Masc.	1	-	24	-				
		Ispra	14	0	9	63	0	4	26	0.40
COLUMBIDAE										
Columba oenas	Stock dove	Masc.	5	<2	150	620				
Columba palumbus	Wood pigeon	Masc.	14	26	380	1100				
		Rant.	6	19	150	353	9	75	194	0.50
TURDIDAE										
Turdus merula	Blackbird	Ispra	7	100	193	292	25	95	192	0.50

Table 5 : Radiocaesium contamination observed in various species of bird collected in 1986 in Sweden (Mascanzoni 1987, 1989a; Reisch 1987), in Finland (Rantavaara et al. 1987) and in Italy (data from the CEC Joint Research Centre in Ispra); the figures are expressed in Bq/kg FM, column N indicates the number of samples, the columns min, ave and max are, respectively, the minimum, average and maximum levels observed, column R gives the average $^{134}Cs/^{137}Cs$ concentration ratio.

III.3. SUMMING-UP AND COMMENTS

III.3.1. Species involved

The forest animal species likely to be hunted as game are mammals and birds.

Among mammals these include numerous ***Cervidae*** (roe deer, red deer, white-tailed deer, reindeer, elk and fallow deer). The roe deer is certainly the most hunted in Europe, while the elk and reindeer are very important game in Sweden and Finland.

Another species making up "big game" is the **wild boar** (*Suidae*). The others come under the heading of "small game" and belong to the family of ***Leporidae*** (rabbit, brown and blue hare).

Among the birds, a large number of ***Anatidae*** (duck, teal, etc.) are hunted. ***Tetraonidae*** (capercaillie, black grouse, willow grouse/ptarmigan, hazel hen) are also hunted as game in regions where they abound. Finally, other species belonging to various families are also likely to be hunted : woodcock, stock dove, wood pigeon and pheasant.

III.3.2. Variations in contamination observed

a) According to geographical region : numerous authors report strong correlation between ground deposition following the Chernobyl accident and game contamination (Danell et al. 1989, Rantavaara et al. 1987, Tataruch et al. 1989). Table 5 even shows that the $^{134}Cs/^{137}Cs$ concentration ratio in the birds analysed is almost identical to that of the fallout, apart from some species for which only a few samples are available and which were collected in lightly contaminated zones. However, these studies generally cover measurements taken in 1986. It is probable that during the following years the influence of the type of soil will have become greater due to its impact on the biological availability of radionuclides in general and radiocaesium isotopes in particular. On the other hand, due to the relatively low contamination level of cultivated crops, animals living in zones where crop farming is widespread are generally much less contaminated than those living in completely wild regions.

b) According to species or "systematic" groups : classification of this type is not easy due to the fact that only a small number of data are available for many species. Furthermore, contamination levels can vary due to a number of other factors (local deposition, age and sex of individuals, season, etc.), thus making it more difficult to compare data.

However, it seems that reindeer (*Rangifer tarandus*) head the list of the most contaminated game species. The average figures quoted in the literature usually exceed 2 000 Bq/kg FM of ^{137}Cs, with a record figure of 80 000 Bq/kg being cited for ^{137}Cs. The persistently high contamination levels of this species are due in part to the important role played by lichens in its diet. These plants are well known for their capacity to retain radioactive atmospheric fallout (Guillitte et al. 1989c and 1990, Sloof & Wolterbeek 1992). Furthermore, the reindeer plays an important role in the food of some human groupings in northern Europe (Laplanders). The lichen-reindeer-man foodchain has been much studied since the Sixties in order to monitor the consequences of fallout from nuclear weapons testing (Hanson 1967).

Other species with very high contamination levels are the badger (*Meles meles*, average 2 200 Bq/kg FM for ^{137}Cs, but for 3 individuals only : Mascanzoni 1987, 1989a), the woodcock (*Scolopax rusticola*, average 1 500 Bq/kg FM for ^{137}Cs in 45 individuals : Mascanzoni 1987, 1989a) and the blue hare (*Lepus timidus*, average 1 120-1 200 Bq/kg FM for ^{137}Cs : Rantavaara et al. 1987, Mascanzoni 1987, 1989a).

The group of Tetraonidae is also fairly contaminated on average : capercaillie (*Tetrao urogallus*), black grouse (*Lyrurus tetrix*) and hazel hen (*Tetrastes bonasia*). The willow grouse and ptarmigan (*Lagopus lagopus* and *L. mutus*) are less contaminated, however. These species normally live in areas with Ericaceae growing on acid soil and this is probably the reason for their high contamination level.

Roe deer seem to be the most contaminated among the *Cervidae* (with the exception of the reindeer). The contamination levels cited frequently exceed 1 000 Bq/kg FM. The other Cervidae occupy the middle rungs of the contamination ladder, as do waterfowl (*Anatidae*).

On the bottom rung of this ladder are the wild boar and pheasant. These animals are often reared in captivity and released into the forest prior to the hunting season. While in captivity these animals are fed on commercial grains, which no doubt explains why their contamination level is much lower than that of wild animals.

c) According to age and sex : several authors have demonstrated **intra-species variability due to age.** In particular, elk calves were studied by Danell et al. (1989). Their average contamination level is 470 Bq/kg FM, whereas that of adults is near to 300 Bq/kg FM. Johanson & Bergström (1989) and Johanson et al. (1989) obtained comparable differences, but less marked, perhaps because the calves they analysed were some five weeks older on average.

Rantavaara et al. (1987) calculated an average transfer factor of 0.0097 m^2/kg for adult elks and 0.015 m^2/kg for calves in summer and autumn 1986. However, Danell et al. (1989) compared the diet of young and adult elk and concluded that there is no appreciable difference in the contamination level of foodstuffs consumed by the two age classes (plants, milk, etc.). The explanation could be that a larger part of the young animals' food intake is incorporated into their growing tissues.

Analysis of data from the CEC Joint Research Centre in Ispra on measurements made in Austria on roe deer (Table 3) show that the young are always more contaminated than adults, both as regards muscles and offal and as regards ^{134}Cs and ^{137}Cs.

Similar differences were observed by yet other authors (cited by Johanson & Bergström 1989). Among the data transmitted by the Radioprotection Division of the Grand Duchy of Luxembourg, two very young does had very high contamination levels (1 078 and 1 200 Bq/kg FM) compared to those of adult animals killed in the same period.

Furthermore, there is also **intra-species variability due to sex.** Danell & al. (1989) showed a difference in contamination between the sexes in elk, with the females being on average more contaminated than males. A similar phenomenon is also observable in the Austrian data on roe deer (Table 3) and chamois (Table 4). This difference, related to sex, is found above all in adults but also exists (although it is not so marked) among the young. Since the size of individuals increases according to the following classification young female < young male << adult female < adult male - these differences are attributable to the principle that contamination level is inversely proportional to size.

d) According to tissue type : All the parts of one and the same animal are not contaminated to the same degree. Among roe deer (see Table 3) and chamois (see Table 4) the following progression was observed in radiocaesium contamination levels : liver < heart < muscle, kidneys << thyroid. As noted before, the thyroid accumulates a large amount of radioactive iodine.

e) According to season : seasonal variation in contamination has also been observed in certain species of game : reindeer (Eriksson 1989) and elk (Danell et al. 1989, von Bothmer et al. 1989). Rantavaara et al. (1987) observed a tendency for transfer factors to decrease during the summer and in early autumn, among both adult and young elk. It is probable that similar differences exist for most species.

According to these authors, the contamination level has two maxima during the year : the first in spring and the second in autumn (September, October). These differences are generally due to seasonal changes in diet, i.e. the animals switching successively from contaminated plants to less-contaminated plants and then back again. The seasonal variations in daily quantities consumed probably also play a role here.

The following two points, often ignored by the authors, may also be of some significance.

- Contamination levels in one and the same plant can vary a lot during the year (see I.7.2.2. of this report). The maxima observed are to be found precisely in April-May, followed by a rapid fall in June-July. Thus, the contamination of game can change during this period of the year without there being any changes in their diet.

- Certain species of fungi have contamination levels greatly exceeding those of plants growing in the same places. Numerous studies of the diet of mammals have shown that a lot of species consume fungi, and sometimes in large quantities (Degrez 1989, Durrieu et al. 1984, Fogel & Trappe 1978, Johanson et al. 1989, Johnson & Nayfield 1970, Rantavaara 1982, Ure & Maser 1982). Since fungi mainly develop in September-October, it seems very probable that their consumption by game goes a long way to explaining the increase in contamination observed during this period (Bakken & Olsen 1989, Henrich et al. 1988, Mascanzoni 1989a).

From the point of view of radioprotection, it is worth noting that the autumn maximum overlaps exactly with the hunting season. This fact has presented some problems in certain cases : Danell et al. (1989) note that contamination of elk made it necessary to discard near on 5% of the 10 445 animals killed in 1986 in the region of Sweden they studied, and that 4% of hunters' associations decided to discontinue hunting in October of that year. The Swedish government gave 746 000 Swedish kronor to the hunters in this region to compensate for the animals discarded.

f) In the course of time : several authors observe that there was hardly any decrease in contamination of game between 1986 and 1988; in some cases it even increased : Johanson & Bergström (1989) and Johanson et al. (1989) observed that radiocaesium concentrations in roe deer were at their highest in 1988. Presumably this phenomenon is due to the fact that contamination is very slow to disappear from plants and soils in natural habitats.

Mascanzoni (1987) reproduces a figure showing that while contamination of reindeer in Sweden was some 2 200 Bq/kg FM in 1964, it was still approximately 1 500 in 1966 and 1 000 in 1968, and that it probably did not drop to 500 Bq/kg FM until the early Eighties. A similar reduction rate is also likely for contamination in natural or semi-natural habitats in Europe following the Chernobyl accident.

III.3.3. Transfer coefficients and factors

On the basis of data available on the diet of this species, von Bothmer et al. (1989) calculated that elk in central Sweden had absorbed nearly 25 000 Bq of ^{137}Cs per day during October 1988. During this same period the activity of this isotope in elk muscle was approximately 800 Bq/kg. From this they deduce a transfer coefficient of 0.03 days/kg. By way of comparison, these authors quote similar coefficients calculated by other authors : 0.0067 days/kg for transfer of ^{137}Cs from grass to cow's milk, 0.24 days/kg for sheep meat and 0.02 days/kg for beef.

Rantavaara et al. (1987) calculate the transfer factors by relating the contamination observed in game (expressed in Bq/kg) to the deposition (expressed in Bq/m^2) measured at the same place. They thus obtain transfer factors expressed in m^2/kg. For example, the transfer factors for moose are higher in spring than in summer and autumn. During this latter period they are also higher in the young than in adults.

III.3.4. Biological half-life and ecological half-life

Biological half-life is the time taken by an animal to eliminate half of a dose of a given radionuclide absorbed at a given moment in time. In general, the duration of the biological half-life depends on several factors (Crossley 1964, Lowe & Horrill 1988, Rantavaara et al. 1987) :

- animal size, with half-life often being longer the larger the animal; Cryer & Baverstock (1972) show that in humans the half-life of ^{137}Cs varies from 10 to 110 days, essentially as a function of weight; Crossley (1964) observed a similar correlation between the biological half-life of Cs and the size of insects;

- on external temperature, with half-life being shorter during the hot season than during the cold season; Holleman et al. (1971) also showed that the half-life of ^{134}Cs in reindeer was 2 to 4 times shorter in summer than in autumn-winter; this influence of external temperature is probably even more marked in cold-blooded animals;

- on internal temperature, with half-life being longer in cold-blooded animals (reptiles, fish).

To sum up, one can say that the more rapid an animal's metabolism, the greater the level of radionuclide elimination. On the other hand, it should be remembered that a rapid metabolism also leads to quicker absorption.

As for ecological half-life, this is the time taken by an animal to reduce by half its contamination while continuing to live in a contaminated environment. The ecological half-life is clearly longer than the biological half-life, because the animals continue to eat contaminated food.

Lowe & Horrill (1988) put at 28 days the ecological half-life of ^{134}Cs in roe deer. They report that other authors calculated a biological half-life for ^{137}Cs of 17 days for *Odocoileus hemionus* and of 33 days for reindeer. They therefore suppose, by comparing sizes, that the biological half-life of Cs in roe deer should be under 17 days. Consequently, the ecological half-life of Cs is probably at least twice as long as its biological half-life.

Using analysis results sent to us by the Radioprotection Division in the Grand Duchy of Luxembourg we calculated an ecological half-life of 51 days for radiocaesiums in roe deer during the June-September 1986 period.

Days	Species	Source
1.5	mice	Richmond, quoted by Crossley (1964)
3.0	rat	id.
7.5	monkey	id.
9.5	dog	id.
17	sheep	Goldman et al. (1965)
17	(roe?) deer	id.
14-21	roe deer (*Capreolus capreolus*)	Lowe & Horrill (1988)
22	id.	Molzahn et al. (1987)
10-35	reindeer (*Rangifer tarandus*)	Lowe & Horrill (1988)
21-35	id.	Hanson (1967)
± 30	moose (*Alces alces*)	Rantavaara et al. (1987)
34	id.	Mirell & Blahd (1989)
10-60	man (children <15 years)	Cryer & Baverstock (1972)
65-110	man (adults)	id.
60	man	Beentjes et al. (1988)
65	man	Hanson (1967)
± 91	man	Goldman (1987)
68	man (summer)	Lowe & Horrill (1988)
130	man (winter)	id.
0.3	man (1st Cs fraction)	Henrichs et al. (1989)
45-210	man (2nd Cs fraction)	id.
2	man (10% of Cs)	Watson (1986)
110	man (90% of Cs)	id.
110	man	Richmond, quoted by Crossley (1964)
115	man	Hayball et al. (1989)

Table 6 : Some half-life values published in the literature for ^{137}Cs.

IV. EDIBLE FOREST FUNGI

IV.1. INTRODUCTION

This chapter contains data collected by us on contamination of forest fungi likely to be consumed by man.

We used the following reference works :

- for the Latin names of fungi : Moser (1983) and Kreisel (1987);
- for the data on radiocontamination of fungi and their interpretation : Aumann et al. (1989), Bakken & Olsen (1989), Battiston et al. (1989), Bellù & Moroder (1986), Bellù (1989), Bersan & Degrassi (1986), Block & Pimpl (1990), Burri & Pallua (1986), Byrne et al. (1979), Byrne (1988), Consiglio et al. (1990), CRIIRAD (1988), Daillant (1987, 1989, 1991), de Meijer et al. (1987, 1988), Der Spiegel (1988), Dietl & Breitig (1988), Dietl (1989), Dighton & Horrill (1988), Eckl et al. (1986), Elstner et al. (1987, 1989), Fauvel (1951), Fischer (1972), Fourré (1988, 1989), Fraiture et al. (1989, 1990), Gans (1986, 1987), Gerzabek et al. (1988), Giovani et al. (1989), Gorham (1959), Govi & Innocenti (1987), Grüter (1964, 1967, 1971), Guillitte et al. (1987, 1989b), Haselwandter (1977, 1978), Haselwandter et al. (1988), Heinrich (1987), Heinrich et al. (1989), Henrich et al. (1988, 1989), Horyna & Řanda (1988), Horyna et al. (1988), Ijpelaar (1980), Johnson & Nayfield (1970), Johnson et al. (1989), Klán et al. (1988), Korky & Kowalski (1989), Kühn et al. (1986), Kuyper (1987), Lambinon et al. (1988), Lambinon (1989), Liiva & Parmasto (1989), Mascanzoni (1989), Mihok et al. (1989), Molzahn et al. (1989, 1990), Mornand (1988), Moser (1972), Muramatsu et al. (1991a, 1991b), Nimis et al. (1986, 1988b), Olsen et al. (1989), Oolbekkink & Kuyper (1989), Paulus & Reisinger (1990), Piérart (1986, 1989), Řanda et al. (1987, 1988, 1989, 1990), Řanda (1988, 1989), Rantavaara (1987), Rivasi (1988), Rogers & Williams (1986), Römmelt et al. (1987, 1988, 1989), Rückert & Diehl (1987), Sansone et al. (1988, 1990), Seeger (1978, 1987, 1989), Seeger & Schweinshaut (1981), Seeger et al. (1982), Société mycologique et botanique du Chablais (1991), Stijve (1967), Strullu (1985), Teherani (1987, 1988), Witkamp (1968), Witkamp & Barzansky (1968).

IV.2. CONTAMINATION INVOLVING RADIOCAESIUM ISOTOPES

Given the large number of bibliographic references and the multitude of species involved, it has not been possible to compile a review of the various species, as we have done for higher plants and game. We have limited this review to the radiocaesium activities observed in one single edible species, *Xerocomus badius*, which is well known and often has very high contamination levels. Afterwards we review data on the other radionuclides observed in fungi.

XEROCOMUS BADIUS (FR.: FR.) GILB.

Syn. : *Boletus badius* (Fr.: Fr.) Fr.

Order : *Boletales*
Family : *Boletaceae*

English : Bay boletus
French : Bolet bai
German : Maronenröhrling
Italian : Boleto baio
Spanish : Boleto rojo-moreno
Dutch : Kastanjeboleet

Contamination levels observed :

Rantavaara (1987) gives, for a sample collected in Finland in 1986, 9.4 Bq/kg FM for ^{134}Cs and 16 Bq/kg FM for ^{137}Cs.

Grüter (1971) had already noted levels of 1 110 Bq/kg FM for bay boletus collected in Germany in 1966. Grüter (1967) reports that this same species had ^{137}Cs activity of 1 308 cpm/g K in 1963 and of 7 560 cpm/g K in 1966.

Diehl et al. (in Seeger 1987) analysed 612 samples of *X. badius* harvested in Germany in 1986; ^{137}Cs contamination varied from 0.2 to 49 862 Bq/kg FM, with an average of 921 Bq/kg.

Dietl & Breitig (1988) observed activity of 509 Bq/kg FM for total ^{134}Cs + ^{137}Cs in bay boletus gathered in Baden-Württemberg (Germany).

Block & Pimpl (1989) analysed a sample of *X. badius* harvested in 1987 in Rhineland-Palatinate (Germany). The ^{137}Cs and ^{134}Cs levels were 314 and 89 Bq/kg FM respectively. A sample of the same species, collected from the same spot in 1988, contained only 226 and 50 Bq/kg FM of these two isotopes. Two samples of the same species collected at another site gave (respectively) 820 and 340 Bq/kg FM in 1986 and 660 and 220 Bq/kg FM in 1987.

Molzahn et al. (1989) analysed 25 samples of this species, collected in Marburg-Biedenkopf district (Germany). Their radiocaesium contamination level was 370 Bq/kg FM on average. Measurements carried out on cuticles alone showed that they had activity 3.1 to 3.3 times greater than that of whole fungi. In 1988, in the same region, the levels ranged from 13 to 1 197 Bq, with an average of 519, in the same region (Molzahn et al. 1990).

Still in Germany, Dietl (1989) harvested *Xerocomus badius* with radiocaesium levels of 7 500 to 80 000 Bq/kg DM. He observed that contamination of the tubes and the flesh of the cap is not much greater than that of the stem. The cuticle is twice as radioactive as the base of the stem.

Gans (1986, 1987) analysed fungi collected in Berlin in 1986. Three samples of bay boletus gave levels for ^{134}Cs ranging from 96 to 127 Bq/kg FM and from 295 to 422 Bq/kg FM for ^{137}Cs.

Rückert & Diehl (1987) analysed the contamination levels of 34 species of fungi harvested in Germany in 1986, including 12 samples of *Xerocomus badius*. The levels of ^{134}Cs in the latter varied from 29 to 1 100 Bq/kg FM, with an average of 219, and those of ^{137}Cs varied from 68.3 to 2 510 Bq/kg FM, with an average of 558.

Paulus & Reisinger (1990) analysed fruit bodies of *X. badius* harvested in the Fichtelgebirge (Germany) from 1985 to 1988. Average radiocaesium contamination, expressed in Bq/kg DM, was 2 100 in 1985, 19 000 in 1986, 29 100 in 1987 and 17 900 in 1988.

As for Bavaria, the worst affected region in Germany, in 1986 Elstner et al. (1987) measured 56 Bq/kg FM for ^{134}Cs and 350 Bq for ^{137}Cs in an old bay boletus, while the levels of the same radionuclides in a young fruit body were 238 and 641 Bq.

Still in Bavaria, Römmelt et al. (1987) measured up to 13 000 Bq of radiocaesium/kg FM in bay boletus. Römmelt et al. (1988) obtained average radiocaesium levels of 1 930 and 5 960 Bq/kg in this same species, with $^{134}Cs/^{137}Cs$ ratios of 0.34 and 0.32. Römmelt et al. (1989) observed activities of 555 Bq ^{134}Cs/kg FM and 2 555 Bq ^{137}Cs/kg FM in this same species. An article in Der Spiegel magazine (1988) reported levels of up to 40 000 Bq/kg in this same region.

A listing from the Ludwig-Maximilians-Universität in Munich (1986) reproduces the results of analyses of 8 samples of bay boletus. The radiocaesium levels range from 303 to 7513 Bq/kg and those for ruthenium from 0 to 83 Bq/kg.

Teherani (1987) measured 1 554 to Bq/kg FM of ^{134}Cs and 3 848 Bq of ^{137}Cs in a *Xerocomus badius* collected in Austria in 1986. Subsequently, the same author (1988) measured levels of 11 to 607 Bq/kg FM for ^{134}Cs and of 67 to 1 598 Bq for ^{137}Cs in the fruit bodies of this species gathered in 1987.

Eckl et al. (1986) calculated the soil-fruit body transfer factors in 1980-1982, putting them at around 30 to 70 for the bay boletus.

The CEC Joint Research Centre in Ispra sent us the following figures for an analysis of a sample of bay boletus collected in Austria in 1986 : 11 100 Bq ^{137}Cs/kg FM and 5 180 Bq ^{134}Cs/kg FM.

Heinrich (1987) measured the activity (total for gamma emitters?) in two samples of *Xerocomus badius* harvested in Austria in 1986. The values obtained are 4 662 and 10 777 Bq/kg FM for the caps and 2 257 and 6 623 Bq/kg FM for the stems. Heinrich et al. (1989) measured 3 737 Bq ^{137}Cs and 1 184 Bq ^{134}Cs/kg FM in a sample of the same species harvested in 1987 in Austria.

Henrich et al. (1989) studied contamination of this species in Austria in 1988. In the Kobernausserwald and the Weinsbergerwald the levels for ^{134}Cs varied between 45 000 and 98 400 Bq/kg DM (74 560 Bq on average) and those for ^{137}Cs between 59 570 and 142 000 Bq/kg DM (109 900 Bq on average).

Finally, still in Austria, Gerzabek et al. (1988) noted in 26 samples of bay boletus an average contamination level of 49 547 Bq ^{137}Cs/kg DM, with a maximum of 147 260 Bq ^{137}Cs/kg DM. Differences, such as those mentioned above, between the levels measured in the same species in the same region are not surprising for Austria, since this is a country where deposition varied greatly from one region to another.

Horyna & Řanda (1988) analysed the ^{137}Cs content of this species in Czechoslovakia. In 1986 it ranged from 1 400 to 16 500 Bq/kg DM, and the authors calculated soil-fungus concentration factors ranging from 7 to 99. In 1987 the levels observed ranged from 3 000 to 50 000 Bq/kg DM.

In 1986 Horyna et al. (1988) measured levels varying from 14 800 to 17 000 Bq/kg DM in bay boletus collected around Prague.

In the same year Klán et al. (1988) analysed samples of this species from Czechoslovakia having some 370 to 30 000 Bq of ^{137}Cs/kg FM. The levels observed in the same region prior to the Chernobyl accident ranged from 240 to 2 440 Bq.

Still in Czechoslovakia, Řanda et al. (1988) measured radiocaesium activity of 20 000 to 50 000 Bq/kg DM. Deposition was as high as 30 000 Bq/m^2 in certain parts of the country. Řanda et al. (1989) analysed bay boletus from Czechoslovakia; average contamination (in Bq ^{137}Cs/kg DM) was 19 283 in 1986 if one disregards one aberrant sample of 540 Bq; it was 34 900 Bq in 1987 and 33 167 Bq in 1988. In these same samples the average $^{134}Cs/^{137}Cs$ ratio changed from 0.38 in 1986 to 0.28 in 1987 and to 0.19 in 1988 as a result of radioactive decay.

Řanda et al. (1989) analysed 18 samples of *X. badius* collected in Czechoslovakia in 1988. The activities measured, expressed in Bq/kg (DM?), vary from 200 to 47 000 for ^{137}Cs (average : 20 514) and from the detection limit to 8 500 for ^{134}Cs (average : 3 724). Comparison of two of these readings with the levels measured in samples of the same species collected in the same places in 1986 and 1987 shows a clear increase in radioactivity : the levels observed in 1988 are 2.6 and 3.4 times higher than those noted in 1986.

Byrne (1988) analysed samples of *Xerocomus badius* collected in Slovenia (Yugoslavia), observing 1 200 to 66 000 Bq/kg DM (on average : 19 000) for the radiocaesiums.

Sansone et al. (1988) harvested bay boletus around Lake Como in Italy. In 1986 two samples measured (in Bq/kg FM) 1 073 and 1 225 for ^{134}Cs and 2 586 and 2 926 for ^{137}Cs. In 1987 4 samples read 381 to 3 236 and 1 184 to 10 309 for the same two isotopes. Sansone et al. (1990) harvested fungi in the same region in 1988 and 1989. The activities recorded for *Xerocomus badius*, expressed in Bq/kg FM, range from 358 to 758 for ^{134}Cs and from 1 694 to 3 515 for ^{137}Cs in 3 samples collected in 1988; for two samples harvested in 1989 the respective figures are 157 and 338 and 1 068 and 2 292.

Daillant (1989) quotes figures from the CEC Joint Research Centre in Ispra (Italy) giving activities of 682 Bq/kg FM for ^{134}Cs and 1 659 Bq/kg for ^{137}Cs in bay boletus.

The above research centre also sent us the results of an analysis of 5 samples of this fungus collected in Italy in 1986 : the levels ranged from 254 to 3 248 Bq ^{137}Cs/kg FM and from 89 to 1 489 Bq ^{134}Cs/kg FM.

The CRIIRAD (1988) gives results of analyses carried out on this same species in the Loire (France). The activities recorded were as high as 12 745 Bq/kg DM for ^{137}Cs in 1987 and between 2 685 and 13 000 Bq in 1988.

Fourré (1988 and 1989) published the results of analyses carried out in France from 1986 to 1988. The ^{134}Cs levels range from 0 to 1 750 Bq/kg FM and those for ^{137}Cs between 90 and 5 700. A sample of bay boletus collected in France in 1990 contained 3 222 Bq of ^{137}Cs and 157 Bq of ^{134}Cs/kg DM (Société mycologique et botanique du Chablais 1991).

De Meijer et al. (1988) published levels of 520 Bq/kg DM for ^{134}Cs and 1 620 Bq for ^{137}Cs in bay boletus in the Netherlands.

In the Grand Duchy of Luxembourg the Radioprotection Division measured levels of 45 to 522 Bq/kg FM for ^{134}Cs and of 98 to 1 149 Bq for ^{137}Cs in 14 samples of *Xerocomus badius* collected in 1986. In 1987 the respective figures in four samples were 148 to 420 and 508 to 1 397 Bq for these two isotopes. In 1988 five samples gave readings of 123 to 631 and 604 to 3 090 Bq respectively.

Finally, in Belgium we collected and analysed, in cooperation with the Faculté Agronomique in Gembloux (Fraiture et al. 1989, Guillitte et al. 1987), 15 samples of *Xerocomus badius* in 1986, which had levels of 103 to 4 900 Bq/kg DM for ^{134}Cs and of 251 to 12 600 Bq for ^{137}Cs. In 1987 an analysis of 91 samples of bay boletus revealed levels of 0 to 6 260 Bq/kg DM for ^{134}Cs (average = 1 260) and of 25 to 21 400 Bq for ^{137}Cs (average = 4 755). In 1988 the corresponding levels for 24 samples were 40 to 3 502 and 176 to 16 185 respectively for the same isotopes. This made it possible to draw up maps charting the activity levels in various species of fungus used as bioindicators of radioactive contamination.

OTHER SPECIES OF EDIBLE FUNGI

The following is an overview of edible wild fungi culled from a study of the relevant bibliographic references.

E = edible, EE = tasty, EEE = very tasty; C = little or no contamination, CC = average contamination, CCC = high contamination.

Boletaceae	Gyroporus castaneus	E	C
	Gyroporus cyanescens	EE	C
	Boletus edulis	EEE	C
	Boletus reticulatus	EE	C
	Boletus aereus	EEE	C
	Boletus pinophilus	EE	C
	Boletus appendiculatus	EE	C
	Boletus regius	EE	C
	Boletus impolitus	E	C
	Boletus calopus	E	C
	Boletus luridus	E	C
	Boletus erythropus	EE	C
	Boletus queletii	E	C
	Leccinum aurantiacum	EE	C
	Leccinum versipelle	E	C
	Leccinum duriusculum	E	C
	Leccinum scabrum	E	C
	Leccinum variicolor	E	C
	Leccinum carpini	E	C
	Leccinum crocipodium	E	C
	Xerocomus subtomentosus	E	C
	Xerocomus chrysenteron	E	CCC
	Xerocomus rubellus	E	CCC

	Xerocomus badius	EE	CCC
	Chalciporus piperatus	E	C
	Suillus luteus	EE	C
	Suillus grevillei	E	C
	Suillus granulatus	E	C
	Suillus bovinus	E	CCC
	Suillus variegatus	E	CCC
Strobilomycetaceae	Porphyrellus pseudoscaber	E	C
Paxillaceae	Hygrophoropsis aurantiaca	E	C
Gomphidiaceae	Chroogomphus rutilus	E	CCC
Hygrophoraceae	Hygrophorus penarius	EE	C
	Hygrophorus nemoreus	EE	C
Tricholomataceae	Clitocybe clavipes	E	CCC
	Clitocybe alexandri	E	C
	Clitocybe geotropa	EE	C
	Clitocybe gibba	E	CC
	Clitocybe nebularis	E?	C
	Clitocybe odora	E	C
	Armillaria mellea	E(E)	C
	Lepista inversa	EE	C
	Lepista irina	E	CC
	Lepista nuda	E(E)	C
	Laccaria laccata	E	CCC
	Laccaria amethystina	E	CCC
	Tricholomopsis rutilans	E	C
	Tricholoma terreum	E	C
	Tricholoma scalpturatum	E	C
	Tricholoma cingulatum	E	C
	Tricholoma equestre	EE	C
	Tricholoma portentosum	E	C
	Melanoleuca cognata	E	C
	Lyophyllum decastes	E	C
	Calocybe gambosa	EEE	C
	Oudemansiella radicata	E	C
	Marasmius oreades	EE	C
	Megacollybia platyphylla	E	CC
	Collybia fusipes	E	C
	Collybia dryophila	E	C
	Mycena pura	?	C
Entolomataceae	Clitopilus prunulus	EE	C
	Entoloma clypeatum	E	C
	Entoloma sepium	E	C
Pluteaceae	Pluteus cervinus	E?	C
	Volvariella speciosa	E	C
Amanitaceae	Limacella lenticularis	E	C
	Amanita vaginata	E(E)	C
	Amanita fulva	E	CC
	Amanita crocea	E	C
	Amanita spissa	E	C
	Amanita rubescens	EE	C
	Amanita citrina	E	CC

Agaricaceae	Agaricus arvensis	EEE	C
	Agaricus macrosporus	E	C
	Agaricus augustus	EE	C
	Macrolepiota procera	EE(E)	C
	Macrolepiota rhacodes	EE	C
Coprinaceae	Coprinus comatus	EE	C
	Coprinus atramentarius	E	C
Bolbitiaceae	Agrocybe praecox	E	C
Strophariaceae	Hypholoma capnoides	E	C
	Kuehneromyces mutabilis	EE	C
	Pholiota squarrosa	E	C
Cortinariaceae	Cortinarius praestans	E	C
	Rozites caperata	EE	CCC
Russulaceae	Russula cyanoxantha	EE	C
	Lactarius deliciosus	E	C
	Lactarius sanguifluus	EE	C
Cantharellaceae	Craterellus cornucopioides	EE	C
	Cantharellus cibarius	EEE	C
	Cantharellus tubaeformis	EE	CC
Sparassidaceae	Sparassis crispa	EE	C
Hydnaceae	Hydnum repandum	EE	C
Polyporaceae	Pleurotus ostreatus	EE	C
	Pleurotus cornucopiae	E	C
	Polyporus squamosus	E	C
	Grifola frondosa	E	C
	Grifola umbellata	E	C
	Meripilus giganteus	E	C
	Fistulina hepatica	E	C
Phallaceae	Phallus impudicus	E	C
Lycoperdaceae	Langermannia gigantea	EE	C
Auriculariaceae	Auricularia auricula-judae	E	C
Morchellaceae	Morchella div. sp.	EEE-E	C
	Mitrophora semilibera	E	C
Pezizaceae	Peziza badia	E	C
	Peziza vesiculosa	E	C

IV.3. OTHER RADIONUCLIDES OBSERVED

In addition to ^{134}Cs and ^{137}Cs, 23 other radioisotopes were observed in fungi by the authors we consulted.

IV.3.1. Strontium (^{90}Sr)

The half-life of ^{90}Sr is 29 years. In contrast to most of the other radionuclides mentioned in this report, this isotope emits beta radiation, not gamma radiation. Seeger et al. (1982) studied the concentration of chemical Sr in 1 169 samples, made up of 433 species of fungus. The levels observed varied from practically 0 to 174.5 mg/kg DM. They also observed a strong correlation between the concentrations of Sr and those of Ca, its chemical homologue. They conclude that the species studied do not accumulate Sr because the exchangeable Sr concentrations observed in 12 samples of soil and litter were not lower than the Sr concentrations measured in the corresponding fungi.

However, slightly preferential absorption of Sr in comparison to Ca does exist to some extent because the Sr/Ca concentration ratio is higher in the fungi than in the soils analysed. The species which absorb the most Sr are not the same as those which accumulate Cs. Here we are dealing mainly with Ascomycetes as well as lignicolous and fimicolous fungi. It is believed that the substrate plays a role : many of the samples with a high Sr content were collected from the edge of metalled roads founded on limestone gravel.

Mascanzoni (1989b) studied the presence of ^{90}Sr in fungi. The activities observed are low (with the averages ranging from 0.52 to 2.68 Bq/kg DM) when compared with the fallout or ^{137}Cs activities (10 to 115 000 Bq/kg DM) recorded in the same samples, this latter isotope having a roughly equivalent half-life.

Römmelt et al. (1989) measured up to 60 Bq/kg DM for ^{90}Sr in fungi from southern Bavaria. They, too, observed that uptake of this isotope by fungi was lower than that of Cs. They note, however, that it is better adsorbed than Cs by organic matter, but less well by clay minerals, which is probably due to it being a divalent cation.

Finally, it should be noted that ^{90}Sr had already been observed in fungi prior to the Chernobyl accident (Grüter 1971).

IV.3.2. Ruthenium (^{106}Ru and ^{103}Ru)

^{106}Ru, which has a half-life of approximately one year, had already been observed in fungi prior to the Chernobyl accident (Grüter 1971). It was detected in the fruit bodies of *Boletus edulis* from around Lake Como (Italy) (Sansone et al. 1988). It has also been observed in fungi collected in Germany (Molzahn et al. 1990).

Very small quantities of ^{106}Ru have also been observed in fungi collected in north-eastern Italy by Battiston et al. (1989). These authors put forward the hypothesis that fungi absorb hardly any of this isotope or, at least, do not concentrate it, which would explain the very low levels observed.

^{103}Ru, whose half-life (39 days) is much shorter than that of ^{106}Ru, has sometimes been detected in fungi. Sansone et al. (1988) observed its presence in *Boletus edulis* from around Lake Como (Italy).

Gans (1986) found ^{103}Ru in the fruit bodies of various species collected in Berlin on 6 June 1986. The concentrations are often near to the limit of detection and never exceed 17 Bq/kg FM.

Teherani studied the presence of this radioisotope in the fructifications of various species collected in Austria in 1986. The activities vary from 3.7 to 148 Bq/kg FM, this latter level being found in a sample of *Macrolepiota procera*.

^{103}Ru was detected in small quantities in numerous samples of fungi collected in 1986 in Finland and analysed by Rantavaara (1987). The concentrations observed were always below 4 Bq/kg FM, with the exception of some fruit bodies of *Gyromitra esculenta*, which probably received direct fallout (collected on 30 April and 12 May 1986). These 7 samples all contained ^{103}Ru, with levels varying from 6.2 to 750 Bq/kg FM.

The analysis results sent to us by the CEC Joint Research Centre in Ispra concerning fungi collected in Italy in 1986 contain a lot of data on ^{103}Ru. However, only some 20 samples have a level exceeding 40 Bq/kg FM. The highest levels are found in non-identified fungi (555 and 337 Bq/kg FM) and also in *Russula* sp. (359 Bq/kg), *Macrolepiota* sp. (quite a number of contaminated samples, highest level = 196 Bq/kg), *Boletus granulatus* (86 Bq/kg) and *Calocybe georgii* (52 Bq/kg). Finally, a sample of *Cantharellus lutescens* contained 155 Bq ^{106}Ru/kg FM.

Finally, a listing from the Ludwig-Maximilians-Universität in Munich giving the results of analyses of 137 samples of fungus collected prior to 15.9.1986 shows that most contain radioactive ruthenium (exact types of isotope not specified). The activities range from 0 to 494 Bq/kg.

IV.3.3. Silver (^{110m}Ag)

^{110m}Ag is a radionuclide with a half-life of 250 days. According to Molzahn et al. (1990), it gives birth to ^{110}Cd, which is doubly toxic radioactively and chemically (it being a heavy metal).

Since 1979 Byrne et al. have shown large differences in the accumulative ability of the various fungus species with regard to Ag (chemical). Among the 32 species analysed by these authors, the highest concentrations were found in agarics (*Agaricus* div. sp. : 10.5 to 133 mg/kg DM), *Calvatia utriformis* (11.4 mg/kg), *Lycoperdon perlatum* (10.2 mg/kg), *Paxillus involutus* (6.7 mg/kg) and *Boletus edulis* (5.8 mg/kg). The accumulative ability of these species is remarkable : one sample of *Agaricus campester* contained 133 mg Ag/kg DM whereas the soil contained less than 0.1 mg/kg.

Distribution of Ag in the various parts of the fruit bodies is uneven : 41.4 mg/kg DM in the tubes, 11.8 mg/kg in the cap and 7.8 mg/kg in the stem of *Boletus edulis*.

Byrne (1988) measured fairly high concentrations of ^{110m}Ag in fungi in Yugoslavia : 50 to 210 Bq/kg DM for *Agaricus campester*, 120 to 565 Bq/kg for *Lycoperdon perlatum* and 205 Bq/kg for *Calvatia utriformis*. On average these concentrations are comparable to those of radiocaesium isotopes. They are not proportional to the concentration of chemical Ag; this is probably due to the fact that the distribution of chemical Ag and of ^{110m}Ag is not identical : one is a function of the nature of the soil whereas the other is a function of radioactive deposition.

Among the fungi analysed by Gans (1986), only *Stropharia squamosa*, *Marasmius oreades*, *Agaricus arvensis* and *Bovista plumbea* (listed in order of increasing contamination) contained ^{110m}Ag. The activities observed did not exceed 13 Bq/kg FM, however.

Molzahn et al. (1990) observed the presence of ^{110m}Ag in fungi gathered in Upper Hesse (Germany) in 1988. The levels recorded varied from 1.1 to 8.6 Bq/kg FM, with the highest levels being found in *Macrolepiota procera*, *Xerocomus badius*, *Lycoperdon perlatum*, *Xerocomus chrysenteron*, *Agaricus campestris* and *Calvatia excipuliformis*.

Sansone et al. (1988) detected ^{110m}Ag in the fruit bodies of *Bovista* and *Lycoperdon* gathered in northern Italy. Just like the previous authors, they observed activities comparable to those of radiocaesium isotopes.

Battiston et al. (1989) also observed ^{110m}Ag in fungi gathered in north-eastern Italy in 1986. The activities varied from 20 to 1 873 Bq/kg DM. The highest levels were found in *Lycoperdon perlatum* and *Clitocybe gibba* and the lowest levels in *Boletus edulis* and *Boletinus cavipes*. However, it is difficult to compare the figures because the samples came from different sites. The ^{110m}Ag/^{137}Cs ratio is 10 to 100 times greater in *Lycoperdon perlatum*, *Agaricus campester*, *Macrolepiota procera* and "*Lepiota pudica*" than in the fallout (where it is from 0.018 to 0.005).

Rantavaara (1987) analysed 205 samples of fungi made up of 37 species harvested in Finland in 1986. Only 22 samples showed traces of ^{110m}Ag; the concentrations never exceeded 30 Bq/kg FM and the $^{110m}Ag/^{137}Cs$ ratio was generally below 0.01. This is probably due to the fact that not many Ag accumulator species were analysed. The only sample of *Agaricus campestris* contained 1.7 Bq ^{110m}Ag/kg, 0.8 Bq ^{137}Cs/kg and no ^{134}Cs.

A listing from the CEC Joint Research Centre in Ispra containing numerous results of measurements carried out in Italy in 1986 contains some information on ^{110m}Ag. The highest levels (only concentrations exceeding 10 Bq/kg FM are given) were found in *Lycoperdon* (13 and 281 Bq/kg), *Agaricus* sp. ("prataiolo", 22 Bq), *Lepiota naucina* (= *Lepiota leucothites*, 89 Bq), *Macrolepiota procera* (8 readings, average = 28 Bq, maximum = 122 Bq), *Clitocybe gibba* (43 Bq), *Clitocybe nebularis*, *Lepista nuda*, *Langermannia gigantea* (10.4 Bq), *Armillaria mellea* (37 Bq), *Boletus edulis*, *Cantharellus lutescens* and *Cortinarius violaceus*. Most of these species are known to accumulate ^{110m}Ag to varying degrees.

Finally, Daillant (1987) notes that ^{110m}Ag is absorbed only by lepiotas, which clearly contradicts the data from the literature quoted above. This author reproduces results of analyses carried out by the TÜV [= Technical Inspection Authority] in Baden and gives levels of 4 Bq/kg and from 7 to 13 Bq/kg (FM?) for lepiotas from France and Germany respectively.

IV.3.4. Tellurium (^{129m}Te and ^{132}Te)

These two isotopes have a short half-life, 33 days and 78 hours respectively. They were observed in several fungi collected in Italy in 1986 (figures from the CEC Joint Research Centre in Ispra), viz. : *Armillaria mellea* (126 Bq ^{129m}Te), *Armillaria tabescens*, *Boletus edulis* (18.5 and 86 Bq ^{129m}Te and 73 Bq ^{132}Te), *Boletus granulatus*, *Clitocybe infundibuliformis*, *Langermannia gigantea*, *Lepiota naucina* (= *Lepiota leucothites*, 22 Bq ^{129m}Te) and *Macrolepiota* sp. (figures expressed in Bq/kg FM, given only if above 10).

On the other hand, these isotopes were observed by Rantavaara (1987) in samples of *Gyromitra esculenta* collected in Finland from 30 April to 12 May 1986 and which probably received direct fallout. The concentrations were 78 Bq ^{129m}Te and from 51 to 4 140 Bq ^{132}Te/kg FM, this latter level being found in a sample taken on 30 April.

IV.3.5. Iodine (^{131}I)

This isotope, which has a very short half-life (8 days), was detected in various species of fungi collected in Berlin on 6.6.1986 (Gans 1986), but in very low concentrations, often at the limit of detection.

Bellù and Moroder (1986) observed it in samples from Bolzano province (Italy). The activities were always below 111 Bq/kg.

Of the 205 samples of fungi analysed by Rantavaara (1987), only the fruit bodies of *Gyromitra esculenta*, an edible species, were contaminated with ^{131}I. The concentrations are sometimes high and vary from 3.6 to 9 170 Bq/kg FM. The most contaminated samples were harvested during the 40 days following the fallout, which explains such high values.

IV.3.6. Cerium (^{141}Ce and ^{144}Ce)

The half-lives of these two radionuclides are 33 and 284 days respectively. Traces of ^{144}Ce were observed in fungi prior to the Chernobyl accident by Eckl et al. (1986) and by Grüter (1971).

Rantavaara (1987) noted traces of ^{141}Ce in a small number of fungi harvested in Finland in 1986. However, the levels are very low (≤ 3.5 Bq/kg FM), with the exception of *Gyromitra esculenta* which received fallout directly and for which rates of 0 to 400 Bq/kg FM were measured. ^{144}Ce was not measured in fungi by this author (although it was measured in vegetables).

The file sent by the CEC Joint Research Centre in Ispra contains four readings for cerium in fungi collected in Italy in 1986 (concentrations given in Bq/kg FM) : *Cantharellus lutescens* (63 Bq ^{141}Ce), *Armillaria tabescens* (7.4 Bq ^{144}Ce) and *Boletus granulatus* (5.9 Bq ^{141}Ce and 7.4 Bq ^{144}Ce).

IV.3.7. Miscellaneous

A number of other radioactive isotopes have been detected in fungi, but not often.

The half-life of ^{125}Sb is 2.76 years. It had already been detected in fungi in 1980-1982 (Eckl et al. 1986) and was found again by Battiston et al. (1989) in the same type of sample. These latter authors put forward the hypothesis that ^{125}Sb absorption by fungi is very low, which would explain the very low rates observed despite the presence of this radionuclide in the soil.

Eckl et al. (1986) observed traces of ^{7}Be, ^{95}Zr, ^{95}Nb, ^{210}Pb, ^{226}Ra and ^{238}U in fruit bodies harvested in 1980-1982.

Daillant (1991) observed concentrations of ^{226}Ra (some 70 to 136 Bq/kg DM) and of ^{210}Pb (approximately 61 and 702 Bq/kg DM) in samples of *Coprinus comatus* and *C. atramentarius* collected near a disposal site for radioactive waste in France. The soil-fungus concentration factor was approximately 0.6 to 0.7.

Grüter (1971) measured ^{106}Rh activity in fungi.

The file sent to us by the CEC Joint Research Centre in Ispra contains a number of readings concerning ^{95}Nb, ^{95}Zr ^{99}Mo, ^{125}Sb, ^{132}I, ^{140}Ba, ^{140}La and ^{226}Ra detected in fungi collected in Italy in 1986. The levels observed are always below 75 Bq/kg FM.

Finally, Rantavaara (1987) occasionally detected ^{95}Nb, ^{95}Zr, ^{99}Mo, ^{140}Ba, ^{147}Nd and ^{239}Np in fungi collected in Finland in 1986. Apart from the first two isotopes listed, for which a small number of samples exhibit some activity (maxima respectively of 18 and 35 Bq/kg FM), these isotopes were detected only in the fruit bodies of *Gyromitra esculenta* which probably received direct fallout. In these particular cases, activities of several hundred Bq/kg FM were sometimes observed and a record level of 1 300 Bq/kg FM was measured for ^{140}Ba in fruit bodies.

IV.4. SUMMING-UP AND COMMENTS

Consumption of wild fungi is fairly widespread among the public. A survey carried out in Spain (Garcia Rollán 1989) shows that 64% of the rural population consumes this type of food; on average 19 times a year, with an average of 301 grams (fresh matter) of fungi per meal, which corresponds - on average again - to 5.7 kg of fresh fungi per mycophagist and per year.

Radioactive contamination of fungi has been the subject of numerous studies, mainly in Europe, following the Chernobyl accident. However, several authors had already demonstrated prior to this date the capacity of fungi to accumulate radionuclides (see references cited in IV.1). *In-vitro* studies have also been carried out to show the accumulative ability of fungi (Johnson et al. 1989, Witkamp 1968).

As in the case of plants and animals, contamination of fungi varies as a function of deposition (Fraiture et al. 1989, Guillitte et al. 1987, Mascanzoni 1989b). However, after 1986 this link is sometimes blurred, due to the growing influence of the nature of the soil and its impact on radionuclide availability.

Many species develop very high levels, exceeding by far those of plants and animals living in the same ecosystems (Bakken & Olsen 1989, Fraiture et al. 1989, Olsen et al. 1989). Certain record levels are enough to make radioecologists' heads reel : up to 2 000 000 Bq/kg DM in Norway (Bakken & Olsen 1989) !

On the basis of this ability to accumulate radiocaesium it should be possible to use fungi to obtain a very concentrated solution of radionuclides. Řanda (1988) refers to such a method, which involves blanching water, ammonium molybdophosphate and a cation exchange resin.

Witkamp & Barzansky (1968) calculated that Cs content in the mycelium of fungi accounts for a not inconsiderable proportion of the radiocaesium in the soil. Olsen et al. (1989) calculated that the mycelium of fungi was present in quantities of 72 to 838 g/m^2 (average : 200 g) in the top 3 cm of soil and that this mycelium contained approximately 1/3 of the Cs present in the soil. This accumulative ability had prompted some people to propose collecting fructifications as a way of decontaminating forest soils. However, Kuyper (1987) estimates that removing all fruit bodies during one year from a given area would decrease the radioactivity of the soil at that particular site by only 1%, whereas radioactive decay reduces it by 2.3%.

Some authors took advantage of fungus contamination to make autoradiographs (Haselwandter 1977, Haselwandter et al. 1988).

Comparing the levels observed in 1986 with those measured prior to the Chernobyl accident reveals a large increase (on average 3 to 4.8 times, according to Haselwandter et al. 1988).

Furthermore, between 1986 and 1989 the species' contamination levels decreased only very slowly, or even increased (CRIIRAD 1988, Paulus & Reisinger 1990, Řanda et al. 1989, Sansone et al. 1988). An increase of this kind was seen especially in species with a deep-growing mycelium (*Boletus edulis*), which had not until then been affected by the main body of the Chernobyl contamination.

In general, species growing in the open (pratincolous) have very little contamination, just like those which grow on a woody substrate (lignicolous) because this substrate absorbed hardly any radioactive fallout.

Among the species growing in soil in forests there are two distinct ecological groups : the humo-terricolous saprophytes and the mycorrhizal fungi. The first of these is made up of fungi whose mycelium feeds on more or less decomposed organic matter (dead leaves, twigs, humus, etc.) located in the upper part of the soil and in the litter. They generally have average contamination levels, but some of them (*Clitocybe clavipes*, for example) can be highly contaminated. Given the location of their mycelium, these fungi are the ones which become contaminated most rapidly following fallout. However, after one or two years their contamination level has fallen sharply, because the litter on which they feed is then much less contaminated.

The other ecological category of fungi, the mycorrhizal species, lives in symbiosis with the trees in the forest and exchanges nutritive substances with the roots. The mycelium of these species is located at different depths according to the species (Figure 9) and is affected that much later by fallout the further it is from the surface.

The species with a deep-growing mycelium (*Boletus edulis*, for example) experienced an increase in contamination during at least five years because the wave of Chernobyl radionuclides - which percolates very slowly through the soil - took some time to arrive at their level (Molzahn et al. 1990).

Other factors also influence fungus contamination : nature of the soil, nature of the forest stand, etc. as we mentioned in the first chapter of this report.

Contamination is not spread evenly over the various parts of the fructifications. Several authors have demonstrated this (Dietl 1989, Fraiture et al. 1989, Haselwandter et al. 1988, Molzahn et al. 1990). The flesh of the cap is on average two times more contaminated than that of the stem. A similar phenomenon was observed for chemical Cs (Seeger & Schweinshaut 1981) and for K, the homologue of Cs (Seeger 1978).

Aumann et al. (1989) discovered that certain pigments (badione and norbadione) found in the cuticle of the cap of *Xerocomus badius* and other related species (*Boletus erythropus*, for example) complex ^{137}Cs and can thus engender an appreciable accumulation of this isotope. Molzahn et al. (1990) measured in *Xerocomus badius* an activity 2.65 times higher in the cuticle than in the flesh; the corresponding figure was only 1.1 for *Boletus edulis*.

Seeger & Schweinshaut (1981) measured the concentrations of chemical Cs in 533 species (1 166 samples) of fungi, and observed large differences in accumulative capacity. The families whose representatives contained the greatest amounts of Cs were the *Cortinariaceae*, the *Clavariaceae*, the *Rhodophyllaceae* and the *Strophariaceae*. The families which were less accumulative were the *Helvellaceae* and the *Lycoperdaceae*. The findings of Byrne et al. (1979) point in the same direction.

The conclusions of these authors tally quite well on radiocaesium levels observed in fungi after the Chernobyl accident, but some differences exist nevertheless. These could be due to the fact that chemical Cs is much better distributed in the soil profile than radioactive caesium, most of which is still contained within the top layers. This difference in distribution leads to a difference in average biological availability, which is much higher in the holorganic horizons. This no doubt explains the fact - as observed by certain authors (Klán et al. 1988) - that fungi absorb ten times less chemical Cs than radiocaesium.

Seeger (1978) studied the concentrations of chemical K in fungi, and found extremely high levels : 1.5 to 117 g/kg DM. The family containing the most is that of the *Coprinaceae*, whereas the *Polyporaceae* contain the least. In actual fact, the K concentrations in fungi are fairly constant, especially if expressed in terms of water content (as suggested by Nimis et al. 1988a) and show no correlation with Cs concentration. During a study of radioactive contamination of fungi in Japan, Muramatsu et al. (1991a) obtained results for ^{40}K parallel to those given above for chemical K.

IV.5. METHODOLOGICAL PRECAUTIONS TO BE TAKEN WHEN USING FUNGI AS BIOINDICATORS

When measuring the radioactivity in a fungus sample the aim is not just to find out whether it is "edible or non-edible". Fungi analysis makes it possible to estimate the degree of contamination of the ecosystems in which they are gathered, and even of neighbouring ecosystems, and thus draw up contamination maps for a whole region (Fraiture et al. 1989, 1990). Thus, fungi can be used as bioindicators of radioactive contamination. In order to use samples to this end, however, it is necessary to adhere to a number of methodological principles, as set out below.

Collect species with a strong tendency to accumulate radionuclides. For radiocaesium isotopes the following can be used : *Cortinarius alboviolaceus, C. anomalus**, *C. armillatus**, *C. brunneus, C. delibutus* and most of the species of the genus *Cortinarius** (except *C. elatior*) and of the genus *Dermocybe**; *Hygrophorus olivaceoalbus, H. pustulatus, Laccaria amethystina, L. laccata, Paxillus involutus, Rozites caperata, Tylopilus felleus, Xerocomus badius* and *X. chrysenteron*. The species followed by an asterisk are those which had the most contamination during analyses of 1 576 samples representing 249 species and were collected in Belgium in 1987 in cooperation with the Faculté des Sciences Agronomiques de Gembloux (Fraiture et al. 1989). Radiocaesium activity in these species often exceeded 10 000 Bq/kg DM. *Russula ochroleuca* accumulates a little less radiocaesium than the above-mentioned species, but is nevertheless of interest because it is very common and is found in a wide range of varying ecological conditions. The bioindicator species for other radioisotopes (in particular ^{110m}Ag) are given in IV.3.

When collecting samples, **carefully note the type of trees constituting the forest canopy** (woody species present, age and density of stand) **as well as the exact location of the fruit bodies gathered** (for example, at the foot of a large beech tree or, on the contrary, away from the base of trees, on the edge of a road, in a patch of moss under a small gap in the forest canopy, etc.). The influence of these factors on contamination of fungi was dealt with in I.4.

Also note down the type of soil and humus : the pH of the soil (limy or acid soil), its composition (sandy, loamy, clayey, peaty soil), rate of decomposition of organic matter (litter thickness, type of humus : mull, moder or mor) and soil moisture content (high or low, permanent or not). These characteristics determine the biological availability of radionuclides in the soil (see I.5.).

Do not collect fruit bodies which are too young or too old. Contamination of these fructifications is lower than that of fungi which have just matured (see I.7.2.1).

Weigh the fresh fruit bodies and then dry them as soon as possible, in order to avoid any weight loss due to evaporation and loss of radionuclides due to loss of cytoplasm from dead cells, and then **reweigh the fruit bodies.** If one wants to know both the contamination level of the fresh matter and that of the dry matter, the two weighing operations need to be carried out, regardless of whether one is measuring the contamination in fresh or dry matter. Fungi contain a lot of water and contamination of the dry matter is normally much higher than that of the fresh matter (10 to 12 times on average).

Incidently, it should be noted that the expressions Bq/kg of fresh weight (FW) and Bq/kg of dry weight (DW), which are often found in the literature, do not make much sense; we should really speak of Bq/kg of fresh matter (FM) or Bq/kg of dry matter (DM).

In order to make data more comparable, we believe it is preferable to **express contamination of fungi in Bq/kg DM** because the water content can vary a lot from one species to another, depending on the relative proportion of cell walls in the total weight of the fruit bodies. The water content can also vary from one fruit body to another in the same species, even within the same fruit body (over time), depending upon climatic conditions.

V. PREVENTIVE MEASURES AIMED AT REDUCING DOSES INGESTED BY THE PUBLIC

V.1. Harvest in regions where fallout was lower

The distribution of fallout sometimes varies greatly nationally speaking. In the case of fallout limited in time (Chernobyl, for example), it is possible to obtain a rapid idea of how fallout is distributed by looking at the pattern of rainfall, which constitutes the main contamination factor (except, perhaps, for immediate closeness to the contamination source). This makes it possible to provide the public with advice tailored to each region.

V.2. Harvest on types of soil in which the biological availability of caesium is low

As we have already seen (I.5), clayey-limy soils are very efficient at fixing caesium and thus rendering the radioactive isotopes of this element less available for plants. By contrast, on acid soils radiocaesium isotopes are much more available to the same plants. If possible, any harvesting should preferentially be done on limy soils and, as far as possible, care should be taken to avoid harvesting on less favourable types of soil, such as peaty soils and sandy podzols.

Furthermore, it has been noted on several occasions that wild plants are more contaminated than plants of the same species which have grown on arable land in the same region. This difference is due to the biological availability of caesium being much lower in arable soils.

It has also been discovered that wild plants are less contaminated when they grow outside forest, in non-cultivated open areas. This could be due to the efficiency of the forest canopy as a collector of dry deposition, or to soil differences.

V.3. Harvest species which do not accumulate contamination

This advice holds particularly good for edible fungi, which include species with a clear tendency to accumulate radiocaesium (*Xerocomus badius*, etc.; see list under IV.2). Furthermore, as a general rule, fungi growing on wood (*Armillaria mellea*, *Kuehneromyces mutabilis*, etc.) and in meadows (*Agaricus arvensis*, *Coprinus comatus*, etc.) have very low contamination, which is also true for most of the Ascomycetes (*Morchella*, *Peziza*, etc.).

V.4. Washing

Washing with clean water can help to reduce significantly contamination in plants and fungi which received direct fallout. Some of the examples given below refer to commercial crops, but the principle is the same for wild plants.

Rantavaara (1987) observed that washing with water a sample of lingonberry (*Vaccinium vitis-idaea*) collected in Finland in 1986 reduced its Cs contamination by 20%. This reduction was 10% for Cs and 30% for Ru in a sample of blackcurrant (*Ribes nigrum*).

Schelenz & Abdel-Rassoul (1986) noted that washing bunches of redcurrants (*Ribes rubrum*) collected in Austria in 1986 reduced ^{134}Cs + ^{137}Cs as well as ^{103}Ru contamination by 20%. After stripping and washing, this reduction was 30% for the radiocaesium isotopes and 75% for ^{103}Ru. It should be noted that Cs activity in the juice of these redcurrants was only 20% of that of the unwashed whole fruit, while that of Ru was practically zero. The same authors also point out that cleaning and washing greatly reduced the radioactivity in meadow fungi.

Kühn et al. (1986) report the result of analysing spinach collected near Hannover (Germany) on 5.5.1986. An unwashed sample contained 384 Bq ^{137}Cs/kg FM and 1 210 Bq ^{131}I/kg FM. A washed sample contained, respectively, 164 and 717 Bq/kg FM of the above-mentioned isotopes.

V.5. Peeling

For items having received direct contamination, peeling is advisable (whenever possible) because it helps to reduce the contamination level substantially.

It is sometimes also useful in cases of indirect contamination, especially when eating bay boletus (*Xerocomus badius*). It should be recalled that Aumann et al. (1989) observed that ^{137}Cs was complexed by the cuticle pigments in this and a number of related species, and was therefore much more concentrated in the cuticle than in the rest of the fungus tissues.

V.6. Boiling in water

Several terms are used in the literature to describe this treatment : blanching, parboiling and boiling. Authors seldom give a detailed description of the methods employed, making it difficult to know exactly what these terms mean and, in particular, how long the process lasted, whether they involve water on its own or salted water, whether the fungi (or vegetables) were chopped up or not, etc. Whatever the case, such treatment can contribute to reducing very significantly the contamination levels of the items concerned.

Kühn et al. (1986) analysed two samples of spinach collected on 5.5.1986 near Hannover, one sample remaining untreated while the other was blanched. They contained respectively 384 and 181 Bq ^{137}Cs/kg FM, as well as 1 210 and 514 Bq ^{131}I/kg FM.

Rantavaara (1987) observed that, after being boiled in water, nettles (*Urtica dioica*) collected in Finland in 1986 contained only 86% of the Ba, 76% of the Ru, 73% of the Te, 48% of the Cs and 21% of the I they contained prior to the treatment.

The same author applied the same treatment to two sets of fungus (*Gyromitra esculenta*), whereas a third set of the same species was given the same treatment twice. The amounts still present in these groups with respect to the initial activity were, respectively, 100% for Zr (just one sample, blanched once), 100, 90 and 90% for Ru, 77, 65 and 65% for Ba, 42, 35 and 35% for Te, 28, 30 and 20% for Cs and 18, 25 and 19% for I.

Rantavaara (1987) also tested the impact of parboiling on fungus contamination. A sample of *Lactarius torminosus* treated in this manner contained only 12% of its initial content in Cs. She noted that carrying out this treatment twice reduced the Cs content to 2.6 and 1.8% of the initial value in *L. torminosus* (2 samples), to 2.5% in *L. trivialis* and to 3.2% in *L. rufus*.

Finally, after subjecting *L. torminosus* to parboiling, salting and soaking, the residual Cs contamination dropped to 0.07% of its initial value. This is an impressive result, but it needs to be checked whether the nutritional value of the fungi thus treated did not decrease accordingly !

The experiments carried out by this author indicate that the types of treatment referred to here (washing, boiling, blanching) do not have the same impact on all the chemical elements. One can draw up the following classification based on the ease with which these elements are extracted by such treatment : I > Cs > Te > Ba > Ru > Zr. Rantavaara indicates that the latter three are linked to compounds of very low solubility and that their isotopes were found in the particulate fraction in air analysed in spring 1986. The results obtained by Schelenz & Abdel-Rassoul (1986) - which we mentioned in the paragraph on washing - indicate, however, that this has an important effect on ^{103}Ru.

Other authors have studied the effect of such treatment. Klán et al. (1988) observed that 80% of the ^{137}Cs in *Xerocomus badius* and *X. chrysenteron* transferred into the extract after 5 minutes in boiling water, and 87% after 20 minutes.

Grüter (1967) reports that a sample of 130 g of *Paxillus involutus* - diced into 1-cm cubes after discarding the gills and the cuticle - lost 64% of its ^{137}Cs activity to the cooking liquid after being boiled in half a litre of water for 30 minutes.

Rohleder (1967, fide Grüter 1971) estimates that fungi lose at least 50% of their ^{137}Cs when boiled in salted water.

V.7. Steeping in salted water

We have already reported the results obtained by Rantavaara (1987) and Rohleder (1967) in using salted water to treat fungi.

Kowalska (in litt.) sent us the following results of an analysis of fungi collected in Poland in 1988 (the figures correspond to the concentrations in ^{134}Cs + ^{137}Cs, expressed in Bq/kg FM; the middle figure is the average, the two others are the minima and maxima).

Species	wet	salt-cured
Boletus erythropus	490 - 1 470 - 4 990	38 - 116 - 230
Boletus edulis	60 - 120 - 215	14 - 16 - 17
Cantharellus cibarius	16 - 60 - 142	5 - 13 - 29

An interesting experiment was carried out by Wahl & Kallee (1986) on game meat. The main idea was to substitute another alkali metal for Cs, viz. sodium (Na), which would be present to a large excess (in the order of 10^{12} times the Cs concentration). In order to do this a leg from a roe buck (*Capreolus capreolus*) killed in Germany in 1986 was placed in a salt solution containing 100 g/l of salt (NaCl) and 3 g/l of KNO_3. After 3 weeks the radioactivity in the meat had dropped from 570 to 70 Bq ^{137}Cs/kg FM and the meat was still very tasty. Similar results were obtained for livers, kidneys and hearts of roe deer.

The same authors report that subsequent experiments showed potassium was not essential for the process. A kilo of roe deer meat was cut up into approximately 50 pieces, to which were added 100 g of crystalline salt and 0.5 l of solution containing 100 g of NaCl/l. Over a period of 5 days the meat was taken from the solution each day and washed with 100 ml of water. The ^{137}Cs activity was then measured in the meat and in the salt solution, and then the meat was re-immersed in one litre of a 1M solution of NaCl (approximately 59 g/l). The radioactivity in the meat dropped to less than 40% of the initial level after 2 days of treatment, and to below 5% after 5 days.

Petäjä et al. (1992) has just published a very interesting study in which they look at the influence of various parameters on the effectiveness of soaking with a view to extracting radiocaesium from meat. They use pieces of reindeer meat weighting 200 g. Water alone is effective (42.2% of the Cs remaining after 2 days in soak), but brine (water + NaCl) is even more effective (38.7% and 35.9% of Cs remaining after 2 days in brine solutions containing 5% and 10% salt respectively). The effectiveness of the treatment also increases with duration (after 7 days in a 10% brine solution only 21.8% of the Cs remained), especially if the brine solution is regularly changed (after 6 days in 10% brine solution only 7.1% of the initial Cs content remained if the brine solution is changed every 2 days).

Temperature also influences the effectiveness of the treatment (36% or 22% of the initial Cs content remains, depending on whether the treatment is carried out at 7-8 °C or at 42 °C), but cooking in brine is less effective than soaking. It is also preferable to use fairly small pieces of meat (a 3 kg piece of meat takes much longer to decontaminate than 200 g pieces).

Finally, the authors note that the taste of the meat is hardly altered by soaking and that its salt content remains acceptable (2.9% to 3.8%) if a 5% brine solution is used. However, its nutritional value may be lowered to a certain extent due to the loss of some of the soluble vitamins and proteins.

To sum up, the authors of this study recommend that, in the event of contamination, radioactive caesium be removed from meat by soaking meat pieces not larger than 200 g for three successive periods of 2 days in 5% brine. Another recommended treatment is the soaking of coarsely ground meat for 17 h in slowly running water (6 l/min). By these methods the caesium content can be reduced by 90% or more of the original.

V.8. Cooking in an oven

Halford (1987) studied the effect of cooking contaminated mallards (*Anas platyrhynchos*) in an oven for one hour at 210-235°C. After cooking, the ^{134}Cs and ^{137}Cs levels had both dropped by 27% in the carcass (= body without skin or viscera), by 10% in the liver and by 0% in the gizzard. The ^{60}Co level had dropped by 7% in the carcass, 49% in the liver and 0% in the gizzard. The ^{110m}Ag level had dropped by 44% in the liver and was not detected in the carcass and the gizzard. Finally, cooking had no effect on the concentrations of ^{131}I, ^{65}Zn, ^{75}Se, ^{203}Hg and ^{51}Cr. The author supposes that this reduction in the concentration of certain nuclides is due to their being in the juices and fats exuded during cooking.

V.9. Preservation

Foods can be preserved in various ways : drying, freezing, preserving after sterilising, preserving in salted water, etc. With the exception of this latter method (see V.7), preservation does not contribute to extracting radioactivity from contaminated foods. However, the radioactivity level does decrease during preservation, due to the process of radioactive decay.

This process, which has a negligible impact in the case of ^{137}Cs, can be very important for radioisotopes with a very short half-life. For example, the activity of ^{131}I would be reduced to less than 10% of its initial level after one month of preservation, to below 1% after 2 months and to below 0.1% after 3 months. Because of this rapid decay rate this method is only of any use during the few months following contamination.

V.10. Summing-up and conclusions

In addition to observing certain rules when harvesting items of food (avoiding contaminated regions, gathering on limy or cultivated soils, avoiding accumulator species), certain processes can be used to reduce the dose of radioactivity ingested during consumption. The principles mentioned below apply in particular to "Chernobyl" types of contamination, i.e. that consisting mainly of relatively soluble isotopes (radiocaesiums and radioiodines, for example).

If harvested food items received direct contamination, they should be amply rinsed in clean water and, if possible, peeled. Vegetables and fungi should be sliced and blanched, i.e. boiled in salted water for 5 minutes, and then amply rinsed in clean water. The cooking and rinsing liquids should be discarded.

Extraction of Cs from meat can also be achieved by cutting the meat into pieces and immersing these in a salt solution (60 to 100 g of salt/l) for several days.

The juice and fat exuded when meat is cooked in an oven contain a non-negligible fraction of the contamination and should be thrown away.

Finally, in the case of very recent contamination involving isotopes with a very short half-life (^{131}I, for example), preserving the food for 2 or 3 months prior to consumption is an extremely effective decontamination method.

VI. CHANGES IN CONTAMINATION OVER TIME

VI.1. Preliminary remarks

The contamination level of a plant or animal living in an environment contaminated by radioisotopes depends on the aggregate effect of several concomitant phenomena. Some of these increase contamination : (1) absorption of radionuclides, mainly via food, but also via respiration, even directly via the epidermis; (2) adsorption of radionuclides on the surface of the organism.

In contrast, other processes help reduce contamination : (1) removal of internal contamination, mainly via various types of excretion, but also via respiration; (2) removal of adsorbed contamination, via resuspension, rain "washoff", and the shedding of leaves by plants or of coat (seasonal moulting) by animals; (3) radioactive decay.

When a healthy organism suddenly finds itself in a contaminated environment the interplay of the phenomena mentioned above leads first of all to a rapid phase of increasing contamination, then to a slower, asymptotic, phase of contamination decrease.

The speed at which these different phenomena occur depends on many factors (see I.6 and III.3 above). The major factor in animals is certainly the metabolic rate. Below we will look at the most important processes involved in changes in contamination in a forest ecosystem over time.

VI.2. Elimination of direct deposition adsorbed on plants

During fallout large amounts of radionuclides were adsorbed by the surfaces of plants. Because of this, in the months following the accident the contamination of samples from various regions corresponded closely to fallout distribution. Linkage with deposition was also shown by the $^{134}Cs/^{137}Cs$ concentration ratio in the samples collected, this being very close to that of the fallout deposited. This strong correlation was observed not only in plants but also in animals, because these feed directly (herbivores) or indirectly (carnivores) on plants.

However, immediately after deposition the amounts adsorbed started to decrease under the impact of several phenomena :

a. Resuspension. A certain proportion of the adsorbed radionuclides becomes detached and is carried away by the wind (Kovar 1990, Moorby & Squire 1963). This proportion may be large during certain forest management operations (felling, etc.). It is very high in the event of fires in contaminated forests. Resuspension does not necessarily lead to a corresponding decrease in the radioactivity of a forest stand, because it can be accompanied by redeposition of resuspended radionuclides from neighbouring stands.

b. Washoff by rain. As we explained in I.4, rainwater progressively washes off the deposit attached to the leaves and branches of trees and carries it to the ground.

c. Transfer to ground via falling leaves/needles/dead wood, etc.

These various processes greatly reduce the radioactivity attached to the surfaces of plants. These account for the large decrease observed in the activity of many samples during the year following the Chernobyl accident. This decrease does not, however, lead to full elimination of the adsorbed radioactivity, and - even a year after the fallout - the plants or the parts of plants which received direct fallout are still more contaminated than the plants or parts of plants which developed after deposition (Nylen & Ericsson 1989).

VI.3. Percolation of contamination through soil

As we stated in I.5, contamination reaching the ground migrates downwards via the litter and the mineral soil. This migration is slow in organic matter and even slower in mineral soil, especially if it contains a lot of clay. This movement of radionuclides in the forest soil takes place in two stages, which have different consequences.

Initially, the radioisotopes move from the non-decomposed litter to the hemiorganic horizons. Here they become much more available to plants, which - apart from certain species of fungus - do not feed off the litter. This no doubt explains why the contamination levels of various plants remained stable or even grew in 1987 and 1988. This phenomenon is particularly frequent in fungi (Fourré 1989, Horyna et al. 1989, Molzahn et al. 1990, Řanda et al. 1989) because these organisms, not having received any direct contamination, did not benefit from the "decontamination by washoff" which we referred to in V.1 above.

Later, the radionuclides migrate slowly towards the mineral horizons where they are progressively adsorbed onto the clay minerals and thus become less available for plants. This reduction in availability varies according to the type of soil. It is much more rapid in soils containing little organic matter (Cremers et al. 1989).

Furthermore, some of the radioactivity in the soils is taken up by the plants and fungi and then redeposited on the surface during the shedding of leaves and the death of herbaceous plants and fungi. This constant turnover of the Cs offsets (at least in part) migration of this element towards the deeper layers of the soil. Furthermore, by this time such migration has become very slow (of the order of 1 to 3 mm per year?), so much so that a state of near-equilibrium has set in.

A number of factors are likely to disrupt this process. The worst disruption is that involving soils subject to erosion - the contamination is carried away with the layers of soil containing it. This situation is very rare in forests, however. Less obtrusive or more localised disruption results from the activity of various mammals which burrow in the soil to make their homes or in search of food. Finally, when present in large numbers, earthworms can have some impact on the distribution of radionuclides in soil.

VI.4. Radioactive decay

This is the only decontamination process not involving removal of radionuclides. Its effectiveness depends on the length of the half-life of the isotopes involved. Although very rapid for ^{131}I (half-life = 8 days), it is much slower for ^{134}Cs (half-life = 2 years), and even slower for ^{90}Sr (half-life = 29 years) and ^{137}Cs (half-life = 30 years). It has no impact whatsoever in the case of radionuclides with a very long half-life, such as ^{239}Pu (half-life = 24 000 years), which - luckily - was practically absent from Chernobyl fallout.

This is the main decontamination factor for isotopes with a short half-life. For the others (^{90}Sr and ^{137}Cs, for example) it is only important once the state of near-equilibrium just mentioned is attained.

VI.5. Medium- and long-term prospects

Following a period in which the situation gradually balances out (2 to 5 years after the fallout?), migration of radionuclides in the soil becomes very slow and the activities involved in turnover stabilise. Due to the effectiveness of fixation by clay minerals and, to a lesser degree, by humus, only a portion of the radiocaesium will be available for plants. This fraction will vary with the type of soil (see I.5). Developments will probably be very slow and mainly governed by radioactive decay.

The authors are agreed on the fact that even years after fallout deposition the bulk of the ^{137}Cs remains confined to the litter, humus and the top few centimetres of the mineral soil (see in particular Molzahn et al. 1990, Ritchie et al. 1972 and Rogowski & Tamura 1970). Schell & Tobin (1989) note that after 30 years the ^{137}Cs from atmospheric nuclear testing had migrated no more than some 16 cm in an ombrogenous peat bog. Squire & Middleton (1966) observed migration of ^{137}Cs in pastureland of 1 to 3 cm in 6 years.

Belli et al. (1989) calculated the mean residence half-time for ^{137}Cs in the top layer (0-5 cm) of various soils in north-eastern Italy. These were 5 (±1) years in natural grassland on old alluvia, 9 (±7) years and 10 (±3) years in two beech woods growing on limy soil, and 20 (±8) years in a mixed stand of pines (Abies) and beech (Fagus) on soil derived from glacial alluvia.

Such developments will no doubt be even slower in northern ecosystems. Bergman & Johansson (1989) note that, prior to the Chernobyl accident, no appreciable change was observed in 20 years in the northern forest ecosystem contaminated by fallout. Furthermore, it is also in the higher latitudes that the residence times of radionuclides in biota are at their highest. An effective half-life of 10-12 years was measured for ^{137}Cs in lichens living in xeric habitats at a latitude of 80° north (Taylor et al. 1985). Hanson (1967) puts at 13 years the half-life of ^{137}Cs in lichens in Alaska.

Finally, a mathematical model drawn up by Van Voris et al. (1989) predicts that forest ecosystems will progressively accumulate radiocaesium during 40 to 50 years, after which regression should occur due to radioactive decay.

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