IAEA-TECDOC-647

Modelling of resuspension, seasonality and losses during food processing

First report of the VAMP Terrestrial Working Group

Part of the IAEA/CEC Co-ordinated Research Programme on the Validation of Environmental Model Predictions (VAMP)

INTERNATIONAL ATOMIC ENERGY AGENCY IAEA

MODELLING OF RESUSPENSION, SEASONALITY
AND LOSSES DURING FOOD PROCESSING:
FIRST REPORT OF THE VAMP TERRESTRIAL WORKING GROUP

IAEA VIENNA, 1992
IAEA-TECDOC-647
ISSN 1011–4289

Printed by the IAEA in Austria
May 1992

FOREWORD

Following the Chernobyl accident and on the recommendation of the International Nuclear Safety Advisory Group (INSAG) in its Summary Report on the Post-Accident Review Meeting on the Chernobyl Accident (Safety Series No. 75-INSAG-1, IAEA, Vienna, 1986), the Agency established a Co-ordinated Research Programme on "The Validation of Models for the Transfer of Radionuclides in Terrestrial, Urban and Aquatic Environments and the Acquisition of Data for that Purpose". The programme seeks to use the information on the environmental behaviour of radionuclides which became available as a result of the measurement programmes instituted in the Soviet Union and in many European countries after April 1986 for the purpose of testing the reliability of assessment models. Such models find application in assessing the radiological impact of all parts of the nuclear fuel cycle. They are used at the planning and design stage to predict the radiological impact of planned nuclear facilities, in assessing the possible consequences of accidents involving releases of radioactive material to the environment and in establishing criteria for the implementation of countermeasures. In the operational phase they are used together with the results of environmental monitoring to demonstrate compliance with regulatory requirements regarding release limitation.

The programme, which has the short title "Validation of Environmental Model Predictions" (VAMP), was started in 1988; it is jointly sponsored by the Division of Nuclear Fuel Cycle and Waste Management and the Division of Nuclear Safety and is also supported by the Commission of the European Communities. There are four working groups within the programme; they are the Terrestrial Working Group, the Urban Working Group, the Aquatic Working Group and the Multiple Pathways Working Group. This is the first report of the Terrestrial Working Group.

PLEASE BE AWARE THAT
ALL OF THE MISSING PAGES IN THIS DOCUMENT
WERE ORIGINALLY BLANK

EDITORIAL NOTE

In preparing this material for the press, staff of the International Atomic Energy Agency have mounted and paginated the original manuscripts and given some attention to presentation.

The views expressed do not necessarily reflect those of the governments of the Member States or organizations under whose auspices the manuscripts were produced.

The use in this book of particular designations of countries or territories does not imply any judgement by the publisher, the IAEA, as to the legal status of such countries or territories, of their authorities and institutions or of the delimitation of their boundaries.

The mention of specific companies or of their products or brand names does not imply any endorsement or recommendation on the part of the IAEA.

CONTENTS

Chapter 1

INTRODUCTION

The Terrestrial Working Group of VAMP aims to examine, by means of expert review, the state-of-the-art in modelling the transfer of radionuclides from the terrestrial environment to man. An aim of the studies is to evaluate the lessons learned and to document the improvements in modelling capability as a result of post-Chernobyl experience. The topics for review were decided upon following the first Working Group Meeting. In making this choice, consideration was given to the relative importance of a given process as a potential contributor to radiation dose to man, the degree of uncertainty which exists in the area because of lack of knowledge and the possibilities presented by Chernobyl for improvement of understanding of the process.

Reviews of three of the identified topics are presented in this report; they are:

Resuspension - the process in which surface deposits of particulate matter are re-entrained into the air by mechanisms such as wind disturbance and man-made activities.

It has been demonstrated that in dry environments, particles can be resuspended for many years after their initial deposition. In the case of radionuclides this could represent a long term pathway of radiation exposure, especially for those long lived radionuclides which present most hazard via the inhalation route. The validity of the resuspension models developed on the basis of the evidence from dry environments in other more vegetated and damper environments had never been adequately tested. The fall-out from the Chernobyl accident affected environments which are mostly in temperate regions and an opportunity for testing resuspension models was presented.

Food processing losses - the removal of radionuclides from the food-chain to man as a result of food processing and culinary preparation.

The significance of the reductions in radionuclide contamination of foodstuffs which result from food processing and culinary preparation has

been recognised for many years. However, the information available on the reduction factors achieved by the various processes and as a function of radionuclide was scarce and poorly documented. The need for realistic dose assessments after the Chernobyl release made modelling deficiencies of this type apparent and stimulated new work in the area.

Seasonality - the varying response to radioactive contamination of food crops and other environmental systems to the time of year when the contaminating event occurs.

The variation of the contamination of environmental materials with season has been most clearly demonstrated in grain crops. Early experiments showed that depending upon the stage of growth of a crop when contaminated, the level of radionuclide concentration in the final grain product can vary by more than a factor of 100. Because of the season of the year when the Chernobyl accident occurred and because the fall-out affected countries in different geographic latitudes, a significant variation of the apparent degree of radionuclide transfer to the same crop type with latitude was seen. This was due to seasonality effects. Pre-existing food-chain models did not represent the seasonality effect very well, if at all. The Chernobyl experience has emphasised the need for it to be taken into account in predictive models.

The following reviews were prepared by the named authors but have had the benefit of a review process which involved the members of the Terrestrial Working Group of VAMP. The versions presented in this document have been revised to take into account the comments received from the Working Group. Publication was finally decided upon at a meeting of the full VAMP Research Co-ordination Meeting in March 1991, and after the approval of VAMP members (by correspondence) in Autumn 1991.

Chapter 2

RESUSPENSION FOLLOWING CHERNOBYL

J.A. Garland, N.J. Pattenden, K. Playford
AEA Environment and Energy,
Harwell Laboratory,
Didcot, Oxfordshire,
United Kingdom

2.1. Introduction

Resuspension comprises the processes which may raise deposited material from the ground. It is thus related to, but in principle distinct from, suspension, where no previous airborne condition is implied, and entrainment, which is suspension caused by the wind. However, the resuspension mechanisms usually include the entrainment of particulate material in winds passing over surfaces, and this can occur over any natural or artificial surface from which particles or droplets may be detached by the wind or other disturbance. Therefore, the distinction between resuspension and suspension is often ignored. The material initially deposited may be changed in its physico-chemical characteristics during its contact with the surface. For example, it might become attached to other particles. In conditions of continuous emission from the primary source, it can be difficult to make quantitative measurements of resuspended material, because of its similarity to its source material before deposition and the relative sizes of the concentrations of the material before deposition and after resuspension.

Resuspension has the potential to cause persistent air concentrations following a brief release, and to redistribute material deposited on the ground. Inhalation hazards, recontamination of previously cleaned surfaces and contamination of crops are possible consequences. Plans for coping with the aftermath of accidents that result in spills or airborne releases, and consequence assessment models, must make allowances for resuspension processes.

Despite many measurements, the prediction of resuspension remains uncertain. Many factors influence resuspension, including various facets of the weather, the surface structure and the nature of the contaminant

TABLE I

SOME FACTORS THAT MAY INFLUENCE RESUSPENSION

Time since deposition
Wind speed
Nature of surface: vegetation, building etc.
Surface moisture
Soil chemistry and texture
Size distribution of contaminant particles
Chemical properties of contaminant
The deposition process: wet or dry, wind speed, precipitation amount etc.
Mechanical disturbance by traffic, agriculture etc.
Depth and method of cultivation
Intensity and frequency of rain
Snow cover or freezing of the surface

(see Table I). Consequently, field measurements show a wide range of behaviour and it is often difficult to associate changes with specific causes.

Data available from observations at American nuclear sites and weapon test sites have been summarised by Sehmel [1] and the wide range of information available on the topic has been described by Sehmel [2] and Nicholson [3]. Generally, arid or semi-desert areas predominate in the results, and a strong dependence on wind speed is apparent. Additional studies have used wind tunnels or controlled applications of tracers to outdoor surfaces to investigate individual resuspension mechanisms (e.g. Garland, [4, 5]. Resuspension by traffic, agriculture and some other types of mechanical disturbance has been demonstrated, as summarised by Nicholson [3]. Experiments designed to investigate such factors have also demonstrated an important reduction in resuspension with time after deposition. Several models have been proposed to describe this effect (see Ref. [6]).

Results from wind tunnel experiments, mechanistic field investigations, and even monitoring results from contaminated areas within nuclear establishments or weapon test sites, may be inappropriate for application in some areas. They may represent too great or too small an area of contaminated surface or the climate, surface roughness, or the density of traffic or other source of disturbance may differ. In principle the effects of these parameters have been investigated and a correction could be attempted for recognised differences, but knowledge is incomplete and uncertainties would be great. This is particularly true for the humid climate of Europe where there have been few investigations of resuspension.

The Chernobyl accident of 26 April 1986 provided a unique opportunity to observe resuspension from a brief event which caused widespread contamination in Europe. This paper attempts to draw together data from several laboratories that have reported measurements following Chernobyl, and to compare the results with expectation based on previous experience.

2.2. Definitions

Three parameters have been used to describe resuspension [3]. The first is the resuspension factor:

$$K\ (m^{-1}) = \frac{\text{Concentration in air, } C_a\ (Bq\ m^{-3})}{\text{Surface deposit, } d\ (Bq\ m^{-2})} \qquad (1)$$

which has the advantage that both numerator and denominator are directly measurable. However, measurement of K may be complicated since the concentration in air may include important contributions from processes other than resuspension. In addition K is likely to be strongly dependent on such details of measurement technique as the height of air sampling above the surface. In evaluating d a decision must be made regarding the depth of surface to be included, and it is not straightforward to decide what depth of soil may contribute to airborne radioactivity. In practice, for prediction of resuspension from an accidental dispersion, it is convenient to use the total amount initially deposited, without allowance for vertical mixing into the soil or for removal by weathering.

K has the drawback that it must depend on the variation of d with distance from the point of measurement. At locations with relatively low

deposit, transport by the wind from areas of high deposit may dominate air concentrations. Although there has been some theoretical investigation relevant to this problem, [7, 8], an appropriate averaging scale for d has not been established.

In principle, the difficulty of spatial variations in deposit is circumvented by the use of the resuspension rate:

$$\Lambda\ (s^{-1}) = \frac{\text{Resuspension flux, R (Bq m}^{-2}\ s^{-1})}{\text{d (Bq m}^{-2})} \qquad (2)$$

Thus, Λ is the fraction removed per second by resuspension. This quantity can often be derived directly in wind tunnel measurements and laboratory studies. However, it can only be readily deduced from field observations in exceptional circumstances. In principle, Λ may be included in atmospheric transport models to describe the movement of contamination from one area to another by resuspension and deposition, as well as to predict air concentrations (eg. Ref. [8]). Such models, with a knowledge of Λ, appear essential for predicting air concentrations at locations downwind of a contaminated area.

The use of the equivalent soil concentration

$$S_E\ (kg\ m^{-3}) = \frac{C_a\ (Bq\ m^{-3})}{\text{Concentration in surface soil, } C_s\ (Bq\ kg^{-1})} \qquad (3)$$

avoids the need to decide what depth of soil to include in the calculation of K. It allows surrogate soil tracers to be used to estimate air concentrations from a radioactive contaminant in soil, but it has not been much used.

The definitions of K, S_E and Λ make no allowance for the effects of the many environmental variables that may influence resuspension rates. Large variations must therefore be expected with wind speed, time after deposition and with the other parameters listed in Table I.

2.3. Available Estimates of Resuspension

The resuspension factor, K, is the parameter that has been used most frequently to describe the results of previous studies, and for predicting the consequences of the contamination of surfaces. Field measurements

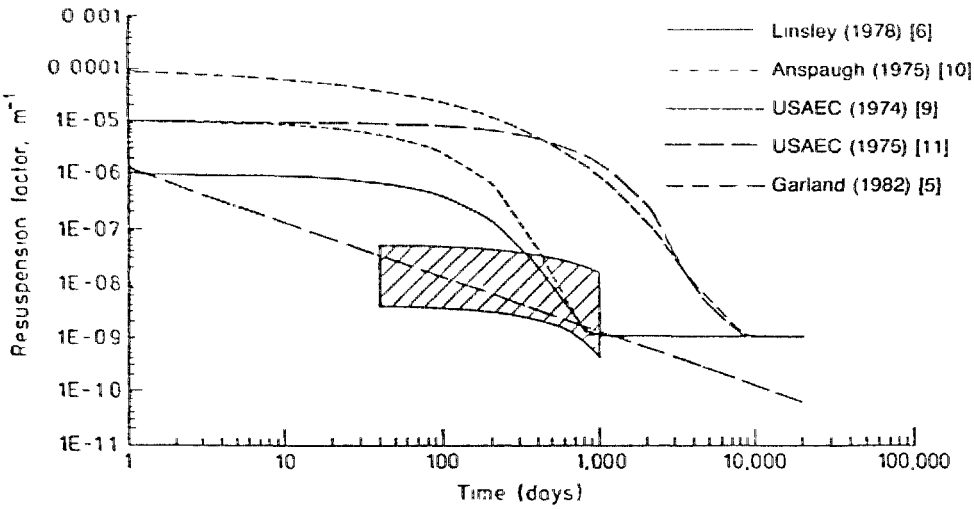


FIG 1 Resuspension factors (K) measured in Europe (hatched area) compared with models. The model relationships are: Linsley $K = 10^{-6} \exp(0.01\ t) + 10^{-9}$; Anspaugh $K = 10^{-4} \exp(-0.15\ t^{1/2}) + 10^{-9}$; USAEC (1974) $K = 10^{-5} \exp(-0.0139\ t) + 10^{-9}$; USAEC (1975) $K = 10^{-5} \exp(-0.00185\ t) + 10^{-9}$; Garland $K = 1.2 \times 10^{-6}/t$; where t is time after deposition in days and K is in m^{-1}.

(e.g. Ref. [1]) suggest that soon after deposition K is of the order of 10^{-6} to 10^{-4} m^{-1}, but over a period of years, K declines to approximately 10^{-9} m^{-1}. A number of empirical models represent this behaviour (see Fig. 1). One model [9] allows K to vary with particle size, ascribing a value to each of several size intervals and setting K inversely proportional to deposition velocity.

The last model listed in Fig. 1 is based on wind-tunnel studies [4, 5]. These studies were in agreement with field experience, in that K increased strongly with wind speed and decreased with time following deposition. K also increased with particle diameter in the range from about 1 to 5 μm. The decrease with time was represented by an inverse-time relationship.

Information on K long after deposition can be gleaned from measurements of nuclear weapon test debris. Air concentrations at ground level have always been subject to some input from the stratospheric reservoir, so that only upper limits on K can be deduced. Data for ^{137}Cs in air over the United Kingdom show that, 13 years after peak deposition, K was less than 2×10^{-9} m^{-1} and, after 22 years, less than 5×10^{-10} m^{-1}. Consideration of radionuclide ratios and trace element concentrations in air and soil has been used to refine limits

further, and Garland [4] and Cambray et al. [12] argue that K cannot have been greater than 7×10^{-11} or 2×10^{-10} m^{-1}, 15 or 18 years after deposition, respectively.

The effect of countermeasures on resuspension is indicated by observations at Palomares. In 1966 two nuclear weapons were destroyed by fire following an accident in the air. The bombs burned on impact with the ground causing plutonium contamination over an area of a few km^2. Subsequently, continuous measurements of plutonium in air were made at four sampling stations in the contaminated zone [13]. Using estimates of the initial level of deposition and the air concentration data, the mean values of K for the first year range from 0.2 to 2×10^{-9} m^{-1}. Soon after the accident top soil was removed from the worst affected areas, and in other areas surface concentrations were alleviated by deep ploughing. Presumably these actions account for the low values of K. During the following four years, air concentrations declined rather erratically at all stations about five-fold. At two stations where measurements continued, the air concentrations remained roughly constant at the reduced level for a further seven years, but rose again in 1977 when cultivation began on a previously unused, contaminated area. These data apply to rather arid conditions in southeast Spain.

2.4. Opportunities from Chernobyl

In the following, the observations after Chernobyl will be examined in terms of K. Generally, only long term mean concentrations are available, and the response of K to many of the parameters in Table I are hidden in values which are appropriate to the climate, rather than the short term weather.

The Chernobyl accident yielded a rare opportunity for the study of resuspension, since it provided what could be considered as a large single pulse of radionuclides of limited duration. This pulse was fairly rapidly dispersed over a large part of Europe. Deposition continued over a few weeks, and was essentially complete by the end of June. However, data given by Cambray et al. [14] shows that over 90 per cent occurred by mid May.

The main contaminated air mass arrived at locations experiencing considerably different climatic regimes, where, for example, in some cases

spring was well advanced and in others was in the early stages. Continental, maritime and mediterranean situations all received a signal from Chernobyl. This brings the possibility of observing the transfer of radionuclides through the environment, including the resuspension process, under a variety of conditions. The comparison of such observations may enable the relative significance of the various conditions to be established.

Data obtained following Chernobyl permit direct estimation of resuspension in many areas and extend the information available on resuspension to climates not previously represented. However, these observations have limitations. They give information on only one radioelement - caesium. The deposition was very variable in some areas, and even where numerous measurements of the amount deposited have been made their representativeness may be in doubt. There is some question as to whether local resuspension is the only source of material measured in the air: contributions due to transport by the wind of resuspended material from more heavily contaminated areas may be significant.

^{137}Cs and ^{134}Cs were the most widely distributed long lived radionuclides from Chernobyl. The ratio between these radionuclides varied within a narrow range about $^{137}Cs/^{134}Cs = 1.6$ [14] and this ratio has been used as evidence that the persistent radiocaesium in the atmosphere did derive from Chernobyl. However, the shorter half-life and smaller quantity of ^{134}Cs released in the accident make ^{134}Cs of minor interest in assessing resuspension.

Information on other radioelements would be of significant interest. Resuspension may be influenced by the physico-chemical behaviour of radionuclides in soil and other environmental compartments. For example, Garland [4, 5] observed an increase in resuspension with particle size. Many other radionuclides were observed in Europe soon after the accident. However, they were mostly short lived or were present in small quantities and gave little opportunity for observing resuspension.

Garger et al. [15] observed the resuspension rates of ^{144}Cs, ^{137}Cs and $^{95}Zr/Nb$ within the 30 km exclusion zone around Chernobyl. Some differences in resuspension rates were observed but there was not a consistent pattern. On some occasions the resuspension rate for ^{137}Cs exceeded that for ^{144}Ce and for $^{95}Zr/Nb$, while in other cases the reverse was true.

The limited data suggest that any differences between radionuclides are much smaller than the two or three orders of magnitude variability between measurements on different occasions. However, these results apply near the release point, where large fuel particles probably dominated deposition for the radionuclides measured, and a different behaviour might apply in other circumstances.

2.5. Data Following Chernobyl

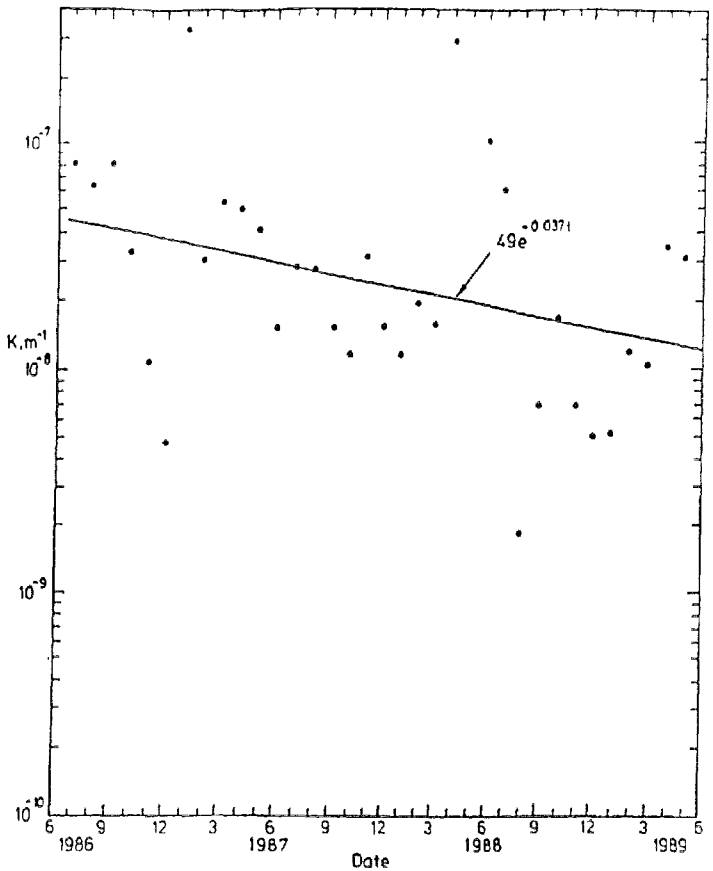
2.5.1. Time series resuspension data

Experimental results for monthly average ^{137}Cs concentrations in near-ground air have been obtained for 14 stations across Europe. Typically, the measurements used high volume air samplers, operating at sampling rates of 1 to 10 m^3 per minute and using high efficiency filters.

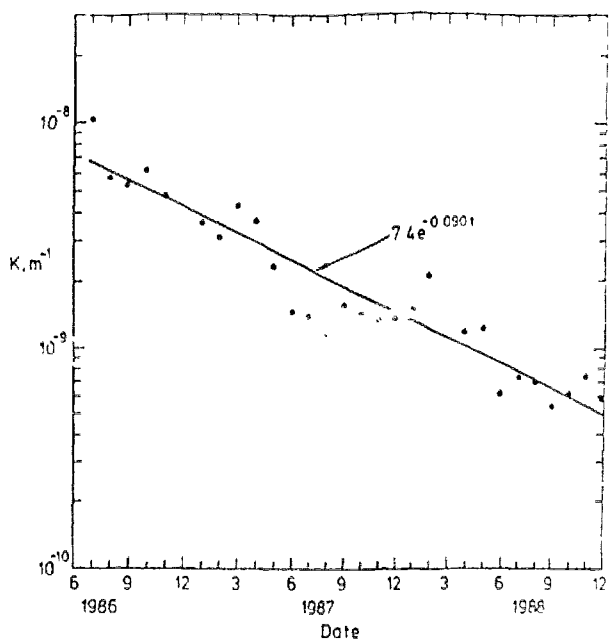
In all cases, measurements of the direct deposit from the Chernobyl accident were available, by either direct deposition (wet and dry) or soil sampling, where the Chernobyl contribution was estimated from the ^{134}Cs concentration. This has enabled the air concentrations to be converted to resuspension factors. In this review, the Chernobyl direct deposit has been assumed to be the deposit measured between the date of the accident and 30 June 1986. After this time the deposit has been assumed to be due to indirect mechanisms, such as resuspension. Thus, the resuspension factor at time T was calculated by dividing the measured air concentration at time T by the amount deposited soon after the accident, with no correction for gains due to deposition after 30 June 1986, or removal by resuspension or by dispersal or transport within the soil.

Examples of the variation of resuspension factors with time are shown in Fig. 2. Each data set has been fitted by an exponential function which is also shown in the figures. A summary of the parameters is given in Table II.

For the 14 sites, the initial deposit varied from 52 to 43700 Bq/m^2. In all cases the resuspension factor, K, decreased with time over the observation period, and the exponential function A exp(-Bt) gave a reasonable fit to the data. Values of the parameter A, corresponding to K in mid-June, 1986, were in the range (3.6 - 49) x 10^{-9} m^{-1}, and values of the parameter B, which specified the exponential decline in resuspension, were in the range 0.026 - 0.124 per month.



CHILTON · ^{137}Cs RESUSPENSION FACTOR (m^{-1})



NEUHERBERG · ^{137}Cs RESUSPENSION FACTOR (m^{-1})

Fig. 2. Examples of data for resuspension of ^{137}Cs from Chernobyl.

TABLE II

^{137}Cs RESUSPENSION FACTOR (K)

$$K(t)\ (m^{-1}) = \frac{\text{Air concentration } (t)\ (Bq\ m^{-3})}{\text{Surface deposit } (Bq\ m^{-2})}$$

Time dependence fitted by exponential function:

$K = A \exp(-Bt)$ (t = time in months after mid-June 1986)

Location	Initial Deposit ($Bq\ m^{-2}$)	Period of Observation	A ($m^{-1}X10^{-9}$)	B ($month^{-1}$)	Reference
Klagenfurt, Austria	43700	1/87-12/89	3.6	0.049	16,17,18,19
Neuherberg, Germany	19700	7/86-12/88	7.4	0.090	20,21
Bregenz, Austria	13200	1/87- 8/89	5.3	0.061	16,17,18,19
Ispra, Italy	11900	7/86- 6/88	5.9	0.026	22,23,24,25
Nurmijärvi, Finland	7100	7/86-12/89	10.0	0.063	26,27,28,29
Vienna, Austria	4000	1/87- 8/89	12.8	0.053	16,17,18,19
Warsaw, Poland	3200	7/86-10/89	15.0	0.062	30, 31
Braunschweig, Germany	2700	7/86-12/88	28	0.124	32, 33
Eskmeals, UK	2500	7/86- 6/89	16.0	0.082	34, 35
Berlin, Germany	2300	7/86-12/88	35	0.099	32, 33
Risø, Denmark	800	7/86-12/88	31	0.106	36, 37, 38
Bornholm, Denmark	620	7/86-12/88	31	0.082	36, 37, 38
Skibotn, Norway	195	7/86-12/88	35	0.060	32, 33
Chilton, UK	52	7/86- 6/89	49	0.037	39,40,41

There is a negative correlation between the initial deposit and A. The Spearman rank correlation coefficient is -0.96 which is significant at the >99.9% level, (see Fig. 3). However, no systematic variation of B with initial deposit is apparent.

The variation of K with deposit has been confirmed in a larger data set in Table III, which includes some stations for which the data were unsuitable for the time trend analysis of Table II. Table III also shows that the dependence of K on deposit has persisted over two years. Moreover, Bjurman et al. [42] observed resuspension at four sites in

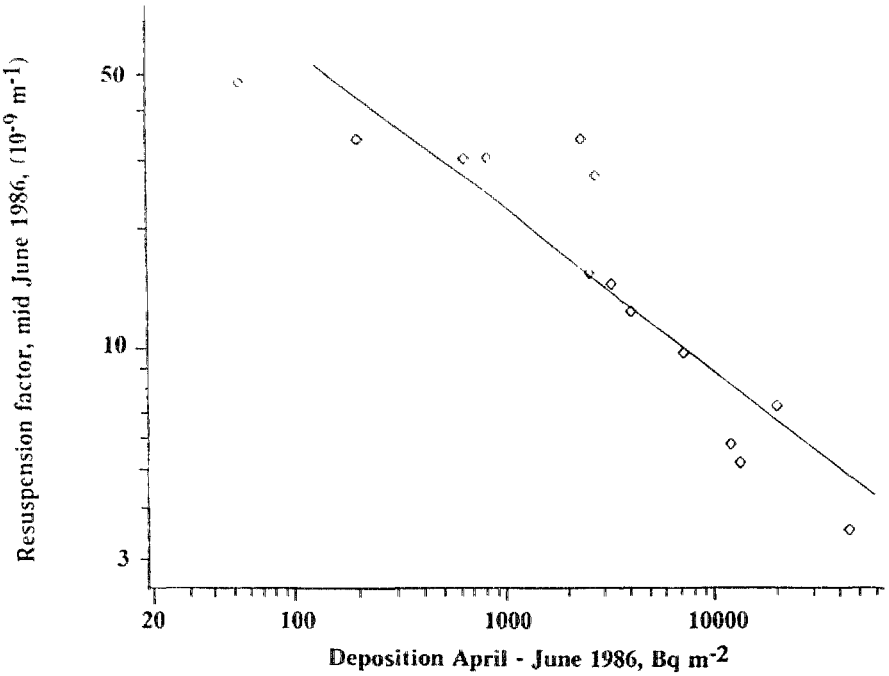


Fig. 3. Variation of the resuspension factor in mid-June 1986 with the amount deposited during April-June 1986.

Sweden. At Gavle, the deposit was 30 to 50 times larger than at the other Swedish sites, and K averaged over July to December 1986 was up to 10 times smaller.

The correlation between A and the initial deposit shows that, although the concentration of ^{137}Cs in air soon after deposition increases with the amount initially deposited, the rate of increase is significantly less than proportional to the deposit. Two possible explanations for this difference have been considered. Generally, different deposition mechanisms operated for large and small deposits, the large deposits being caused by wet deposition in heavy rain while the small deposits were due mainly to dry deposition. A real difference in resuspension factors, consistent with the observations, might be due to a greater availability for resuspension of material deposited dry than for wet deposited material. However, a 'memory' of a year or more of the circumstances of deposition would be required, and this seems improbable. The second explanation involves long range transport of resuspended material. This would increase the observed concentration in areas of low deposit, but reduce it in areas of high deposit, and produce an apparent variation in

TABLE III

^{137}Cs AIR CONCENTRATIONS AND MEAN RESUSPENSION FACTORS

Site	Initial Deposit (Bq m^{-2})	Mean air concentration (μBq m^{-3})		Resuspension factor (10^{-9} m^{-1})	
		7/86-6/87	7/87-6/88	7/86-6/87	7/87-6/88
Neuherberg, Germany	19700	96	27	4.9	1.4
Ispra, Italy	11900	69	54	5.8	4.5
Nurmijärvi, Finland	7100	68	23	9.6	3.2
Lerwick, UK	4600	2.7	1.4	0.6	0.3
Warsaw, Poland	3200	39	14.5	12	4.5
Braunschweig, Germany	2700	46	9.9	17	3.7
Eskmeals, UK	2500	37	7.8	15	3.1
Berlin, Germany	2300	57	19.6	25	8.5
Conlig, UK	2200	6.7	2.6	3.0	1.2
Eskdalemuir, UK	820	9.5	3.3	12	4.0
Risø, Denmark	800	18.5	3.6	23	4.5
Bornholm, Denmark	620	14.4	4.4	23	7.1
Skibotn, Norway	195	6.3*	3.3	32	17
Tromso, Norway	175	2.6	1.7	15	9.7
Orfordness, UK	110	10.3	4.0	94	36
Compton, UK	105	29	3.0	280	29
Chilton, UK	52	3.1	2.7	60	52
Harwell FC, UK	52	3.8	1.8	73	35
Harwell CM, UK	52	69	6.2	1300	120
Milford Haven, UK	18	1.5	0.7	80	40

* 11 months data

Data sources as for Table II and Garland and Cambray [43] for additional UK stations.

resuspension factor. Very simple modelling of the transport process suggests that such an explanation is plausible.

In the case of the Chilton data set, corresponding to one of the smallest deposits, there is some evidence, albeit inconclusive, of a spring peak in the air concentration. A spring peak is characteristically

observed in the effects of debris from past nuclear weapon tests, and is explained by the return of material from a stratospheric reservoir. Thus there is a suggestion that a small fraction of the Chernobyl release was injected into the stratosphere. (Aoyama, [44], reported similar observations in Japan.) The Chernobyl material returned from the stratosphere would be difficult to identify at locations which had received large deposits because of the relatively large contribution from local resuspension. Hence, there would be more chance of observing a stratospheric effect at Chilton than at most other sites. Stratospheric return would also reduce the differences in air concentration between sampling sites, and might contribute to the lack of proportionality between air concentration and deposit.

2.5.2. Traffic

Measurements of ^{137}Cs in air at Risø (Aarkrog, et al. [36]) show a pronounced weekly cycle during July and August 1986. Concentrations were high during weekdays and low at weekends. Later this pattern disappeared. The weekly periodicity was attributed to resuspension by traffic of Chernobyl deposit on nearby roads, which were busy on working days but quiet at weekends. After August, the Chernobyl deposit had presumably been removed from roads or fixed to the road surface and was no longer available for resuspension.

A comparison between two sites at Harwell Laboratory also suggested that traffic is important. One sampling site (Harwell CM, in Table III) is located in a car park and the second (Harwell FC), a kilometre away, in a field with no roadway within a few hundred metres. A 20-fold difference in air concentration was observed during the year July 1986 - June 1987, and the difference persisted, though somewhat diminished, in the following twelve month sampling period. Although the car park and the neighbouring roads are busy for only one or two hours each day, the highest post-Chernobyl resuspension factors were observed at this locality.

2.5.3. Scale length for resuspension

Large differences between neighbouring sites show that variations in air concentrations occur on a scale not exceeding some tens of kilometres. Indeed, where local variations in traffic occur, concentrations may vary substantially in a kilometre or so. Further indications of the length

scale for resuspension are found in the data for selected sites. Weekly measurements for ^{137}Cs in air at the island of Bornholm and Risø were compared (Aarkrog et al. [37]) with the frequency of wind speed and direction. At both sites, there was a positive correlation of air concentration with winds from the east, and negative correlations with westerly winds. Both sites are coastal with land to the east and water to the west, and the correlations support the view that the source of resuspension is dominantly nearby land areas, within a range of about 20 km (the distance across Bornholm). No correlations with wind speed were revealed.

Low resuspension factors were found at Lerwick (Table III) which is near the east coast of a strip of land only 5 km wide on the island of Shetland.

2.5.4. Comparison with models

The range of the exponential function fitted to the values of K is shown in Fig. 1, along with models previously used to provide a description of resuspension. The models show a wide divergence over the time interval occupied by the data. To some extent this is determined by the different circumstances and uses intended for the models. The Chernobyl data tend to support the models predicting lower rather than higher concentrations, but the limited time range of the Chernobyl data currently available must be kept in mind. In addition, most of the data sets reflect conditions where disturbance by man and vehicles is slight. People working out of doors are liable to generate dust locally and to be exposed to higher concentrations in consequence, and Linsley [6] suggested that an increase of an order of magnitude in predicted concentrations may be appropriate.

2.5.5. Deposition of resuspended ^{137}Cs

Garland and Cambray [43, 45] noted two unexpected features of deposition measurements following Chernobyl. Total apparent deposition velocities (i.e. the flux to a deposition collector funnel divided by the air concentration) ranged up to about 3 m s^{-1}, and the amount of ^{137}Cs collected during July 1986 to June 1987 at some sites was comparable to that deposited in May–June 1986. In comparison, Aarkrog [37] noted that at Risø and Bornholm the amount deposited in 1987 was just 2.1 per cent of the initial deposit.

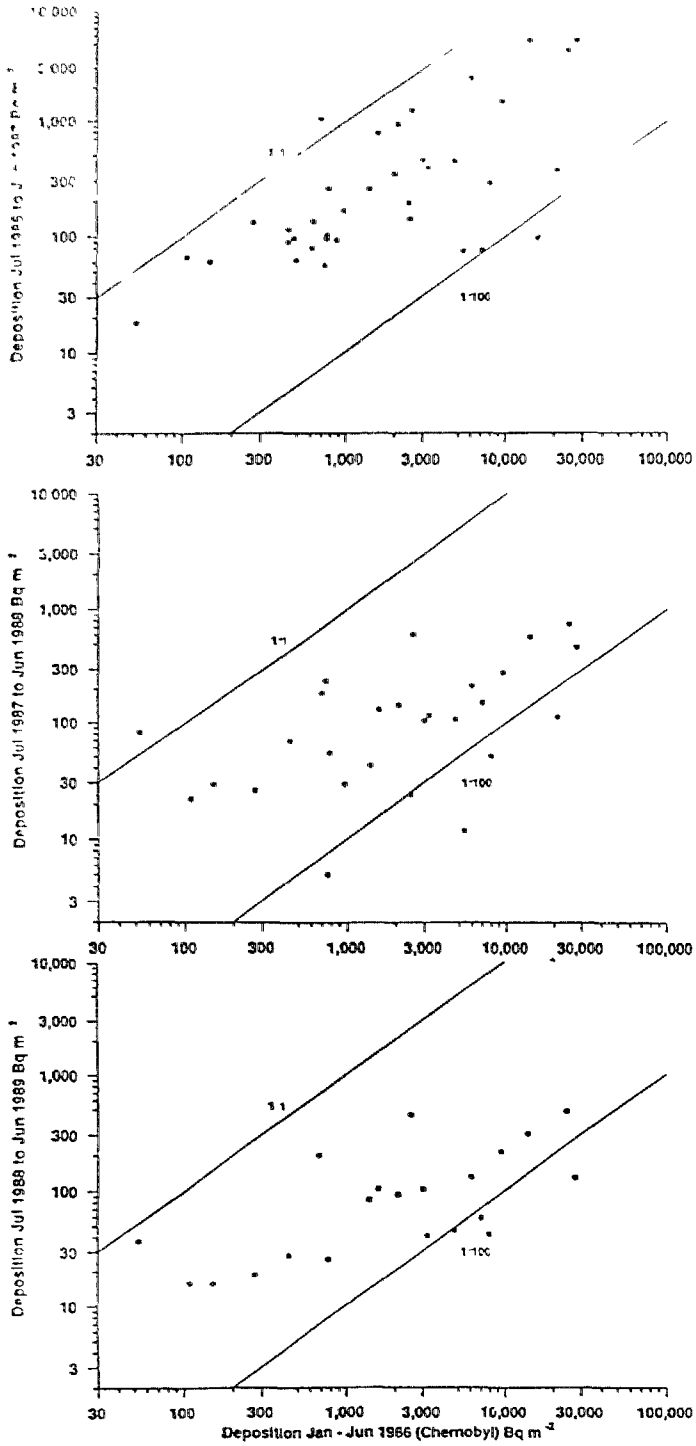


Fig. 4. The amount of resuspended material deposited during (i) July 1986–June 1987;
(ii) July 1987–June 1988;
(iii) July 1988–June 1989.

The deposition of resuspended ^{137}Cs over a period of three years, at several sites in Europe, is displayed in Fig. 4. A decrease with time is apparent, as is also the wide variability in the ratio to the amount originally deposited. There is no apparent trend in this ratio with the amount initially deposited, but the wide scatter in the results may have obscured such a trend.

It seems probable that the observed deposition involves material resuspended very locally, as a result of rain splash, wind or mechanical disturbance. Particles of at least several tens of microns diameter would explain the high apparent deposition velocities. Air samplers are prone to undersample such large particles and their sedimentation velocity is of order tens of centimetres per second. The explanation of apparent deposition velocities of order 1 m s^{-1} probably involves an exaggeration of the true deposition velocity as a result of biased measurement of the air concentration. The very variable deposition velocities and fractions re-deposited suggest that local characteristics of surface condition and vegetation cover have a marked effect on the suspension rate, or that the amount collected may be strongly influenced by the size and height and the exposure of the deposition collector.

2.6. Significance of Resuspension

2.6.1. Air concentration due to resuspended particles

Models which describe resuspension in terms of the resuspension factor, K, may readily be used to compare the predicted inhalation dose due to resuspended material with the dose caused by the passage of the plume resulting directly from a release of airborne material. This section sets out the results of such a comparison for the resuspension models introduced earlier (Fig. 1) and also considers the corresponding comparison derived from field measurements following Chernobyl.

It is convenient to make these comparisons by calculating the ratio A_R/A_I, where

- A_R is the air dose (that is, the integral of concentration in air over time) due to resuspended material and
- A_I is the air dose due to the initial passage of the cloud of airborne material.

If V_T is the total deposition velocity due to dry and wet deposition processes, the quantity deposited during the initial passage of the cloud is given as $D_I V_T$ and the subsequent concentration in air due to resuspension will be $D_I V_T K(t)$ where K(t) is the resuspension factor at time t following the initial passage of the cloud.

Thus, for each resuspension factor model, the ratio

$$\frac{A_R}{A_I}(T_1,T_2) = V_T \int_{T_1}^{T_2} K(t)dt \qquad (4)$$

gives a measure of the relative importance of resuspension during the period T_1 to T_2 in comparison with the initial passage of the cloud. The ratio may be calculated for any value of T_1 and T_2 within the range of validity of the models listed in Fig. 1, while the measured air concentrations following Chernobyl can be used to calculate A_R/A_I directly for times, T, exceeding about 2 months. The appropriate value of V_T for this comparison requires some consideration. Following Chernobyl, V_T ranged from ~0.1 mm s^{-1} in areas where dry deposition occurred to ~0.1 m s^{-1} in areas where heavy rain coincided with the arrival of the plume. For the calculations, a value of 0.01 m s^{-1} has been used as a representative value. It should be remembered that larger values of V_T may occur both where heavy rain enhances deposition, or where very large particles are present, and that A_R/A_I is proportional to V_T. Table IV shows the values of A_R/A_I calculated for the resuspension models for times up to 50 years, probably an upper limit of the time for which the model can be considered valid. The table also includes corresponding ratios determined from measurements of ^{137}Cs in air for a number of locations where sufficient data is available to determine A_R and A_I. Some caution is advised in applying these ratios for radiological assessment purposes, since the size distributions and behaviour on inhalation of the initial and resuspended aerosols, may differ substantially.

The values shown in the table emphasise the large differences between the various models. Some predict resuspension doses similar to or greater than the dose due to the original cloud, while others predict resuspension doses which are quite insignificant in this comparison. The measurements generally indicate that the latter resuspension models are closer to the truth. Low values of the ratio were observed even where rain resulted in high deposition.

TABLE IV

THE RATIO OF RESUSPENDED AIR DOSE A_R TO DIRECT AIR DOSE A_I FOR FIVE K-MODELS AND DETERMINED DIRECTLY FROM MEASUREMENTS FOLLOWING CHERNOBYL

Model or Location	A_R/A_I for time intervals			
	0 to 365 d	0 to 50 a	60 to 425 d	425 to 790 d
Anspaugh	5.9	7.6	3.8	0.84
USAEC 1974	0.61	0.64	0.27	0.002
USAEC 1975	2.3	4.7	2.1	1.0
Linsley	0.083	0.10	0.045	0.0025
Garland	0.006	0.010	0.0021	0.00064
Observations:				
Lerwick, UK			0.1	0.005
Milford Haven, UK			0.01	0.005
Eskdalemuir, UK			0.002	0.0007
Compton, UK			0.026	0.003
Orfordness, UK			0.002	0.0008
Tromso, Norway			0.002	0.001
Chilton, UK			0.003	0.003
Harwell CM, UK			0.06	0.005

2.6.2. Deposition on crops

It is well established that a fraction of material deposited from the atmosphere is retained on the leaves and other aerial parts of plants (Chamberlain and Garland, [46]): thus direct contamination of pasture grass and leaf vegetables results whenever toxic or radioactive substances are deposited. Here commonly used models are applied to estimate the possible effect of resuspended material in contaminating crops.

The fraction, f, of material intercepted by the crop canopy can be approximated by

$$f = 1 - \exp(-\mu B), \qquad (5)$$

(Chamberlain, [47]), where B is the dry weight of the above-ground portion of the crop per unit area of ground and μ is an empirical parameter, which may be a function of the physico-chemical properties of the material deposited, the mode of deposition, the structure of the crop canopy and the weather.

Observed values of μ range from 0.2 to 4 $m^2\ kg^{-1}$ (Chamberlain and Garland, [46]). There is some evidence that μ decreases with increasing particle size. If the inference is correct that the deposition of resuspended particles is dominated by large particles, then values of μ in the range 0.2 to 1.0 $m^2\ kg^{-1}$ are probably applicable.

It is convenient to note that, for $\mu B < 0.3$, $\exp(-\mu B) \simeq (1-\mu B)$ so that (5) may be simplified to

$$f \simeq \mu B. \qquad (6)$$

Then the fraction intercepted per unit dry weight of crop is

$$f/B \simeq \mu. \qquad (7)$$

Material retained on leaves may be lost by processes such as wash-off by rain, erosion of leaf cuticle, blow-off and die back of older leaves. Several observations have shown that the consequence is an exponential decrease of contaminant from crop surfaces:

$$M/M_o = \exp(-t/\tau), \qquad (8)$$

where M_o and M are the quantities retained on the crop initially and after time, t, and τ is an empirical time constant. During the growing season, τ ~15 days.

It is possible to use the simple models represented in equations (7) and (8) to estimate the ratio, d_R/d_I, of the ingestion dose due to resuspension to the ingestion dose due to the initial contamination event. The timing of the contamination event in relation to the development and harvest of the crop is crucial. Hence we consider two examples.

First, consider a hypothetical crop which is harvested and consumed at a constant rate over an extended growing season for T days after the initial deposition. Then the dose ratio will be approximately:

$$\frac{d_R}{d_I} = \frac{\mu_R \tau_R RT}{\mu_I \tau_I} \qquad (9)$$

assuming that T is several times τ_R.

The parameters μ and τ are as defined in equations (5) and (8) and the suffixes R and I refer to the resuspended material and initially deposited material, respectively. Because of the particle size considerations mentioned above, μ_R may be similar to or less than μ_I; we consider $\mu_R = \mu_I$ and $\mu_R = 0.1\ \mu_I$.

R is the fraction of deposited material that is resuspended and deposited again each day. During the year from July 1986 to June 1987 the fraction of ^{137}Cs from Chernobyl that was re-deposited was in the range 0.03 to ~1.0. Then R may be of order 10^{-4} to 3×10^{-3} d^{-1}, perhaps with a most probable value of about 3×10^{-4} d^{-1}.

Using the values indicated above for the parameters in equation (9) the ratio d_R/d_I for growing season of 100 days is expected to be about 10^{-2} with a possible range of about 10^{-3} to 10^{-1}.

This is the dose ratio that would result during the first growing season after deposition, assuming that the growing season continued for 100 days after initial deposition. Deposition, and the dose ratio, in the second growing season might be several times smaller (see Fig. 4) and would add a small fraction to d_R/d_I.

In an alternative scenario, initial deposition occurs T_c days before harvest of a seasonal crop. Various loss processes deplete the fraction of the initial deposit that remains on the crop at harvest to $\mu_I \exp(-T_c/\tau_I)$, while the accumulation and loss of resuspended material throughout the growing season results in a contribution $\mu_R R \tau\{1-\exp(-T_c/\tau_R)\}$.

In this case

$$\frac{d_R}{d_I} = \frac{\mu_R \tau_R R\{1-\exp(-T_c/\tau_R)\}}{\mu_I \exp(-T_c/\tau_I)}. \qquad (10)$$

For short T_c this ratio approximates μ_R R T_c/μ_I and for $T_c > \tau_R$, d_R/d_I approaches μ_R τ_R R/μ_I $\exp(-T_c/\tau_I)$. If Tc is as long as 100 days this ratio may be substantially greater than unity, as a result of the attenuation of the initial deposit.

These sample calculations suggest that resuspension may cause a modest increase in the total ingestion dose through crop contamination. Resuspension may sustain concentrations in a crop which would have been cleansed by field loss processes and may necessitate countermeasures over extended periods of time after the initial contamination event.

It is important to appreciate that the calculations have used models beyond their usual range of application, and the results must be regarded as tentative.

2.7. Conclusions

The measurements reviewed here have demonstrated that resuspension may result in measurable concentrations in air for a period of at least three years following deposition in climates common in Europe. The resuspension factors (K) derived from measurements in different regions of Europe all show a reduction with time after the initial deposit. The deposits varied by a factor of 800. However, with the exception of one or two sites where traffic or agriculture cause substantial disturbance shortly after deposit, K varied by a factor of 14, viz. (3.6 - 49) . 10^{-9} m^{-1}. There is a strong negative correlation between the direct Chernobyl deposit and K. A possible explanation relates to the transport of resuspended airborne activity from areas of high deposit to those of lower deposit. The reduction with time of K can be fitted with an exponential function, with the decay time constant within the range 0.03 - 0.12 per month. With few exceptions K was similar at sites with similar deposits in contrasting climatic regions.

There is evidence that the source of the resuspended material is mainly the area around the sampling site. In the first months after the accident, mechanical disturbance of the deposit in this area, e.g. by road traffic, could provide a large additional contribution to the resuspension. At a site which received a low direct deposit, there is inconclusive evidence of a small contribution to the air concentration from material returned from the stratosphere.

The air dose due to resuspension has been compared to the air dose due to the initial contaminating event at a number of measuring sites. In all cases the air dose due to resuspension during the first two years after the accident was less than 10 percent (and at most sites no more than about 1 percent) of the initial air dose.

The deposition of resuspended material has also been considered. The fraction of the deposited material that was resuspended and deposited again during the first year after the initial contaminating event ranged from 0.01 to 1.0. The potential for contaminating crops and uncontaminated surfaces may be significant.

ACKNOWLEDGEMENTS

The authors thank the Central Electricity Generating Board (now Nuclear Electric) and the Department of the Environment of the UK for financial support. The views expressed in the paper are those of the authors and do not necessarily represent those of the funding bodies, nor UK Government Policy.

REFERENCES

[1] SEHMEL, G.A., Transuranic and Tracer Simulant Resuspension, BNWL-SA-6236, Pacific North West Laboratory, Richland, Washington (1977).

[2] SEHMEL, G.A., Particle resuspension: a review, Environ. Int. 4 (1980) 107-127.

[3] NICHOLSON, K.W., A review of particle resuspension, Atmos. Environ. 22 (1988) 2639-2651.

[4] GARLAND, J.A., Resuspension of Particulate Matter from Grass and Soil, AERE-R 9452, HMSO, London (1979).

[5] GARLAND, J.A., Resuspension of Particulate Material from Grass, Experimental Programme 1979-1980, AERE-R 10106, HMSO, London (1982).

[6] LINSLEY, G.S., Resuspension of the Transuranium Elements - A Review of Existing Data, NRPB-75, HMSO, London (1978).

[7] GARLAND, J.A., 'Some recent studies of the resuspension of deposited material from soil and grass', Precipitation Scavenging, (PRUPPACHER, H.R., SEMONIN, R.G., SLINN, W.G.W., Eds.), Elsevier, New York (1983) 1087-1097.

[8] HORST, T.W., Estimation of air concentrations due to the suspension of surface contamination, Atmos. Environ. 12 (1978) 797-802.

[9] UNITED STATES ATOMIC ENERGY COMMISSION, Proposed Final Environment Statement, Liquid Metal Fast Breeder Reactor Programme, Part II G-19, WASH-1535, USAEC, Washington, DC (1974).

[10] ANSPAUGH, L.R., SHINN, J.H., PHELPS, P.L., KENNEDY, N.C. Resuspension and redistribution of plutonium in soil, Health Phys. 29 (1975) 571-582.

[11] UNITED STATES ATOMIC ENERGY COMMISSION, Reactor Safety Study - An Assessment of Accident Risks in US Commercial Nuclear Power Plants - Appendix IV, E-13, WASH-1400, USAEC, Washington, D.C. (1975).

[12] CAMBRAY, R.S., PLAYFORD, K., LEWIS, G.N.J., Radioactive Fallout in Air and Rain: Results to the End of 1981, AERE-R 10485, HMSO, London (1982).

[13] IRANZO, E., SALVADOR, S., IRANZO, C.E., Air concentrations of ^{239}Pu and ^{240}Pu and potenti radiation doses to persons living near Pu-contaminated areas in Palomares, Spain, Health Phys. 52 (1987) 453-461.

[14] CAMBRAY, R.S., CAWSE, P.A., GARLAND, J.A., GIBSON, J.A.B., JOHNSON, P., LEWIS, G.N.J., NEWTON, D., SALMON, L., WADE, B.O., Observations on radioactivity from the Chernobyl accident, Nucl. Energy 26 (1987) 77-101.

[15] GARGER, E.K., ZHUKOV, G.P., SEDUNOV, Yu.S., Estimating parameters of wind lift of radionuclides in the zone of the Chernobyl Nuclear Power Plant, Soviet Meteorology and Hydrology No.1 (1990) 1-5.

[16] HENRICH, E., WEISZ, J., ZAPLETAL, M., FRIEDRICH, M., HAIDER, W., Radiocaesium in Austrian Precipitation Samples After Chernobyl, BALUF-STS-87-8, Federal Food Control Institute, Vienna (1987).

[17] HENRICH, E., FRIEDRICH, M., ZAPLETAL, M., HAIDER, W., Bewertender Bericht über die Messung der Radionuklidkonzentration in Aerosolen 1987, BALUF-STS-AERO 87, Federal Food Control Institute, Vienna (1988).

[18] HENRICH, E., FRIEDRICH, M., HAIDER, W., Bewertender Bericht über die Messung der Radionuklidkonzentration in Aerosolen 1988, BALUF-STS-AERO 88, Federal Food Control Institute, Vienna (1989).

[19] HENRICH, E., Private communication (1990).

[20] HOETZL, H., ROSNER, G., WINKLER, R., Ground depositions and air concentrations of Chernobyl fallout radionuclides at Munich-Neuherberg, Radiochem. Acta 41 (1987) 181-190.

[21] HOETZL, H., Private communication (1989).

[22] DOMINICI, G., Misure di Radioattivita Ambientale Ispra 1986, EUR 11348 (1987).

[23] DOMINICI, G., Misure di Radioattivita Ambientale Ispra 1987, EUR 11821 (1988).

[24] DOMINICI, G., Misure di Radioattivita Ambientale Ispra 1988, EUR 12223 (1989).

[25] CAZZANIGA, R., DOMINICI, G., MALVICINI, A., SANGALLI, E., Chernobyl: A Year After, EUR 12175 (1989).

[26] SINKKO, K., AALTONEN, H., MUSTONEN, R., TAIPALE, T.K., JUUTILAINEN, J., Airborne Radioactivity in Finland after the Chernobyl Accident in 1986, Suppl. 1 to Annual Report STUK-A55 (1987).

[27] SAXEN, R., TAIPALE, T.K., AALTONEN, H., Radioactivity of Wet and Dry Deposition and Soil in Finland after the Chernobyl Accident in 1986, Suppl. 2 to Annual Report STUK-A55, (1987).

[28] AALTONEN, H., Radionuclides in Ground-Level Air - Quarterly Results of Air Surveillance Programme - 1st, 2nd, 3rd and 4th quarters 1987, Finnish Centre for Radiation and Nuclear Safety (1987-1988).

[29] AALTONEN, H., Radionuclides in Ground-Level Air: Quarterly Results of Air Surveillance Programme - 1st, 2nd, 3rd and 4th quarters 1988: 1st, 2nd, 3rd and 4th quarters 1989, Finnish Centre for Radiation and Nuclear Safety (1988-1990).

[30] JAGIELAK, J., PIETRUSZEWSKI, A., WOLOSZYN, Z., ZAWANOWSKI, K., GARLINSKI, K., KOZUB, M., Effective dose equivalent to average individuals in Warsaw after the Chernobyl accident, Radiat. Prot. Dosim. 20 4 (1987) 243-247.

[31] JAGIELAK, J., Private communication (1990).

[32] KOLB, W., Radionuclide Concentration in Ground-Level Air from 1986 to 1987 in North Germany and North Norway, Physikalisch Technische Bundesanstalt PTB-Ra-21 (1988).

[33] KOLB, W., Private communication (1989).

[34] PATTENDEN, N.J., CAMBRAY, R.S., PLAYFORD, K., Studies of Environmental Radioactivity in Cumbria, Part 16: Trends in Radionuclide Concentrations in Coastal Airborne and Deposited Materials 1978-1987, AERE-R 12617, HMSO, London (1989).

[35] PLAYFORD, K., Private communication (1990).

[36] AARKROG, A., BOTTER-JENSEN, L., CHEN QING JIANG, DAHLGAARD, H., HANSEN, H., HOLM, E., LAURIDSEN, B., NIELSEN, S.P., SOGAARD-HANSEN, J., Environmental Radioactivity in Denmark in 1986, Risø National Laboratory Report Risø -R-549 (1988).

[37] AARKROG, A., BOTTER-JENSEN, L., CHEN QING JIANG, DAHLGAARD, H., HANSEN, H., HOLM, E., LAURIDSEN, B., NIELSEN, S.P., SOGAARD-HANSEN, J., Environmental Radioactivity in Denmark in 1987, Risø National Laboratory Report Risø -R-563 (1989).

[38] AARKROG, A., Private communication (1989).

[39] CAMBRAY, R.S., PLAYFORD, K., LEWIS, G.N.J., CARPENTER, R.C., Radioactive Fallout in Air and Rain: Results to the End of 1987, AERE-R 13226, HMSO, London (1989).

[40] CAMBRAY, R.S., PLAYFORD, K., LEWIS, G.N.J., CARPENTER, R.C., Radioactive Fallout in Air and Rain: Results to the End of 1988, AERE-R 13575, HMSO, London (1991).

[41] CAMBRAY, R.S., Private communication (1989).

[42] BJURMAN, B., FINCK, R., ARNSTING, R., LARS-ERIK de G., JAKOBSSON, S., VINTERSVED, I., Resuspension Measurements Second Half Year 1986, National Defence Research Institute Report FOA C 20678-9-2 (1987).

[43] GARLAND, J.A., CAMBRAY, R.S., 'Resuspension after Chernobyl', Aerosols, Their Generation, Behaviour and Application, 3rd Annual Conf. of the Aerosol Society (Proc.Conf. West Bromwich, 1989).

[44] AOYAMA, M., Evidence of stratospheric fallout of caesium isotopes from the Chernobyl accident, Geophys. Res. Lett. 15 (1988) 327-330.

[45] GARLAND, J.A., CAMBRAY, R.S., 'Deposition resuspension and the long-term variation of airborne activity from Chernobyl', Impact des Accidents d'Origine Nucléaire sur l'Environnement, (4e Symp. Int. de Radioécologie de Cadarache, 1988) Tome 1, CEN/Cadarache (1988).

[46] CHAMBERLAIN, A.C., GARLAND, J.A., Interception of Radioactive Fallout by Vegetation, AERE R-13826, HMSO, London (1991).

[47] CHAMBERLAIN, A.C., Interception and retention of radioactive aerosols by vegetation, Atmos. Environ. 4, (1970) 57-58.

Chapter 3

THE INFLUENCE OF FOOD PROCESSING AND CULINARY PREPARATION ON THE RADIONUCLIDE CONTENT OF FOODSTUFFS: A REVIEW OF AVAILABLE DATA

H. Noordijk
National Institute of Public Health,
Bilthoven, Netherlands

J.M. Quinault
Commissariat à l'energie atomique,
Centre d'etudes nucléaires de Cadarache,
Saint-Paul-lez-Durance, France

3.1. Introduction

3.1.1. Background

Radioactive contamination of the environment may result in the increased radiation exposure of human beings due to their ingestion of radionuclides in food. In the chain of transfer processes which leads from the deposition of radionuclides on to soils and plant surfaces to their presence in the diet, food processing and culinary preparation are the last processes which can effect the radionuclide content of foodstuffs. In the years before the Chernobyl accident the effects of food processing and culinary preparation on the radionuclide content of foodstuffs received comparatively little attention although for some food products the reductions can be substantial. Studies and investigations in this field have instead focussed mainly on the processes of transfer of radionuclides in the components of the human food chain.

Knowledge of the effects of food and processing and culinary preparation is needed when assessing the radiation dose to man from the ingestion of contaminated foodstuffs. Appropriate allowances have to be made for the reductions produced by food processing* in order that doses are not systematically overestimated. A knowledge of the effects of special food processing or decontamination techniques is needed in order to estimate their effectiveness as possible countermeasures for reducing the

---

*In the remainder of this paper 'food processing' is used as a general term to include industrial processing and domestic food preparation.

dose to man following the contamination of the food chain with radionuclides in the aftermath of an accident.

Measurements of the transfer of radionuclides during food processing are comparatively scarce. Most measurements date back to the sixties and only a few attempts have been made to review this literature [1-6]. These reviews are often limited to selected food classes. Also, the comparability of data is hindered by the variety of definitions used to define changes in radionuclide content of the food. Since the Chernobyl accident in 1986 new data have become available especially on the behaviour of radiocaesium.

3.1.2. Industrial and domestic scale processing

Food processing methods vary considerably from foodstuff to foodstuff and also between the industrial and the domestic scale (Fig. 1). A food product may be processed quite differently when grown locally in a garden and consumed by the grower and his family as compared with the large scale commercial production of the same food product followed by factory processing. The resulting losses in the two cases may be significantly different. In the case of milk, modern processing techniques provide several alternative products each of which may contain different concentrations of the contaminating radionuclide (Fig. 2). This may be contrasted with the small scale consumption of whole milk.

3.1.3. Data needed for radiological assessments

Depending upon the nature of the radiological assessment, different types of information are needed on food processing losses. For assessments of dose to identified groups of individuals living at particular locations, the ideal approach would be to obtain local information on food processing losses specific to the foods consumed by the group. This approach would minimise the predictive error associated with the food processing component of the dose calculation. However, this is rarely feasible and national or regional compilations of the type contained in this report are usually used. Nevertheless, the potential error in the use of tabulated processing loss factors can be reduced by checking that the sources of information on which the loss factor is based are relevant to the circumstances of the dose assessment, for example, whether the food is locally grown and prepared or is subject to industrial processing.

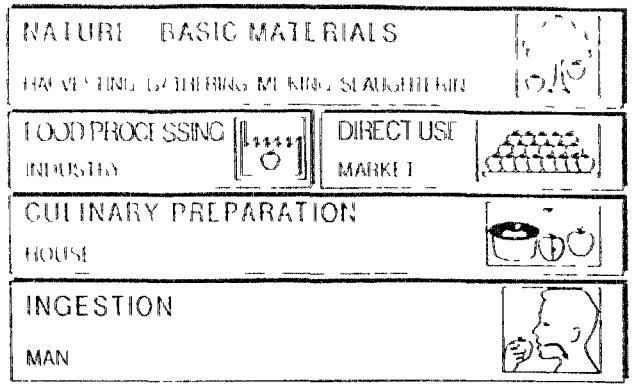
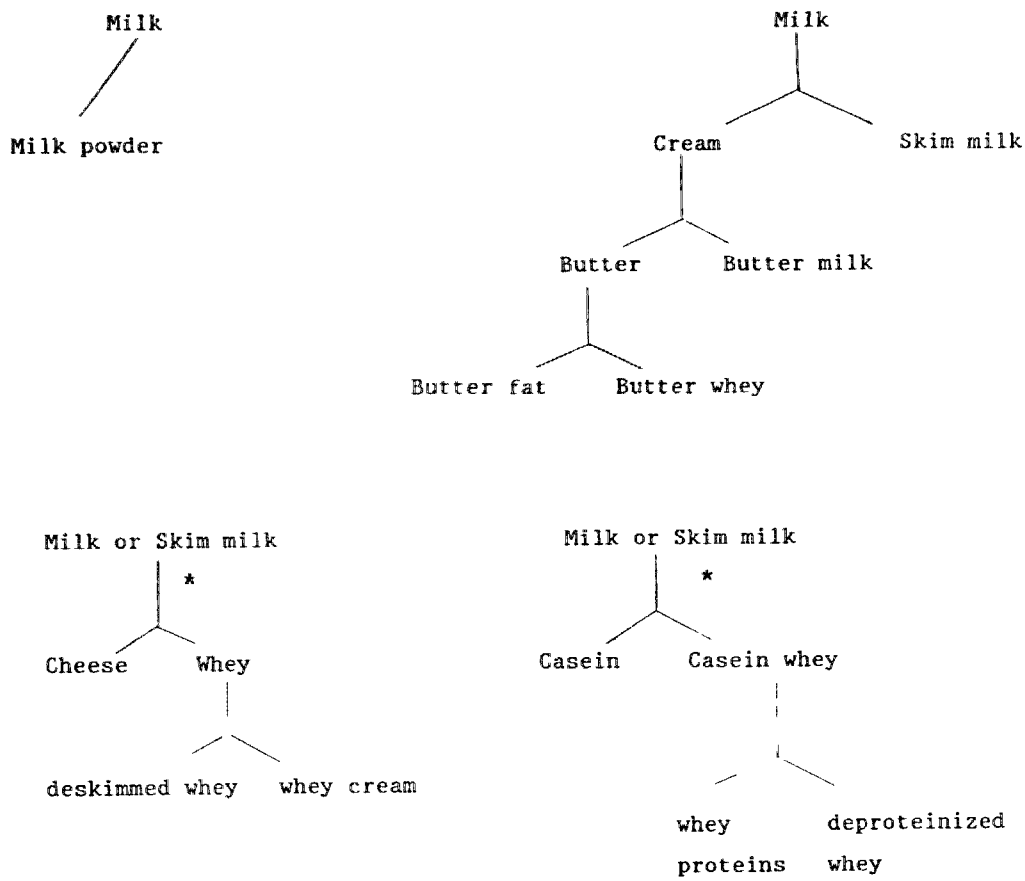


Fig. 1. Different types of food processing.



* rennet or acid may be used to start the coagulation procedure.

Fig. 2. The fractionation of milk into products and by-products in the dairy industry.

In more generic dose assessments such as those used in planning studies where doses to hypothetical critical groups are being predicted, or in the evaluation of collective dose, regional or nationally averaged values of the loss factor may be appropriate. In one approach to collective dose evaluation it may not be necessary to identify the exposed population group but merely to assume that all food produced in a particular area is consumed. In this case an overall food processing loss factor could, in principle, be used. This is an averaged factor representing the total loss between the activity in the raw food product and that in the consumed material. However, in practice, the available information tends to be at the process level and is in the form of several separate loss factors between the raw product and consumption by man.

3.1.4. Objectives and scope

It is the aim of this paper to present a survey of the information available on radionuclide behaviour during food processing. It expands on an earlier study supported by the Commission of the European Communities (CEC) [1] and takes into account new data. Most of these were presented at the Seminar on "Radioactivity Transfer during Food Processing and Culinary Preparation" organised by the Commissariat à l'Energie Atomique (CEA) and the CEC at Cadarache, France, in 1989. In addition, the paper takes into account the comments of the VAMP Terrestrial Working Group. The data presented are reported in terms of a single definition of the change in the radionuclide content of food as a result of processing. A distinction is made between normal food processing procedures and special decontamination procedures which might be of importance in emergency situations.

If not stated otherwise, all data presented in this review are based on experiments which reflect contamination in food products at a time after an accident, when contamination has been to some extent relocated and is no longer predominantly external. Thus, the data on plant contamination used in this report are generally derived from root uptake experiments, while those on food from animal origin are derived from in vivo studies. Shortly after an accident the contamination will be predominantly on the exterior parts of the product. For vegetables and fruit, separate values are available for contamination which is predominantly on external parts. These data have rather large ranges. Data which refer to the situation shortly after an accident are not available for other classes of food. However, the use of data, which refer to internal contamination, will lead,

in the case of fresh contamination, to underestimation of the decontaminating effect of food processing since newly deposited external contamination will be easier to remove than aged and relocated contamination.

3.2. Definitions

Several definitions are used in the literature to define changes in the radionuclide content of food due to processing. Most definitions are equal to or closely related to one of the three definitions presented here.

(A) The change in radionuclide content due to food processing can be defined by the food processing retention factor, F_r, which is the total amount of a radionuclide in processed food divided by the total amount of this radionuclide in the original raw food (Bq processed per Bq raw). Thus, F_r is the fraction of the quantity of the radionuclide which remains in the food after processing. All data presented in this review are based on this definition.

(B) The effect of food processing on the radionuclide content of food may also be quantified by the ratio of radionuclide concentrations, measured in fresh raw and fresh processed food. (Bq/kg fresh processed per Bq/kg fresh raw).

(C) The third type of definition is based on a ratio of radionuclide concentrations, measured in dried raw and dried processed food. (Bq/kg processed dried per Bq/kg raw dried).

The F_r value is considered to be the best way to define changes in radionuclide content of food in this review because (a) a considerable amount of literature data is already defined in this way, and (b) conversion to F_r values appears to be possible for nearly all collected literature data which is not the case for the other definitions.

In order to avoid misinterpretations, the definition (A) is illustrated with reference to Cs and Sr. The F_r value of 0.4 for Cs boiled meat indicates that only 40% of the Cs present in raw meat is retained after boiling, 60% is removed in the boiling liquid (Table I). In the case of dairy products (Table II) the yield of each product is important. In the

TABLE I

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR MEAT

Underlined data denote best estimates

Raw Material	Method of processing	Sr		Cs		I	Ru	Pe
Meat of mammals (cow, pig sheep, deer rabbit)	Boiling meat	<u>0.5</u>	0.4-0.9	<u>0.4</u>	0.2-0.7	0.6	0.3	0.7
	Boiling bone		0.999		0.2-0.3	0.98	0.7	1.0
	Frying, roasting or grilling meat		0.8		0.5-0.8			0.4-0.7
	Mincing meat				0.4			
	Microwave baking				0.4-0.5			0.4-0.7
	Pickling wet				0.1-0.7			0.9
	dry				0.8			
	Marinating				0.1-0.6			
	Sausage production				0.4-1.0			
Birds	Boiling meat		0.5					
	Baking meat				0.7-0.8			
Fish	Boiling meat		0.9	<u>0.7</u>	0.2-0.9			0.5-0.9
	Frying meat				0.8-0.9			0.7-0.8

References: [3, 6, 29, 42-58]

tables, the yield is presented under the "processing efficiency" P_e (kg prepared product/kg raw material). For example, an F_r value of 0.61 for Sr in goat cheese indicates that 39% of the Sr is removed by the conversion of goat milk to cheese, but, due to the 12% yield of cheese, the concentration of Sr in goat cheese is 0.61/0.12 = 5 times the concentration of Sr in goat milk.

It must be stressed that all F_r values referring to extraction procedures such as cooking, frying, etc., are only valid when the extraction liquid is removed and not used for other culinary purposes. The same applies to fractionation processes which take place in eg. the dairy industry or during the milling of cereals. As most fractions are used for human consumption, the use of F_r values should be supported with a thorough knowledge of the destination of the different fractions. For example, the by-product of cheese-making, whey, was formerly regarded as waste or used to feed animals. Nowadays, a large part of the produced whey is used as an additive for food for humans. Similarly, all fractions of the milling process are usually used to produce food. In these cases, an F_r value of 1.0 should be used.

TABLE II

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR DAIRY PRODUCTS

Underlined data denote best estimates

Product	Sr		Cs		I	P_e	
Cream	<u>0.07</u>	0.04-0.25	<u>0.05</u>	0.03-0.16	0.06-0.19	<u>0.08</u>	0.03-0.24
Skimmilk	<u>0.93</u>	0.75-0.96	<u>0.95</u>	0.85-0.99	0.81-0.94	<u>0.92</u>	0.76-0.97
Butter	<u>0.006</u>	0.0025-0.012	<u>0.01</u>	0.003-0.02	0.035-0.01	<u>0.04</u>	0.03-0.05
Buttermilk	<u>0.06</u>	0.03-0.07	<u>0.05</u>	0.02-0.13	0.05-0.13	<u>0.04</u>	0.03-0.14
Butterfat		0.001-0.002	<u>0.00</u>	0.00-0.00	0.02	<u>0.04</u>	0.04-0.04
Milk powder		1.00		1.00	1.00		0.12
Cheese *							
goat		0.61		0.07-0.15	0.08-0.14	<u>0.12</u>	0.08-0.17
cow rennet		0.025-0.80	<u>0.07</u>	0.05-0.23	<u>0.20</u> 0.11-0.53	<u>0.12</u>	0.08-0.18
cow acid		0.04-0.08		0.11-0.12	0.22-0.27	<u>0.10</u>	0.08-0.12
Cottage cheese rennet		0.07-0.17		0.01-0.05			
Cottage cheese acid		0.22					
Yogurt				0.34			
Whey *							
rennet		0.20-0.80		0.73-0.96	0.47-0.89	<u>0.90</u>	0.70-0.94
acid		0.70-0.90		0.75-0.90	0.60-0.73		0.82
Casein *							
rennet		0.10-0.85		0.01-0.08	0.02-0.12		0.03-0.06
acid		0.05-0.08		0.01-0.04	0.03-0.04		0.01-0.06
Casein whey*							
rennet		0.08-0.16		0.77-0.83	0.69-0.82	<u>0.76</u>	0.73-0.79
acid		0.67-0.86		0.83-0.84	0.78-0.80	<u>0.78</u>	0.75-0.79
Milk **							
ion exchange	<u>0.1</u>		<u>0.01</u>		<u>0.1</u>		

COMMENTS

* Separate values are given for the rennet and acid coagulation procedures

**Decontamination of milk by ion exchange on commercial scale

References: [3-5, 7-41]

3.3. Effects of Normal Food Processing

3.3.1. Data analysis

The quality and reliability of all data on radionuclide behaviour during food processing was checked to the extent possible before they were incorporated in the survey. Firstly, it is required that experimental or test procedures should produce an artificial contamination of the food products which is comparable to the contamination which would occur following an accidental release of radionuclides. Special attention was given to speciation and to the spatial distribution of the radionuclides in food. Also, simulated processing procedures need to be a realistic reflection of the procedures normally applied to food before it is consumed. Information on modified procedures may, however, be relevant in order to know the effectiveness of a planned decontamination method. Special decontamination procedures are included in the survey only when (a) they result in a lower contamination of the processed food than in the case of normal food processing and (b) when the food retains its nutritional value and is not unacceptably changed in taste and appearance.

The data was converted to F_r values and ordered on the basis of the type of food, the method of processing and the type of radionuclide studied. The large amount of information from more than a hundred articles and reports required a grouping of the different processing modes in order to present the data in a useful format for subsequent applications. The tables present ranges of F_r values for different foods and processing techniques. Where possible, processing efficiencies P_e are also included.

3.3.2. Data variation

In general, the data within each separate combination of product, procedure and radionuclide show a large variation. This variation may have different causes. Variation may be caused by differences in the yield, P_e. For example, the first row of data in Table I show the range in F_r values of Sr for cream. This range in F_r values is nearly identical to the range in P_e. Observed concentrations of Sr in cream are close to those in raw milk, with much less variation in them than the F_r values in Table I suggest. For dairy products and cereals the differences in yield are an important cause of variation. Variation in the F_r values of the other products, however, seem to be mainly caused by differences in the application of particular food preparation methods. In fact, the way in

which people prepare their food is a very important and specific part of their cultural background. Some of the most important aspects of the preparation are the temperature, the additives used, the time spent on each step in the procedure and the mechanical treatments such as removal of inedible parts, peeling, slicing, etc. Also industrial processes may vary from country to country, leading to more or less refined products. For external contamination, the time between contamination and harvest will cause variation in F_r values. In this case, climatological conditions or agricultural techniques, such as the addition of water by spraying rather than by irrigation, may also influence the F_r values.

Best estimate values of F_r and P_e are given whenever the data are adequate to permit these judgements. The most important features are that many data from different regions are available, that the range in the vast majority of the data is much smaller than the total range observed and, that extremes are not typical for certain regions. Regarding the large variation in P_e and F_r, the use of P_e values to convert F_r to concentration ratios is not recommended for extreme values. However, for best estimates or mean values the use of P_e will give a good indication of expected concentration ratios.

3.3.3. Dairy products

A considerable amount of data are available on the behaviour of the nuclides Sr, Cs and I during milk processing (Table II) (Figure 2). In this case F_r represents the fraction of the radionuclides which remains in the processed food. However, as under normal circumstances nearly all by-products are used for other food products - even the whey of cheese production is used nowadays for beverages and other foodstuffs - only a small amount of radioactivity will be removed from the diet by milk processing. Following an accident, however, milk processing may be modified deliberately so as to reduce intakes.

The distribution of the radionuclides over the various products is influenced by the yield of each product. For example, a higher yield of cream will lead to a proportionally higher F_r value for cream if the increase in the yield does not affect the concentration of the radionuclide in the cream.

The coagulation process appears to have a considerable effect on the transfer of Sr to cheese. F_r values for Sr are quite unpredictable in

the case of rennet coagulation, but often a large transfer to cheese is observed. Coagulation by an acidifying procedure appears to lead to a much lower Sr transfer to cheese. For all nuclides, the variation in F_r values for cheese production by acid coagulation is smaller than by rennet coagulation.

TABLE III

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR *LOWER SEA ORGANISMS*

Organism	Method of processing	F_r values	P_e
crevette	cooking	Ra:0.04-0.5 Pb:0.0-0.4 Po:0.04-0.8	0.35
mussels	Washing and removal of flesh	Ra:0.01 Pb:0.5 Po:0.02	0.25
algues	alginate production	Tc:0.02 Ru, Rh 0.07 Sr:0.6	0.04
	satiagum production	Ru, Rh:0.04 Co:0.04	0.08

References: [59-61]

3.3.4. Meat

Tables I and III give data on meat, fish and other marine derived food sources. Data on meat processing do not appear to be influenced by animal type. Radionuclides in fish might behave somewhat differently when compared to mammals, although too few data are available on fish processing to draw any firm conclusions.

3.3.5. Vegetables and fruit

F_r values for vegetables and fruit appear to depend highly on the type of product and the method of processing (Table IV). The variability

TABLE IV

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR VEGETABLES AND FRUIT

Data are based on total contamination of the plant

Plant	Method of processing	Sr	Cs	Other Nuclides	P_e
Spinach	Washing	0.4-1.0	0.6		1 0
	Washing & blanching	0.4-1.0	0.5-0.6		0.8
	Cooking & rinsing	0.9	1.0	Co 0.9	0.7
	Canning	0.5	0.2		0.7
	Freezing	1.0			0 7
Lettuce	Removing inedible parts		0.5		0.7
	Blanching	0.3-0.9	0.1-0.6		0.5
Cabbage	Marinating		0.9	Ru 0.5	0.9
	Washing	0.3	0.9		1.0
	Washing & blanching	0.4-1.0	0.1-1.0		1.0
	Cooking & rinsing	0.8			0.7
	Freezing	0.2-0.9	0.7		0.7
	Canning	0.4	0.2		0 7
Cauliflower	Peeling		0.5		0.7
Beans	Washing	0.1			1.0
	Blanching	0.3-1.0	0.6-0.9		0.9
	Canning	0.3-0.8	0.4-1.0		
	Froth flotation	0.4-0.6	1.0		
	Brine grading	0.6	1.0		
Tomatoes	Washing	0 7			1.0
	Peeling & slicing	0.7			0.9
	Canning	0.8			
	Frying	0.5			0.5
Onions	Peeling + washing + boiling	0.5			
Mushrooms	Boiling	0.7-0.9	0.2-0.5		0.6
	Boiling in 2% NaCl	0.2	0.2		
	Canning	0.5	0.5		
	Parboiling		0.1-0.4		
	Soaking of dried mushrooms		0.1-0.2		
	Parboiling + salting + soaking		0.00		
Cucumbers	Pickling		0.15		
	Canning	0 35	0.06		
Peaches	Peeling	0 5			0.9
	Canning	0.5			
	Lye peeling	0.09	0.03		
Strawberries	Rinsing	0.7	0.6		1.0
Berries	Making Puree		0.6-0.8		0.6-0.8
	Rinsing		0.8		

References [4, 29, 61-86]

TABLE V

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR VEGETABLES AND FRUIT

Data are based on external contamination only

Plant	Method of Processing	Sr	Cs	I	Other Nuclides	P_e
Spinach	Washing	0.2	0.2-0.9	0.07-0.8	Ru:0.4-0.8	1.0
	Washing + blanching	0.4 0.7	0.2-0.9	0.6-0.7	Ru:0.5-0.8	0.8
	Cooking + rinsing			0.4		0.7
Lettuce	Washing		0.2-1.0	0.1-0.5	Ru:0.2	1.0
	Removing inedible parts		0.1-0.4	0.1-0.4	Ru:0.01-0.3	0.7
Cabbage	Removing inedible parts		0.9	0.5	Ru:0.7-1.0	0.8
	Washing	0.07	0.09	0.4		1.0
	Washing + blanching	0.3	0.2-0.7			0.7
	Cooking + rinsing			0.2-0.5		0.7
Cauliflower	Peeling		0.05-0.2	0.03	Ru:0.02	0.7
Beans	Washing			0.7		1.0
	Blanching	0.3	0.3	0.2		0.9
	Froth flotation	0.4	0.4			
	Brine grading	0.4	0.4			
Tomatoes	Washing			0.5		1.0
	Boiling			0.2		0.7
Onions	Removing inedible parts		0.2	0.2	Ru:0.2	0.9
	Washing		0.3	0.2		1.0
Mushrooms	Boiling in 2% NaCl		0.3			
Berries	Rinsing		0.8-0.9		Ru:0.8-1.0	
	Making purée				Ru:0.7	0.6-0.8
	Boiling		0.3-0.5	0.2	Te:0.3-0.7 Ba:0.6-0.9 Zr:1.0	

References: [29, 61-86]

of the data may be a result of differences in processing time and temperature, in other specific aspects of the processing, and in the rigidity of the cellular structure of the vegetable product, etc. For example, a much more efficient removal was observed on several products for steam blanching when compared to water blanching [78]. The high Sr removal from beans by washing might be due to large amounts of soil attached to the product, but no explanation was given in the relevant report [76].

Data on externally contaminated plants are presented separately (Table V). The scatter in these data is even higher when compared to internally contaminated plants.

TABLE VI

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR ROOT CROPS

Raw material	Method of processing	Sr	Cs	Other radionuclides	Pe
Potato	Boiling with peel	0.9-1.0	0.8-0.9	Po:0.4-0.7	0.9
	Peeling	0.5-0.9	0.6-0.8	Po:0.3-0.5 Pu, Am:0.1-1.0	0.8
	Peeling & boiling	0.7-0.8	0.6		0.8
	Frying	0.6			0.6
	Microwave boiling unpeeled		0.8		0.8
	Microwave boiling peeled		1.0		0.6
	Canning	0.7	1.0		0.6
	Decontamination	0.5	0.05-0.2	Ru:0.5	
Carrot	Scraping + washing + boiling	0.8			0.8
	Peeling	0.7	0.5	Pu, Am:0.4	0.8
	Cooking unpeeled		0.5-0.8		0.8-0.9
	Microwave cooking unpeeled		0.7		0.8
	Microwave cooking peeled		0.5		0.7
Beetroot	Peeling	0.8	0.4-0.7	Pu, Am:0.45	0.8
	Cooking unpeeled		0.3-0.7		0.9
	Cooking + peeling		0.3		0.8
	Microwave cooking unpeeled		0.4		0.75
	Microwave cooking peeled		0.3		0.7
Parsnip	Peeling	0.7	0.6	Pu:0.3	
Swede	Peeling	0.65	0.6	Pu:0.7	

References: [29, 62, 63, 69, 72, 73, 76, 77, 82, 83, 84, 86, 87, 88]

3.3.6. Root crops

The effects of food processing are generally small for root crops with the exception of beetroot (Table VI). Peeling may be very efficient in removing actinides, probably due to the removal of soil particles attached to the peel combined with the extremely low uptake of actinides from the soil by plants.

3.3.7. Cereals

F_r values for milled cereals are generally lower than 0.5 (Table VII). The values depend on the yield of the product, as observed with milk products. The conclusion which has sometimes been drawn that transfer of Cs and Sr to 'dark' wheat flour is lower than transfer to 'white flour' is not justified. Concentrations of Cs and Sr in 'dark' flour are higher than concentrations in 'white' flour, but the yield P_e of 'dark' - or second quality - flour is much lower in these experiments. Removal of actinides by milling is also high.

3.3.8. Drinks

The limited data in Table VIII indicate that normal treatment and processing of beverages has only modest effects on reducing the

TABLE VII

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR CEREALS

Raw Material	Method of Processing	Sr	Cs	Pu, Am	P_e
Wheat grain	Milling to white flour	0.09-0.5	0.2-0.6	0.1-0.2	0.7
	Milling to dark flour	0.1-0.2	0.05-0.1		0.05-0.1
	Milling to semolina		0.15-0.5		0.2-0.3
	Milling to bran	0.6-0.9	0.5-0.6		0.1-0.2
	Cooking wheat sprouts		0.9		1.8
	Shredding or puffing wheat		0.1-0.15		0.9-0.95
Durum Wheat grain	Milling to flour		0.1-0.6		0.08-0.8
	Milling to groat + groatdust		0.3-0.4		0.6-0.7
	Milling to bran		0.4-0.5		0.2
Rye grain	Milling to white flour	0.6	0.3-0.6	0.2	0.6-0.8
	Milling to dark flour		0.2		0.1
	Milling to bran		0.35-0.7		0.15-0.4
	Cooking rye sprouts		0.8-0.9		1.9-2.4
Barley grain	Milling to white flour	0.5	0.2-0.6	0.1-0.2	0.6-0.8
	Milling to semolina		0.35		0.1
	Milling to bran		0.4		0.4
Oats grain	Milling to white flour	0.3	0.4	0.4	0.4
Pasta	Cooking		0.8-0.9		2.2

References: [16, 29, 62, 67, 86, 89-99]

TABLE VIII

FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR DRINKS

Raw material	Method of processing	Sr	Cs	Other radionuclides	P_e
Surface water	Conventional treatment to tap water	1.0	0.7	Ru:0.3 Co:0.4 I:0.8	
Tea	Brewing 2 minutes		0.4		
	Brewing 8 minutes		0.6		
Herb tea	Brewing		0.04-0.6		
Berries	Pressing to juice		0.8-0.9		0.8-0.9
	Steaming to juice		0.2-0.6		0.3-0.7

References: [29, 61, 68 100-104]

radionuclide content, although it is recognized that special drinking water treatments can be very effective for some elements.

3.4. Decontamination

3.4.1. Introduction

In cases of extensive contamination of the environment with radionuclides, processes for the decontamination of foodstuffs may be considered as a countermeasure in order to reduce doses to the population. Although food processing takes place in the household and in industry, changes in household food preparation methods should only be recommended in

very extreme situations because of possible public reactions, social consequences and the lack of the possibility to control the implications of such advice. However, changes in industrial food processing methods may be feasible and lead to a lower transfer of radionuclides to the population.

In general some decrease in the transfer of radionuclides to the population may be obtained by applying selective processing to certain types of products. In some cases, optimised decontamination procedures may decrease food contamination considerably. The feasibility of changes in food processing methods will depend strongly on the product and food technology.

The success of efforts to decontaminate food is, in general, related to the rigidity of the cellular structure. Consequently, decontamination is more efficient for foods of animal origin than for vegetable products, and the results for liquids such as milk are rather successful.

3.4.2. Milk

For the decontamination of milk without losses in nutritional value and deterioration of the taste, ion exchange appears to be the most promising method [14]. The F_r values presented here refer to milk decontamination on a commercial scale (Table II). Better results have been reported from research on a laboratory scale especially for caesium. Other methods, based on coprecipitation or electrodialysis led to a serious deterioration of the taste of milk. Recently, a method based on complexation of caesium by modified Prussian Blue was developed to decontaminate whey on a commercial scale [104]. It is not yet known whether this method will have adverse effects on taste, appearance or nutritional value of the milk. In the dairy industry it is also possible to shift production towards products which are contaminated to a lesser extent by radionuclides. However, consideration has to be given to the economic value of the by-products. Butter production, sometimes suggested, would lead to contaminated buttermilk, which has an important economic value. A better strategy would be the production of certain types of cheese or refined products such as caseine. The remaining whey is of less economic value and can be regarded as waste.

3.4.3. Meat

Several decontamination procedures have been applied to meat, often resulting in poor decontamination combined with quite unfavourable

consequences for the taste and appearance of the meat. Certain ways of pickling or marinating may, however, result in contamination levels of Cs which are reduced by about an order of magnitude, while the taste remains good [50, 56].

3.4.4. Root crops

In the case of root crops decontamination procedures may lead to an F_r value of about 0.1 or even lower for Cs [63] but the effects of decontamination are limited for S_r.

3.4.5. Vegetables

For vegetables, a number of decontamination procedures have been applied, most often based on immersion in a decontamination liquid. The results are, however, not promising and consequences for taste and appearance not favourable. Unexpected effects were observed, such as a decreasing permeability of the cellular structure at higher temperature, leading to lower decontamination [88]. However, some efforts were successful; mushrooms can be decontaminated effectively by a combination of salting and cooking [75].

No important results are known on decontamination of cereals, other drinks than wine or of sea organisms.

3.5. Conclusions

The effects of processing on the behaviour of radionuclides depend on the radionuclide, on the type of product and on the method of processing. In general, food processing may halve the amount of radionuclides present in the food. Processing effects are rather small for root crops whereas milling cereal grains to flour will often remove about 70% of the radioactivity. Although attempts to apply decontamination procedures were often not successful, several procedures have been developed which may remove about 90% or more of the radioactivity for certain combinations of radionuclides and products. In the case of the processing of milk, meat and potatoes, available data allow a rather detailed insight into the effects of processing. The effects of processing on vegetables and fruits are rather unpredictable, especially when the radionuclides are adsorbed on the surface of the plants.

REFERENCES

[1] NOORDIJK, H., 'Literature review on radionuclide behaviour during food processing', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989).

[2] COMMISSION OF THE EUROPEAN COMMUNITIES, Underlying Data for Derived Emergency Reference Levels, Post-Chernobyl Action, EUR 12553 EN, Commission of the European Communities, Luxembourg (1991).

[3] ANNEKOV, B., Radiobiology and radioecology of farm animals, Radiobiol. Radioekol. Sel'Shokhoz Zhivotn. (1973) (in Russian).

[4] ARNAUD, M.J., 'The removal and/or reduction of radionuclides in the food chain', Radionuclides in the Food Chain, ILSI Monographs, Springer Verlag, Berlin (1988) 195-213.

[5] McENRI, C.M., MITCHELL, P.J., CUNNINGHAM, J.D., 'The transfer of radiocesium from whole milk and milk products', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Seminar Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 251-273.

[6] WAGNER, H., Uebergang von radioaktiven Stoffen vom Futter in das Fleisch von Schlachttieren, Fleischwirtschaft 68 (1988) 656-664.

[7] ASSIMAKOPOULOS, P.A., IOANNIDES, K.G., PAKOU, PARADOPOULOU, C.V., Transport of the radioisotopes iodine-131, cesium-134 and cesium-137 from the fallout following the accident at the Chernobyl nuclear reactor into cheese and other cheesemaking products, J. Dairy Sci. 70 (1987) 1338-1343.

[8] ANTILA, V., KANKARE, V., PAHKALA, E., PAAKKOLA, O., RANTAVAARA, A., Radionuklidien siirtyminen meijeriprosesseissa eri maitovalmisteisiin ja maidon fraktioihin, Meijeriteollisuus 3 (1987) 36-37.

[9] BALES, R.E., HICKEY, J.L.S., 'Commercial processing of milk for concurrent removal of cationic and anionic radionuclides', Radioisotopes and Radiation in Dairy Science Technology (Proc. Sem. Vienna, 1966), IAEA, Vienna (1966) 121-145.

[10] BUMA, T.J., MEERESTRA, J., Transfer of radiostrontium from milk to cheese and whey, Nature 202 (1964) 310-311.

[11] CALAPAJ, G.G., ONGARO, D., Sul comportamento del ^{90}Sr e del ^{137}Cs nei processi di burrificazione e caseificazione del latte, Minerva Nucleare 9 (1965) 93-98.

[12] CIGNA, A., MARIANI, A., SPADONI, M.A., TOMASSI, G., Research on radiocontamination transport in food chains: ^{90}Sr in Parmigiano-Reggiano cheese during 1957 to 1963, Nucl. Sci. 23 (1969) 297.

[13] DUBROVINA, S.V., BELOVA, O.M., Transfer of strontium-90 from milk into several milk products by different technological processes, Gig. Sanit. 28 (1963) 105-108 (in Russian).

[14] EASTERLY, D.G., BROOKS, J.R., HASUIKE, J.K., WEAVER, C.L., Development of Ion Exchange Processes for the Removal of Radionuclides from Milk, Techn. Report RO/EERL 71-1, US Environmental Protection Agency (1971).

[15] FOOKS, J.H., TERRILL, J.G., HEINEMANN, B.H., BALDI, E.J., WALTER, H.E., Evaluation of full scale strontium removal system for fluid milk, Health Phys. 13 (1967) 279-286.

[16] FRIFDLI, C., GEERING, J.J., LERCH, P., 'Strontium-90 measurements in Switzerland', Impact des Accidents d'Origine Nucléaire sur l'Environnement, (4e Symp. Int. de Radioécologie de Cadarache, 1988) Tome 2, CEN/Cadarache (1988).

[17] KANDARAKIS, J.K., ANIFANTAKIS, E.M., Distribution of ^{131}I, ^{134}Cs and ^{137}Cs in Ewes Milk during Processing into Different Products, Deltio-Ethnikes Epitropes Galaktos 3, (1986) 20-23 (in Greek).

[18] KANKARE, V., ANTILA, V., PAHKALA, E., RANTAVAARA, A., PAAKKOLA, O., Cesium 137:N Ja 134:N Siirtyminen Maidon Fraktioihin. Posteri Eleintarvikepaivilla, Food Congress, Helsinki (1987).

[19] KANKARE, V., ANTILA, V., PAHKALA, E., RANTAVAARA, A., PAAKKOLA, O., Transfer of Cesium-137 and Cesium-134 into Milk Fractions, Kemia-Kemi 14 (1987).

[20] KIRCHMANN, R., ADAM, V., VAN PUYMBROECK, S., 'Radiocontamination des dérivés du lait de vache', Radioisotopes and Radiation in Dairy Science Technology (Proc. Sem. Vienna, 1966), IAEA, Vienna (1966) 189-201.

[21] LAGONI, H., Dekontamination von Milchprodukten mit Hilfe von Molkereitechnischen Verfahren, Strahlenschutz 86 (1965).

[22] LENGEMANN, E.W., Distribution of radiostrontium and radiocesium in milk and milk products, J. Dairy Sci. 45 (1962) 538-539.

[23] MARSHALL, R.O., SPARLING, E.M., HEINEMANN, B., BALES, R.E., Large scale fixed bed ion-exchange system for removing iodine-131 and strontium-90 from milk, J. Dairy Sci. 51 (1965) 673-678.

[24] McENRI, C., MITCHELL, P.I., CUNNINGHAM, J.D., An experimental study of the transfer of radiocesium from raw whole milk to a wide range of milk products produced by the Irish dairy industry, Nuclear Energy Board Interim Report (1988) 1-28.

[25] KERKHOF-MOGOT, M.F., VERINGA, H.A., MOL, J.J., The processing of whole milk powder from milk contaminated with ^{131}I and its consequences for the environment, Health Phys. 47 (1984) 644-648.

[26] OBINO, A.M., Radiocontamination de l'homme par la chaine alimentaire: le cas des produits laitiers dans les différentes régions de la CEE en 1977, CEPN EURATOM SC/14-099-78-P.S.A.F. (1979) 49-51.

[27] OMOMO, Y., TSUGO, T., Distribution of Radioactive Strontium and Cesium in Milk, J. Agric. Chem. Soc. (Japan) 37 (1963) 725-728 (in Japanese).

[28] PIRHONEN, T., UUSI-RAUVA, E., RANTAVAARA, A.N., RAURAMAA, A., The radioactivity of milk and milk products in Finland, Meijeritieteellinen Aikakauskirja 45 (1987) 62-75.

[29] RANTAVAARA, A.H., 'Transfer of radionuclides during processing and preparation of foods; Finnish studies since 1986', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 69-94.

[30] RAYMOND, A.E., WILLIAMS, G.W., Effect of the evaporation and powdering processes on the iodine-131 content of milk, Radiol. Health Data Rep. 5 (1964) 70-72.

[31] REAVEY, T.C., Distribution of radionuclides in products from two milk processing plants, J. Environ. Health 27 (1965) 809-817.

[32] VRIGASOV, A., MARINOV, V., Distribution of strontium-90 and cesium-137 in dairy products in the processing of ewe's milk, Veterinarnomed. Nauki 5 (1968) 83-86 (in Russian).

[33] WALKER, J.P., EDMONDSON, F.L., Studies on ion exchange resins for the removal of radionuclides from milk, Health Phys. 16 (1969) 85-61.

[34] WALTER, H.E., SADLER, A.M., EASTERLY, D.G., EDMONDSON, L.F., Pilot plant fixed-bed ion exchange resin system for removing iodine-131 and radiostrontium from milk, J. Dairy Sci. 50 (1967) 1221-1225.

[35] WALTER, H.E., 'Pilot plant and commercial scale development of processes for removing radionuclides from milk', Radioisotopes and Radiation in Dairy Science Technology (Proc. Sem. Vienna, 1966), IAEA, Vienna (1966) 99-110.

[36] WHELLER, S.M., FLEET, G.H., ASHLEY, R.J., Effect of processing upon concentration and distribution of natural and iodophor-derived iodine in milk, J. Dairy Sci. 66 (1983) 187-195.

[37] WHITE, M.M., MOGHISSI, A.A., Transfer of ^{131}I from milk into cheese, Health Phys. 21 (1971) 116-118.

[38] WILSON, L.G., BOTTOMLEY, R.C., SUTTON, P.M., SISK, C.H., Transfer of radioactive contamination from milk to commercial dairy products, J. Soc. Dairy Technol. 41 (1988) 10-13.

[39] WOOD, G.M., WILSON, L.G., BOTTOMLEY, R.C., SUTTON, P.M., SISK, C.H., 'Transfer of radioactive contamination from milk to commercial dairy products', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 275-294.

[40] YASHCHENKO, V.F., Transfer of strontium-90 from milk into milk products, Veterinariya 8 (1969) 93-95 (in Russian).

[41] YASHCHENKO, V.F., Some regularities in the passage of ^{90}Sr from milk to milk products during commercial processing, Gigiena i Sanitariya 36 (1971) 108-109 (in Russian).

[42] BARTLEY, J.C., RABER, E.F., Effect of roasting on radiostrontium in fresh ham, J. Am. Diet. Assoc. 37 (1960) 466-467.

[43] BELL, M.C., BUESCHER, R.G., Effect of bone on Sr-89 and Ca-45 in beef roasts, J. Am. Diet. Assoc. 39 (1961) 567-568.

[44] BELOVA, O.M., Transfer of strontium-90 from meat to broth when applying different technological processes and treatments. Gigiena i Sanitariya 31 (1966) 111-112 (in Russian).

[45] BELOVA, O.M., DIBOBES, J.K., DUBROVINA, Z.V., Correlation of strontium-90 content in raw and thermally treated food products, Gigiena i Sanitariya 32 (1967) 40-42 (in Russian).

[46] DUBROVINA, Z.V., BELOVA, O.M., Changes in the strontium-90 content of food products after cooking, Gigiena i Sanitariya 29 (1964) 40-43 (in Russian).

[47] EYMAN, L.D., Changes in ^{137}Cs concentration in fish flesh during preparation for human consumption, Health Phys. 28 (1975) 475-477.

[48] GERNON, G.D., Removal of radiocesium from beef, Nature 203 (1964) 1189-1190.

[49] HALFORD, D.K., Effect of cooking on radionuclide concentration in waterfowl tissues, J. Environ. Radioact. 5 (1987) 229-233.

[50] HECHT, H., Dekontamination radioaktiv belasteten Wildbrets mittels Beizverfahren, Fleischwirtschaft 67 (1987) 250-256.

[51] HEMBRY, F.G., BELL, M.C., Sr^{90} phytate and Sr^{90} chloride in sheep and swine and roasting effect on Sr^{90} in meat, J. Anim. Sci. 25 (1966) 199-202.

[52] MEYER, B., FORRESTER, J., Effects of three cooking methods on cesium-134 content of beef from orally dosed steers, Food Technol. 16 (1962) 110-112.

[53] PAKULO, A.G., Changes in the Cs^{137} concentration of fish by culinary preparations, Gigiena i Sanitariya 36 (1971) 49-51 (in Russian).

[54] PAKULO, A.G., The influence of different food preparation procedures on the transfer of ^{137}cesium from crawfish to men, Gigiena i Sanitariya 46 (1981) 86-87 (in Russian).

[55] STEGER, U., BURGER, A., ZIEGLER, W., WALLNOEFER, P.R., Verteilung von Cs-134 und Cs-137 bei der küchentechnischen Verarbeitung verschiedener Lebensmittel, Dtsch. Lebensm.-Rundsch. 83 (1987) 85-88.

[56] WAHL, R., KALLEE, E., Decontamination puts mean in a pickle, Nature 323 (1986) 208.

[57] WOOD, G.M., CLARK, S.A., WILSON, L.G., SUTTON, P.M., 'The effect of processing on the radiocesium content of lamb'. Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 403-408.

[58] WORSECK, M., NIEPEL, J., KRüGER, I., Dekontamination von Fleisch. Monatsh. Veterinärmed. 25 (1970) 439-440.

[59] VAN GELDER, E., HURTGEN, G., KIRCHMANN, R., 'Etude expérimentale de la contamination résiduelle des aliments préparés à partir des fruits de mer', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 445-455.

[60] MASSON, M., GERMAIN, P., EYSSANTEER, B., GRAUBY, A. 'Evolution de la radioactivité de produits marins des côtes de la Manche au cours de la transformation agroalimentaire (algues) ou au cours de la préparation culinaire (moules)', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 457-468.

[61] BOVARD, P., BENARD, P., DELMAS, J., GRAUBY, A., 'Transfert des produits de fission dans la vigne et le vin' (C.R. Séances Acad. Agric. Fr.) (1968) 989-995.

[62] DELMAS, J., GRAUBY, A., 'Influence des technologies de transformation agroalimentaire sur la radioactivité des aliments', Foodstuffs Interv. Level Following Nucl. Accid. (Proc. Sem. Luxembourg), Commission of the European Communities, Luxembourg Eur. 11232 (1987) 183-197.

[63] ENDRES, O., FISCHER, E., Untersuchungen zur Dekontamination von Gemüse, Dtsch. Lebensm.-Rundsch. 65 (1969) 1-5.

[64] FEDERAL DRUGS ADMINISTRATION, Strontium-90 in food at intermediate stages of preparation for canning and freezing, Radiol. Health Data Rep. 5 (1964) 222-226.

[65] GRUETER, H., Radioactive fission product Cs-137 in mushrooms in West Germany during 1963-1970, Health Phys. 20 (1971) 655-656.

[66] HISAMATSU, S., TAKIZAWA, Y., ABE, T., Reduction of ^{131}I content in leafy vegetables and seaweed by cooking, J. Radiat. Res. 28 (1987) 135-140.

[67] LAUG, E.P., Temporal and geographical distributions of strontium-90 and cesium-137 in food, Radiol. Health Data Rep. (1963) 448-455.

[68] MURAMATSU, Y., UCHIDA, S., SUMIYA, M., YOSHIDA, S., OHMOMO, Y. 'Decontamination of radioiodine from waste and vegetables', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 341-349.

[69] PAULUS, K., Beurteilung pflanzlicher Lebensmittel nach Behandlungen zur Beseitigung der radioaktiven Kontamination, Z. Lebensmitt.-Untersuch. 142 (1969) 17-24.

[70] PAULUS, K., Dekontamination von mit ^{137}Cs and ^{85}Sr kontaminiertem Blattgemüse (Salat, Kohl, Spinat). Z. Lebensmitt.-Untersuch. 138 (1969) 329-334.

[71] PAULUS, K., Dekontamination von mit ^{137}Cs and ^{85}Sr kontaminiertem Blattgemüse (Salat, Kohl, Spinat). Z. Lebensmitt.-Untersuch. 139 (1969) 7-12.

[72] PAULUS, K., Dekontamination von pflanzlichen Lebensmitteln, insbesondere von Obst und Gemüse, sowie damit zusammenhängende Probleme. Kontamin. Decontamin. Lebensmitt. (Proc. Conf. Karlsruhe, 1974) 197-228.

[73] PAULUS, K., Die Dekontamination von mit radioaktiven Stoffen oberflächig verseuchtem Gemüse und Obst. Ind. Obst-u. Gemüseverwert. 53 (1968) 85-88.

[74] PAULUS, K., Zur Dekontamination von Rote Beete, Kohlrabi und Rosenkohl. Ind. Obst-Gemüseverwert. 53 (1968) 635-638.

[75] RANTAVAARA, A., Radioactivity of vegetables and mushrooms in Finland after the Chenobyl accident in 1986, Suppl. 4 to Annual report STUK-A55, Finnish Centre for Radiation and Nuclear Safety, Helsinki (1987).

[76] RALLS, J.W., MAAGDENBERG, H.J., GUCKEEN, T.R., MERCER, W.A., Removal of radioactive strontium and cesium from vegetables and fruits during preparation for preservation, J. Food Sci. 36 (1971) 653-656.

[77] RALLS, J.W. PRIMBSCH, S., GUCKEEN, T.R., MAAGDENBERG, H.J., RINEHART, J., LAMB, F.C., MERCER, W.A., Distribution of strontium and calcium in major vegetable and fruit crops and criteria for use of fallout-contaminated foods. Radiol. Health Data Rep. 8 (1967) 355-358.

[78] RALLS, J.W., MAAGDENBERG, H.J., GUCKEEN, T.R., MERCER, W.A., Removal of radioactive strontium and cesium from certain vegetables and fruits during normal preparation for preservation, Isot. Radiat. Technol. 6 (1969) 146-149.

[79] ROHLEDER, K., Ueber die radioaktive Dekontamination von Speisepilzen durch Blanchieren, Ind. Obst-Gemüseverwert. 52 (1967) 64-66.

[80] ROHLEDER, K., Untersuchungen über die Aufnahme radioaktiver Stoffe durch Grünkohl aus dem Boden und aus der Atmosphäre und Versuche zur Dekontamination, Z. Lebensmitt.-Untersuch. 149 (1972) 223-227.

[81] THOMPSON, J.C., HOWE, M., Retention and removal of ^{131}I from contaminated vegetables, Health Phys. 24 (1973) 345-351.

[82] THOMPSON, J.C., Sr removal in vegetables prepared for home consumption, Health Phys. 11 (1965) 136-137.

[83] WEAVER, C.M., HARRIS, N.D., Removal of radioactive strontium and cesium from vegetables during laboratory scale processing, J. Food Sci. 44 (1979) 1491-1493.

[84] WILKINS, B.T., BRADLEY, E.J., DODD, N.J., The effects of culinary preparation on radionuclides levels in vegetable foodstuffs, Radiat. Prot. Dosim. 20 (1987) 187-190.

[85] Werkdocumenten CCRX-rapport: De radioactieve besmetting in Nederland t.g.v. het kernongeval in Tsjernohyl, Part 1, Chapter 12, Ministry of Housing, Physical Planning and Environment, Leidschendam (1986).

[86] BRADLEY, E.J., POPPLEWELL, D.S., WILKINS, B.T., 'The influence of simple culinary preparation on the radionuclide content of vegetables', Radioactivity Transfer during Food Processing and

Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 311-324.

[87] PAULUS, K., Zur Dekontamination totalkontaminierter Kartoffeln, Z. Lebensmitt.-Untersuch. 139 (1969) 282-287.

[88] PERKINS, H.J., STRACHAN, J., Decontamination of potato tubers containing cesium-137, Science 144 (1964) 59-60.

[89] APOSTOLATOS, G. HADJIANTONIOU, A., 'Cesium contamination transfer in wheat products', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 377-400.

[90] ARAPIS, G., MARTI, J.M., KOUSKOUTOPOULOS A., KARANDINOS, M., IRANZO, E., 'Low radioactivity contribution of cereals to diet due to their specific composition and processing'. Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989).

[91] BUNZL, K., KRACKE, W., Soil to plant transfer of ^{239} + ^{240}Pu, ^{238}Pu, ^{241}Am, ^{137}Cs and ^{90}Sr from global fallout in flour and bran from wheat, rye, barley and oats, as obtained by field measurements, Sci. Total Environ. 63 (1987) 111-124.

[92] BUNZL, K., KRACKE, W., Transfer of ^{137}Cs and ^{90}Sr to flour, bran and straw from wheat, rye, barley and oats during the years 1982, 1986 (reactor accident at Chernobyl) and 1987 in field measurements, Z. Lebensmitt.-Untersuch. (in press).

[93] LIU, D.J., ROBBINS, G.S., POMERANZ, Y., Composition and utilization of milled barley products, Cereal Chem. 51 (1974) 309-316.

[94] LOFTI, M., NOTARO, M., AZIMI-GARAKANI, D., PIERMATTEI, S., TOMMASINO, L., Loss of radioactivity in cooked spaghetti, Sci. Total Environ. 79 (1989) 291-293.

[95] MUELLER, H., Radioaktivitäts-Kontamination von Brotgetreide nach Tschernobyl, Mühle-Mischfuttertech. 123 (1986) 392-393.

[96] OHMOMO, Y., SUMIYA, M., UCHIDA, S., MURAMATSU, Y., YOKOSUKA, S., OBATA, H., YAMAGUCHI, S., 'Transfer of radioiodine into rice grains', Impact des Accidents d'Origine Nucléaire sur l'Environnement, (4e Symp. Int. de Radioécologie de Cadarache, 1988) Tome 1, CEN/Cadarache (1988).

[97] PFEIFER, V.F., PEPLINSKI, A.J., HUBBARD, J.E., Strontium-90 in plant parts and milling fractions from a 1963 Illinois wheat, Radiol. Health Data Rep. 5 (1964) 283-284.

[98] RIVERA, J., Sr-90 in U.S. wheat and flour produced in 1962 and predictions of levels in the 1963 crop, Fallout Program Quarterly Summary Report HASL-140, Office of Technical Services, Department of Commerce, Washington, D.C. (1964) 276-282.

[99] VOIGT, G., MUELLER, H., PROEHL, G., PARETZKE, H.G., 'Cesium activity distribution in cereals after milling processes', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem.

Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 351-360.

[100] CASTRO, J., SANCHO, A., VEIGA, E., DIAS YUBERO, J., MURIAS, B.S.F., 'Transfert de Cs-134 et Cs-137 dans les infusions d'herboristerie', Radioactivity Transfer during Food Processing and Culinary Preparation' (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 205-218.

[101] GEDIKOGLU, A., SIPAHI, B.L., OSKAY, T., KURUGLU, M., Chernobyl radioactivity in Turkey, Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 193-204.

[102] MASSCHELEIN, W.J., 'Elimination d'isotopes radioactifs par les filières conventionnelles de traitement des eaux potables', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 167-192.

[103] MIRIBEL, J., DELMAS, J., 'Utilisation de traceurs activables pour l' étude de devenir de la contamination radioactive au cours de la vinification', Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 219-227.

[104] GIESE, W., SCHIMANSKY, K., KLUGE, K., ROINER, F., Radiocesium transfer to whey and whey products: whey decontamination on an industrial scale, Radioactivity Transfer during Food Processing and Culinary Preparation (Proc. Sem. Cadarache, 1989), Commission of the European Communities, Luxembourg (1989) 295-308.

Chapter 4

SEASONALITY

A. Aarkrog
Risø National Laboratory,
Roskilde, Denmark

4.1. Introduction

4.1.1. Seasonality

The impact of environmental releases of radionuclides on the human food chain depends, among other things, on the time of the year when the release occurs. The term seasonality, as used in this report, means the varying response to radioactive contamination of environmental samples according to the time of the year, when the contamination occurs. Seasonality effects may be masked, enhanced or confused by other seasonal changes and to assist in clarification a seasonality factor is introduced. The seasonality factor, S, allows seasonality to be quantified; it is defined in Annex 1. When $S > 0$ seasonality is present and when $S = 0$, no seasonality is observed. Annex 1 also contains an example of the application of S to seasonality in agricultural crops.

Agricultural ecosystems

The most obvious effects of seasonality are seen in relation to direct deposition on the surface of crops. When the contamination has reached the ground, and indirect contamination (root uptake) becomes the dominant pathway, seasonality is usually no longer relevant. Seasonality is of special importance in connection with the contamination of cereals. In general it may be said that a deposition event during winter, when the fields lie fallow and the domestic animals are stabled, will result in a significantly lower radiological impact than if a similar contamination were to take place in the summer before harvest. Seasonality is generally of greater importance in temperate regions than in the subtropics where soils may be cultivated throughout the year.

Seminatural ecosystems

Natural and seminatural ecosystems e.g. forests may also show seasonality. Decidious trees contaminated during the winter will retain

less activity than in case of a summer contamination, when the leaves will retain more of the activity from the air. Hence the forest will be most contaminated if the contamination occurs in the summer. However the effect on the forest floor will be delayed until leaf fall, which is not the case for a contamination during the winter. Coniferous trees will show less seasonality than decidious trees. But a delay of the contamination of the forest floor will also be seen here.

Aquatic ecosystems

In general, seasonality has not been considered in the context of aquatic pathways. However, in the case of freshwater fish living in a frozen lake when the contamination occurs, there will be a delayed and lesser response to the contamination compared with an ice-free lake. There might also be some influence due to plankton. If it is assumed that the contamination occurs during a plankton bloom, then much of the activity may be captured and enter the food-chain of the fish. However, if the radioactive debris is not captured by plankton, it will reach the bottom sediments becoming more or less unavailable to the biota in the lake and the infinite time integral of activity in the fish will be lower than in the case where there is a plankton bloom. Contamination of drinking water derived from groundwater will show no seasonality. However, drinking water coming from surface water may (as discussed above for fish) show a seasonality, due either to plankton bloom or to ice. Radioactivity in drinking water may also be influenced by snow melting. In this case, when the snow melts, contamination of a snow-covered area may affect drinking water derived from a river receiving the meltwater. Contamination with long lived radionuclides during the summer would probably have fewer implications because the activity then would more easily be retained by the soil.

4.1.2. Seasonal variability

Seasonality should not be confused with "seasonal variability". Foodproducts produced throughout the year, e.g. milk, may show seasonal variability which occurs even though the radioactive deposition is evenly distributed throughout the year. Thus, ^{137}Cs concentrations may be seen to be significantly higher in summer milk than winter milk, if the cows are grazing in summer, but fed with beets during winter (Fig. 1). Permanent pastures may also show a seasonal variation in their radionuclide

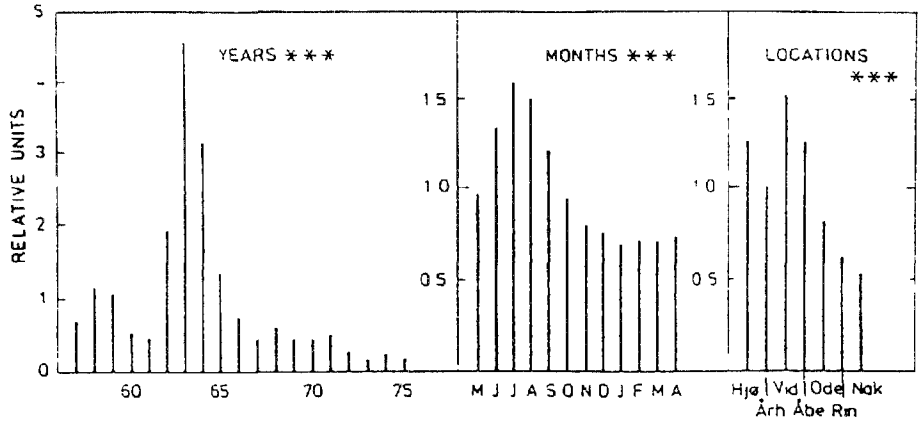


Fig. 1. The variation of pCi Cs-137 $(g\ K)^{-1}$ in Danish dried milk collected monthly at 7 dried milk factories from May 1959 to April 1976. The bars indicate the levels relative to the grand mean 18.5 pCi $(g\ K)^{-1}$ (= 1 at the relative scales) [41].

concentrations due to variations in the uptake and loss of radionuclides throughout the growing season. In the case where only seasonal variability is present with no seasonality, the seasonality factor, S is zero.

Agricultural practice

Agricultural practice may have seasonal aspects. Ploughing, harrowing or harvesting giving rise to resuspension may, for example, contaminate neighbouring fields. In case of an evenly distributed fallout rate throughout the year this resuspension will enhance the contamination of crops in neighbouring fields. Variations in crop contamination from this phenomenon belongs to seasonal variability. If the deposition of the activity is unevenly distributed in time and if the resuspension from treatment of neighbouring fields occurs prior to the contamination only a minor additional effect from resuspension will be seen. If however the treatment of the fields takes place after the contamination event a more pronounced additional contamination is expected. The seasonality effect is then either diminished (if contamination took place in winter season) or enhanced (if the contamination occurs in summer). This illustrates how seasonal variability may influence seasonality.

Seminatural ecosystems

Karlèn et al.[1] have shown seasonal variability in the ^{137}Cs content of roe-deer in Sweden. A peak is observed in August-September due to a high consumption of mushrooms in these months by the roe-deer.

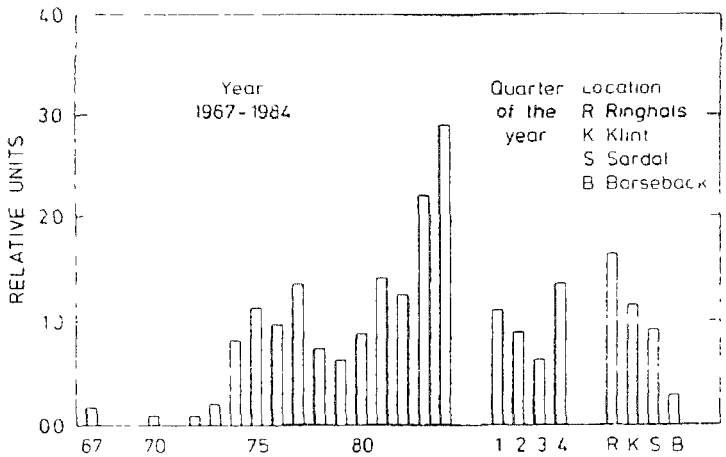


Fig. 2. Time variation of Tc-99 in Fucus in the Danish Straits 1967-1984 [42].

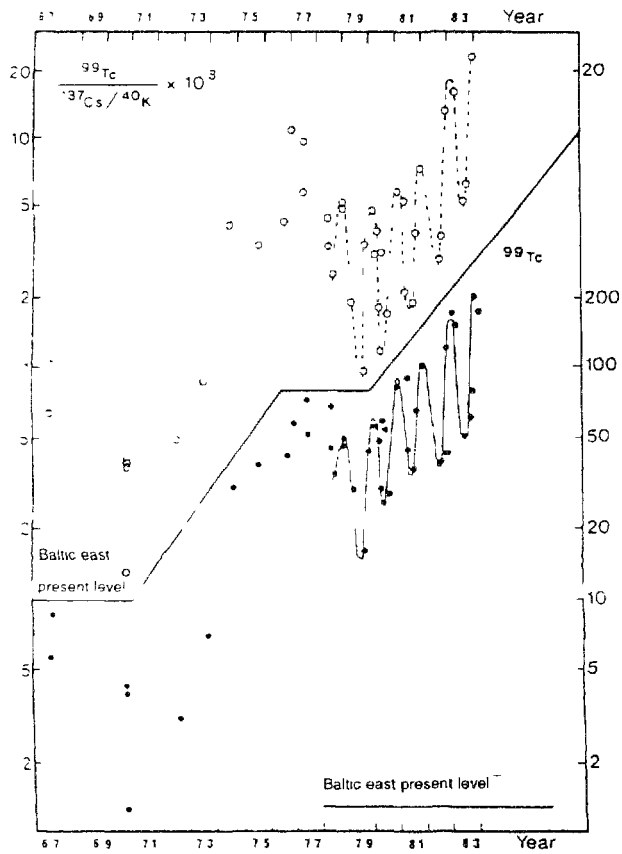


Fig. 3. Activity concentration of Tc-99 in Fucus at the west coast of Sweden (56.76°N, 12.63°E) during 1967-84. In the upper part of the figure the Tc-99/(Cs-137/K-40) ratio is given [43].

Marine environment

Seasonal variability is also observed in the marine environment, where e.g. ^{99}Tc (Fig. 2) and ^{137}Cs (Fig. 3) in brown algae (Fucus vesiculosus) from the Baltic area show marked seasonal variations. The winter values for ^{99}Tc are 2-3 times higher than in the summer, however, for ^{137}Cs, the summer values are 1.5 times higher than those in the winter. These seasonal variations are primarily influenced by physiological factors, but seasonal variations in the oceanographic conditions may also play a role.

Global fallout

Environmental contamination due to global fallout from nuclear weapons testing in the atmosphere has shown a pronounced seasonal variation with relatively low fallout rates during autumn and winter and a fallout peak in spring (Fig. 4). This seasonal pattern influences the levels in environmental samples and thus gives rise to seasonality. However, natural as well as anthropogenic factors may, as it has been shown above, superimpose on this seasonality.

4.2. Pre-Chernobyl Data and Models

4.2.1. Experimental studies

Middletons experiments

The first studies on seasonality were carried out by Middleton [2] in the late fifties. He studied the direct contamination of wheat, potato, cabbage and sugar beet after the application of ^{89}Sr and ^{137}Cs in a fine spray at different stages in the growth of the plants. He found (Tables I-III) that the extent to which ^{137}Cs was concentrated in the edible tissues greatly exceeded that of ^{89}Sr. Usually the concentration of ^{89}Sr or ^{137}Cs in each crop varied by less than an order of magnitude irrespective of the state of growth of the crop at the time of contamination. The ^{89}Sr content of wheat grain, however, was considerably lower when the plants were contaminated before the ears emerged. In later experiments, Middleton and Squire [3] observed that the maximum concentrations of ^{89}Sr and ^{137}Cs in the cabbage hearts and ^{137}Cs in potato tubers were observed after contamination occurred in the middle of the season.

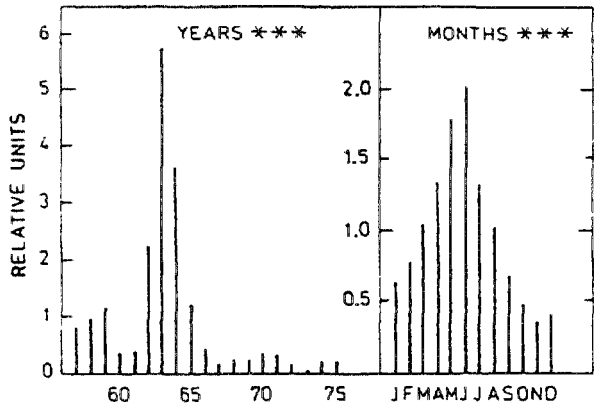


Fig. 4. The annual and monthly variation of Sr-90 in ground-level air collected at Risø 1957-1975. The bars indicate the air concentrations relative to the grand mean: 5.31 fCi m^{-3} (= 1 at the relative scales) [41].

TABLE I. FOLIAR CONTAMINATION OF WHEAT WITH ^{89}Sr AND ^{137}Cs [2]

Experimental data			Extrapolation to field conditions			
	Days from planting to contamination	Content of grain as percentage initial retention by plant	Initial retention on foliage		Final content in grain	
			μc/m²	μc/kg dry wt	μc/m²	μc/kg dry wt
(a) ^{89}Sr						
1957	100	0·04	330	1044	0·14	0·6
		1·62				
	105	1·07	259	563	2·88	11·2
		2·96				
	149	0·74	273	339	2·00	7·8
		2·82				
	Sig. diff.	*0·53*				
1958	86	0·12	400 ± 64	883	0·48	1·5
		0·90				
	107	1·26	470 ± 68	627	5·92	18·7
		2·08				
	107†	0·15	—			—
		1·73				
	136	1·79	540 ± 68	461	9·68	30·6
		2·27				
	Sig. diff	*0·27*				
(b) ^{137}Cs						
1957	100	8·8	204	499	17·9	58·4
	105	8·2	301	576	24·6	80·3
	149	3·2	289	325	9·2	30·1
	Sig. diff.	3·3				
1958	86	5·2	446 ± 84	986	23·2	58·8
	107	10·2	523 ± 41	559	53·3	135·3
	107†	5·4	—		—	
	136	6·8	585 ± 52	432	39·7	100·7
	Sig. diff.	2·3				

† Ears protected from spray.

TABLE II. FOLIAR CONTAMINATION OF CABBAGE WITH ^{89}Sr AND ^{137}Cs [2]

Experimental data			Extrapolation to field conditions			
Days from planting to contamination		Content of heart as percentage initial retention by plant	Initial retention on foliage		Final content in heart	
			μc/m²	μc/kg dry wt.	μc/m²	μc/kg dry wt
(a) ^{89}Sr						
1957	32	0·33	81	3,000	0·27	0·84
	88	1·04	86	688	0·89	2·82
	Sig. diff.	N.S.				
1958	58	0·18	85 ± 9	2,180	0·15	0·27
	72	0·34	292 ± 26	3,073	0·99	1·83
	87	0·16	309 ± 40	1,119	0·50	0·91
	99	0·11	281 ± 16	1,227	0·31	0·56
	Sig. diff.	0·15				
(b) ^{137}Cs						
1957	32	0·91	42	1,720	0·4	1·3
	88	1·51	87	617	1·3	4·3
	Sig. diff.	N.S.				
1958	58	1·15	141 ± 26	2,850	1·6	3·6
		1·02				
	72	6·93	281 ± 20	2,754	19·5	43·4
		1·80				
	87	4·92	309 ± 36	1,240	15·2	33·7
		1·62				
	99	3·92	280 ± 44	1,143	10·7	23·8
		1·61				
	Sig. diff.	*0·28*				

The final concentration in the heart as a percentage of the amount initially retained by the plant was experimentally determined; the values for initial retention and final content of the heart under field conditions after a deposition of 1 millicurie per square metre have been calculated.

Final sampling 120 and 113 days after planting in 1957 and 1958 respectively.

Where a logarithmic transformation was necessary for statistical analysis the transformed values are in italics. Significant differences are shown at the 5 per cent level.

Aarkrog's grain studies

In the sixties and early seventies, Aarkrog [4–7] studied the direct contamination of cereal grain (rye, barley, wheat and oats) applying radioisotopes of Sr, Cs, Mn, Ce, Cr, Fe, Co, Zn, Hg, Pb, Sb, Ru, Be, Na, Cd and Ba as a fine spray during different stages in the growth of the crops. The results of these studies with respect to seasonality were summarized in a number of heuristic models [8] (Figs. 5 and 6).

It appears that two important factors influence the contamination of grain: initial retention and translocation from the vegetative parts of the plants to the seeds. Initial retention is fairly independent of which radionuclide is involved, whereas translocation depends strongly on the radioelement.

TABLE III. FOLIAR CONTAMINATION OF POTATO WITH ^{89}Sr AND ^{137}Cs [2]

Experimental data			Extrapolation to field conditions			
Days from planting to contamination		Content of tuber as percentage initial retention on plant	Initial retention foliage		Final content in tubers	
			μc/m²	μc/kg dry wt	μc/m²	μc/kg dry wt
(a) ^{89}Sr						
1957	55	0 046	480	9,600	0 22	0 094
	69	0 038	650	8,666	0 25	0·11
	Sig diff.	N S				
1958	47	0 14	270	2,570	0·38	0·13
	74	0·03	320	2,602	0 096	0 034
	97	0 02	580	4,461	0 12	0·041
	Sig. diff	N.S.				
(b) ^{137}Cs						
1957	55	25	480	6 760	120	49
	69	29	520	6 753	151	62
	Sig diff.	N S				
1958	47	45	240	2.400	108	32
		1 63				
	74	14	430	3,435	63	19
		1 09				
	97	21	390	4 193	82	24
		1 22				
	Sig diff	*0 28*				

The final concentration in the tuber as a percentage of the amount initially retained by the plant was experimentally determined the values for initial retention and final content of the tuber under field conditions after a deposition of 1 millicurie per square metre have been calculated.

Final sampling 125 and 129 days after planting in 1957 and 1958 respectively.

Where a logarithmic transformation was necessary for statistical analysis the transformed values are shown in italics. Significant differences are shown at the 5 per cent level.

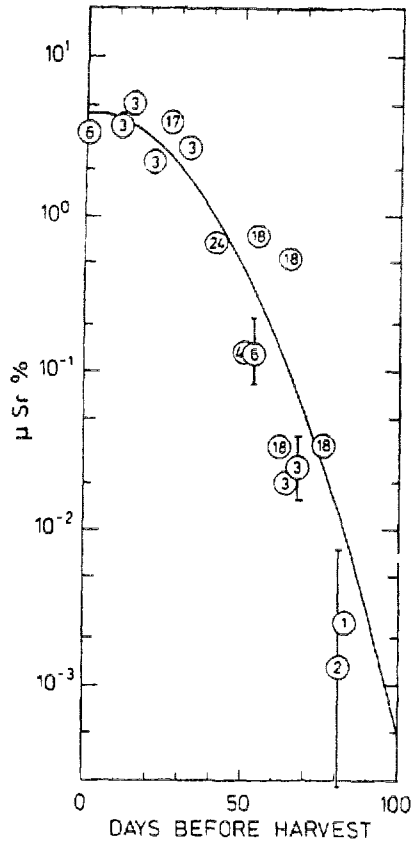


Fig. 5. Percentage of radiostrontium, applied at various dates to 1 m² barley field, recovered per kg of mature grain at harvest assuming no radioactive decay. Number of determinations and 1 S.E. are shown (unless S.E. is less than the radius of the circle). Curve calculated from $\mu(t)=4.5\times10^{-2}\ e^{-0.00095(t-2)^2}$ [8].

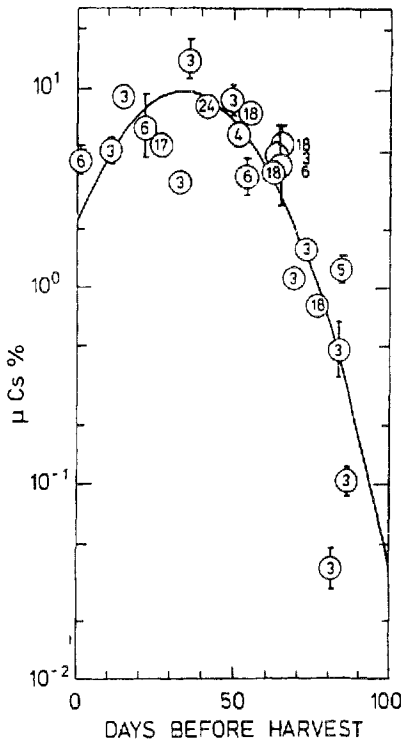


Fig. 6. Percentage of radiocaesium, applied at various dates to 1 m^2 barley field, recovered per kg of mature grain at harvest assuming no radioactive decay. (Number of determinations and 1 S.E. are shown.) Curve calculated from $\mu(t)=9.8\times10^{-2}\ e^{-0.0013(t-34)^2}$ [8].

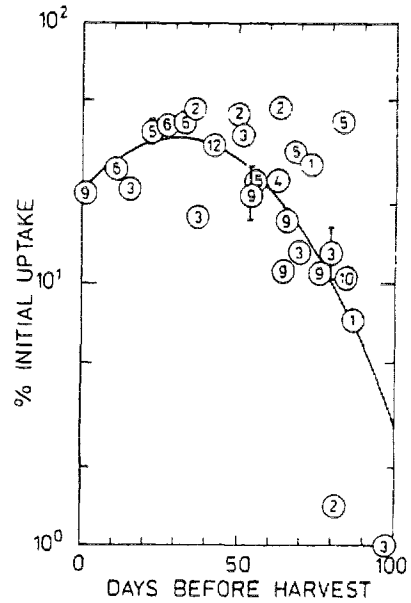


Fig. 7. Percentage of initial uptake in whole crops (above-ground parts) when a barley field at various dates is contaminated by radionuclides. The total crop yield at harvests in 0.8 kg dry matter m^{-2}; data are based upon measurements with radioisotopes of Sr, Cs, Co, Mn, Sb, Zn and Fe; number of determinations and 1 S.E. are indicated. Curve calculated from % initial uptake = $36\ e^{-0.00052(t-30)^2}$ [8].

The percentage of initial uptake 3 months before harvest is about 5% of the activity deposited over a field with barley crops, and 1 month before harvest it reaches a maximum of 36% (Fig. 7). If the field is contaminated with, for example ^{137}Cs, 3 months before harvest (1 kBq m^{-2}), the mature grain contains 2 Bq $^{137}CS\ kg^{-1}$; if the contamination occurs 1 month before harvest, 100 Bq $^{137}Cs\ kg^{-1}$ is found in the mature grain (Fig. 6). In the case of ^{90}Sr, the corresponding concentrations would have been 0 Bq kg^{-1} and 20 Bq kg^{-1}, respectively, demonstrating that ^{137}Cs is translocated to a much greater extent than ^{90}Sr (Fig. 5).

The highest levels in grain are expected when the contamination takes place in the final month before harvest. The lowest levels will be seen if the fields are contaminated before sprouting.

Delmass' fruitstudies

Delmas et al. [9] have studied experimentally the contamination of orange fruits with ^{90}Sr and ^{137}Cs. For the same initial direct contamination with the two radionuclides during the flowering of the orange trees, the ^{137}Cs contamination of the fruit flesh was an order of magnitude higher than that of ^{90}Sr. The difference was due to the higher translocation of ^{137}Cs than of ^{90}Sr to the flesh. If the contamination were to have occurred just prior to the orange harvest, the difference would have been far less.

4.2.2. Models

At a CEC seminar in Dublin in 1983 on the transfer of radioactive materials in the terrestrial environment subsequent to an accidental release to the atmosphere, a number of papers dealt with seasonality viz [10-14]. The consequences of seasonality were illustrated by model calculations.

Simmonds' model

The paper by Simmonds [13] concluded that for green vegetables and meat from sheep, season of the year has little effect, but for grain (Table IV) and milk (Fig. 8) or meat from cattle there can be large variations in concentration. Such variations are greater for short-lived radionuclides such as ^{131}I than for the longer-lived ^{137}Cs (Fig. 9) and ^{90}Sr. The variations in the total activity in the diet following

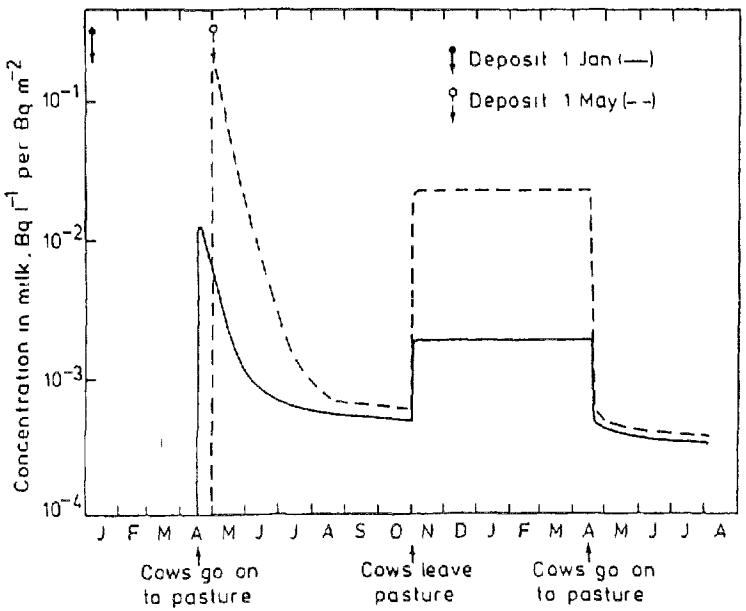


Fig. 8. The concentration of Cs-137 in milk as a function of time following an accidental release at two different times of year [13].

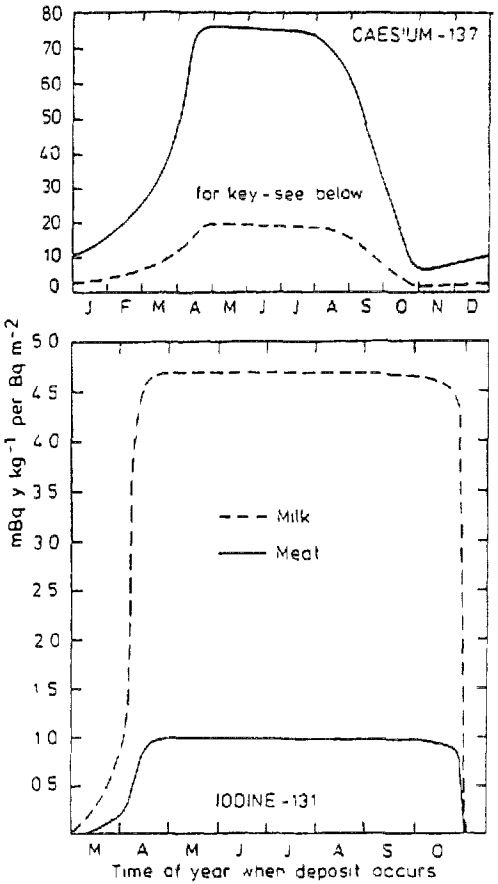


Fig. 9. Integrated concentrations in milk and meat following a single deposit of Cs-137 and I-131 at various times of the year [13].

TABLE IV. TIME INTEGRAL OF CONCENTRATION IN GRAIN PRODUCTS FOLLOWING A SINGLE DEPOSIT AT VARIOUS TIMES BEFORE HARVEST [13]

Time of deposit before harvest / Nuclide	Time integral of concentration[a] (Bq y kg^{-1} per Bq m^{-2})				
	7d	30d	60d	90d	Winter[b]
Sr-90	$1.7\ 10^{-2}$	$1.6\ 10^{-2}$	$4.5\ 10^{-3}$	$2.0\ 10^{-3}$	$1.4\ 10^{-3}$
Ru-106	$1.7\ 10^{-3}$	$6.8\ 10^{-4}$	$3.0\ 10^{-4}$	$2.1\ 10^{-4}$	$1.2\ 10^{-4}$
I-131	$4.8\ 10^{-4}$	$2.5\ 10^{-4}$	$1.1\ 10^{-5}$	$2.8\ 10^{-7}$	0.0
Cs-134	$2.3\ 10^{-2}$	$8.5\ 10^{-2}$	$4.8\ 10^{-2}$	$1.6\ 10^{-2}$	$3.1\ 10^{-5}$
Cs-137	$2.8\ 10^{-2}$	$1.0\ 10^{-1}$	$5.9\ 10^{-2}$	$2.0\ 10^{-2}$	$4.4\ 10^{-4}$

Notes

a) The integral of concentration in processed grain, integrated over all time.

b) Deposition occurs when the fields are fallow before ploughing and seeding.

TABLE V. TOTAL INTAKES VIA FOOD FOLLOWING A SINGLE DEPOSIT AT VARIOUS TIMES OF THE YEAR [13]

Time of deposit / Nuclide	Total intake via food[a] (Bq per Bq m^{-2})			
	January	April	August	October
I-131	0.17	0.42	1.7	1.6
Cs-137	4.8	9.9	27	7.9
Sr-90	4.9	5.3	7.5	5.0

Note

a) The total intake of activity from all food at a critical group intake rate integrated over all time

deposits at different times of the year are generally smaller than for individual foods, but are again greater for ^{131}I than for longer-lived radionuclides (Table V).

Müller's model

Müller et al. [10] shows in a model calculation (Fig. 10) that a contamination with ^{137}Cs in May (in Germany) results in an integrated intake from food which is about 60% of that seen if a similar contamination were to take place in June, but 5 times higher than if the contamination were to occur in winter (November-March).

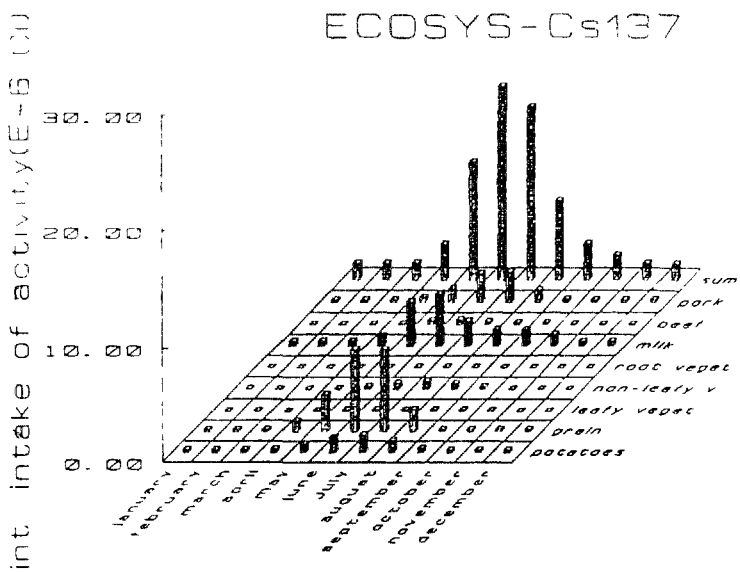


Fig. 10. Integral intake of activity by man over 50 years after deposition of 1 Ci/km^2 of Cs-137 as function of month of release. The total intake ("sum") is split up into the different foodstuffs [10].

AGRID

In a Finnish model study [15] it was found that the expected values of collective 30-year doses in the case of dormant season and growing season seem to differ by a factor on the order of 10. In the first-year doses, the difference would be larger, a factor of about 100. The Finnish AGRID model includes seasonal effects by considering growing season and dormant season separately. But the model makes no seasonal distinctions within the growing season. Table VI shows the variation in the doses according to the AGRID model.

PATHWAY

The PATHWAY model [16-18] was developed to estimate radionuclide ingestion by humans exposed to fallout originating at the Nevada Test Site (NTS) during the 1950s. As NTS is an arid region PATHWAY does not consider wet deposition. The model considers seasonal changes in the biomass of vegetation and animal diets, as well as specific plowing and crop harvest dates; thus the integrated radionuclide intakes by humans are dependent on the seasonal timing of deposition. The model does not consider translocation. Table VII shows the PATHWAY model calculations for selected

TABLE VI CONDITIONAL EXPECTED VALUES OF THE COLLECTIVE EFFECTIVE DOSES (man Sv) DUE TO THE FK-2 RELEASE CATEGORY [15]

	Release during		On an average during the whole year
	The growing season	The dormant season	
Plume γ			$1.7 \cdot 10^{2}$
Fallout γ			$1.8 \cdot 10^{1}$
Inhalation			$2.1 \cdot 10^{2}$
Milk	$9.8 \cdot 10^{3}$	$8.0 \cdot 10^{2}$	$3.3 \cdot 10^{3}$
Meat	$2.6 \cdot 10^{3}$	$4.7 \cdot 10^{2}$	$1.1 \cdot 10^{3}$
Vegetables	$1.0 \cdot 10^{3}$	$5.8 \cdot 10^{1}$	$2.1 \cdot 10^{2}$
Grain	$4.0 \cdot 10^{3}$	$1.8 \cdot 10^{2}$	$8.1 \cdot 10^{2}$
Roots	$5.0 \cdot 10^{1}$	$4.1 \cdot 10^{1}$	$4.3 \cdot 10^{1}$
Nutrition total	$1.7 \cdot 10^{4}$	$1.5 \cdot 10^{3}$	[$5.4 \cdot 10^{3}$]
Total			$7.5 \cdot 10^{3}$

TABLE VII INTEGRATED INTAKE BY ADULT MALES[a] OF VARIOUS RADIONUCLIDES FROM ALL FOODS PER UNIT FALLOUT DEPOSITION VERSUS SELECTED FALLOUT DATES [17]
(units are Bq per Bq m^{-2})

Radio-nuclide	1 Mar (Tesla)	17 Mar (Annie)	25 Apr (Simon)	19 May (Harry)	24 Jun (Priscilla)	24 Jul (Kepler)	31 Aug (Smoky)	7 Oct (Morgan)
^{140}Ba	1.6×10^{-2}	1.6×10^{-2}	2.2×10^{-2}	2.6×10^{-2}	4.5×10^{-2}	2.7×10^{-2}	1.5×10^{-2}	1.0×10^{-2}
^{143}Ce	1.1×10^{-3}	4.2×10^{-4}	1.0×10^{-3}	1.5×10^{-3}	2.4×10^{-3}	2.4×10^{-3}	1.2×10^{-3}	6.2×10^{-4}
^{144}Ce	3.7×10^{-2}	5.5×10^{-2}	1.5×10^{-1}	2.0×10^{-1}	5.3×10^{-1}	5.5×10^{-2}	4.2×10^{-2}	4.3×10^{-2}
^{136}Cs	1.0×10^{-1}	1.1×10^{-1}	1.2×10^{-1}	1.4×10^{-1}	1.9×10^{-1}	1.6×10^{-1}	1.1×10^{-1}	1.2×10^{-1}
^{137}Cs	2.0	2.0	3.0	2.6	3.3	1.4	1.6	2.4
^{131}I	2.1×10^{-2}	2.0×10^{-2}	3.3×10^{-2}	4.9×10^{-2}	9.1×10^{-2}	1.0×10^{-1}	5.5×10^{-2}	3.7×10^{-2}
^{133}I	4.2×10^{-4}	1.5×10^{-4}	3.7×10^{-4}	1.1×10^{-3}	2.0×10^{-3}	2.9×10^{-3}	1.5×10^{-3}	7.5×10^{-4}
^{135}I	7.8×10^{-6}	2.6×10^{-6}	6.9×10^{-6}	1.5×10^{-5}	2.6×10^{-5}	3.4×10^{-5}	1.8×10^{-5}	8.7×10^{-6}
^{99}Mo	3.7×10^{-3}	1.9×10^{-3}	3.9×10^{-3}	5.5×10^{-3}	9.2×10^{-3}	1.0×10^{-2}	5.3×10^{-3}	3.0×10^{-3}
^{147}Nd	1.3×10^{-2}	1.3×10^{-2}	1.8×10^{-2}	2.0×10^{-2}	3.4×10^{-2}	2.0×10^{-2}	1.2×10^{-2}	7.3×10^{-3}
^{239}Np	2.7×10^{-3}	1.1×10^{-3}	2.6×10^{-3}	3.4×10^{-3}	5.4×10^{-3}	5.2×10^{-3}	2.7×10^{-3}	1.4×10^{-3}
^{239}Pu	4.0×10^{-2}	6.0×10^{-2}	2.1×10^{-1}	2.9×10^{-1}	8.1×10^{-1}	6.0×10^{-2}	4.7×10^{-2}	4.9×10^{-2}
^{105}Rh	1.3×10^{-3}	4.8×10^{-4}	1.2×10^{-3}	1.7×10^{-3}	2.7×10^{-3}	2.8×10^{-3}	1.4×10^{-3}	7.2×10^{-4}
^{103}Ru	6.1×10^{-2}	7.4×10^{-2}	8.2×10^{-2}	9.0×10^{-2}	1.6×10^{-1}	5.9×10^{-2}	5.2×10^{-2}	6.9×10^{-2}
^{106}Ru	1.5×10^{-1}	1.8×10^{-1}	2.6×10^{-1}	3.1×10^{-1}	6.8×10^{-1}	1.4×10^{-1}	1.6×10^{-1}	2.3×10^{-1}
^{89}Sr	5.8×10^{-2}	6.6×10^{-2}	1.1×10^{-1}	1.3×10^{-1}	2.4×10^{-1}	1.0×10^{-1}	6.5×10^{-2}	6.1×10^{-2}
^{90}Sr	2.5×10^{-1}	2.4×10^{-1}	6.0×10^{-1}	6.0×10^{-1}	1.1	2.0×10^{-1}	1.6×10^{-1}	1.8×10^{-1}
^{91}Sr	4.5×10^{-5}	1.6×10^{-5}	4.0×10^{-5}	6.9×10^{-5}	1.1×10^{-4}	1.2×10^{-4}	6.0×10^{-5}	3.0×10^{-5}
^{132}Te	4.5×10^{-3}	2.4×10^{-3}	4.6×10^{-3}	5.7×10^{-3}	9.0×10^{-3}	8.4×10^{-3}	4.5×10^{-3}	2.6×10^{-3}
^{97}Zr	2.6×10^{-4}	9.1×10^{-5}	2.3×10^{-4}	3.7×10^{-4}	5.8×10^{-4}	6.0×10^{-4}	3.0×10^{-4}	1.5×10^{-4}

[a] Values for adult females average 35% less than males

fallout dates. The ^{137}Cs intake varies e.g. by a factor of 2.3 in the period March-October, while ^{90}Sr varies by a factor of nearly 7.

4.3. Post-Chernobyl Data and Models

4.3.1. Observations

In general, the Chernobyl accident confirmed qualitatively the assumptions on seasonality made prior to the accident.

European ^{131}I and ^{137}Cs data

In the case of milk (Fig. 11), it was thus seen, as expected, that the radioiodine concentrations were low in northern Europe where the grazing season was just at its beginning, while they were significantly higher in southern Europe where the cows had been grazing for months. This latitudinal effect, which is due to seasonality, was also seen for ^{137}Cs as shown by UNSCEAR [19]. The effect was most evident for grain (Fig. 12) as was anticipated.

In Denmark (Fig. 13) the relative ^{137}Cs concentrations of the grain species (rye, barley, wheat, oats) were compared with those observed in the years prior to the Chernobyl accident. It appeared that the rye levels were an order of magnitude higher than those seen in the other species after the Chernobyl accident. This observation reflects

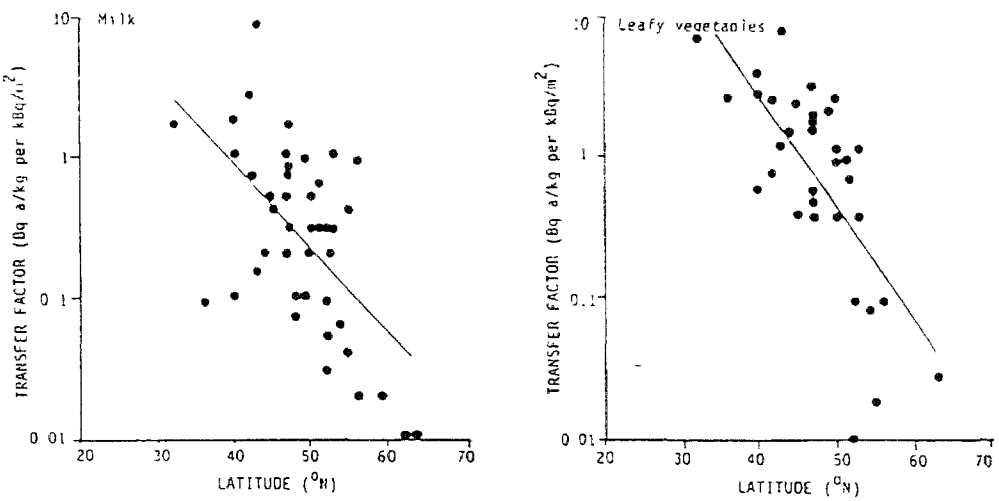


Fig. 11. Integrated concentrations of I-131 in milk and leafy vegetables per unit I-131 deposition density [19].

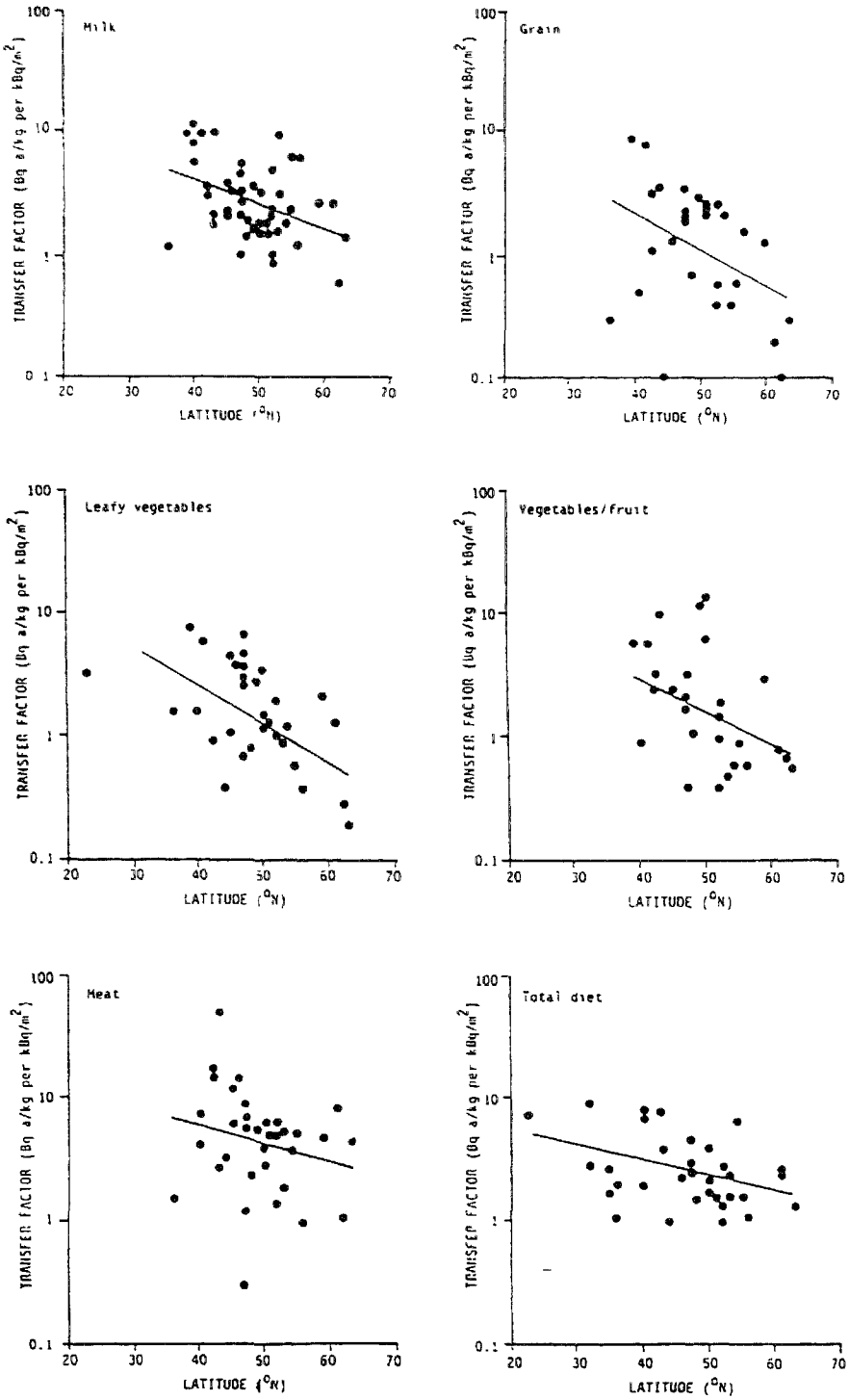


Fig. 12. Integrated concentrations of Cs-137 in foods and total diet in the first year after the accident per unit Cs-137 deposition density [19].

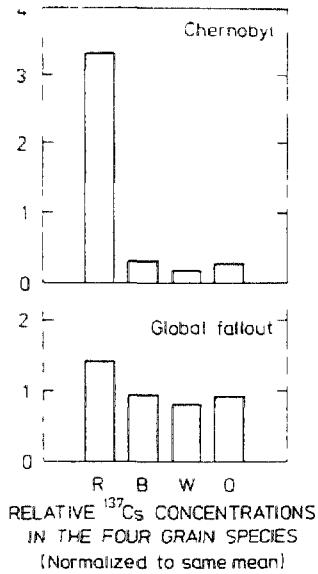


Fig. 13. Relative Cs-137 concentrations in Danish grain, normalized to a mean of 1. The global fallout samples are based on observations in the period 1962-1974. The mean concentration in Danish grain in 1962-1974 was 7x1 Bq Cs-137 kg^{-1}. In 1986 the mean level was 3x3 Bq Cs-137 kg^{-1}. R = rye, B = barley, W = wheat and O = oats [44].

the precosity of rye compared with the other species, resulting in a higher retention of ^{137}Cs by the more developed rye crops when the fallout from Chernobyl was deposited. UNSCEAR [19] calculated the transfer factors (Bq ^{137}Cs a kg^{-1} per kBq ^{137}Cs m^{-2}) in various diets after the Chernobyl accident. It appeared that the factors were usually lower than those observed previously for global fallout ^{137}Cs, especially for northern Europe. One reason for this was seasonality, which in particular influences the levels for grain and for which the transfer factors post-Chernobyl were 1-2 orders of magnitude lower than those observed for global fallout. Another reason could be a lower availability of the Chernobyl ^{137}Cs than of that from global fallout with respect to contamination of crops. But seasonality seems to be the most important factor.

Fruit seasonality

Berries and nuts showed relatively high radiocesium concentrations after the Chernobyl accident. It has been shown experimentally that radiocesium deposited on the leaves of fruit bushes is translocated to the

berries and the content of radiocaesium in the berries generally increases with time. In an experiment carried out by Kopp et al. [20], it was found that in red currants 0.6% of the deposited ^{134}Cs activity was transported to the unripe fruit and 3.4% to the ripe berries; only minute amounts of ^{85}Sr had been transferred to the edible parts, however . In bilberries the corresponding figures for ^{134}Cs were 0.7% (unripe) and 1.8% (ripe) and for gooseberries 4.4% (unripe) and 3.7% (ripe).

4.3.2. Models

The principal terrestrial foodchain models known to consider seasonality are ECOSYS and FARMLAND.

ECOSYS

ECOSYS (Fig. 14) [21] was developed for the rapid prediction of radiation exposure after single releases of radionuclides to the atmosphere. In order to accomplish this for vegetation, the model takes morphological and physiological changes of the plants during their growing period into consideration.

For the modelling of seasonality of plant contamination, the processes deposition, interception, translocation, growth dilution and weathering are considered. Dry and wet deposition are modelled separately. The amount of activity retained by the plant is characterized by the leaf area index (Fig. 15). This index is a suitable parameter for modelling because the leaf area is the boundary layer for dry as well as for wet

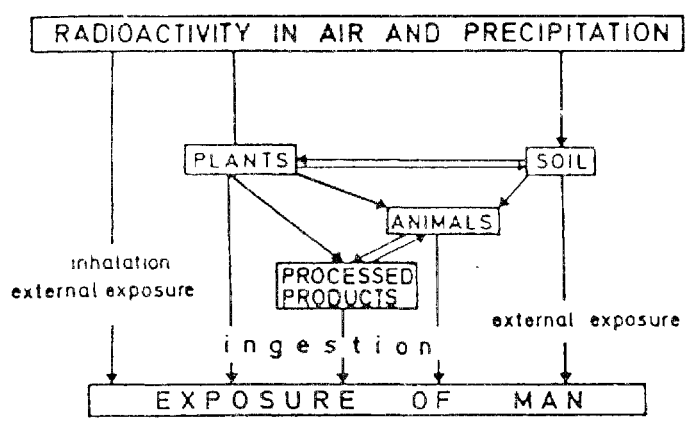


Fig. 14. Structure of the dynamic model ECOSYS [21].

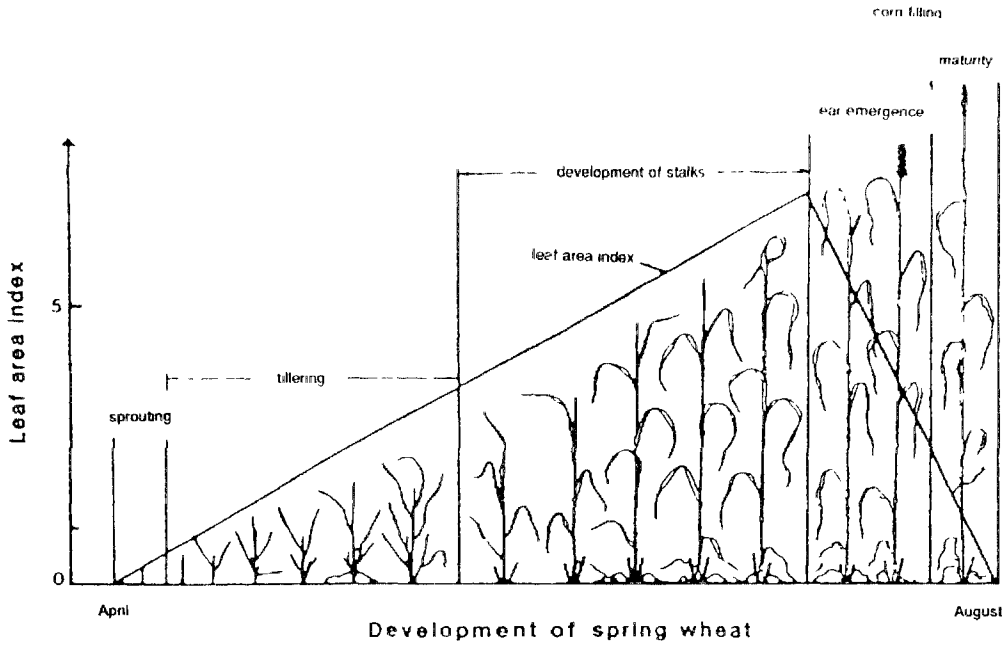


Fig. 15. Leaf area index of spring wheat during the growing period and its stages of development [21].

deposited radionuclides from the atmosphere. Data on the leaf area index are available from investigations on the growth characteristics of agricultural crops [22-26]. For the interception of wet deposition the amount of rainfall is another very important parameter [21,27] (Fig. 21).

The modelling of translocation is based on data from Middleton [2,3], Aarkrog [4 -7], Voigt et al. [28] and Ludwieg [29] (Fig. 16).

For pasture grass growth dilution is modelled as a season-dependent process. The overall field loss of activity due to growth dilution and weathering varies by not more than a factor of 1.5 between May and October.

The version of ECOSYS which was available immediately after the Chernobyl accident was able to predict the contamination of foodstuffs quite well starting from the initial retention on plants. There were some discrepancies concerning the activity in different varieties of grain (winter and spring wheat, rye, winter and spring barley, oats) because no differentiation between species was taken into account. The differences in the morphological and physiological status of winter and spring varieties are very pronounced at beginning of May. Whereas winter varieties are

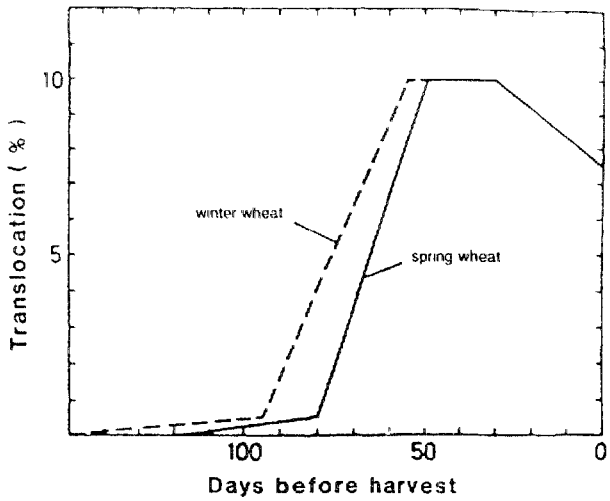


Fig. 16. Translocation of caesium from foliage to grain and its dependence on the period between deposition and harvest [21].

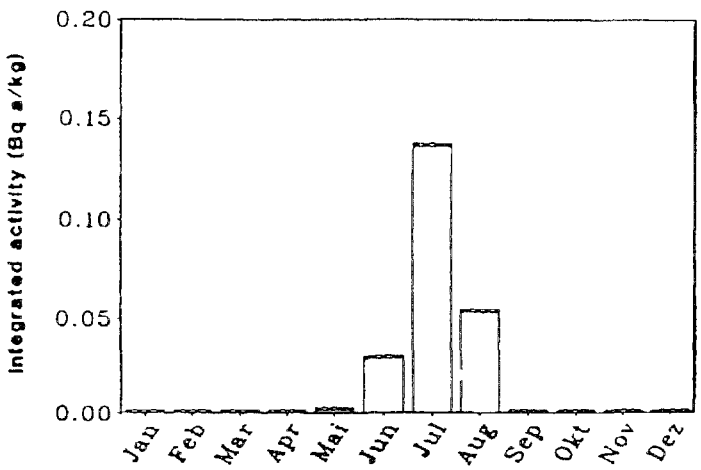


Fig. 17. Integrated Cs-137 activity concentration of spring wheat and its dependence on the month of deposition (dashed part: contribution of root uptake) [21].

normally well developed at this time, spring varieties have only recently emerged and the plants are still relatively small at the beginning of May. This leads to substantial differences in the intercepted activity and to different fractions being translocated from the foliage to the grain.

At later times of deposition (e.g. 1st June or 1st July) these differences would not have been as pronounced as they are at the beginning of May.

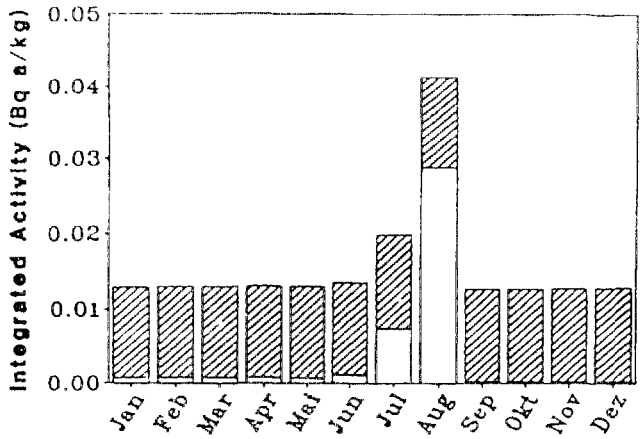
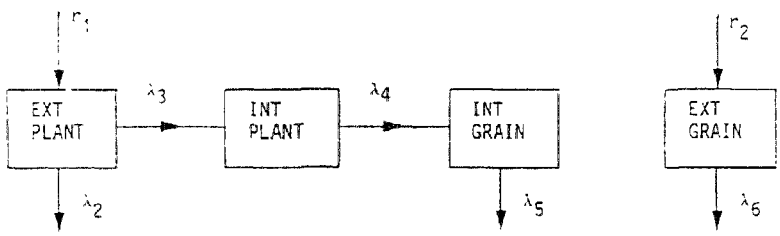


Fig. 18. Integrated Sr-90 activity concentration of spring wheat and its dependence on the month of deposition [27].



Notes

a) The concentration in grain is obtained by summing the contributions from external and internal grain compartments.

b) r_1 and r_2 represent interception of the deposit by the whole plant and by the grain respectively. They have values of 0.3 and 0.012 respectively for all elements.

c) λ_2 and λ_6 represent losses due to weathering from the whole plant and the grain respectively. $\lambda_2 = 4.95\ 10^{-2}\ d^{-1}$ and $\lambda_6 = 4.82\ 10^{-2}\ d^{-1}$ for all elements.

d) The values of λ_3, λ_4 and λ_5 are obtained by fitting to experimental data and are element dependent. The values for strontium and caesium are:-

Transfer Coefficient	Strontium d^{-1}	Caesium d^{-1}
λ_3	$3.7\ 10^{-2}$	$3.4\ 10^{-2}$
λ_4	$6.9\ 10^{-2}$	$6.4\ 10^{-2}$
λ_5	$4.5\ 10^{-1}$	$5.2\ 10^{-2}$

Fig. 19. Revised model for the deposition of radionuclides onto grain plants [13].

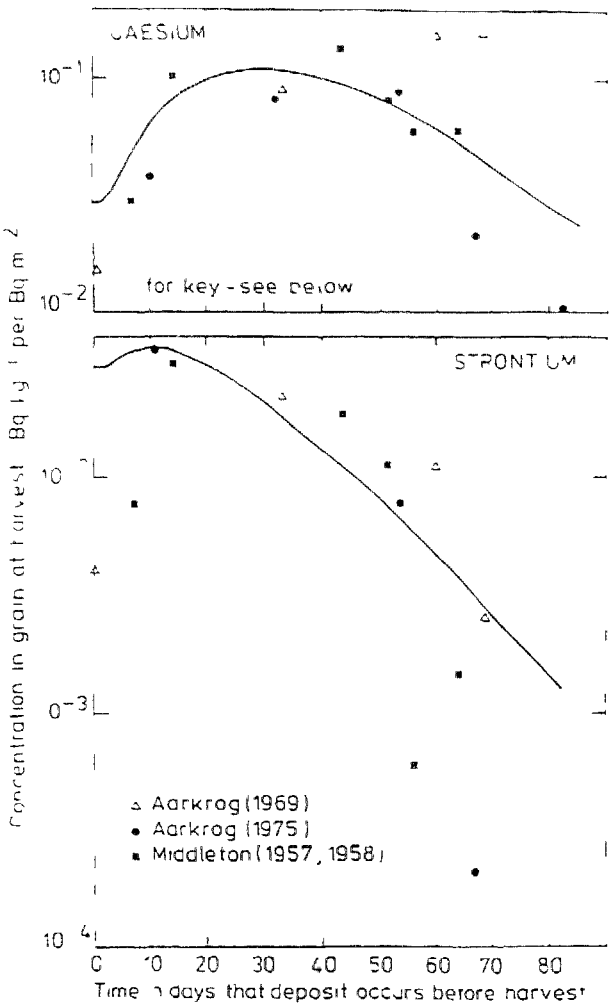


Fig. 20. Comparison of model and experimental results for single deposits of caesium and strontium on grain plants at various times before harvest [13].

FARMLAND

The National Radiological Protection Board's (NRPB) foodchain model FARMLAND [30-32] has been developed over a number of years and it includes seasonality (Fig. 19). The model and its parameter values were derived from global fallout observations as well as from experimental measurements, notably Middleton [2], Middleton and Squire [3] and Aarkrog [4-8] (Fig. 20). Furthermore compilations of data, notably those of Ng [33] and International Union of Radioecologists (IUR) [34] have been applied.

From a seasonal viewpoint FARMLAND predicted the transfer of radionuclides to food following Chernobyl reasonably well. In particular,

FARMLAND's predictions of a decline in cow's milk concentrations following the deposit followed by an increase when the animals were fed on stored feed was observed following Chernobyl. However, the release occurred at a single time of year and data on the stage of growth of crops from locations other than that on which the model is based are difficult to obtain. Therefore, the testing of the seasonal aspects of the model has, so far, not been as rigorous as is desirable.

In testing the validity of FARMLAND with Chernobyl data some discrepancies were noted with results for particular farms in the United Kingdom. This was due mainly to differences between the general assumptions regarding agricultural practices adopted in FARMLAND and the actual practices at the individual farms. Also there were differences in some cases when the deposition occurred in rainfall as FARMLAND does not take account of the rainfall rate, neither does it distinguish between wet and dry deposition.

Risø-models

The curves shown in Fig. 20 are to some extent similar to those in Figs. 5 and 6. These models were made from least squares fitting of the normal distribution curve to the experimental data [8]. The model for Cs (Fig. 6) was applied to the Chernobyl observations in 1986 in Denmark by Aarkrog [35]. The model predicted 4.2 Bq ^{137}Cs kg^{-1} grain but the observed level was 0.3 Bq ^{137}Cs kg^{-1}. One reason for this discrepancy was a more rapid field loss of the Chernobyl ^{137}Cs than that found during the experiments because intense rain in Denmark removed a good deal of the initially deposited Chernobyl debris. This effect has been shown by Müller and Pröhl [27] (see Fig. 21) who found that the interception factor can vary from nearly one hundred down to only a few percent.

A simpler empirical model for the contamination of grain in Denmark has been based on global fallout data collected since 1959 (Table VIII). This model assumes that the ^{137}Cs deposition in May-August is decisive for the contamination of the grain at harvest time in August-September. The model gave good predictions prior to the Chernobyl accident; but the predictions in 1986 for Danish grain came out 1-2 orders of magnitude higher than those actually observed. The reason for this was that the empirical models in Table VIII assume that the fallout is evenly distributed over the period May-August, but the majority of the Chernobyl

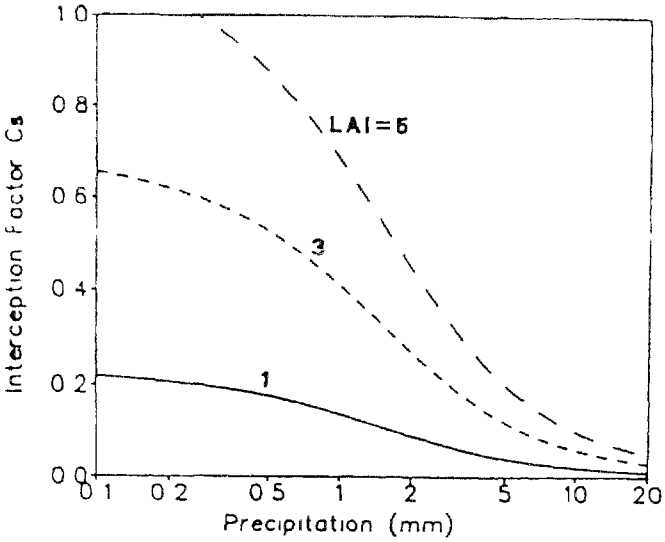


Fig. 21. Interception factor for wet deposition of caesium on grass with different leaf area indices (LAI) as used in ECOSYS [27].

TABLE VIII PREDICTION MODELS FOR ^{137}Cs IN CEREAL GRAIN [41]
(unit for radioecological sensitivity pCi ^{137}Cs kg^{-1}a per mCi ^{137}Cs km^{-2})

No	Sample	Area	Period	^{137}Cs-activity, pCi $kg^{-1}_{(1)}$	Sensitivity	r
1	Rye	Denmark	1962-76	$79d'_{1\,(May-Aug)} + 5\ 0d'_{i-1}$	47	0 9991***
2	- " -	- " -	- " -	$85d'_{1\,(May-Aug)}$	46	0 9957***
3	Barley	- -	- " -	$59d'_{1\,(May-Aug)}$	32	0 9957***
4	Wheat	- " -	- " -	$52d'_{1\,(May-Aug)}$	28	0 9857***
5	Oats	"	- " -	$50d'_{1\,(May-Aug)}$	27	0 9952***

debris was deposited in May, where the initial retention by the sparsely developed crops was low. Hence, although these empirical grain models have considered seasonality they were not sophisticated enough for dealing with the Chernobyl input.

OSCAAR-CHRONIC

Matsuzuru [36] has also proposed models that take seasonality into account. A foodchain model is incorporated in the computer code system OSCAAR-CHRONIC which evaluates chronic exposure due to radionuclides remaining in a Japanese environment after a reactor accident. The model is an extension of the methodology used in the U.S. CRAC code [37]. Four kinds of important Japanese crops are selected: leafy vegetables, cereals,

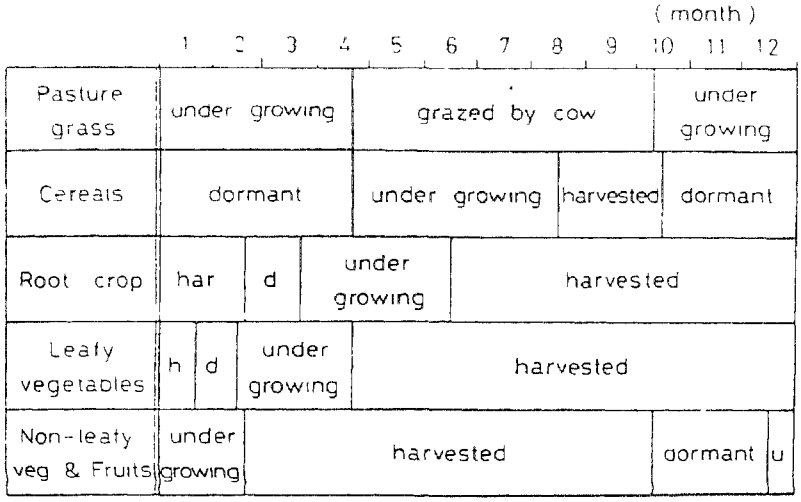


Fig. 22. The Model of Agricultural Practice [36].

rootcrops and fruits (including non-leafy vegetables). It is assumed that the year can be divided into 3 and 4 periods for the various crops as shown in Fig. 22 and time dependent functions are postulated for the intake of radionuclide for the various periods. Translocation is not included in the model, hence the model seems to underpredict ^{137}Cs intake from cereals (Table IX). The parameter values employed in the model are Japanese as far as productivity density of grass/crops, growing periods and consumption rate of foodstuffs are concerned. Foreign data are applied for the other parameters such as interception factors, transfer rates, time delay to consumption and so on. Most of the values are those used in the CRAC code and others are taken from NRPB data sets.

Figures 23 and 24 show the results of the model calculations for milk and cereals, respectively, for Japanese conditions. The models were

TABLE IX. RATIOS OF PREDICTED-TO-OBSERVED CONCENTRATIONS OF ^{137}Cs IN WINTER GRAIN (w) AND SPRING GRAIN (s) BASED ON GEOMETRIC MEANS OF ALL MEASUREMENTS [38]
(GM: geometric mean, GSD: geometric standard deviation)

MODEL	T ROSKILDE (w)	T ROSKILDE (s)	S GEEL (s)	C TOKAI (w)	H TRANVIK (w)	N MUNICH (w)	L LOVIISA (w)	L LOVIISA (s)	GM	GSD
DETRA		8.0	7.5					7.5	7.7	1.0
SIRATEC	0.26	0.52	0.2	0.029	0.14	0.43	0.13	0.18	0.18	2.4
FARMLAND	3.3	21	3.8				1.6	0.65	3.1	3.6
ECOSYS*	1.3	0.13	0.087	0.055	0.63	0.96		0.03	0.20	4.0
SPADE 2	3.7	6.6		0.13	8.7	4.9	9.7	0.014	1.5	12.7
TERFOC	0.029	0.027		0.49	0.0052	0.17	0.019	0.013	0.036	4.7
CHRONIC	0.029	0.026		0.49	0.0059	0.17	0.019	0.015	0.036	4.6
RAGTIME	7.1			0.91	2.2	6.7	1.8		2.8	2.4
CHERPAC	0.34	0.85		4.1	7.3E-6	7.96	0.028	0.00033	0.049	176

* Calculations for ECOSYS are grain specific: listed are P/O ratios for wheat.

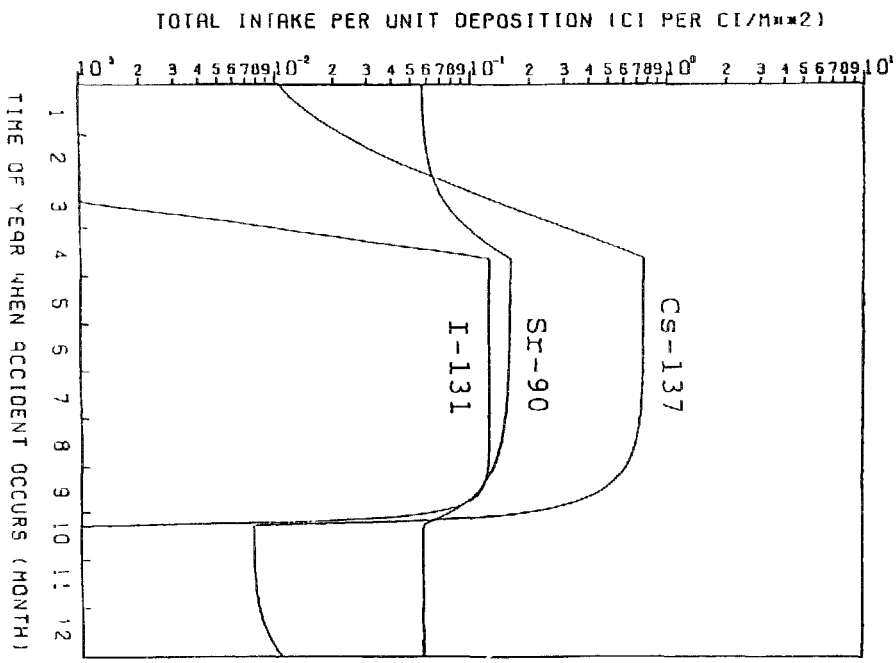


Fig. 23. Total intake from milk [36].

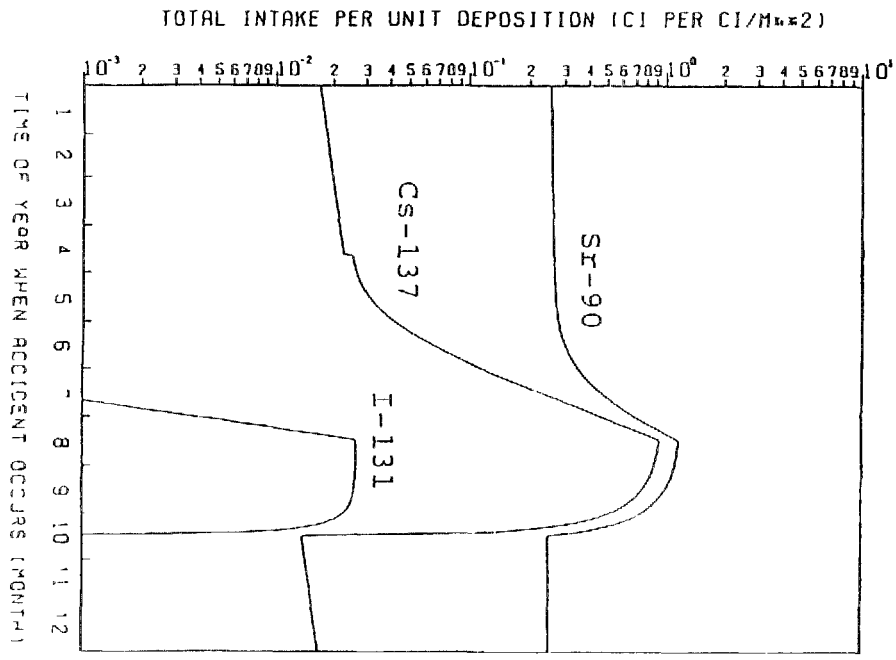


Fig. 24. Total intake from cereals [36].

developed before the Chernobyl accident. In the BIOMOVS study the model was applied to the Chernobyl scenario for ^{131}I and ^{137}Cs.

BIOMOVS international model validation study

The BIOMOVS group [38] compared a number of model predictions of ^{137}Cs in grain with the actual concentrations found after the Chernobyl accident in 1986 (Table IX). Three of the models have been described above (ECOSYS, FARMLAND and CHRONIC). None of the model predictions were close to observed values for all locations tested and there was considerable variability in the model predictions. For this scenario, the BIOMOVS group did not attempt to analyze the reasons for predictive variations and discrepancies because of the generally poor quality of the information available from the sites tested.

The BIOMOVS study concludes that it is difficult to find reasons for the discrepancies between the model predictions and the observations. However, it concluded that they may be associated with a strong seasonal factor and the absence of site specific information on the state of maturation of the plants at the time of the accident.

4.4. Need for Model Improvement

4.4.1. ECOSYS

After the Chernobyl accident the ECOSYS model was improved in various ways. Dry and wet deposition is now modelled separately, the leaf area index during the growing period of each plant species is considered and a differentiation between all six major cereal species cultivated in Europe is made.

Furthermore, realistic consideration of animal diets especially for beef and dairy cattle, has been introduced [39]. The radionuclide content of animal feed is, as mentioned earlier, subject to seasonal variability (Section 4.1.2.) due to the differing growth intensities of pasture grass during the growing season. The supply of fresh pasture grass is highest at the start of the growing period and decreases towards the end of the growing period. This means that increasing amounts of fresh pasture have to be substituted by other feedstuffs such as maize, beet, beet leaves etc. Also for a reliable prediction of the activity in milk and meat in the winter following the deposition, the real site-specific feeding regimes

have to be taken into account. It is also necessary, for a realistic prediction of the activity in winter fodder, to have information about the period when hay and silage are prepared, which may vary for climatic reasons.

4.4.2. FARMLAND

With regard to FARMLAND, this model has, according to NRPB, not been modified or "fine-tuned" in any way to take account of the Chernobyl data. However, some particular parameter values may be altered in the light of data obtained after Chernobyl, although care is required not to place too much reliance on a single source of data. The comparison of FARMLAND predictions with post-Chernobyl measurements generally strengthens NRPB's belief in the validity of the model for use in generic radiological assessments, the use for which it was intended. For this reason some of the detailed site specific modifications, such as can be incorporated into the ECOSYS model, are not included. However, it is recognised that further development of FARMLAND is desirable to enable the effect of deposition in varying wet and dry conditions to be taken into account for more detailed applications.

4.4.3. Summary

The Chernobyl accident, which was a major short time input of radionuclides into the biosphere, demonstrated that no existing models (at the time of the accident) were able adequately to predict the effect of seasonality. A number of models have considered the importance of the leaf area index and translocations (as described by Pröhl et al. [21]), but as shown by Müller and Pröhl [27] the meteorological conditions (mm precipitation) also had a significant impact on the interception factor. Furthermore, the meteorological conditions in the time after the deposition had an effect on the field loss, i.e. rate of decrease of the activity in the plants. Field loss is influenced by weathering processes, e.g. washoff, and also by plant growth dilution. All these elements have to be included in models if they are to yield reliable model predictions for single event radioactive contamination. In order to describe the seasonality process adequately, models have to be site specific.

At any particular time of the year the stage of plant development in different regions of the world varies widely. The time of harvest is also variable. (Müller and Pröhl [27] (Fig. 25)). Both of these factors can

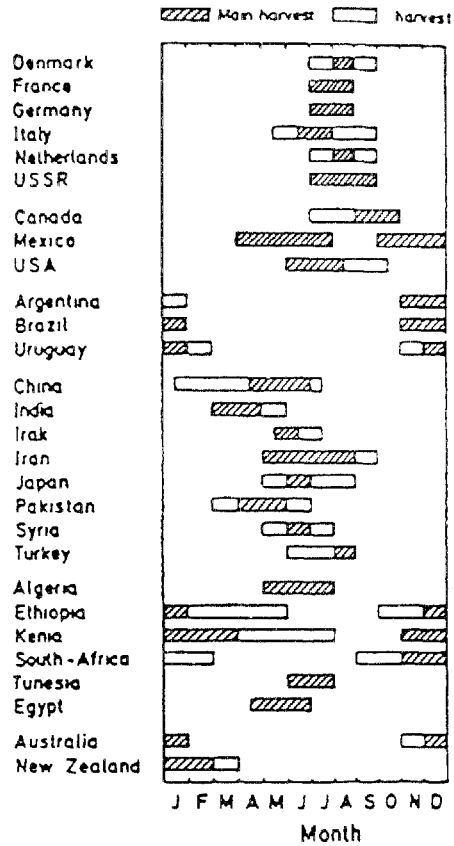


Fig. 25. Times of wheat harvest in some countries [45].

strongly influence the radionuclide content of crops; other factors, such as harvesting methods may also have an influence.

The BIOMOVS study [38] has shown that the more complex dynamic models do not necessarily give more reliable predictions than simple models. However in the case of a short time input of radionuclides into the environment, simple steady- state models for predicting the contamination of cereals will not suffice and dynamic models are required. If the contamination is "chronic" (e.g. as has been the case for global fallout) a simple model will probably be just as good as the complex one. The Riso model shown in Table 8 has thus proved to produce very reliable predictions throughout the years of global fallout ^{137}Cs in Danish cereals.

4.5. Conclusion and Recommendations

Seasonality may have a strong influence on the radiological impact of environmental radioactive contamination. Seasonality is influenced by a number of environmental factors. In the terrestrial soil-to-plant-animal system the ratio between direct and indirect (root uptake) contamination of the plants is of major importance. Seasonality is pronounced if direct contamination is the dominating pathway. The degree of translocation of radionuclides to the fruit bodies of plants also influences seasonality. High translocation normally reduces seasonality. Contamination of cereal grains with ^{137}Cs and ^{131}I shows a pronounced seasonality, whereas ^{90}Sr in root crops shows little or no seasonality.

In aquatic ecosystems ice cover, plankton bloom and runoff during snow melting may influence seasonality. Natural and seminatural ecosystems show generally lower seasonality than agricultural systems. Temperate zones show higner seasonality than subtropical and tropical regions.

There is a need for a better understanding of the processes that influence seasonality, especially those connected to meteorology, e.g., washoff. The physicochemical behaviour of contaminants with respect to initial uptake and translocation in crops needs further study. Information on seasonality in connection with fruit and vegetable contamination is particularly sparse. Seasonality effects also require further investigation in connection with aquatic pathways (e.g., fish, drinking water). Finally, natural and seminatural ecosystems should be further examined with respect to seasonality.

In order to obtain reliable predictions of the radiological consequences of short term contamination of crops, the model applied should be site specific and be capable of taking account of changes in interception characteristics with plant growth, the effects of translocation, of rain intensity and of the physicochemical characteristics of the contaminant.

Annex 1

Definition of the Seasonality Factor, S

The seasonality factor S is the coefficient of variation (CV) of the monthly infinite time integral concentrations of a given radionuclide in a given sample observed throughout twelve months of a year.

A monthly deposition of 1 Bq m^{-2} in month (i) of radionuclide (p) results in an infinite time concentration integral in sample item (a) of I_{iap} Bq $Kg^{-1}yr$.

The variation of I_{iap} between the 12 months of the year is a measure of the degree of seasonality. This variation is quantified as the coefficient of variation CV, i.e. the relative standard deviation of the twelve I_{iap} values, which have an annual mean of I_m. Hence

$$S \equiv CV = \frac{\sqrt{\dfrac{\sum_{i=1}^{12} (I_m - I_{iap})^2}{(12 - 1)}}}{I_m}$$

If CV > 0 seasonality is present. If CV ≃ 0 no seasonality is observed.

Application of S

Pröhl [40] has calculated a variety of I_{iap} values (see e.g. Figs. 17 and 18) based on a time dependent simulation model SINK, which is closely related to the ECOSYS model (Pröhl et al. [21]). These I_{iap} values are site specific for the average conditions in southern Germany and should thus be considered only as examples.

Below are summarized some S values calculated from Pröhl [40]

	Sr-90	I-131	Cs-137
MILK	0.4	1.1	0.8
WHEAT	0.5	2.6	1.9
ROOT VEGETABLES	0	-	1.2
LEAF VEGETABLES	0.4	0.2	0.5

It is evident (Figs. 17 and 18) that the degree of seasonality depends strongly on the ratio between root uptake and direct contamination of the crops. If this ratio is low the seasonality is high and vice versa. However, other factors, e.g. translocation of certain radionuclides play an important role, in particular in the case of fruits and cereals.

It should further be noticed that for certain sample types, e.g. cereal grain, there may be even monthly seasonality variation, but for practical reasons the definition of the seasonality factor has been based on monthly data.

REFERENCES

[1] KARLEN, G., JOHANSON, K.J., BERGSTROEM R., 'Seasonal variation in the activity concentration of ^{137}Cs in Swedish roe-deer and their daily intake, Vol. 58, J. Environ. Radioact. 14 (1991) 91-103.

[2] MIDDLETON, L.J., Radioactive strontium and cesium in the edible parts of crop plants after foliar contamination, Int. J. Radiat. Biol., 4 (1959) 387-402.

[3] MIDDLETON, L.J., SQUIRE, H.M., Further studies of radioactive strontium and cesium in agricultural crops after direct contamination, Int. J. Radiat. Biol. 6 (1963) 549-558.

[4] AARKROG, A., On the direct contamination of rye, barley, wheat and oats with ^{85}Sr, ^{134}Cs, ^{54}Mn and ^{141}Ce, Radiat. Bot. 9 (1969) 357-366.

[5] AARKROG, A., LIPPERT, J., Direct contamination of barley with ^{51}Cr, ^{59}Fe, ^{58}Co, ^{65}Zn, ^{203}Hg and ^{210}Pb, Radiat. Bot. 11 (1971) 463-472.

[6] AARKROG, A., Direct contamination of barley with ^{7}Be, ^{22}Na, ^{115}Cd, ^{125}Sb, ^{134}Cs and ^{133}Ba. Aspects of Research at Risø. A collection of papers dedicated to Professor T. Bjerge on his seventieth birthday. Risø Report No. 256, Risø National Laboratory, Roskilde (1972) 163-175.

[7] AARKROG, A., Radionuclide levels in mature grain related to radiostrontium content and time of direct contamination, Health Phys. 28 (1975) 557-562.

[8] AARKROG, A., Translocation of radionuclides in cereal crops. Ecological Aspects of Radionuclide Release. Special Publication Series of the British Ecological Society No. 3, Blackwell Scientific Publications, Oxford (1983) 81-90.

[9] DELMAS, J., BOVARD, P., GRAUBY, A., DISDIER, R., BLONDEL, L., GUENNELON R., 'Contamination directe expérimentale de l'oranger par le ^{90}Sr et le ^{137}Cs', Radioecology Applied to the Protection of Man and his Environment (Proc. Symp. Rome, 1971), Commission of the European Communities, Luxembourg (1972).

[10] MUELLER, H., EISFELD, K., MATTHIES, M., PROEHL, G., 'Contamination of crops by irrigation with radioactively contaminated water', The Transfer of Radioactive Materials in the Terrestrial Environment Subsequent to an Accidental Release to Atmosphere (Proc. Sem. Dublin, 1983), Commission of the European Communities, Luxembourg (1983).

[11] NAIR, S., 'A sensitivity study of ingestion population dose following a hypothetical atmospheric release from a nuclear power station', The Transfer of Radioactive Materials in the Terrestrial Environment Subsequent to an Accidental Release to Atmosphere (Proc. Sem. Dublin, 1983), Commission of the European Communities, Luxembourg (1983).

[12] ROBERTS, J.A., 'Terrestrial pathways model for NUCRAC - SAI', The Transfer of Radioactive Materials in the Terrestrial Environment Subsequent to an Accidental Release to Atmosphere (Proc. Sem. Dublin, 1983), Commission of the European Communities, Luxembourg (1983).

[13] SIMMONDS, J.R., 'The influence of the season of the year on the agricultural consequences of accidental releases of radionuclides to the atmosphere', The Transfer of Radioactive Materials in the Terrestrial Environment Subsequent to an Accidental Release to Atmosphere (Proc. Sem. Dublin, 1983), Commission of the European Communities, Luxembourg (1983).

[14] GOVAERTS, P., KRETZSCHMAR, J.G., MERTENS, I., 'Application of the "DOSDIM"-model to assess doses due to deposited materials subsequent to an accidental atmospheric release', The Transfer of Radioactive Materials in the Terrestrial Environment Subsequent to an Accidental Release to Atmosphere (Proc. Sem. Dublin, 1983), Commission of the European Communities, Luxembourg (1983).

[15] PARTANEN, J.P., SAVOLAINEN, I., Significance of contaminated food in collective dose after a severe reactor accident, Health Phys. 50 (1986) 209-216.

[16] KIRCHNER, T.B., WHICKER, F.W., Validation of PATHWAY, a simulation model of the transport of radionuclides through agroecosystems, Ecol. Modelling 22 (1984) 21-44.

[17] WHICKER, F.W., KIRCHNER, T.B., PATHWAY: A dynamic food-chain model to predict radionuclide ingestion after fallout deposition, Health Phys. 52 (1987) 717-737.

[18] BRESHEARS, D.D., KIRCHNER, T.B., OTIS, M.D., WHICKER, F.W., Uncertainty in predictions of fallout radionuclides in foods and of subsequent ingestion, Health Phys. 57 (1989) 943-953 .

[19] UNITED NATIONS, Sources, effects and risks of ionizing radiation, Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), UN, New York (1988) 648 pp.

[20] KOPP, P., GOERLICH, W., BURKART, W., ZEHNDER, H.J., Foliar Uptake of Radionuclides and their Distribution in Plant (personal communication) (1989).

[21] PROEHL, G., MUELLER, H., JACOB, P., PARETZKE, H.G., 'The dynamic radioecological model ECOSYS - A tool for the management of nuclear accidents' consequences' (4e Symp. Int. de Radioécologie de Cadarache, 1988) Tome 1, CEN/Cadarache (1988).

[22] PETR, J., et al., Ertragsbildung bei landwirtschaftlichen Kulturpflanzen, Deutsch. Landwirtschaftsverlag, Berlin (1986).

[23] GEISLER, G., Ertragsphysiologie von Kulturarten des gemässigten Klimas, Verlag Paul Parey, Berlin und Hamburg (1983).

[24] NOESBERGER, J., Die Analyse der Ertragsbildung von Pflanzen, Schweizer Landwirtschaftliche Monatshefte, 48 (1970) 325-345.

[25] NOESBERGER, J. Einfluss der Bestandesdichte auf die Ertagsbildung bei Mais, Z. Acker-Pflanzenbau 133 (1971) 215-232.

[26] FRIEDRICH, G., NEUMANN, D., VOGL, M. Physiologie der Obstgehölze, Akademie Verlag Berlin (1986).

[27] MUELLER, H., PROEHL, G., 'The role of seasonal, climatic and meteorological conditions in modifying nuclear accidents

consequences', The Influence of Seasonal Conditions on the Radiological Consequences of a Nuclear Accident (Proc. NEA Workshop, Paris, 1989) OECD/NEA, Paris (1989).

[28] VOIGT, G., PROEHL, G., MUELLER, H. Experiments on the seasonality of the caesium translocation in cereals, potatoes and vegetables, Radiat. Environ. Biophys. 30 (1991), 295-303.

[29] LUDWIEG, F.Z., Die Aufnahme von Cs-137 durch Kartoffelblätter, Z. Pflanzenernaehr. Bodenk. 99 (1962) 109-194.

[30] SIMMONDS, J.R., LINSLEY, G.S., JONES, J.A., A general model for the transfer of radioactive materials in terrestrial foodchains, Harwell, NRPB-R89, HMSO, London (1979).

[31] SIMMONDS, J.R., CRICK, M.J., Transfer parameters for use in terrestrial foodchain models, Chilton, NRPB-M63, HMSO, London (1982).

[32] SIMMONDS, J.R., The influence of season of the year on the transfer of radionuclides to terrestrial foods following an accidental release to atmosphere, Chilton, NRPB-M121, HMSO, London (1985).

[33] BOONE, F.W., NG, Y.C., PALMS, J.M., Terrestrial pathways of radionuclide particulates, Health Phys. 41 (1981) 735-747.

[34] INTERNATIONAL UNION OF RADIOECOLOGISTS, 5th report of the workgroup on soil-to-plant transfer factors. RIVM, Bilthoven (1987).

[35] AARKROG, A., et al., Environmental radioactivity in Denmark in 1986, Risø-R-549, Risø National Laboratory, Roskilde (1988).

[36] MATSUZURU, H., VAMP-Terrestrial Working Group, Questionnaire No. 1 (1989).

[37] NUCLEAR REGULATORY COMMISSION, Calculation of Reactor Accident Consequences, WASH-1400, NUREG 75/014, USNRC Washington, D.C. (1975).

[38] KOEHLER, H. PETERSON, S.-R., HOFFMAN, F.O. (Eds.), Multiple model testing using Chernobyl fallout data of I-131 in forage and milk and Cs-137 in forage, milk, beef and grain, BIOMOVS Technical Report Scenario A4, National Institute of Radiation Protection, Stockholm (1991).

[39] KIRCHGESSNER, M., Tierernaehrung, DLG-Verlag, Frankfurt/Main (1987).

[40] PROEHL, G., Modellierung der Radionuklidausbreitung in Nahrungsketten nach Deposition von Strontium-90, Cäsium-137 und Jod-131 auf landwirtschaftliche genutzte Flächen, GSF-Bericht 29/90, Gesellschaft für Strahlen-und Umweltforschung, Neuherberg (1990).

[41] AARKROG, A., Environmental studies on radioecological sensitivity and variability with special emphasis on the fall-out nuclides ^{90}Sr and ^{137}Cs, DSc. Thesis, University of Copenhagen, Risø-R-437 (1979), Risø National Laboratory, Roskilde (1979).

[42] AARKROG, A. (Ed.), Bioindicator studies in Nordic waters, Nordic Liaison Committee for Atomic Energy, Risø National Laboratory, DK-4000 Roskilde, Denmark (1985) 22.

[43] HOLM, E., RIOSECO, J., MATTSSON S., 'Technetium-99 in the Baltic Sea', Technetium in the Environment, (DESMET, G., MYTTENAERE, C., Eds.) Elsevier, London, New York (1986) 61.

[44] AARKROG, A., The radiological impact of the Chernobyl debris compared with that from nuclear weapons fallout, J. Environ. Radioact. 6 (1988) 151-162.

[45] FRANKE, G., Nutzpflanzen der Tropen und Subtropen, Vol. II, S. Hirzel Verlag, Leipzig (1984).

LIST OF TERRESTRIAL WORKING GROUP MEMBERS ATTENDING THE 1990-1991 MEETINGS

Aarkrog, A.	Risø National Laboratory P. O. Box 49 DK-4000 Roskilde Denmark
Barchudarov, R.	Institute of Biophysics Ministry of Public Health Zhivopisnaya, 46 123182 Moscow Russian Federation
Benes, P.	Technical University of Prague Brehová 7 115 19 Prague 1 Czechoslovakia
Bergman, R.	National Defence Researh Institute ABC Research Department FOA ABC - skydd S-901 82 Umea Sweden
Bilo, M.	Institut für Radioagronomie Forschungszentrum Jülich GMBH Postfach 1913 D-5170 Jülich 1 Germany
Chamberlain, A.	1 Fifth Road Berkshire RG14 6DN United Kingdom
Colgan, T.	Nuclear Energy Board 3 Clonskeagh Square Dublin 14 Ireland
Desmet, G.	Commission of the European Communities XII/F/1 Rue de la Loi, 200 B-1049 Brussels Belgium
Dovlete, C.	Institute of Environmental Research and Engineering Environmental Radioactivity Laboratory Postal 11, C.P. 11-2 Bucharest Romania
Eriksson, A.	Swedish University of Agriculture, Sciences, Department of Radioecology Box 7031 S-750 07 Uppsala Sweden

Erlandsson, B.	Department of Nuclear Physics University of Lund Sölvegatan 14 S-22362 Lund Sweden
Ettenhuber, E.	Institut für Umweltüberwachung des Staatlichen Amtes für Atomsicherheit und Strahlenschutz Waldowallee 117 1157 Berlin Germany
Firsakova, S.	All-Union Institute of Agricultural Radioecology 246020 Gomel Barykina 305 E Russian Federation
Frissel, M.	Torenlaan 3 6866 BS Heelsum Netherlands
Fujimoto, K.	Division of Nuclear Safety International Atomic Energy Agency Wagramerstrase 5, P. O. Box 100 A-1400 Vienna Austria
Fulker, M.	British Nuclear Fuels plc Building B433 Sellafield Seascale Cumbria CA 20 1PG United Kingdom
Fülop, N.	National Research Institute Radiobiology and Radiohygiene P. O. Box 101 H-1775 Budapest Hungary
Garland, J.A.	AEA Environment and Energy Harwell Laboratory Didcot Oxfordshire OX11 ORA United Kingdom
Gerzabek, M.	Austrian Research Centre Seibersdorf A-2444 Seibersdorf Austria
Giess, P.	Imperial College of Science and Technology Reactor Centre Silwood Park Buckhurst Road Ascot, Berkshire SL5 7PY United Kingdom
Henrich, E.	BALUF (Federal Food Control and Research Institute Radiation Protection Department Berggasse 11 A-1090 Vienna Austria

Holm, E.	Department of Radiation Physics Lund University Lasarettet S-221 85 Lund Sweden
Howard, B.	Institute of Terrestrial Ecology Merlewood Research Station Grange-over-Sands, Cumbria LA11 6JU United Kingdom
Létourneau, C.	Atomic Energy Control Board P. O. Box 1046. Station "B" Ottawa, K1P 5S9 Canada
Linsley, G. (Scientific Secretary)	Division of Nuclear Fuel Cycle and Waste Management Internatinal Atomic Energy Agency Wagramerstrasse 5, P. O. Box 100 A-1400 Vienna, Austria
Myttenaere, C. (Chairman)	International Union of Radioecologists 4 Place Croix du Sud B-1348 Louvain-la-Neuve Belgium
Noordijk, H.	National Institute of Public Health (RIVM) Laboratory for Radiation Research P. O. Box 1 NL-3720, BA Bilthoven Netherlands
Pietrzak-Flis, Z.	Central Laboratory for Radiological Protection Department of Radiation Hygiene Ul. Konwaliowa 7 Warsaw 03-194 Poland
Prister, B.S.	Ukrainian Branch of the All-Union Institute of Agricultural Radioecology 7 Mashinostroiletnaya St. Chabany, Kiev Ukraine
Pröhl, G.	Gesellschaft für Strahlen und Umweltforschung Institute für Strahlenschutz Ingolstädter Landstrasse 1 D-8042 Neuherberg Germany
Quinault, J.M.	CEA France CEN Cadarache 13108 Saint Paul lez Durance Cedex France
Rantavaara, A.	Finnish Centre for Radiation and Nuclear Safety P. O. Box 268 SF-00101 Helsinki Finland

Rauret, G.	Laboratori de Radiologia Ambiental Universitat de Barcelona Avda. Diagonal 647 Barcelona 08028 Catalunya Spain
Sansone, U.	ENEA/DISP Via Vitaliano Brancati 48 I-00144 Rome Italy
Schell, W.R.	University of Pittsburgh Department of Environmental and Occupational Health Graduate School of Public Health Pittsburgh, Pennsylvania 15261 United States of America
Shaw, G.	Imperial College Reactor The Reactor Centre Silwood Park, Ascot, SL5 7PY United Kingdom
Sobotovich, E.	Institute of Geochemistry and Physics of Minerals Academy of Sciences of Ukraine Palladin av. 34 Kiev 252680 Ukraine
Strand, P.	National Institute of Radiation Hygiene P. O. Box 55, 1345 Osteras Norway
Thompson, L.	Ministry of Agriculture, Fisheries and Food Ergon House c/o Nobel House 17 Smith Square London, SW1P 3JR United Kingdom
Vetrov, V.	Division of Nuclear Techniques in Food and Agriculture International Atomic Energy Agency Wagramerstrasse 5, P. O. Box 100 A-1400 Vienna Austria
Wirth, E.	Institute of Radiation Hygiene Federal Health Office, BGA Ingolstädter Landstrasse 1 D-8044 Neuherberg Germany
Zombori, P.	Hungarian Academy of Sciences Central Research Intitute for Physics Health Physics Department P. O. Box 49, Budapest Hungary

92-01268