APPENDIX F

Changes in the Levels of Radioactive Fall-out and the Resulting Radiation Doses to Man in the United Kingdom

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INTRODUCTION

In the period covered by the Council's 1956 report, that is, up to the spring of 1956, the radioactive debris falling in the United Kingdom had arisen from a large number of small nuclear explosions, chiefly in Nevada, and from a few large nuclear explosions mainly in the Pacific, especially in March, 1954 It was recognized that the doses of radiation to persons from the deposition of debris arising from the smaller nuclear explosions would be far outweighed by those from the larger explosions. The material reaching the U.K. from the latter at that time was found to have an apparent age (determined from the ratio of strontium 89 to strontium 90) which ranged from 7 to 14 months. In these circumstances the dose contributed by the isotopes of relatively short life (say, two months or less) was much less important than the dose from the long-lived isotopes such as caesium 137 and strontium 90.

From the autumn of 1956 the pattern of testing changed. A high proportion of the explosions carried out were of megaton size and took place in higher latitudes in the northern hemisphere. As a result the short-lived isotopes became relatively more important and, with the heavy testing of nuclear devices in the Arctic in October, 1958, the contribution of radioactive fall-out to the background dose-rate in air in the open rose in the spring of 1959 so that, for a period of a month or two, it amounted to some 30 per cent of the natural background. This rise in dose-rate can be attributed to two main causes, namely, the increase in the rate of testing and the shorter time during which the fission products from the tests in the autumn in northern latitudes, particularly in the Arctic, have remained airborne.

CHANGES IN THE CONCENTRATION OF ARTIFICIAL RADIOACTIVITY IN AIR

Continuous measurements have been made over the past seven years of the concentration of radioactivity due to fission products in air at ground level at Chilton, Berks. (Peirson *et al.*, 1960a), and the results are given in Fig. 1F. This shows the average monthly concentrations of caesium 137 and zirconium 95, measured by a gamma spectrometer after shorter-lived activities had decayed, and the total beta radioactivity due to fission products. The total beta radioactivity is of little value in the assessment of potential hazard because it is compounded of short-, medium- and long-lived nucleides, a few only of which are taken up by the body: it is much influenced by recent explosions when short- and very short-lived nucleides greatly increase its amount; and it is not indicative of the important long-term hazard from what are considered the most undesirable nucleides—strontium 90, caesium 137 and carbon 14. It is important, therefore, to scrutinize the levels of specific nucleides, for example, zirconium 95, strontium 90 and caesium 137, and thereby to make an overall assessment.

Fig. 1F shows clearly the regular seasonal variation, particularly of the long-lived Cs–137 activity. This is thought to be mainly due to the mode of transfer of the radioactivity from the stratosphere (see p. 84). Nuclear explosions of megaton size occurred fairly regularly throughout 1957 and 1958 (U.S. Joint Committee on Atomic Energy, 1959a, p. 2517) and the general increase in fission-product concentration in air in 1958 and early 1959 reflects this increased rate of testing.

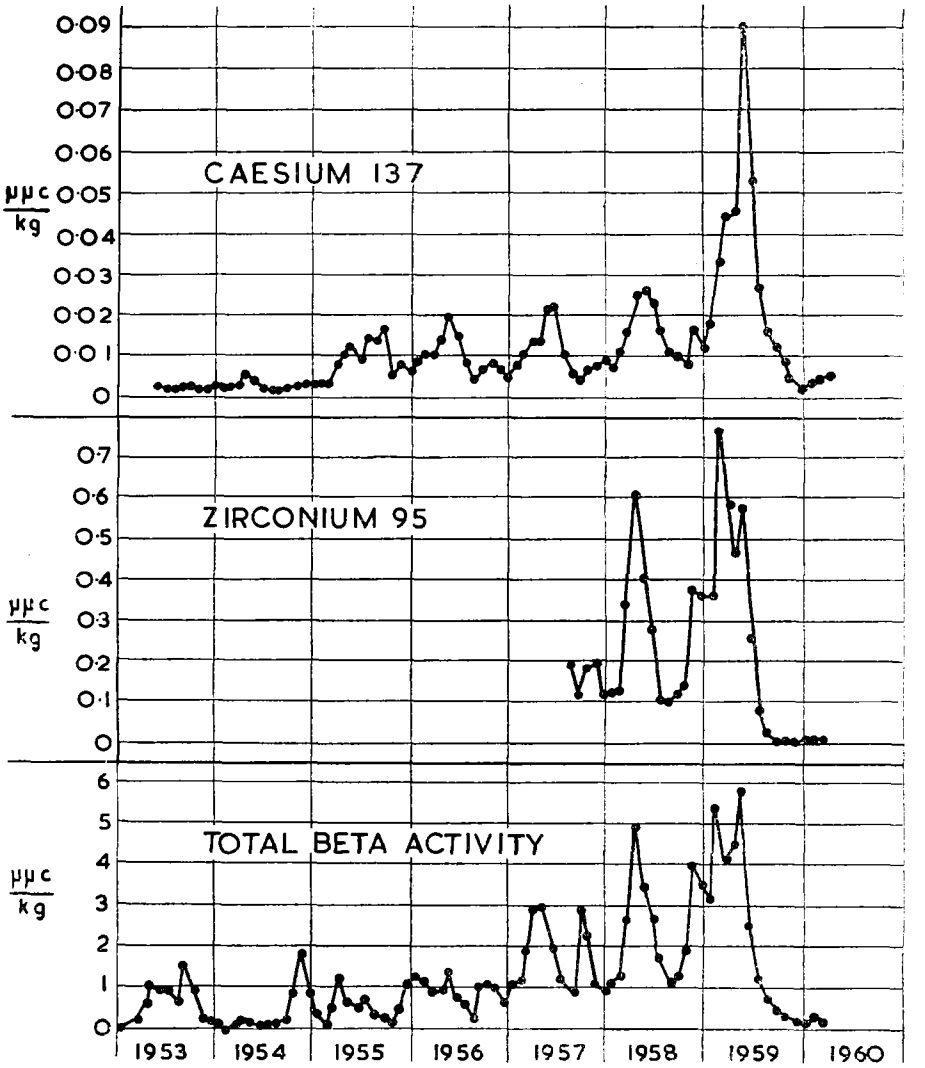

FIG. 1F.—Concentration of fission-product radioactivity in air at ground level at Chilton, Berks. (from Peirson *et al.*, 1960).

CHANGES IN THE RATE OF DEPOSITION OF LONG-LIVED RADIOACTIVITY

The rate of fall-out of strontium 90 in rain has been measured systematically at Milford Haven, Pembrokeshire, since 1954 (Stewart *et al.*, 1957, 1959; Crooks *et al.*, 1959a, 1960) and the results are summarized in Table 1F. The ratios of caesium 137 to strontium 90 and of strontium 89 to strontium 90 were also determined in the material deposited each month. Fig. 2F shows the apparent age of the strontium 90 at time of deposition calculated from the

Sr–89/Sr–90 ratio. The same workers have also made measurements over recent years at six other places in the United Kingdom and eighteen places in other parts of the world. The results show a maximum deposition in the 30°–50°N latitudes, relatively very small deposition in tropical latitudes, and an intermediate level in southern latitudes (see Figs. 5 (a) and 5 (b) of Stewart *et al.*, 1957). This pattern of deposition has been widely recognized (United Nations Scientific Committee, 1958) and is supported by extensive soil surveys (Alexander *et al.*, 1960).

The rate of fall-out of long-lived fission products in the U.K. changed little from the autumn of 1954 up to the spring of 1958. Between May, 1958, and July, 1959, it was about three times higher than for the previous three years. The increase in the summer of 1958 was due partly to heavy rainfall, which collected more of the debris from the atmosphere, and partly to increased testing. In late 1958 and early 1959 about half the enhanced fall-out was identified as originating mainly from the series of tests in October, 1958. From the estimates of the Ministry of Defence (1959), it appears that about the same quantity of fission products was injected into the stratosphere in 1957 and 1958 as during all the previous years combined.

TABLE 1 F

Change in the rate of fall-out of strontium 90 at Milford Haven

Period	Number of months in period	Total Sr-90 fall-out (mc/km²)	Average Sr-90 fall-out rate (mc/km²/month)
1954, May-December	8	1·97	0·25
1955, January-December	12	2·42	0·20
1956, January-December	12	2·47	0·21
1957 January-December	12	2·60	0·22
1958, January-April	4	0·98	0·25
1958, May-December	8	4·40	0·55
1959, January-July	7	4·74	0·68
1959, August-December	5	0·99	0·20
1960, January-April	4	0·62	0·16

Notes: (i) The rate of fall-out for the various periods is shown graphically in Fig. 1, p. 31.

(ii) The apparent age of the debris deposited in shown in Fig. 2F.

(iii) The measured ratio of Cs-137 to Sr-90 was found to be about 1·7 except for the last two periods when it tended to fall slightly.

There is considerable evidence that the deposition of the radioactivity at ground level in the United Kingdom occurs principally in rain and a comparison of the levels of fall-out at the six monitoring stations in the U.K. shows that the deposition at the various stations is roughly proportional to the rainfall (Stewart *et al.*, 1959). As a result of this relationship, it is possible to obtain a reasonable estimate of the deposition at any site in the U.K. from a knowledge of the local rainfall, although this relationship is not necessarily true for other climatic zones.

TRANSFER OF THE DEBRIS FROM STRATOSPHERE TO TROPOSPHERE

Direct measurements of the radioactivity in the stratosphere have been made in recent years by means of high-flying balloons (Machta and List, 1959) and jet aircraft (Feely, 1960). These studies have provided valuable new information about the distribution of the radioactive debris which is produced and the manner in which it reaches the earth's surface. It was pointed out by Stewart *et al.* (1957) that the behaviour of the debris closely matched that of the ozone formed by the effects of ultra-violet radiation in the upper atmosphere. The concentrations both of ozone and of the long-lived fission products in the air in the U.K. show a seasonal variation (see Fig. 1F) with a maximum in the spring and a minimum in the autumn. Recently seasonal fluctuations have also been observed (Anderson *et al.*, 1960b) in the concentration of the naturally-occurring isotope beryllium 7 produced by the effects of cosmic radiation in the upper atmosphere (Cruikshank *et al.*, 1956). These observations are in accord with a model for the stratospheric-tropospheric interchange of ozone and water vapour proposed by Brewer (1949, 1960) and Dobson (1956). Moreover, the distribution of fall-out, with a maximum in the northern temperate latitudes and a minimum at the equator, is also in accordance with the model.

It now appears that the radioactive debris from nuclear explosions reaches the lower atmosphere (from which it is readily deposited in rain) mainly by the sinking of air in the polar regions in the winter and by some diffusion in the middle latitudes where there is a discontinuity or step in the height of the tropopause layer which lies between the stratosphere and the troposphere. This mechanism would explain the marked seasonal variation of concentration in the air and of fall-out rate observed in parts of the northern hemisphere over several years (see Fig. 1 F). Moreover, on this basis, debris released into the stratosphere at high altitudes in tropical latitudes may take a considerable time, running into years, before it is in a position to be readily transferred to the lower atmosphere. On the other hand, debris released into the lower stratosphere in high latitudes, particularly in the autumn, would be transferred very quickly to the troposphere and heavy deposition would rapidly occur.

Another factor which undoubtedly affects the time of residence is the size of the explosion. Fall-out in the period up to mid-1956 originated largely from very large explosions in tropical latitudes. The heavy fall-out in early 1959, however, followed rapidly upon the firing in the autumn of 1958 of a considerable number of nuclear explosions in Arctic latitudes. The mean residence time of this debris has been estimated by Feely (1960) as only six to thirteen months, whereas the residence time of debris injected in tropical latitudes from very large explosions had earlier been estimated as ranging up to a few years (Stewart *et al.*, 1957). This change in the pattern of testing has greatly influenced the nature of the deposit which, from tests in Arctic latitudes in the autumn of 1958, has thus involved a high proportion of short-lived activity.

EXTERNAL RADIATION DOSE FROM DEPOSITED MATERIAL

Dose-rate in air in the open

It has been shown that in certain months in 1959 in the U.K. the level of external radiation in air in the open resulting from the deposition of short-lived

nucleides in fall-out ranged up to some 30 per cent of the natural background (Peirson and Salmon, 1959; Spiers, 1959; Vennait, 1960); this is much higher than the corresponding dose-rates in the period up to the Council's previous report in 1956. Spectrometric analysis by Peirson and Salmon (1959) of the gamma radiation from soil samples has shown that some 80 per cent of the dose-rate from the fission products arose from zirconium-niobium 95; the remainder was contributed largely by ruthenium 103 and rhodium 106 and the long-lived caesium 137. However, these measurements relate to dose-rate at one point in time. For the purpose of estimating the possible biological effect it is necessary to know the integrated tissue dose over an appropriate number of years. For genetic effects, for instance, the integrated dose over the 30-year period of one generation is probably the appropriate index. Accordingly, short periods of relatively high activity due to short-lived nucleides do not have corresponding significance.

The direct measurement of the dose-rate resulting from fall-out presents considerable difficulty and few direct measurements have been reported except for the period in 1959 when the additional dose-rate was at its greatest value. Accordingly, it is necessary to calculate this from the systematic measurements of deposition, and a knowledge of the fast fission-yields of the various nucleides.

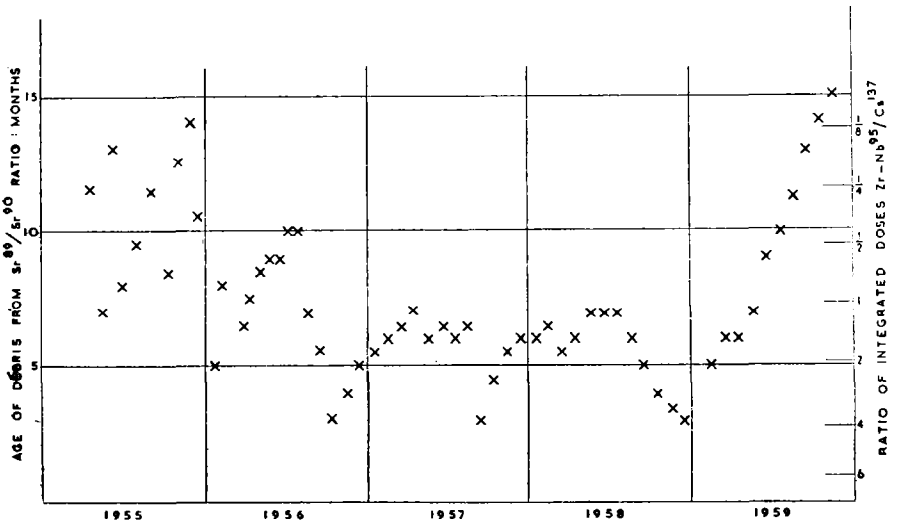

FIG. 2F.—Apparent age of fall-out debris at the time of deposition and relative dose contribution from $Zr\text{-}Nb^{95}$ and Cs^{137}.

The apparent age of the radioactive debris at the time of deposition can be determined from the Sr–89/Sr–90 ratio. Fig. 2F gives the data for debris deposited in the period May, 1954, to December, 1959, and also shows the ratio of the integrated dose from zirconium-niobium to that from caesium 137.*

*It is assumed that the levels of the short-lived nucleides are not affected by rain. Moreover, rain would not be expected to leach out the deposited caesium 137 from undisturbed ground sufficiently to affect the dose-rate appreciably. However, from roadways and other covered areas the deposited activity is known to be carried down drains and on agricultural land cultivation distributes the deposited activity through the soil to the depth of cultivation. In view of these considerations the assumption has been made, for the purpose of calculating the gamma dose-rate, that 5 per cent per annum of the deposited caesium 137 is removed from the superficial layer of the ground: thus the apparent half-life of the caesium 137 is about 10 years.

In the period up to July, 1956, as has already been pointed out, the debris had an apparent age ranging between seven months and fourteen months, and the external dose was thus contributed principally by the caesium 137. From about August, 1956, up to May, 1959, the age of the debris deposited was such that the integrated dose from the zirconium-niobium 95 would be about one and a half times that from caesium 137, except for three periods, namely, October–December, 1956, September–November, 1957, and September, 1958–February, 1959, during which the proportion of zirconium-niobium was such that the integrated dose would range up to four times the integrated dose from the caesium 137 in the deposit.

TABLE 2 F

Values of strontium 90 fall-out rate and fission-product age used for calculation of the external dose-rate

Reference date	†Rate of deposition of Sr-90 (mc/km²/month)	†Apparent age of fall-out (months)
Jan. 1956	0·22	12
Jan. 1958	0·22	6
July 1959	0·53	6
Jan. 1960	0·27	12

†Average over preceding months.

The probable contribution of the principal gamma-active fission products to the external dose-rate on various dates has been calculated below. For this purpose four dates have been selected in the interval between May, 1954, and January, 1960, for which in the immediately preceding period the fall-out rate of long-lived activities was broadly constant. The dates were 1st January, 1956, 1st January, 1958, 1st July, 1959, and 1st January, 1960. The accumulated fall-out of caesium 137 on these dates was calculated from the monthly measurements of fall-out. The dose-rate from the short-lived fission products was calculated in the following manner.

The values used in the calculation are given in Table 2F. The apparent age of the fall-out at time of deposition, expressed in round figures, was obtained from Fig. 2F. The fission yields for uranium 238 for 14 MeV neutrons (Cuninghame, 1955) were used to obtain the amounts of fission products relative to strontium 90 initially produced. From the apparent age of the fall-out, the amount of each fission product deposited on the ground associated with the strontium 90 and caesium 137 deposited was calculated. For the gamma-emitting fission products, other than caesium 137, the physical half-life was used in calculating the accumulation at the end of each period. (In some instances this is very close to an equilibrium level.) The dose-rate at one metre above a flat surface produced by the gamma radiation from each of the deposited fission products was then calculated. Scatter of the gamma radiation was taken

into account in these calculations by using the build-up factors given by Goldstein and Wilkins (1954). The specific dose-rates in air in millirads per year for a deposit of 1 millicurie per km^2 were taken as:

Cs–137	0·11	Ru–103	0·091
Zr–95	0·129	Ce–141	0·008
Nb–95	0·137	Ce–144 }	0·009
Rh–106	0·031	Pr–144 }	

TABLE 3 F

Dose-rates in air and in gonads on various dates resulting from external radiation from fall-out

Nucleides	Calculated values (mrads/yr)				*Measured values (mrads/yr)
	1 Jan. 1956	1 Jan. 1958	1 July 1959	1 Jan. 1960	July 1959
Cs-137	0·8	1·6	3·1	3·4	4·4
Zr-95 — Nb-95 ...	1·4	10·2	24·4	4·8	2·1
Ru-103 & Rh-106 ...	0·9	2·4	5·3	3·1	1·4
Ce-141 & Ce-144 ...	0·2	0·4	0·9	0·6	0·3
Total dose-rate in the open	3·3	14·6	33·7	11·9	27·1
Estimated mean gonad dose-rate† ...	0·4	1·8	4·2	1·5	—

* From Peirson and Salmon (1959).

† The effective shielding factor for gonads appropriate to U.K. conditions was assumed to be 8.

The results of these calculations are given in Table 3F. The dose-rates calculated relate to Milford Haven, Pembrokeshire, where the basic measurements were made. The rainfall at Milford Haven is about 37 in./year and the dose-rates at other places in the U.K. will be roughly proportional to the local rainfall. Calculations of this nature are of necessity based on a large number of approximations so that high accuracy cannot be attained. However, some detailed measurements made by Peirson in July, 1959, given in the last column in Table 3F, agree in general with the calculated values. Only in the case of the small contribution from ruthenium 103 and rhodium 106 does a large discrepancy occur and there may well be some leaching of the ruthenium and rhodium. The estimates of the total dose-rate from fission products can also be compared with measurements made by Spiers (1959) and Vennart (1960) at Leeds and Sutton respectively. When allowance is made for differences in rainfall, the highest values in both localities would correspond to approximately 37 mrads per year at Milford Haven; Vennart's results show, moreover, that some decrease had occurred before July and that the values continued to decline thereafter. Estimates of the total dose-rate due to fission products made by direct measurements and calculation are thus broadly in accord, and it may therefore be concluded that the data in Table 3F give a reasonable picture of the changing contribution of different gamma-emitting fission products to the total dose-rate in the open during the past 4 years.

Provided that no further nuclear explosions take place the contribution of zirconium-niobium should continue to decrease and in future the radiation dose will be primarily due to caesium 137. It is not possible to make any precise prediction of the future levels of external radiation, but some estimates have been made of the dose-rates in the future from the caesium 137 on the following assumptions: 18 months for the half-life of residence in the stratosphere, 10 years for the effective half-life of the caesium on the ground, and 0·34 mc/km^2/month (average of values for August–December, 1959) as the rate of deposition of caesium 137 at the beginning of 1960. The calculations show that the dose-rate from caesium 137 may be expected to rise slightly over the next few years due to further deposition from the stratospheric reservoir and later to decline to a level of 2·6 mrads/year in the open in 1970 and thereafter to decay with the effective half-life of 10 years.

Shielding factors and corresponding tissue doses

The tissue doses incurred from external radiation from fall-out are dependent upon the average shielding factors appropriate to houses and work places. The average shielding factor for the inside of a house, which depends on the siting and areas of rooms, doors and windows and the presence of neighbouring buildings, is difficult to assess. Stewart *et al.* (1955), using graphite-walled condenser chambers and a large-sector array of cobalt-60 sources, measured the shielding factor for a number of positions in an isolated two-storey brick building with walls 9 inches thick. For positions clear of direct radiation through the windows, protection factors of 22 and 13 were obtained in corner rooms on the ground and first floors respectively and up to 35 in a ground floor room with only one outside wall. In circumstances where direct radiation through a window occurred, the protection factor at head level was about 8. The gamma radiation from cobalt 60 (used by Stewart) is more penetrating than that from the fission products with which we are concerned in fall-out. The possibility of additional dose from the contamination of the insides of buildings has been investigated by Booker (1960), who measured the contamination from zirconium-niobium 95 of rugs and furnishings and showed that the dose from this cause was some 3–4 per cent of that outside.

From Stewart's measurements, it appears that a shielding factor of 20 is appropriate for a typical dwelling in the U.K. After allowance has been made for the time spent out-of-doors, a value of 5 for the effective shielding factor for buildings in the U.K. was adopted in the present calculations. A further factor of 1·6 is applied to allow for the shielding of gonads, bone and bone marrow by body tissues, as determined by Spiers from experiments with ' phantoms ' (see Appendix D). The corresponding gonad dose-rates resulting from external radiation from fall-out are shown in the last line of Table 3F. Although the tabulated values are for dose-rate it is important to recognize that it is the accumulated dose over the 30-year period of one generation which is relevant for assessing the genetic significance. The gonad dose-rates shown may be compared with measurements by Spiers (1959) who concluded that the additional dose-rate in the period following tests conducted at the very high rate of autumn 1958 may have been as little as 2 mrads/year ranging up to a possible value of 10 mrads/year.

ENTRY OF RADIOACTIVITY INTO THE BODY BY INHALATION

Quantity of radioactivity inhaled

Peirson *et al.* (1960a) have measured the concentration in air at ground level of the principal gamma-emitting fission products and the total beta activity. The average daily intake of various isotopes by breathing has been calculated from these data and is shown in Table 4F for the years 1955–1959. The values for strontium 90 are calculated from measurements of caesium 137; the ratio Cs–137/Sr–90 in the fall-out material was observed to have been about 1·7.

It will be seen that the average daily inhalation of strontium 90 during the six months January–June, 1959, was calculated to be 0·65 μμc/day. Similar measurements of the concentration of some prominent fission products in air during May–December, 1959, were made by Anderson *et al.* (1960a, 1960d), who later obtained a mean value of 0·64 μμc/day for strontium 90 for May and June in close agreement with the calculated figure given in Table 4F.

TABLE 4 F

Average daily inhalation of fall-out nucleides in various years

	Micro-microcuries per day						
	1955	1956	1957	1958	1959 Jan.-June	1959 July-Sept.	1959 Oct.-Dec.
Total beta fission products	13·3	20	32	57	100	17	4
Cs-137	0·22	0·24	0·26	0·36	1·1	0·43	0·12
Sr-90 (calc.)	0·13	0·14	0·15	0·21	0·65	0·25	0·07
Zr-95	—	—	3·9*	6·2	12	0·8	0·06
I-131	—	—	—	3†	—	—	—
Pu-239 (calc.)	0·0020	0·0021	0·0023	0·0032	0·010	0·0038	0·0011

*August-December. †October-November.

The apparent age of the fission-product mixture can be assessed approximately from the Zr–95/Cs–137 ratio and also derived from Sr–89/Sr–90 ratio (Fig. 2F). It is then possible to compile a list of the proportions of various isotopes present in the debris. This would be reasonably accurate in the range of the half-lives of strontium 89 (50 days) and zirconium 95 (65 days) but may be in error for the very short-lived isotopes. However, from such a list it can be deduced that, for the period of highest air activity (January–June, 1959):—

(*a*) The quantities of strontium 90 and caesium 137 inhaled are small. The total amount of strontium 90 inhaled per day was but a few per cent of that ingested (see p. 94) and relative amounts taken up by the body by each route will be in comparable ratio.

(*b*) The average level in air calculated for cerium 144, for which the uptake by the body from the diet is very small, was about 1/200 of the permissible average concentration for the population at large (I.C.R.P., 1959).

(*c*) The average level calculated for other isotopes in air was less than 1/700 of the permissible average concentration for the population at large.

The last line in Table 4F gives calculated values for the daily inhalation of plutonium 239. These were based on radiochemical analyses of air filters obtained in the period May–August, 1959, giving the ratio of Cs–137 to Pu–239 (Peirson *et al.*, 1960a). The average ratio for 15 samples was 113 which corresponds to a yield of approximately 0·5 atoms of Pu–239 per fission. It has been assumed that this ratio is valid throughout all the previous fall-out.

It will be seen that during the period of highest contamination of the air, namely, January–July, 1959, the plutonium inhaled in the U.K. was about 10^{-2} μμc per person per day. This is 1/40 of the 0·4 μμc/day permissible as the average daily intake of the population as a whole (I.C.R.P., 1959). The dietary intake can be neglected since there is no evidence of any concentration of plutonium in foodstuffs and the fractional uptake from the gut into the body is only 3×10^{-5} (I.C.R.P., 1959). Indeed, by ingestion an occupational worker is permitted an average daily intake of as much as 100,000 μμc/day.

Measurements made by Anderson *et al.* (1960b) show that, in February, 1960, the amounts of the main gamma-emitting nucleides in air had fallen to those values given in Table 5F, which includes not only fission products but also some naturally-occurring radio-elements produced by cosmic radiation. The isotope of antimony (Sb–125, half-life 2·4 years) was prominent, probably owing to fissions caused by 14 MeV neutrons. At this time the levels of the nucleides listed are similar to that of caesium 137. It is interesting that the levels of the gamma-ray-emitting fission products had, by February, 1960, fallen below that of beryllium 7, a relatively short-lived material continuously produced by natural processes of interaction between cosmic radiation and the upper atmosphere. However, the intake of these materials is some 10,000 times lower than the continuous intake of the decay products of radon which is normally present in air at the level of about 10^{-1} μμc/litre. Small amounts of material incorporated into nuclear devices as tracers, for example Rh–102 and W–185, can also be detected in the atmosphere.

TABLE 5 F

Comparison of concentrations in air in February, 1960, of fission products with nucleides produced naturally by cosmic radiation

	Half-life	Concentration in air (μμc/litre)	Average daily inhalation Feb. 1960 (μμc/day)
Fission product			
Ru-106	370 d	4×10^{-6}	0·08
Sb-125	2·4 y	$2{\cdot}5 \times 10^{-6}$	0·05
Cs-137	30 y	$\sim 10^{-6}$	0·02
Ce-144	285 d	7×10^{-6}	0·14
Zr-95	65 d	$< 10^{-6}$	0·02
Cosmic-ray-produced nucleides			
Be-7	53 d	2×10^{-5}	0·4
Na-22	2·6 y	10^{-8}	0·0002
P-32	14 d	5×10^{-7}	0·01
P-33	24 d	2×10^{-7}	0·004
S-35	87 d	10^{-6}	0·02

Particles and inhaled activity

Details were given in the 1956 report of the nature of radioactive particles found in the atmosphere; these had arisen mainly from small explosions occurring in Nevada. They were often in the region of 1 micron in size and were composed largely of fused silica with entrapped fission products. The particles which we now describe are from the explosions of megaton size and are thus likely to contain less extraneous matter.

Autoradiographs of recent dust samples collected by means of an electrostatic precipitator have been obtained by Mayneord *et al.* (unpublished data). They found that the maximum concentration of particles containing fission products occurred in May, 1959; the concentration at that time was approximately 1 particle in each 70 litres of air at ground level, the activity of the particles apparently falling mostly in the range 0·1–1 μμc/particle. It will be seen that this is consistent with the measurements of gross activity in Table 4F. There may have been many more particles of lower radioactivity, and in the period immediately following the heavy testing in October, 1958, instances of particles of much higher specific activity ranging up to 1,000 μμc/particle have been reported by French and German investigators (Schriftenreihe des Bundesministers für Atomkernenergie und Wasserwirtschaft, 1959).

Mayneord *et al.* (unpublished data) have made observations on the beta activity of human pulmonary lymph nodes. In a series of 40 such measurements made during May–June, 1959, there appeared to be a small amount of activity detectable beyond that due to the natural potassium 40, but it was not greater than 1μμc/lymph node, and the specific activity of the lung ash was similar to that of the lymph nodes. In pulmonary lymph nodes collected in 1960, they were unable to detect any increased beta activity over that due to potassium 40.

One of the most difficult problems at the present time is to assess the significance of radioactive particles in the air. The difficulties arise from both physical and physiological considerations. Such particles are inhaled into the lungs but their retention or elimination is determined by a wide range of phenomena depending upon such factors as particle size, chemical nature, density and solubility of the particles, as well as the frequency and depth of breathing. Even if the particle is caught in the respiratory tract it may be removed by ciliary action and eventually swallowed. Some particles trapped in the lungs may remain there for relatively long periods of time, perhaps even weeks or years, but, on the other hand, many will move to other parts of the lungs, to the local lymph nodes or perhaps, by the systemic circulation, to other parts of the body.

In the immediate neighbourhood of a radioactive particle there is a localized high dose with a very complex but limited micro-distribution. Correspondingly the absorbed energy is concentrated upon only a relatively small number of cells, this concentration of absorbed energy being formidable but being of extremely limited volume. As the particles move about they carry their radiation field with them and, although we can make an approximate estimate of the total dose in the vicinity of a fixed particle, it is clearly very difficult to estimate the biological effects of particles probably moving in very complex ways. A

particle from the atmosphere with a diameter of perhaps 1 μ and having a total radioactivity of 1μμc of high-energy beta rays might irradiate at a given time a group of cells whose radius is 500 cell diameters. The total mass of material within the range of the beta particles is about 0·5 gramme but the increase in mean dose throughout the irradiated cells would be only about one-third of natural background. However, cells extremely near the particle for a long time may receive amounts of energy which would almost certainly be destructive. We might perhaps estimate that a relatively small number of cells, of the order of a thousand, might be killed, but it must be remembered that the human body is destroying many millions of times this number per day in natural processes. It is not, therefore, at all certain that the destruction or damage of small groups of cells has any significance to the body or to single organs.

ENTRY OF RADIOACTIVITY INTO THE BODY BY INGESTION

Most of the fission-product elements are not absorbed to any significant extent from the gut. This is, however, not true for strontium and caesium and ingestion is the most important route by which these elements, particularly Sr–89, Sr–90 and Cs–137 from fall-out, gain access to the body. The same situation could occur with iodine 131. However, the relatively small quantities which have been deposited, due to its short half-life, make its passage through the food chain relatively unimportant in an assessment of the overall effects of fall-out in the U.K.

Strontium 90 *in the Food Chain*

Preliminary surveys

Investigations of the extent to which materials in the food chain were being contaminated with strontium 90 were initiated in 1954 by the Atomic Energy Authority. At that time the measurement of the strontium 90 content in soil, diet and particularly in human bone necessitated stretching to the limit the existing methods of radiochemical analysis, and technical development was therefore necessary before full-scale investigation could be undertaken. The first publications were concerned with the levels of radioactivity in soils, vegetation, sheep bones and milk for the year 1955 although isolated results were available for 1954. More comprehensive investigations were made in 1956 (Bryant *et al.*, 1956, 1957), and preliminary information from this source was given in the Medical Research Council's 1956 report. The work was extended and expanded in 1957 (Bryant *et al.*, 1958a, 1958b).

The chemical similarity between strontium and calcium and the nature of the average diet in Great Britain led to the conclusion that the intake of strontium 90 by the average person, and particularly child, would be determined largely by contamination of milk and milk products; these were therefore sampled in several parts of the country. In 1957 other articles of diet were also examined from four localities and it was possible for the first time to consider the average content of strontium 90 in the total diet; this was estimated as 5·5 μμc/g Ca (Bryant *et al.*, 1958c).

Several conclusions which emerged from this work were of particular importance in guiding further studies. Milk was shown to provide nearly 60 per cent of the total strontium 90 in the average mixed diet of the population. A sequence of dried milk samples from a large milk depot in an area of moderate rainfall showed that during the period 1955–1957 the levels in milk rose only slightly and remained in the range of from 4 to 5 μμc/g Ca despite a steady rise in the accumulated deposition of strontium 90; this was the first direct evidence that the rate of fall-out, as opposed to the accumulated total in the soil, might be an important factor in determining the contamination of human diet. Over the major part of the country the strontium 90 content of herbage and milk bore a general proportionality to rainfall, but in localized wet hill areas the levels of strontium 90 both in milk and herbage exceeded those observed in drier parts of the country to a considerably greater extent than could be attributed to differences in the deposition of fall-out, as calculated from the rainfall.

Systematic surveys

In 1958 considerably extended investigations were commenced with the objectives of obtaining more precise information both on the mean content of strontium 90 in the average diet of the total population and on the maximum level which any individual might receive in areas where the contamination of diet was above average. The new programme was planned by a committee nominated jointly by the Agricultural and Medical Research Councils and carried out by the former body. The methods adopted and the results for 1958 and 1959 have been reported by the Agricultural Research Council (1959, 1960a, 1960b). With assistance from the Ministry of Agriculture, Fisheries and Food and the corresponding Departments in Scotland and Northern Ireland, elaborate sampling schemes were devised to obtain representative samples of foods which constitute the major sources of strontium 90 in diet. In the country-wide milk survey samples were collected and combined, according to a statistical plan, from depots which handle over 40 per cent of the country's total milk production. Grain, green vegetables and potatoes were also sampled on an extensive scale. In addition, some samples of eggs, cheese and tea were examined.

The country-wide average levels of strontium 90 in milk were 7·0 and 9·8 μμc/g Ca for 1958 and 1959 respectively. A comparison of the results for 1958 with those reported for 1957 by Bryant *et al.* (1958b), suggests relatively little difference between these two years, but in 1959 there was a rise of 40 per cent; a comparable increase for a depot in South Wales was observed in an independent investigation by Anderson *et al.* (1959, 1960c).

The estimated total daily intake of strontium 90 is given in Table 6F and is derived from the average diet with small added contributions from drinking water and inspired air. The ratios of strontium 90 to calcium in the daily total intake were 6·1 and 9·3 μμc/g Ca in 1958 and 1959 respectively. There is evidence that the country-wide average was not exceeded by more than 50 per cent in any large section of the country. These figures may be compared with the estimate of 5·6 μμc/g Ca for 1957 made by Bryant *et al.* (1958c).

TABLE 6F

Estimation of mean daily intake of strontium 90 *by members of the population of the United Kingdom,* 1958 *and* 1959

Source of the Sr-90	Estimated Sr-90 intake (μμc/day)	
	1958	1959
Diet(i)		
Milk, cream and cheese	4·20	5·57
Root vegetables	0·35	0·58
Leaf and other vegetables, fruit, etc.	0·54	0·88
Flour and cereals of all types	0·75	1·32
Eggs, meat, fish and other sources	0·32	0·55
Diet total	6·16	8·90
Drinking water(ii)	0·23	0·43
Tea(i)	—	0·45
Air breathed(iii)	0·16	0·30
Total daily intake ...	6·55	10·08

The mean daily intake of calcium is estimated as 1·084 g/day (iv). Hence the ratio of Sr-90 to calcium in the total daily intake was—

1958: 6·1 μμc/g
1959: 9·3 μμc/g

Notes: (i) From Agricultural Research Council (1959, 1960b).

(ii) From Crooks *et al.* (1959b), assuming intake of 1 litre per day.

(iii) From Table I, Peirson *et al.* (1960a), assuming breathing rate of 2×10^7 cc per day and assuming 25 per cent of the inhaled strontium 90 is immediately exhaled. (See also Table 4F, p. 89).

(iv) Data provided by Ministry of Agriculture, Fisheries and Food.

To estimate the maximum levels of strontium 90 from world-wide fall-out to which any individual may be exposed, the Agricultural Research Council also carried out investigations in a number of localized wet hilly areas where a transfer of strontium 90 through the food chains is known to be particularly high. The mean level in milk observed in these localities ranged up to nearly eight times the national average. A calculation, based on most pessimistic assumptions, suggests that the maximum level of strontium 90 which could occur in the diet of any individual was about 80 μμc/g Ca in 1959; there is no evidence, however, that this level was in fact attained by any individual.

Experimental studies

Simultaneously with surveys of the current levels of strontium 90 in diet, experimental studies have been carried out to investigate the mechanisms whereby strontium 90 passes through food chains, and particularly to assess the relative extent to which the contamination of diet depends on recently deposited material, that is to say on the rate of fall-out, as opposed to the accumulated total in the soil.

Because many factors can cause seasonal variations in the extent to which strontium 90 is absorbed by plants, precise information on the extent to which the contamination of diet is determined by the rate as opposed to the accumulated total cannot be obtained by examining relationships between the levels observed in dietary surveys and the pattern of fall-out during the relatively short period for which investigations have been in progress. More precise information is provided by experimental studies. In the United Kingdom results are now available for large-scale controlled field experiments with either natural fall-out or tracers (Agricultural Research Council, 1960a; Milbourn *et al.*, 1959, 1960; Morgan and Stanbury, 1959), as well as for experiments on soils and plants in the laboratory (Middleton, 1959; Morgan, 1959a and 1959b; Russell and Squire, 1958; Russell *et al.*, 1958). These results provide some basis for interpreting the situation in the United Kingdom (Burton *et al.*, 1960; Russell, 1958, 1960). Related studies have been undertaken under the contrasting climatic conditions of other countries, notably the United States. The Food and Agriculture Organization of the United Nations (1960) has recently published the comprehensive report of an expert committee drawn from many countries.

Mechanisms of dietary contamination

Plants become contaminated by several mechanisms (Food and Agriculture Organization of the United Nations, 1960). Some fall-out may be directly retained by leaves and by inflorescences (foliar and floral contamination), some may be washed down and lodge near the base of the plant where it will subsequently be absorbed close to but above the true soil surface (plant-base absorption), and the remainder passes into the soil from which it may enter plants through the roots. Whereas the extent of foliar and floral contamination depends on the extent of fall-out during the growth of the tissue, contamination by root-uptake is related to the accumulated total in the soil, though in a complex manner. Plant-base absorption occupies an intermediate position in this respect since in perennial plants strontium 90 may be retained for appreciable periods in the plant-base reservoir. Moreover, strontium 90 entering by the foliar or floral routes is unaccompanied by calcium; that which is absorbed from the soil has been effectively diluted to a greater or lesser extent by soil calcium; material from the plant-base reservoir may be slightly diluted with calcium from the surface soil though this will not necessarily be the case. No hard-and-fast boundary can be drawn between the foliar, plant-base and soil routes of entry; however, their contrasting characteristics with regard both to the age of strongtium 90 which can enter and its dilution with calcium make it important to distinguish between them.

Experimental studies have shown that the relative importance of these mechanisms of entry varies greatly, depending on many factors and especially on the growth-form of plants, on the climate and on the characteristics of the soil. Largely in consequence of plant-base absorption, the ratios of strontium 90 to calcium in perennial pasture grasses often greatly exceed those in the leaves of annual plants. If contamination occurs after the emergence of the ears of grain crops, floral entry may cause the ratios of strontium 90 to calcium in grain to exceed greatly those in all other articles of diet.

It appears that there are no large inherent differences in the ability of plants to absorb strontium relative to calcium through their roots from the soil but the

slow penetration of strontium 90 into the soil has caused the roots of shallow-rooted plants to be exposed to a higher ratio of strontium 90 to calcium than those of deep-rooted species. Cultivation of the soil thus can reduce the extent of absorption of strontium 90 by shallow-rooted plants because of more thorough mixing. The quantity of freely soluble calcium on the soil can influence the ratio of strontium 90 to calcium in plants in two ways; low soil calcium reduces the calcium content of plants and also increases the absorption of strontium 90 from the soil. If, however, soils contain moderate or high levels of calcium, these effects are small; and most productive soils in the United Kingdom are in these categories.

The extent to which strontium 90 enters into plants from soil in the U.K. has been studied in a series of field experiments for a number of years in different parts of the country (Milbourn *et al.*, 1959; Milbourn, 1960). It is therefore possible to make some estimate of the levels of strontium 90 in plants and hence in milk which would be expected in consequence of the accumulated total of strontium 90 from world-wide fall-out hitherto present in soil. Burton *et al.* (1960) deduce that in 1958 the quantity of strontium 90 in milk was some five times greater than could be attributed to absorption of strontium from the soil into cattle fodder. This implies that about 80 per cent of the strontium 90 in milk was due to direct contamination of the fodder, either foliar or plant-base. A comparable conclusion was reached in 1959 (Burton, 1960). The approximate nature of these assessments was emphasized and it is apparent that large quantitative differences may occur between different localities and on different occasions.

Much qualitative information provided by diet surveys also shows the importance of recently deposited strontium 90 in determining the levels of strontium hitherto observed in diet in the U.K. A comparison of the ratios of strontium 89 to strontium 90 in milk during 1958 (Agricultural Research Council, 1959) with the ratios simultaneously observed in rain (Stewart *et al.*, 1959; Crooks *et al.*, 1959a) indicated that the apparent delay period in the passage of strontium 90 from rain to milk was about one to two months during the summer. The situation appeared to be comparable in North America (Russell, 1960). Evidence of the importance of the recently deposited strontium 90 in determining the contents of green vegetables and flour has also been obtained (Agricultural Research Council, 1959).

When the Medical Research Council prepared its report in 1956, little information was available on the mechanism whereby strontium 90 passes from the atmosphere into human diet and it was prudent therefore to accept provisionally the more cautious assumption that the accumulated total in the soil might be the predominant factor. It is, however, now apparent that the amount of strontium 90 from fall-out which has entered into human diet in the U.K. hitherto has been determined mainly by the quantity deposited in the recent past and not by the accumulated deposition in the soil. There is, however, no valid basis for predicting how rapidly the present decreasing levels of fall-out will be reflected in diet. Not only can climatic factors exert a large effect but a lag period, which cannot be accurately estimated, will be caused by the consumption of stored food. Cattle subsist for a significant part of the year on hay and other materials produced in the previous summer and important components of human diet, especially flour and tinned foods, may be stored for a considerable period before consumption.

If, on the other hand, future injections of strontium 90 into the stratosphere were to maintain fall-out at levels comparable to those which have occurred in the past, only a relatively slow increase would occur in contamination of diet. However, whatever be the future pattern of fall-out, the increasing magnitude of the accumulated deposit relative to recent fall-out will cause the contribution from the soil to increase. The extent to which diet will be contaminated from this source should be small relative to that caused by direct contamination in periods of high fall-out but it will decrease only slowly with time because of the slow loss of strontium 90 from soil.

When it was considered that the contamination of diet with strontium 90 might be due mainly to absorption from the soil, it was reasonable to conclude that there would be large differences in the average ratio of strontium 90 to calcium between different diets, depending on the extent to which dietary calcium is derived from milk. Under these circumstances the discrimination factor of about 10 against strontium relative to calcium in the passage from the diet of cattle to milk (Comar *et al.*, 1957; Food and Agriculture Organization of the United Nations, 1960) could cause the mean ratio of strontium 90 to calcium in a normal European diet, in which over 50 per cent of the calcium is provided by milk, to be some five times greater than that in diets low in milk, for example, in rice-eating countries. So long, however, as the rate of fall-out is of major importance in determining the levels in diet, this situation will not arise; the effect of the discrimination factor against strontium in its passage to milk can be offset in large measure by the enhanced direct contamination of permanent pastures in which cattle graze. Thus, hitherto in the U.K. only small differences have been observed between the ratios of strontium 90 to calcium in milk and in leafy vegetables (in 1958, 7 and 8·7 respectively and in 1959, 9·5 and 14·3) (Agricultural Research Council, 1959, 1960b). Although data from many parts of the world are meagre, there is no reason to conclude that differences in the composition of diet between different countries are at present the major cause of variation in the mean ratio of strontium 90 to calcium consumed.

Caesium in the Food Chain

Unlike strontium 90, caesium 137 is usually retained in the soil in forms which make it largely inaccessible to plants. The caesium 137 in diet is thus attributable to entry through the above-ground parts of plants by the alternative routes referred to in the preceding section (Food and Agriculture Organization of the United Nations, 1960). Its distribution within plants, however, can contrast markedly with that of strontium 90 which has been absorbed in the same manner. Unlike strontium, caesium is readily redistributed within plants upon which it falls (Middleton, 1959).

Caesium is also relatively readily transferred from the diet of cattle to milk (Garner, 1960) and it is known that milk and meat are the two main foods in which it enters the human diet. In Western countries milk appears to be the most important source (Langham and Anderson, 1959). The Atomic Energy Authority (Booker, 1957, 1959) have reported the levels of caesium 137 in milk for a number of localities in the United Kingdom; more recently similar observations have been made by Mayneord *et al.* (1959) and Anderson *et al.* (1959, 1960c) from a single locality. Sampling has not been sufficiently extensive for the average level in the total milk supplies of the country to be

estimated from these data but, since the more extensive series of samples are from areas where the rainfall is appreciably above the national average, the results are also likely to be above average. From early 1956 up till October, 1957, when the Windscale accident occurred, there was little evidence of change and the values seldom exceeded 50 μμc/g K. (Milk contains about 1·6 gramme potassium per kg.) In October, 1957, the Windscale accident led to sharp increases, sometimes by threefold even as far away as South Wales, but the values returned within a few weeks to approximately the previous level. In the spring of 1958, however, a general increase occurred and until the middle of 1959 values were in the range of 100–150 μμc/g K in an area of moderately high rainfall. The changes in levels of caesium 137 in milk during the past four years thus again more clearly reflect variations in the extent of recent deposit rather than in the accumulated total.

RADIATION DOSES TO TISSUES RESULTING FROM INTERNALLY ASSIMILATED ISOTOPES

Strontium 90

The analyses of human bones for strontium 90, which have been carried out on an increasing scale by the Woolwich Outstation of the Atomic Energy Research Establishment, provide direct evidence of the rise in the human bone levels. Furthermore, in 1959 additional analyses (unpublished) have been made by an independent group in Glasgow (Graham, Lenihan, Macdonald and Warren). Where the series overlap there is general agreement. The results of the Woolwich Laboratory have been published *seriatim* by Bryant and his colleagues (1956, 1958a, 1959a, 1959b) and the results for the period 1955–1958 have been reviewed by Bailey *et al.* (1960). With the latest serial report by Arden *et al.* (1960), together with the further unpublished results available up to the end of May, 1960, from this Laboratory and from Glasgow, a more extensive amount of information is available and the data are given in Fig. 3F.

It is not possible to choose human bone for assay on a statistically random basis. The samples were obtained at routine autopsy by pathologists in various parts of the country but especially in the Western and Northern regions of higher than average rainfall. Material from infants and young people was specially requested, since the turnover of bone is greatest in the growing skeleton; and the femur was selected as the standard bone for assay. In the femur of adults, which consists largely of dense bone, the calcium may well be replaced more slowly than in other bones having a greater proportion of spongy bone; however, in the very young there seems to be little difference in this respect between bones.

The newborn

Until 1957 few analyses were made of the bones of infants who were stillborn or who died within the first few days or weeks of life. It appears, however, that between 1956 and the middle of 1958 the values for these bones changed relatively little and were usually around 0·5 μμc/g Ca. In the latter part of 1958 they rose appreciably and in 1959 the average was just over 1 μμc/g Ca. The trend in the past three years thus appears comparable with the rate of fall-out of strontium 90, but the levels are about 8–10 times lower than those measured in or deduced for the mixed diet of adults.

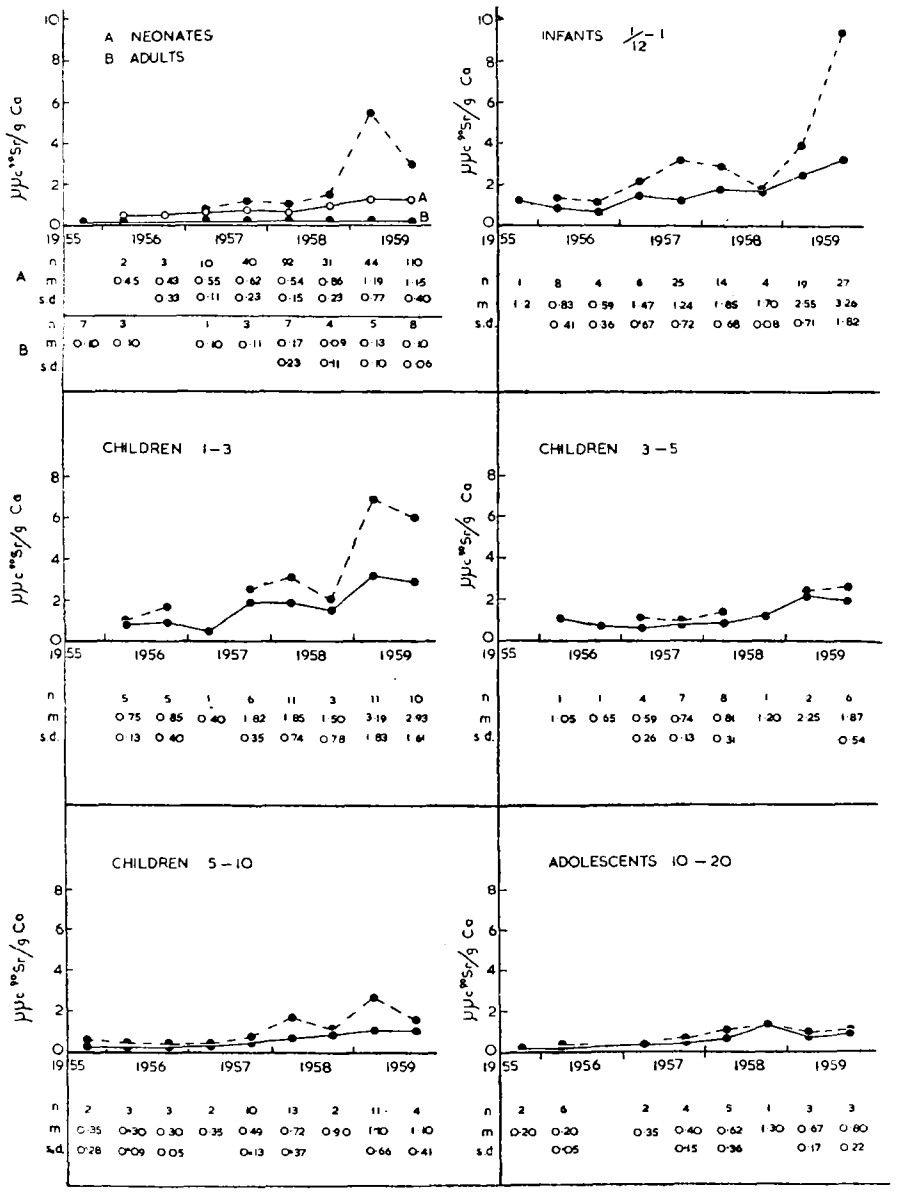

FIG. 3F.—The concentration (μμc per gramme of calcium) of strontium 90 in human bone, 1955-1959.

Notes: (i) In each age group the mean values are linked by a continuous line and the maximum values by an interrupted line. In the first category the lower continuous line (B) relates to the bones of adults and no maxima are drawn as they differ so little from the mean; the upper continuous line (A) relates to the mean values and the interrupted line to the maximum values for stillborn infants whose bones are formed indirectly from the maternal plasmas.

(ii) The subscript tables for each category give: (n) the number of individuals; (m) the mean value in μμc per g calcium; (s.d.) the standard deviation of the values in μμc per g calcium.

The data may be interpreted as follows. The foetal bones at the time of ossification receive supplies of calcium and strontium from the maternal plasma, but across the placenta there is a preferential transfer of calcium, relative to strontium, of some two-fold. The ratio of strontium to calcium in the mother's plasma is about one-quarter of that in her diet. The product of the two factors results in the infant's bones having a strontium: calcium ratio 8 to 10 times lower than the mother's diet.

From the evidence of the newborn it appears that the concentration of strontium 90 in adults' plasma was relatively constant in 1956-1958 and rose only in the latter part of that year to a new level.

Infants

The values for the bones of infants between one month and one year of age have been, in general, considerably higher than those of the newborn though at any one time the values have been rather scattered. Between 1956 and 1959 there was a progressive rise in the average value from about 0·75 to about 3 $\mu\mu$c/g Ca with some indication that the rise was most marked in 1959. A likely explanation is that the rise in 1959 is related to the increased rate of fall-out which preceded it; whereas the increase in values during the early period, when the rate of fall-out was fairly constant, can be attributed to the fact that an appreciable proportion of the diet of the infant population often consists of materials such as dried milk and cereals which are produced a considerable time before they are consumed. In 1956 and 1957 these components of diet would have been still relatively uncontaminated. Because the diets of infants are so variable, some being based on breast milk, some on fresh cows' milk and some on dried milk, no precise interpretation is possible, though on this account the variability of results at any one period is readily understandable.

Children aged 1 *to* 3 *years*

Fewer samples have been available from children between 1 and 3 years of age. The mean values, which reached about 3 $\mu\mu$c/g Ca in 1959, have been similar to those of infants. The diets of these children are transitional between those of infants and adolescents.

Older children and adolescents

Progressively decreasing values have been observed with increasing age. For children aged 3 to 5 the mean value was approximately 2 $\mu\mu$c/g Ca in 1959 while the corresponding figures for children aged 5 to 10 and for adolescents aged 10 to 20 were about 1 and 0·7 $\mu\mu$c/g Ca respectively. The numbers in all age groups were small owing to the low death rate of children over the age of 1 year and the data are insufficient for the rates of increase in different age groups to be compared with precision. Steadily increasing values are, however, apparent in all groups. This is to be expected because of the progressive growth and re-modelling of their bones. Starting with virtually uncontaminated bone before 1954 they have added strontium 90 and replaced uncontaminated with contaminated bone salt each year. Though the levels have been rising with the years, current values are still well below those of infants and very young children all of whose bone has been formed during the period of fall-out.

Adults

The relatively few assays of adult femora which have been carried out give considerably lower values than for any other group. This indicates that only a small part of the bone salt has been renewed since fall-out commenced. The recorded results have been constant around 0·1 μμc/g Ca from late 1955 to late 1959. As noted above, however, the levels in adult plasma, as deduced from the results on bone of the newborn, may well have been about 1 μμc/g Ca from 1956 to 1958 and some 2 μμc/g Ca in 1959. Direct analyses of two large plasma pools obtained from blood drawn between November, 1958 and March, 1959 gave results of 1·1 and 1·4 μμc of Sr–90 per gramme of calcium (Harrison *et al.*, 1960).

Caesium 137

The estimation of internal contamination with this material can be performed on the living subject, since the short-lived barium 137, to which caesium 137 is transformed on disintegration, emits a gamma ray of energy 0·66 MeV with a branching ratio of 0·91. The procedure involves placing the washed and externally cleansed subject in a 'whole-body counter'. The emission of gamma rays from the body is detected by scintillation counters, the outputs of which are fed to multi-channel pulse height analysers. The procedure permits the assessment of contamination not only with caesium 137 but with natural radioactive materials, but is very time-consuming both to the subject and to the operator. This, combined with the cost and complexity of the equipment, means that only relatively few measurements can be made.

Limited numbers of such measurements made between 1956 and 1958 at the Atomic Energy Research Establishment have already been reported by Rundo (1957, 1958a, 1958b). Further results for 1958 and 1959 are to be published by this laboratory and from the Radiological Protection Service.

Until the middle of 1958 there was but little change in the general level of measured values. The mean values expressed as μμc Cs–137 per gramme of potassium rose from 33 to 43 and periodic measurements in the same individuals showed only a slight tendency to rise. Since then it would appear that values have risen somewhat further to some 50–70 in the same units. If it be assumed that Cs–137 was uniformly distributed in soft tissue, the dose in soft tissue would be between 1 and 1·5 millirads per year.

Iodine 131

Few determinations of the radiation dose to human thyroids due to iodine 131 in fall-out have been made in this country. These measurements can only be made during periods of fall-out of fresh fission products because iodine 131 (half-life, 8 days) decays rapidly before it is deposited on the ground. Robertson and Falconer (1959) from measurements on 10 individuals in London obtained an average value of dose to a human thyroid of 6 mrads in a 4-week period from mid-December, 1958, to mid-January, 1959. Calculations based on the experimental observations of Garner (1960) and Cox and Morgan (1959) indicate that with similar fall-out at another season of the year the resulting doses in human thyroids might have been 10 times greater. Cox's and Morgan's results also show that in 1958 there were only two periods in which considerable iodine deposition occurred, namely, March–April and November. These correspond

with the peaks in sheep thyroid activity found by Robertson and Falconer. The values for thyroid dose are small compared to the permissible thyroid dose (1,000 mrads/yr) even to large populations proposed by the International Commission on Radiological Protection.

THE DISTRIBUTION OF NATURAL AND ARTIFICIAL CARBON 14 AND THE RESULTING TISSUE DOSES

Natural Carbon 14

Natural radiocarbon is produced as a result of bombardment of the earth's atmosphere by cosmic rays. Primary cosmic radiation, predominantly protons with energies well above 10^9eV, causes complete nuclear disruption of atoms in the atmosphere. As a result of this process about 2 neutrons/cm^2/sec (Gabbe, 1958) are produced in the atmosphere in the higher geomagnetic latitudes and a somewhat lower number near the magnetic equator. Most of the neutrons produced in the atmosphere will be captured by nitrogen nuclei in the reaction

$$^{14}N + n = {}^{14}C + p + 620 \text{ keV}.$$

The product of this reaction is carbon 14 with a half-life of 5568 $\pm$ 30 years (Strominger *et al.*, 1958).

Libby (1955) has shown from his work on radiocarbon dating that it is most probable that the intensity of cosmic radiation has remained at its present level for at least 20,000 years; therefore natural radiocarbon is in radioactive equilibrium, that is, production is balanced by radioactive decay.

The natural radiocarbon produced in the atmosphere is rapidly oxidized and as carbon dioxide takes part in the natural carbon cycle, which has been considerably studied in recent years. As a result of biological processes and a circulation of the oceans, the carbon contained in the atmosphere, in living matter, and in the oceans changes its distribution at a rate that is rapid on a geological time-scale and reasonably short compared with the mean life of carbon 14. This widely distributed material forms the exchange-reservoir in which the naturally produced radiocarbon is in equilibrium. Table 7F (derived from Craig (1958) and Arnold and Anderson (1957) shows the estimated distribution of carbon and the natural radiocarbon content in this exchange-reservoir.

TABLE 7F

Distribution of natural carbon and carbon 14

Source	Mass of carbon (g/cm^2 of the earth's surface)	Normal inventory of carbon 14 (atoms)
Atmosphere	0·125	34×10^{27}
Biosphere (terrestial)	0·06	16×10^{27}
Humus	0·215	58×10^{27}
Surface waters of ocean (above thermocline)	0·15	41×10^{27}
Remainder of ocean (deep ocean water)	7·35	2000×10^{27}
Total	7.90	$\sim 2150 \times 10^{27}$

A small amount of radiocarbon will be lost from the exchange-reservoir by precipitation of carbonates in the ocean and by the formation of peat, coal and oil. Some stable carbon will also be released to the exchange-reservoir by dissolution of the carbonates and by burning of the carbonaceous fuels but it has been estimated that these processes have little effect on the radiocarbon inventory of the exchange-reservoir.

Dose to tissue from naturally-occurring carbon 14

The dose to tissue from natural carbon 14 can be calculated from the absolute disintegration rate of natural carbon 14, which may be taken as 14 $\pm$ 1 d.p.m./g carbon (Hayes, 1955; Suess, 1955) and the average disintegration energy of 50 $\pm$ 5 keV (Moljk and Curran, 1954). If the carbon content of human tissue is taken as 18 per cent (I.C.R.P., 1959), then the annual dose to the whole body due to natural carbon 14 is about 1 millirad.

Artificial Carbon 14

During the last decade radiocarbon has also been produced by the escape into the atmosphere of neutrons produced in nuclear explosions. The quantity of radiocarbon produced can be greater for certain fusion devices than for fission devices for a given energy release. It also depends on whether the explosion of the device occurs in the air or on a land or ocean surface. Consequently there is uncertainty in estimating the total number of artificial radiocarbon atoms which have been added to the carbon exchange-reservoir. Libby (1959) has estimated the total yield for all nuclear devices exploded up to the end of 1958; and, from data about the consequent neutron production given by Leipunsky (1957), the *maximum* number of carbon 14 atoms which could have been produced is estimated as 94×10^{27} atoms. However, Machta gives a figure, based on U.S. test experience, of 2×10^{26} atoms of carbon 14 per megaton of total (fission and fusion) weapon-yield for an air burst and 1×10^{26} atoms for a ground burst (U.S. Joint Committee on Atomic Energy, 1959a, p. 2191). The estimated uncertainty of these values is given as a factor of 2. If the data of the Ministry of Defence (1959) for the energy released in surface and air-burst fission and fusion explosions are used, a value is found for the carbon 14 produced of 24×10^{27} atoms with a range of uncertainty from 12 to 48 $\times 10^{27}$ atoms. Hagemann *et al.* (1959) concluded from experimental data that 25×10^{27} atoms of carbon 14 had been added by 31st October, 1958, to that naturally present in the carbon cycle; and their value is used in the present calculation.

From experimental data on the distribution of the artificially-produced carbon 14 given in the Hearings of the U.S. Joint Committee on Atomic Energy (1959b) it can be deduced that 0·265 tons or 12×10^{27} atoms had been produced by March, 1958, and the amount added from 1st March, 1958, up to the termination of testing in 1958 was 0·29 tons or 13×10^{27} atoms. The figures given for the distribution of the artificial radiocarbon for March, 1958, have accordingly been increased to allow for the additional production during the period 1st March—31st October, 1958, and the calculated distribution becomes that shown in Table 8F. These values may be compared with the amounts of naturally-occurring carbon 14 given in Table 7F.

In 1959 there was considerably more artificial carbon 14 in the northern hemisphere than in the southern hemisphere, due to the greater number of

weapon tests in the northern hemisphere. In the absence of further tests, the concentration in the two hemispheres will tend to equilibrate in the next few years (Broecker and Walton, 1959). Experimental data are available which show that the mean stratospheric residence time for carbon 14 is about 5 years (Craig, 1958). Carbon 14 in the stratosphere will thus mix with the tropospheric air causing an initial increase in tropospheric concentration. Carbon dioxide is

TABLE 8F

Estimated distribution of artificial carbon 14 *at mid-*1959

Reservoir	Inventory of artificial carbon 14 at mid-1959 (atoms)
Stratosphere	15×10^{27}
Troposphere	8×10^{27}
Surface ocean	2×10^{27}
Biosphere	$0{\cdot}3 \times 10^{27}$
Total	$\sim 25 \times 10^{27}$

transferred from the troposphere into the biosphere and surface waters of the oceans, so eventually the carbon 14 concentration in the troposphere will decrease. The rates of transfer of carbon dioxide between the various reservoirs are known only approximately and consequently the changes in tropospheric concentration can be predicted only roughly. In summary, the future trend in carbon 14 content of the troposphere, due to the artificial carbon, may be considered in four phases:—

(*a*) A continuing rise for the next two or three years, reaching about 40 per cent above normal when all the artificial carbon 14 is thoroughly mixed with the natural content of 34×10^{27} atoms.

(*b*) A fall with a half-period of several years to 25 per cent above normal due to exchange with the carbon reservoir of the biosphere and surface waters of the ocean.

(*c*) A further fall with a longer half-period of about twenty years (Craig, 1958), down to 1 per cent above normal as a result of mixing with the carbon reservoir in deep oceans.

(*d*) The final fall to the original level existing before nuclear weapon-tests with a half-period equal to that of the radioactive decay of carbon 14.

Any further nuclear weapon-tests will, of course, alter the amount and distribution of artificial carbon 14 and affect the values given above.

Dose to tissue from artificial carbon 14

Broecker *et al.* (1959) have shown that the rising concentration of the artificial carbon 14 in humans lags behind the rising concentration in the troposphere by about one year. Thus, for the calculation of the radiation dose received by humans over any future period, it is necessary to know the concentration of carbon 14 in the troposphere during that period.

Calculations similar to those of Plesset and Latter (1960) show that, as a result of the processes referred to in (*a*) and (*b*) above, the tropospheric carbon 14 content in the next thirty years may be expected to be on the average about 30 per cent above the normal value, and the extra dose in that period will thus be

$$1 \times \frac{30}{100} \times 30 \text{ mrads} = 9 \text{ mrads.}$$

In the following 30-year period, the additional dose may well be about 4 mrads, falling off with a half-period of approximately 20 years, the additional dose over the first few generations thus being about 15 mrads.

The total extra dose added by the artificial carbon 14 which has so far been produced may be roughly calculated by assuming that 25×10^{27} atoms have been added to the natural carbon 14 reservoir of 2100×10^{27} atoms for a period equal to the mean life of carbon 14, that is, 8,000 years. Therefore the total added dose is

$$1 \times \frac{25 \times 10^{27}}{2100 \times 10^{27}} \times 8000 + 15 = \sim 110 \text{ mrads}$$

delivered over many thousands of years. This value may be compared with the total gonad dose due to fall-out of fission products which is estimated to be about 25 mrads for the current 30-year period commencing 1954 with very little dose after that. The gonad dose from the natural background is about 3,000 millirads in every 30-year period.

Genetically Effective Dose from Carbon 14

Genetic significance of carbon 14 lies in the fact that carbon atoms make up about 37 per cent of the desoxyribonucleic acid (D.N.A.) which is an important constituent of chromosomes and is associated with the genes. Hence if a carbon 14 atom becomes incorporated in a D.N.A. molecule and later disintegrates, then the D.N.A. molecule may be damaged not only by the ionizing beta particle emitted and the recoiling nucleus, but also by the transmutation of the carbon 14 atom to nitrogen 14 and in this way a viable gene mutation might be produced. These effects may be additional to those produced in the usual way by ionizing radiation.

Totter *et al.* (1958) have made calculations of the biological hazard from carbon 14 and conclude that 'subject to large uncertainty, the transmutation-effect of carbon 14 atoms contained in the genetic material of the human body could lead to about the same number of genetic mutations as the radiation effect from carbon 14'. Pauling (1958) on the other hand has suggested that it is unlikely to amount to more than 10 per cent of the total effect.

Clearly more experimental data are needed before a critical assessment of this factor can be made. Equally clearly, since natural carbon 14 delivers about 30 mrads per generation of 30 years, and the carbon 14 from weapon-tests already conducted may be expected to give a total dose of about 120 millirads spread over many thousands of years, the added effect in individual generations will be very small, since the total dose of ionizing radiation per generation is some 3,000 millirads from natural background.

SUMMARY OF TISSUE DOSES RESULTING FROM NUCLEAR EXPLOSIONS

The various contributions to tissue dose-rate derived in the foregoing sections of this appendix are summarized in Table 9F; the estimates for 1962 and 1970 are necessarily very rough, but are necessary in order to enable the appropriate integrated dose to be derived, as explained below.

TABLE 9F

Summary of the various contributions to tissue dose-rate resulting from fall-out

	Millirads per year					
	Jan. 1956	Jan. 1958	July 1959	Jan. 1960	Jan. 1962 (vi)	Jan. 1970 (vi)
External radiation						
Dose-rate in internal tissues (i)	0·4	1·8	4·2	1·5	0·5	0·3
Internal radiation						
Dose-rate to whole body from Cs-137 (ii), (iii)	0·6	1·0	1·5	1·5	0·5	~0
Dose-rate to whole body from C-14 (iv)	~0	0·1	0·2	0·3	0·5	0·2
Dose-rate to new bone from Sr-90 in bone (v)	2·0	4·5	8·1	(8·1)	?	?
Dose-rate to bone marrow associated with new bone from Sr-90 in bone	0·7	1·5	2·7	(2·7)	?	?

Notes: (i) The external dose-rates are derived from Table 3F and from paragraph 1 on page 88.

(ii) The cumulative dose (whole body) after 1960 from internal Cs-137 is estimated to be about 3 millirads, if the stratospheric residence half-life is taken as $1\frac{1}{2}$ years.

(iii) There is some suggestion that Cs-137 may be somewhat more highly concentrated in bone than in soft tissue.

(iv) The values for C-14 are derived on the basis that the cumulative dose over the 30 years 1954-1984 is estimated as 10 millirads and the maximum dose in any year 0·5 millirad.

(v) Levels for Sr-90 in bone for January, 1960, assumed to be the same as for July, 1959.

(vi) Estimated dose-rates if no further tests.

The dose-rates in individual tissues on the various dates may thus be arrived at by adding appropriately the figures in Table 9F and the results obtained are given in Table 10F.

For the purpose of estimating the possible biological effect of the additional doses incurred in tissue as a result of the nuclear explosions which have already occurred, it is necessary to estimate the integrated tissue dose over an appropriate number of years. For genetic effects, for instance, the integrated dose over the 30-year period of one generation is probably the appropriate index. For somatic effects the appropriate period would depend on the particular effect under consideration.

The integrated gonad doses expected in the 30-year periods commencing 1954, 1960 and 1984 from external radiation and from internal Cs–137 and C–14

TABLE 10F

Tissue dose-rates from fall-out from nuclear explosions occurring before the end of 1958

	Millirads per year					
	Jan. 1956	Jan. 1958	July 1959	Jan. 1960	Jan. 1962	Jan. 1970
Gonads	1·0	2·9	5·9	3·3	1·5	0·5
New bone	3·0	7·4	14·0	(11·4)	?	?
Bone marrow associated with new bone	1·7	4·4	8·6	(6·0)	?	?

Note: The shielding of bone and bone marrow from external radiation is known to be similar to that for gonad tissue, from the experiments of Spiers and Overton (see Appendix D, pages 70-71).

(tests being assumed discontinued from November, 1958) are given in Table 11F. The integrated dose quoted for the 30-year period commencing 1984 is obviously only a rough estimate but is included mainly to indicate that a considerable reduction in the dose-rate should by then have occurred and also the general manner in which the relative proportions of the different components to the total dose will change with time.

TABLE 11F

Estimated gonad doses from fall-out over 30*-year periods. Nuclear test explosions assumed discontinued from November*, 1958

	Estimated dose (millirads)		
	Period commencing 1954	Period commencing 1960	Period commencing 1984
External radiation from fall-out (i) ...	16	8	~1·5
Caesium 137 within the body (ii)...	9	4·5	~0
Carbon 14 within the body (iii) ...	10	10	~5
Total dose from fall-out ...	35	22·5	~6·5

Notes: (i) External radiation figures derived from graphical integration of data in Table 9F and using the criteria on page 88.

(ii) Cs-137 internal doses derived from numerical integration of data in Table 9F.

(iii) C-14 doses derived from data on page 105.

It will be seen that the integrated gonad dose for the 30-year periods commencing 1954, 1960 and 1984 may be about 1·2 per cent, 0·75 per cent and 0·2 per cent of the integrated dose from the natural background, which is about 3,000 millirads in 30 years.

The gonad dose from carbon 14 in the first 30-year period is about one-third of that from all other activity produced from the nuclear explosions, but

thereafter it becomes the predominant source. The integrated dose from carbon14 may well be over twice that from the fission products, without allowance being made for any additional genetic effect due to the transmutation of the carbon 14 on decay.

Doses in bone and bone marrow

The dose-rates in bone and bone marrow at specific dates have been given in Table 10F. We have emphasized above that the index of the biological hazard is not the dose-rate at a particular time but the dose accumulated in a period dependent upon the specific effect under consideration. We have very little information at present on the appropriate integrating period for doses in bone or in bone marrow. However, we give in Table 12F below the dose accumulated in bone and bone marrow in the two-year periods 1956-57 and 1958-59. These have been derived by graphical integration of the dose-rates given in this appendix. The accumulated dose from the natural background in bone marrow in the two-year periods is also shown in Table 12F.

TABLE 12F

Bone and bone-marrow doses accumulated in two-year periods, resulting from fall-out

	Accumulated dose (millirems)	
	1956-1957	1958-1959
New bone	9	20
Bone marrow associated with new bone	5	12
Natural background dose in bone marrow (from Table 2D)	180	180

It will be seen that the average bone-marrow doses from fall-out in the two-year periods are about 3 per cent and 7 per cent of the integrated doses from the natural background in the same periods. The corresponding figures for the accumulated dose in new bone would be a somewhat higher percentage of the natural background, even when allowance is made for the fact that the natural background in bone is some 30 per cent higher than that in bone marrow (see 1956 report, p. 113).

Situation in other countries

This assessment has been concerned predominantly with the situation in the United Kingdom. The situation in other countries may differ from that in the United Kingdom for a number of reasons. In the first place, the deposition of world-wide fall-out varies markedly with the latitude: the deposition in the southern hemisphere, and particularly in the tropics, has been considerably lower than in the north temperate latitude (where the U.K. is situated). On the other hand, the shielding factors afforded by houses may not be as high as in the United Kingdom and differences in the composition of diet may have caused some relative enhancement in the dose to bone and bone marrow, for the same level of fall-out. At the present time, however, it does not appear

that these differences have been great. These points need to be borne in mind if the expected situation in the United Kingdom is used as a guide to possible situations elsewhere.

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Annexe

Possible Levels of Fall-out Under Conditions of Continued Weapon Testing

We have been asked by the Committee to consider the possible situation in the event of continuance of nuclear test explosions. Nuclear devices can be extremely diverse; they can be exploded in many different ways and places from deep in the earth to many miles above it. It would therefore be unrealistic and misleading to attempt to forecast the pattern of future explosions (if any) and the consequent tissue doses which might result.

We have, however, considered a hypothetical situation, namely, that the rate of fall-out and its composition had continued to be similar to that in the twelve months ending April, 1959, when the annual rate of deposition was the highest hitherto measured. We cannot exclude the possibility that, due to the nature of the material which was deposited following the test explosions at high latitudes in 1958, dry deposition may not have been fully allowed for in the measurements. We have necessarily assumed that the behaviour of the fission products over long periods after they have been deposited can be validly predicted from observations and from experimental studies carried out in the relatively short period of the last 5 years, although, in fact, this is not known with certainty. We have considered the expected dose-rate to the population in the United Kingdom. In other countries markedly different situations might occur because of differences in the rate of fall-out, in the composition of diet, in agricultural conditions and in social habits. We have, where possible, directed attention to these matters.

Though we are satisfied that we can assess reasonably the situation at present and in the near future (Appendix F), we will draw attention in each section of this annexe to the limitations of our knowledge of certain factors relating to late effects and we do not exclude the possibility that factors of which we have as yet no knowledge may also be important.

Assumed pattern of fall-out

The highest annual rate of fall-out so far observed in the U.K. was for the 12 months ending April, 1959. The average monthly rate of deposition during this period was about 0·6 mc/km^2 for strontium 90 and 1·0 mc/km^2 for caesium 137 associated with an annual rainfall of about 40 in. An indefinite continuation of fall-out at this rate would lead to an accumulated deposit of 290 mc Sr-90/km^2 and 520 mc Cs-137/km^2 if the only means of loss were radioactive decay. In subsequent sections other mechanisms of loss are taken into account.

We have further assumed that the average age of the current deposit would be that observed during the year ending April, 1959, that is, approximately 6 months.

Dose-rate from external radiation

The dose-rate from external radiation arises mainly from the Cs-137 and the shorter-lived gamma-emitting nucleides deposited on the ground surface.

In considering dose from Cs-137 account must be taken of its loss from the ground surface. We have assumed this in effect to be 5 per cent per year. It is based on a limited number of measurements of dose-rate above the ground surface following experimental applications of caesium 137 and calculations from the observed distribution of caesium 137 in the soil profile. It is assumed that removal of caesium 137 from roofs, roads and other surfaces by rain would be similar to its removal from the soil. In addition, over long periods the rebuilding of houses and resurfacing of roads would cause further reductions.

The dose in air in the open attributable to external radiation from caesium137 would be as in Table 13F. The table shows values of dose-rate for assumed losses of 0, 2·5 per cent and 5 per cent per year. It is notable that, though we consider 5 per cent already an underestimate of the loss, the dose-rate would not be markedly changed by our making a still more cautious assumption of 2·5 per cent per year. The accumulated doses for the first and second generations and for equilibrium conditions are also shown.

TABLE 13F

Calculated Dose-rates in Air

Calculated dose-rates in air in the open at a height of 1 *metre over undisturbed soil on the basis of assumptions detailed in the text*

Loss by processes other than decay	Dose-rate (mrads/year)			Total dose (mrads/30 years)		
	Years after starting time: 10	50	max	1st gen.	2nd gen.	max.
0	12	38	58	490	1,110	1,740
2.5 per cent/year	11	25	27	390	720	810
5 per cent/year	10	19	19	330	540	570

Note: The doses for tissues such as gonads and bones would in the United Kingdom be about one-eighth of the values quoted above on account of shielding (Appendix F; see page 88).

The dose-rate from shorter-lived radioactive nucleides deposited on the ground would rise rapidly to an equilibrium-value of about 40 mrads per year in air, corresponding with about 5 mrads per year in tissue. This is, as expected, a little higher than the 30·6 mrads/year in air which may be derived in Table 3F for 1st July, 1959. However, this dose-rate depends critically on the average age of the fission products at the time of deposition which has been assumed to be 6 months. For example, if the average age were reduced to 3 months the dose-rate attributable to deposited Zr-95, Nb-95 would increase threefold.

Dose-rate from carbon 14 *within the body*

Appendix F indicates that over half the accumulated total of 25×10^{27} atoms of carbon 14 accumulated from nuclear explosions so far was produced in the years 1957–58, and accordingly we assume an average annual production rate of 7×10^{27} atoms/year. This will mix quickly with the 34×10^{27} atoms of natural carbon 14 in the atmosphere and fairly quickly with the additional 57×10^{27} atoms in the biosphere (excluding humus) and surface layers of the oceans (Table 7F). However, the increments would be depleted with a half-life of about 20 years by diffusion into the deep ocean carbonate pool (estimated as 2000×10^{27} atoms) and by slow accumulation in humus, the mean life of which is thought to be of the order of 500 years (Craig, 1958).

If the carbon 14 produced each year were uniformly distributed in the atmosphere, biosphere (excluding humus) and the surface layers of the ocean (see Table 7F) and the dose-rate from natural carbon 14 is ~ 1 mrad/year as calculated in Appendix F, the dose-rate would build up to an equilibrium value of about

$$\frac{7 \times 10^{27}}{91 \times 10^{27}} \times \frac{20}{0{\cdot}693} \times 1 = 2{\cdot}2 \text{ mrads per year.}$$

However, since the source is in the atmosphere (one-third of the rapidly exchangeable reservoir), the actual level would be somewhat higher than this (but less than 7 mrads per year corresponding to mixing with the atmospheric carbon 14 only). The actual value in this range depends on the exchange rates between the various reservoirs and more elaborate calculations by Dolphin (1960), based on the six-compartment model of Plesset and Latter (1960), suggest that the ' equilibrium ' dose-rate in about 100 years (assuming tests continue for this period at the average rate specified) might be about $3\frac{1}{2}$ mrads per year.

The estimated dose-rates are closely dependent upon the value assumed for the rate of transfer of carbon dioxide from the surface waters of the oceans to the deep ocean carbonate pool. It is assumed, following Craig (1958), that this corresponds with a half-life of 20 years. The calculated dose-rates are proportional to this figure, which is only known roughly. If the rate of contamination of the atmosphere were continued over some thousands of years the dose-rate would increase slowly as the deep-ocean carbonate pool gradually becomes contaminated and would reach an ultimate equilibrium level of about 25 mrads/year in the course of many tens of thousands of years.

Dose-rate from strontium 90 *in bone*

Bone and bone marrow receive radiation from internal sources of which strontium 90 localized in bone is the most important. The complexity of the

factors which determine the relationship between the level of strontium 90 deposited on the ground and the level in bone have been referred to earlier in Appendix F. The relationship between the present levels of strontium 90 in the environment and those in bone cannot be used to predict situations in the distant future. Whereas at the present time the recent deposit is the major factor controlling the entry of strontium 90 into the diet, the contribution from the accumulated total in the soil will become increasingly important as its magnitude increases.

To determine the latter it is necessary first to estimate the quantity of strontium 90 which will be present in agricultural land at equilibrium and secondly to estimate the ratio of strontium 90 to calcium in vegetation and diet to which this would give rise.

(i) *Quantity of strontium* 90 *in soil at equilibrium.* While we assume that 290 mc/km^2 would have been the equilibrium level of the deposited activity, the quantity present in the soil and available for absorption by crops would, however, be appreciably less because of the removal of strontium 90 by previous crops, by ' run-off ', by leaching to below the rooting depths of plants and possibly also because slow chemical changes may somewhat reduce its availability. Thus 1 per cent or more of the strontium 90 in the soil may be removed by crops each year. Moreover, there is evidence that recently deposited material may be considerably removed in ' run-off '. For example, the available measurements made in the United Kingdom during 1958 (Agricultural Research Council, 1959) indicate that the strontium 90 content in undisturbed soil to the depth of four inches was on average only about two-thirds of the total deposit determined from analysis of rain. However, because of many variables this figure is open to considerable error.

It appears reasonable to conclude that the effective loss of strontium 90 by all processes, other than the decay of radioactivity, will amount to at least 5 per cent per annum, and certainly be in excess of 2·5 per cent. Agricultural data on the loss of the homologous calcium support this conclusion. On the basis of these estimates the total in the soil at equilibrium available for absorption by plants would be about 100 and 150 mc/km^2 of strontium 90 respectively. The former figure has been used for purposes of calculation and the latter for indicating the possible error in the assessment.

(ii) *Relationship between the strontium* 90 *content of soil and the ratio of strontium* 90 *to calcium in plants.* Experimental data summarized by Burton *et al.* (1960) suggest that, after the radioactivity is reasonably mixed into soil, an average value of the ratio of strontium 90 to calcium in plants for England and Wales as a whole is approximately 1 $\mu\mu$c Sr-90/g Ca when the content of the soil is 1 mc/km^2. Values approaching 4 times this figure could be obtained when the calcium content of the soil is extremely low. At equilibrium with 100 mc/km^2 in soil the ratio of strontium 90 to calcium in plants as the result of absorption from the soil should be about 100 $\mu\mu$c/g Ca. The ratios of strontium 90 to calcium in milk and animal tissues can be derived from the estimates of discrimination factors (Food and Agriculture Organization of the United Nations, 1960; Agricultural Research Council, 1959). The resulting values are used in

Table 14F to calculate the ratio of strontium 90 to calcium in the major components of the average mixed diet in the United Kingdom due to absorption from the soil.

Strontium 90 *in diet from direct contamination*

In addition to absorption from soil, account has to be taken of the contribution from direct contamination. Following Burton *et al.* it has been assumed that this represents some 80 per cent of the total strontium 90 observed in agricultural produce during 1959. This assumption in fact leads to some overestimate of the direct contamination of the vegetable component of diet since the value for 80 per cent was calculated for pasture grasses (and hence milk), the direct contamination of which is likely to be greater than that for other vegetable foods, but no direct information is available. The resulting estimates of the

TABLE 14F

Dietary Contamination Calculations

Possible eventual content of strontium 90 *in the average United Kingdom diet on the assumptions that fall-out continued at the rate of* 0·6 *mc/km²/month and that* 5 *per cent of the strontium* 90 *in soil were lost each year by means other than radioactive decay*

	Daily intake of calcium (g)	Sr-90 from soil		Sr-90 from direct contamination (μμc/day)	Total daily intake (μμc)
		(μμc/g Ca)	(μμc/day)		
Milk and milk produce	0·591	12	7·1	4·5	12
Vegetables including cereals	0·134	100	13·4	2·2	16
Animal produce and other sources ...	0·056	24	1·3	0·4	2
Mineral calcium added flour	0·243	—	—	—	—
Total	1·024				30

strontium 90 content of the total mixed diet are shown in Table 14F. The minor contributions of strontium 90 from water and air have been ignored and no separate consideration has been given to milk, as the estimated ratio of strontium 90 to calcium in it is less than that in the mixed diet.

Table 14F suggests that the expected ratio of strontium 90 to calcium in the average mixed diet at equilibrium will be about 30 μμc Sr-90/g Ca. With the discrimination factor of 4 against strontium 90 relative to calcium in passage from diet to bone this corresponds to a ratio of about 7·5 μμc Sr-90/g Ca in bone; this would give rise to a dose-rate of about 20 mrads per year in bone and about 7 mrads per year in bone marrow. While there is no objective basis for assessing the errors of this estimate, alternative calculations on bases which at present appear unreasonably pessimistic have not increased the mean by a factor of more than 2 to 3.

It must be emphasized that a calculation of the same type would lead to very different values for conditions of agriculture and diet other than those in the United Kingdom. For a given quantity of strontium 90 and calcium in the soil the mean ratio of strontium 90 in the total diet is largely dependent on the amount of dietary calcium derived from vegetable foods. In some countries up to 70 per cent of the dietary calcium comes from vegetables (Food and Agriculture Organization of the United Nations, 1960) and this circumstance could increase the ratio of strontium 90 to calcium in the total diet by a factor of about 3. Also, the addition of mineral calcium to flour in the United Kingdom, a practice not followed in many countries, lowers the mean ratio in the diet by about 20 per cent. Furthermore, the calcium content of soils in Britain is in most areas high and the average is higher than in many countries. It is possible that soil conditions in other countries could enhance the mean ratio in diet by a factor of 2 or more. It would seem probable, therefore, that values of bone-dose up to perhaps six times the United Kingdom level might eventually occur in other countries if the levels of fall-out were the same. It is emphasized that this statement is not applicable to the present situation; as shown earlier the direct contamination of pastures now largely offsets the discrimination factor in the passage of strontium 90 and calcium to milk.

Dose-rate from internal caesium 137

Because caesium 137 is relatively unavailable to plants after its entry into the soil, the present levels of caesium 137 in human tissues may be regarded as dependent on the rate of fall-out. This does not, however, exclude the possibility that an appreciable contribution due to absorption from the soil may occur when the total caesium content in the soil greatly exceeds that now observed. Field experimental data, such as those obtained for strontium, are not available to guide such an assessment and it is possible only to suggest an upper limit of the contribution of the soil. It has been shown that, depending on the nature of soils, caesium is absorbed by plants less than one-tenth, and sometimes less than one-fiftieth, as readily as strontium. This suggests that the magnitude of the absorption of caesium 137 from the soil relative to that due to direct contamination should, at equilibrium, be less than one-tenth of the corresponding relationship for strontium 90. The figures in Table 14F, therefore, point to the conclusion that at equilibrium the contribution of caesium 137 from the soil will not be more than 50 per cent of that due to direct contamination, and will probably be less. Since the dose-rate in man in 1959 was estimated to be about 1·5 mrads per year it is assumed that the dose-rates from this source at equilibrium should not exceed 2·5 mrads per year.

Conclusions

In Table 15F we summarize our estimates of the tissue-doses which might have occurred in the United Kingdom if fall-out had continued for an indefinite period at the same rate as that observed in the year May, 1958–April, 1959. At equilibrium the dose-rate in gonads might have been about 14 mrads per year or one-seventh of that from the natural background, whereas in bone it might have been about 34 mrads per year or one-third of that from the natural background and in bone marrow 20 mrads per year or one-fifth of that from the natural background.

TABLE 15F

Calculations of Tissue Dose-rate

Estimated radiation dose-rates at equilibrium in the United Kingdom on the assumption that fall-out had continued indefinitely at the rate of 0·6 *mc/km²/month for strontium* 90 *and other fission products corresponding to an age of six months at the time of deposition and to carbon* 14 *being produced at the rate of* 7×10^{27} *atoms/year*

Source	mrads/year at equilibrium			Years taken to reach 90 per cent of equilibrium
	gonads	bone	bone marrow	
External				
Caesium 137	2½	2½	2½	45
Short-lived fission products ...	5	5	5	1
Internal				
Strontium 90	—	20	7	45
Carbon 14*	3½	3½	3½	100
Caesium 137	2½	2½	2½	45
Approximate total	14	34	21	

* Allowance may have to be made for other possible effects arising from the transmutation of carbon 14 in the tissue.

In considering the weight which should be attached to this assessment it must again be emphasized that our calculation has been made for the hypothetical situation of fall-out at a constant rate. It is known that the magnitude of fall-out, the mean delay-period before its deposition and its latitudinal distribution may all vary greatly depending on the amount of fission products released and on the site and altitude of release. Our estimate must, therefore, in no sense be regarded as the prediction of the possible consequences of continued weapons-testing.

In our calculation we have endeavoured to make use of all relevant information now available and have adopted assumptions which appear cautious in the light of the information now available. However, since there is not direct information as to the manner in which the long-lived nucleides strontium 90 and caesium 137 will behave for long periods after their deposition on the earth's surface, great uncertainty arises. The possibility must therefore be borne in mind that, if fall-out were to have continued at the rate postulated in our calculation, the eventual level of radiation to which the population would be exposed might differ considerably from that which we have calculated. There is no objective basis for estimating the probable error of our assessment and, although the balance of information now available suggests that tissue doses may have been over-estimated, we cannot exclude the possibility that our assessment is several fold in error in either direction.

Finally, we draw attention to the fact that our assessments apply only for the United Kingdom. In other countries differences in the levels of fall-out, in the degree of shielding from external radiation, in dietary habits and in agricultural conditions could cause considerable differences. We estimate, for example, that in countries where vegetables are the chief source of dietary calcium and the soil is poor and low in calcium, the average dose to bone could, for the same level of fall-out, perhaps actually reach six times that in the United Kingdom. However, it must be recalled that the level of fall-out has been shown to be highest in the northern temperate latitudes and considerably lower in the southern hemisphere, and particularly in the tropics.

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