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PROTRACTED EXPOSURE TO FALLOUT: THE RONGELAP AND UTIRIK EXPERIENCE†

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Abstract—From June 1946 to August 1958, the U.S. Department of Defense and the U.S. Atomic Energy Commission (AEC) conducted nuclear weapons tests in the Northern Marshall Islands. On 1 March 1954, BRAVO, an above-ground test in the Castle series, produced high levels of radioactive material, some of which subsequently fell on Rongelap and Utirik Atolls due to an unexpected wind shift. On 3 March 1954, the inhabitants of these atolls were moved out of the affected area. They later returned to Utirik in June 1954 and to Rongelap in June 1957. Comprehensive environmental and personnel radiological monitoring programs were initiated in the mid 1950s by Brookhaven National Laboratory to ensure that body burdens of the exposed Marshallese subjects remained within AEC guidelines. Their body-burden histories and calculated activity ingestion rate patterns post-return are presented along with estimates of internal committed effective dose equivalents. External exposure data are also included. In addition, relationships between body burden or urine-activity concentration and declining continuous intake were developed. The implications of these studies are:

(1) the dietary intake of ^{137}Cs was a major component contributing to the committed effective dose equivalent for the years after the initial contamination of the atolls;

(2) for persons whose diet included fish, ^{65}Zn was a major component of committed effective dose equivalent during the first years post-return;

(3) a decline in the daily activity ingestion rate greater than that resulting from radioactive decay of the source was estimated for ^{137}Cs, ^{65}Zn, ^{90}Sr and ^{60}Co;

(4) the relative impact of each nuclide on the estimate of committed effective dose equivalent was dependent upon the time interval between initial contamination and rehabilitation; and

(5) the internal committed effective dose equivalent exceeded the external dose equivalent by a factor of 1.1 at Utirik and 1.5 at Rongelap during the rehabitation period.

Few reliable ^{239}Pu measurements on human excreta were made. An analysis of the tentative data leads to the conclusion that a reliable estimate of committed effective dose equivalent requires further research.

INTRODUCTION

SUBSEQUENT to World War II, the United States carried out several series of atmospheric tests of nuclear weapons in the Northern Marshall Islands between the years 1946 and 1958. On 1 March 1954 at Bikini Atoll, BRAVO, the first of six nuclear-weapons tests in the Castle series, was detonated. Due to an unanticipated wind shift, the BRAVO device produced substantial surface contamination on inhabited atolls up to 500 km east of Bikini within a 5000 km^2 area. The contaminated region was cucumber-shaped and falling bomb debris was visible on Rongelap Atoll from 5 to 10 hr after detonation (Gl62; Sh57).

Following a fallout alert by a Navy mon-

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itoring team stationed at nearby Rongerik Atoll, the 64 residents of Rongelap Atoll and an additional 18 Rongelapese who were gathering food nearby at Sifo Island, Ailinginae Atoll, were removed to Kwajalein Atoll, some 300 km to the south on 3 March 1954. On March 3 and 4, the more distant 157 Utirik Atoll residents were moved. During the first few weeks and at least once every year from 1957 to the present, a Brookhaven National Laboratory (BNL) medical team, organized by the AEC (and its successor organizations) and the Department of Defense, has regularly conducted medical examinations to monitor the health and to evaluate the radiobiological status of persons affected by tropospheric fallout from the BRAVO nuclear test.

Reports of their findings including whole-body counting data and urine activity concentration data are available in Cr56, Du56, Du57, Wo59, Co56, Co58, Co59, Co60, Co62, Co63, Co65, Co67, Co70, Co75 and Co80a. These reports may be consulted in order to easily follow the information presented here. Estimates of the initial body burdens of internal emitters were presented in Co55, Coh56 and Coh60 and will not be discussed here. A reassessment of thyroid absorbed dose from the initial 1954 exposure is currently being made and will be reported in a separate study. Since April 1978, the bioassay program and whole-body counting studies have been performed by members of the Safety and Environmental Protection Division of BNL. Reports of their findings may be found in Gr77a, Gr77b, Le80a, Le80b, Mi80, Mi81 and Na80. The report by Lessard (Le80b) contains more detail on the development of the equations used here.

The Utirik and Rongelap inhabitants were returned to their home atoll in June 1954 and in June 1957, respectively. The earlier repatriation of Utirik Atoll was based on the low measured level of external radiation exposure over a three-month observation period. The Utirik population was subsequently examined by a Brookhaven medical team during 1957; 144 people received comprehensive physical examinations.

In 1957, the Rongelap inhabitants were also returned to their atoll to occupy new homes, community structures and other facilities which had been constructed during their three-year residence at Majuro and Kwajalein Atolls. Following the 1957 medical survey, measurements were made on two men from Utirik Atoll using the whole-body counter at Argonne National Laboratory (ANL). Radiochemical analyses of their urine samples were also made. Four persons from Rongelap Atoll also visited Argonne for whole-body counting in 1957. In addition, pooled urine samples from both atoll populations were analyzed radiochemically for ^{137}Cs and ^{90}Sr. The body burdens measured at ANL were corrected for 10 days of biologic elimination and radiologic decay to estimate the body burden while living on the atoll.

Starting in May 1958, Conard and Cohn (Co59), measured whole-body levels of ^{137}Cs, ^{65}Zn and ^{60}Co in about 100 Rongelap adults, adolescents and juveniles as part of the Brookhaven medical examination program. A portable whole-body counter with a standard chair geometry in a shielded steel room was employed (Coh63). Whole-body counts were obtained in the Rongelap and Utirik populations in 1959 (Coh60), 1961 (Co62), 1965 (Co67), 1974 (Co75) and 1977 (Co80a). The counting geometry was converted to a scanning type shadow-shield geometry starting in 1965 (Co67). Urine samples were also collected in these surveys and in additional medical surveys conducted in intervening years. The samples were analyzed for their radiochemical content by both USNRDL and the NYO–AEC Laboratories.

From 1978 to the present time, whole-body counting measurements were performed with the bed-type shadow shield whole-body counter (Co67). In 1980, a standard chair geometry was once again used. All three counting systems were intercalibrated and also calibrated against the large BNL 54-detector whole-body counting facility to ensure consistency of the whole-body counting data over the past 28 yr.

A summary of the sequence of events affecting the whole-body and urine activity measurements on the Rongelap and Utirik people is given in Fig. 1. The detonation of BRAVO in 1954 was followed by the evacuation of Rongelap Atoll at 2.2 days post-detonation and then Utirik Atoll at 3.5 days post-detonation. After a three-month wait, the Utirik people returned in June 1954 and after three years Rongelap Atoll was rehabilitated and occupied in June 1957.

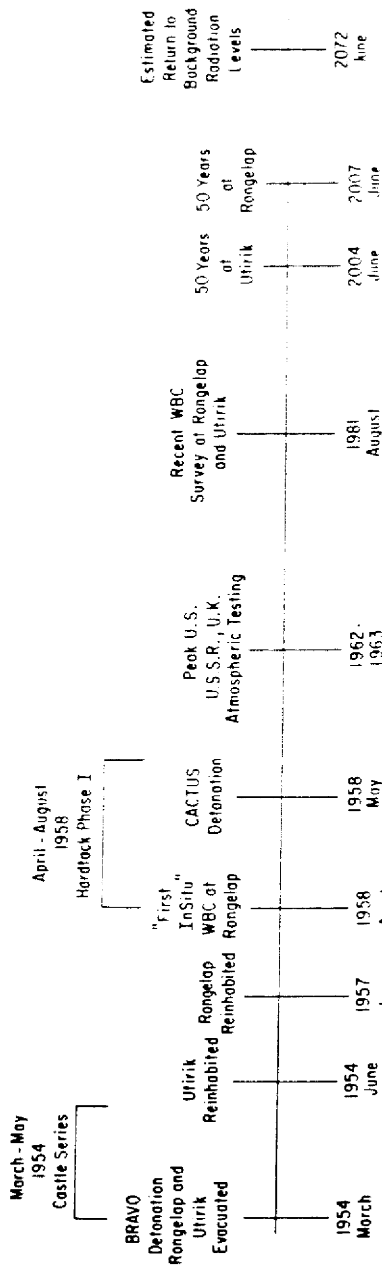

FIG. 1. Sequence of events at Rongelap and Utirik Atolls

Shortly after the Rongelap people's return, the first *in situ* whole-body counting survey was performed in 1958. The Hardtack series of nuclear tests in 1958 was the final above-ground tests to be performed by the United States in the Marshall Islands. Worldwide atmospheric testing of nuclear devices at other locations continued and peaked during the early 1960s. During the period 1958 through 1981 a total of eight whole-body counting surveys at Rongelap and five whole-body counting surveys at Utirik were performed.

METHODS

Body-burden data and urine activity concentrations

Adult average body-burden data and urine-activity concentration data were used as input quantities to equations which related them to activity intake rates. These input data were obtained from Conard's medical reports (Co56; Co58; Co59; Co60; Co62; Co63; Co67; Co70; Co75; Wo59) and from recent surveys performed by members of the BNL Safety and Environmental Protection Division. The methods used to obtain the recent body-burden data were presented by Miltenberger (Mi80). The most recent average data obtained for adult body burden at Rongelap and Utirik are presented here. These data were obtained in April 1978, August 1979 and August 1981.

In the cases of ^{137}Cs, ^{60}Co and ^{65}Zn, direct body-burden measurements were made. In the cases of ^{90}Sr and ^{239}Pu, urine-activity concentrations were measured and then converted to body-burden estimates. This was done by relating the activity in urine to the activity in the total body. For ^{90}Sr and ^{239}Pu, this involved the use of derived quantities which are developed in the next section.

Derived quantities

An equation was developed to relate the activity in the urine or whole body to the activity taken in by ingestion of contaminated food and fluids. To select an appropriate model for this relationship, the body-burden history and the history of activity in vegetation and soil were examined. Activity concentrations of ^{137}Cs, ^{129}I and ^{90}Sr in surface soil on Rongelap and Utirik Atolls were observed to decline with time at a rate greater than radioactive decay from 1954 to the present (Ne77; Ne79; Br82). Activity concentrations of ^{137}Cs and ^{90}Sr in vegetation were observed to decline at a rate greater than that predicted by radioactive decay alone (Ne77; Ne79). Body burdens and urine activity concentrations were observed to increase rapidly and to decline slowly throughout the residence time of persons at Rongelap and Utirik Atolls (Co75; Le80b). These observations led to the selection of a declining continuous intake model.

An exponential decline in the amount of activity ingested each day from the dietary sources was assumed. The following general equations were derived (Le80b). They may be applied to each nuclide of concern.

$$\lambda P^{\circ} = \frac{\left(\frac{UU_s}{f_u}\right) - q^{\circ}\left(\sum_i k_i \chi_i' \mathrm{e}^{-(\lambda + k_i)t}\right)}{f_1\left(\sum_i \frac{\chi_i k_i}{k_i - k}\left(\mathrm{e}^{-(\lambda + k)t} - \mathrm{e}^{-(\lambda + k_i)t}\right)\right)}, \quad (1)$$

or

$$\lambda P^{\circ} = \frac{q - q^{\circ}\left(\sum_i \chi_i' \mathrm{e}^{-(\lambda + k_i)t}\right)}{f_1\left(\sum_i \frac{\chi_i}{k_i - k}\left(\mathrm{e}^{-(\lambda + k)t} - \mathrm{e}^{-(\lambda + k_i)t}\right)\right)}, \quad (2)$$

where t = time post-onset of intake (time from day of return of each atoll population), d; λ = instantaneous fraction of atoms decaying per unit time, d^{-1}; P° = initial daily atom ingestion rate on day of return, atoms s^{-1}; k_i = instantaneous fraction of atoms removed from compartment i in the body by biological processes, d^{-1}; χ_i = compartment i deposition fraction; χ_i' = the number of radioactive atoms in compartment i relative to the number in all compartments on the day of return (some persons returned with body burdens); U = 24-hr or 1-l. urine-activity concentration at any time post-return, Bq l^{-1}; U_s = subject urine excretion rate, l d^{-1}; f_1 = fraction of element transferred from GI tract to blood; f_u = fraction of element reaching extracellular fluid that is excreted through the urine pathway; k = instantaneous fraction of atoms removed from the atom ingestion rate per unit time, d^{-1}, due to factors other than radioactive decay; q = body burden at any time post-return, Bq; and q° = body burden on the day of return, Bq.

Using adult average data, two consecutive urine or body-burden measurements were used to estimate the unknown value of k, a rate constant describing the removal of radioactivity in diet items. This yielded $n - 1$ estimates of k where n was the number of measured adult average data points for body burden or urine-activity concentration during the residence interval. An average value of k was assigned for the entire residence interval during which activity was measured. After the average k was obtained, an estimate of the atom ingestion rate on day of return was calculated based on a value for adult average body burden or urine-activity concentration and the time since day of return. This generated n values of the atom ingestion rate on day of return where n was again the number of adult average data points for body burden or urine-activity concentration.

As indicated by equations (1) or (2), a single exponential relationship was used to model the decline of radioactivity in diet items. Use of these equations led to an estimate of the dietary removal rate constant, k, over the entire residency interval. The average percent decrease in the yearly activity ingested was determined from this dietary rate constant as follows:

$$\% = 100\left(1 - \mathrm{e}^{-(k + \lambda)t}\right), \quad (3)$$

where % = average percent decline in the atom ingestion rate during the residence interval. The definitions of the other quantities in equation (3) were the same as previously given. The value of t was taken at 365 days and the percent reflected the average yearly decline averaged throughout the interval during which a nuclide was observed in people. Thus for ^{137}Cs, the average was for a period of 24 yr at Rongelap and 27 yr at Utirik.

In the development of the three equations, several assumptions were made. For instance, decay of nuclides during transit through the stomach and small intestine was assumed to be negligible relative to their decay within the systemic organs. This was because of the long half-life of the nuclides relative to the transit time through the upper portion of the gastrointestinal tract. Urine activity and body-burden data were assumed to represent instantaneous values rather than incremental values. This was because the sampling periods for whole-body counting and urine collection were very short relative to the intake period. Additionally, one 1. or 24-hr urine samples (which ever was less) were collected. This reduced the influence of biological variation of activity concentration between morning and evening voids. Since urine-activity concentration data were used in equation (1), urine excretion rates which were dependent on sex were adopted from data in *ICRP Publication 23* (ICRP74).

Values for the quantities not measured directly and used in equations (1) and (2) were taken from the literature (ICRP59; ICRP68, ICRP69; ICRP74; ICRP79; Ki78). Cobalt was assumed to be in the form of an organically complexed compound, therefore f_1 was set at 0.3 (ICRP79). The value of f_1 for ^{239}Pu was taken as 10^{-4} (ICRP79). The longest term rate constant for ^{137}Cs was found to be a function of body mass. The value of this rate constant was adjusted for a 60 kg body mass according to formulas given by Miltenberger (Mi81). The single uptake whole-body retention functions given below for adults were based on ICRP models and were not corrected for radioactive decay. These functions, which were used for making an estimate of adult intake, were:

$$^{60}\mathrm{Co}:\ 0.5\,e^{-1.4t} + 0.3\,e^{-1.2\times10^{-1}t} + 0.1\,e^{-1.2\times10^{-2}t} + 0.1\,e^{-8.7\times10^{-3}t};$$

$$^{137}\mathrm{Cs}:\ 0.1\,e^{-3.5\times10^{-1}t} + 0.9\,e^{-8.3\times10^{-3}t};$$

$$^{65}\mathrm{Zn}:\ 0.24\,e^{-3.5\times10^{-2}t} + 0.76\,e^{-1.7\times10^{-3}t};$$

$$^{90}\mathrm{Sr}:\ 0.73\,e^{-3.3\times10^{-1}t} + 0.10\,e^{-2.3\times10^{-2}t} + 0.17\,e^{-2.5\times10^{-4}t};$$

$$^{55}\mathrm{Fe}:\ 1.0\,e^{-3.5\times10^{-4}t};\ \text{and}$$

$$^{239}\mathrm{Pu}:\ 0.45\,e^{-1.9\times10^{-5}t} + 0.45\,e^{-4.8\times10^{-5}t} + 0.1\,e^{-0.33t},$$

where t was in days. An average 60-kg adult body mass was chosen based on the values observed for male and female adult body weights in the study populations (Con56; Co58; Co59; Co60; Co62; Co63; Co67; Co70; Co75; Le80b).

An estimate of body burden for ^{239}Pu and ^{90}Sr was based on use of both equations (1) and (2). The average dietary removal rate constant, k, was first determined using equation (1) and sequential urine-activity concentration data. Once the average k was determined, equations (1) and (2) were set equal to each other and the body burden was calculated for each urine measurement. After the body burden was determined, an estimate of P° was made using equation (1) and the average value for k. In this way an average value for P° was obtained from all the urine data.

To obtain the 50-yr cumulated intake, equation (2) was solved for q and the righthand side of the equation was integrated over an ingestion interval of 50 yr. Total intakes were related to committed effective dose equivalents by using conversion factors "committed effective dose equivalent per unit activity ingested" given by the International Commission on Radiological Protection (ICRP79). The committed effective dose equivalent per unit activity ingested given by ICRP was multiplied by 1.17 to correct for body mass differences to yield committed effective dose equivalent per unit activity ingested by a Marshallese adult.

Statistical analysis of data

The adult average standard deviation for ^{137}Cs, ^{65}Zn or ^{90}Sr atom ingestion rate on the day of return, P°, and the dietary removal rate constant, k, were determined from a set of calculated values derived from a set of adult body-burden measurements and equation (2). The standard deviation for the adult average 50-yr cumulated intake was determined by propagation of error techniques involving first and second order partial derivatives and partial cross derivatives (Be69). To estimate the error, partial cross derivatives and partial derivatives were determined for k and P° with respect to the 50-yr cumulative intake.

Since only one measurement for adult average ^{55}Fe body burden was available, the relative standard deviation of the adult average P° was assumed to equal the relative standard deviation of individual adult P°'s which were determined from the 1970 individual adult ^{55}Fe body burdens. Only two values for the set of adult average ^{60}Co and ^{239}Pu body burdens were available and therefore the same method was employed to obtain adult average standard deviations for k and P'.

External radiation exposure

The external radiation exposure rate data were measured by many individuals and an explanation of their methods can be found in their reports (Ch60; He65; Gr77b; JCAE57; Ti81; USPHS59). A value of 2.8×10^{-8} Gy in tissue of interest per nC kg^{-1} (0.73 rad/R) measured in air at 1 m above the surface was used to convert their data to absorbed dose in tissue. This factor was based on several considerations. First, the planar source represented by the flat atoll was assumed to be an exponential distribution of ^{137}Cs activity with depth in soil, typical of aged fallout (Be70). The nature of this source caused minimal variation of absorbed dose with depth of organ; however, the difference in the number of electrons per gram of air and per gram of tissue necessitated a correction. Secondly, since the atolls presented a varying exposure rate environment, absorbed dose was adjusted for living pattern variations. Both of these considerations combine to give the above factor used to convert external exposure to absorbed dose in tissue. Specific details on the adjustment for living pattern variation were given by Miltenberger and Greenhouse (Gr77b).

RESULTS

Body burden data and urine activity concentrations

The average body-burden data for adults since their return to Rongelap and Utirik Atolls are presented in Tables 1 and 2. In these tables, the zero day or day of return for Utirik was nearly 1000 days before the zero day or day of return for Rongelap. Directly measured body burdens were listed for ^{60}Co, ^{65}Zn and ^{137}Cs. For ^{137}Cs, an initial rise in body burden and a subsequent general decline was apparent. These data were plotted in Fig. 2 along with their standard deviation and standard error.

Conversion of adult average ^{90}Sr and ^{239}Pu urine activity-concentration data was done as indicated in the methods section to derive a body burden for these nuclides. Average data were listed in Tables 1 and 2 and plotted in the case of ^{90}Sr (see Fig. 3). The body burdens listed for ^{55}Fe were obtained from Beasley (Be72). The

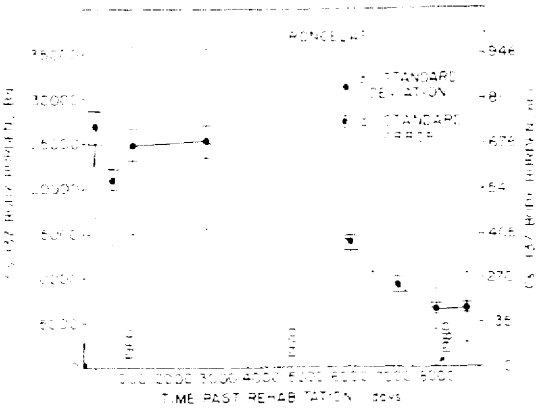

FIG. 2. Cesium-137 body burden for Rongelap adults.

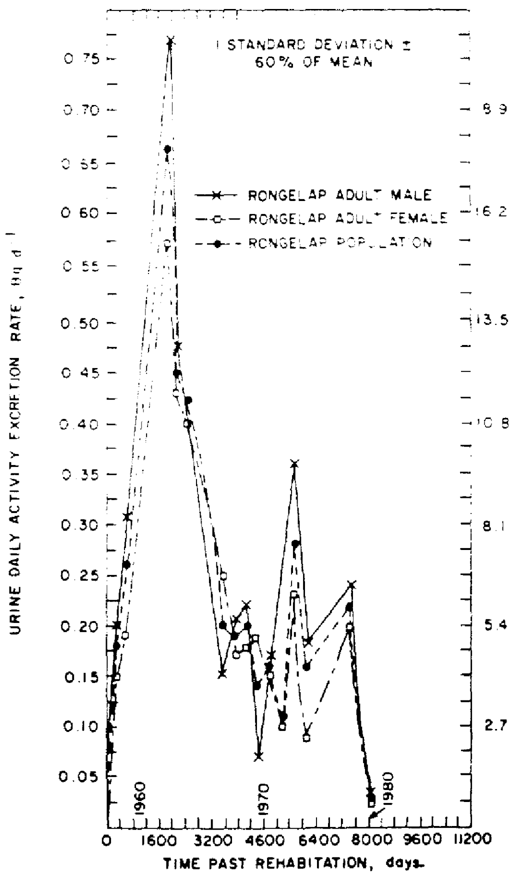

FIG. 3. Strontium-90 urine activity excretion rate for Rongelap adults.

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Table 1. Average radionuclide burden and time since rehabilitation for Rongelap adults

	Adult Males (>15a)		Adult Females (>15a)		Adults (>15a)			
	Body Burden Bq	Number of Individuals	Body Burden Bq	Number of Individuals	Body Burden Bq	Number of Individuals	Time Post Rehabitation Days	Year
^{60}Co	1.1×10^{0}	(A)	6.3×10^{-1}	(A)	9.3×10^{-1}	(A)	0	1957
	3.7×10^{2}	37	2.9×10^{2}	37	3.3×10^{2}	74	1370	1961
	9.3×10^{1}	45	7.4×10^{1}	45	8.1×10^{1}	90	2831	1965
^{65}Zn	1.9×10^{3}	4(B)	(C)	(C)	(C)	(C)	0	1957
	2.3×10^{4}	17	6.4×10^{3}	8	1.8×10^{4}	25	244	1958
	1.6×10^{4}	30	1.4×10^{4}	12	1.5×10^{4}	42	304	1958
	2.3×10^{4}	32	1.9×10^{4}	27	2.1×10^{4}	59	639	1959
	3.5×10^{3}	38	3.1×10^{3}	23	3.4×10^{3}	61	1370	1961
^{55}Fe	1.6×10^{4}	28	1.5×10^{4}	32	1.5×10^{4}	60	4626	1970
^{90}Sr	7.0×10^{0}	(A)	5.2×10^{0}	(A)	6.3×10^{0}	(A)	0	1957
	1.7×10^{1}	11	1.1×10^{1}	4	1.4×10^{1}	15	304	1958
	4.7×10^{1}	24	2.9×10^{1}	16	4.1×10^{1}	40	639	1959
	6.3×10^{1}	9	2.5×10^{1}	4	5.1×10^{1}	13	1370	1961
	3.0×10^{2}	13	1.8×10^{2}	15	2.4×10^{2}	28	1696	1962
	2.1×10^{2}	12	1.9×10^{2}	13	1.9×10^{2}	25	2100	1963
	2.1×10^{2}	11	2.0×10^{2}	7	2.1×10^{2}	18	2466	1964
	7.7×10^{1}	12	1.6×10^{2}	12	1.3×10^{2}	24	3561	1967
	1.5×10^{2}	11	1.2×10^{2}	11	1.3×10^{2}	22	3927	1968
	1.6×10^{2}	11	1.3×10^{2}	13	1.5×10^{2}	24	4292	1969
	5.5×10^{1}	9	1.5×10^{2}	11	1.1×10^{2}	20	4657	1970
	1.4×10^{2}	8	1.2×10^{2}	7	1.3×10^{2}	15	5022	1971
	9.6×10^{1}	5	6.7×10^{1}	7	9.6×10^{1}	12	5388	1972
	3.2×10^{2}	4	2.1×10^{2}	7	2.5×10^{2}	13	5753	1973
	1.7×10^{2}	10	8.5×10^{1}	4	1.5×10^{2}	14	6118	1974
	2.5×10^{2}	26	(C)	(C)	(C)	(C)	7579	1978
	3.7×10^{1}	25	2.8×10^{1}	19	3.3×10^{1}	44	8057	1979
^{137}Cs	5.2×10^{2}	(A)	3.1×10^{2}	(A)	4.1×10^{2}	(A)	0	1957
	2.9×20^{4}	38	1.9×10^{4}	13	2.7×10^{4}	51	304	1958
	2.9×10^{4}	47	1.5×10^{4}	49	2.1×10^{4}	96	639	1959
	3.5×10^{4}	37	1.7×10^{4}	37	2.5×10^{4}	74	1370	1961
	3.5×10^{4}	44	1.8×10^{4}	45	2.5×10^{4}	89	2831	1965
	1.8×10^{4}	22	1.1×10^{4}	24	1.4×10^{4}	46	6118	1974
	1.1×10^{4}	30	7.0×10^{3}	21	9.3×10^{3}	51	7213	1977
	6.7×10^{3}	19	5.6×10^{3}	18	6.3×10^{3}	37	8057	1979
	6.7×10^{3}	36	7.0×10^{3}	30	6.7×10^{3}	66	8813	1981
^{239}Pu	6.4×10^{1}	3	3.8×10^{2}	8	3.2×10^{2}	11	5753	1973
	4.1×10^{1}	3	(C)	(C)	(C)	(C)	7030	1976

A ≡ Number of individuals not recorded.
B ≡ Measured at Argonne National Laboratory.
C ≡ No females measured.

methods used to derive ^{55}Fe body burdens from blood measurements were given in Be72.

The most recent whole-body counting data available (1981) are presented in Table 3. Analysis of the data indicated that ^{137}Cs adult average body burdens at Rongelap and Utirik were from 40 to 90 times greater than those of a comparison population at Majuro, a southern atoll which received little fallout from testing (Le80c). The ^{40}K levels and corresponding potassium content were in close agreement with naturally-occurring values developed from data in *ICRP Publication* 23 (ICRP74).

Due to the paucity of early activity measurements in Utirik residents (see Table 2), their ^{60}Co, ^{239}Pu and ^{55}Fe body burdens were estimated by comparing nuclide ratios for Utirik and Rongelap residents. The measured body

Table 2. Average radionuclide burden and time since rehabitation for Utirik adults

	Adult Males (>15a)		Adult Females (>15a)		Adults (>15a)			
	Body Burden Bq	Number of Individuals	Body Burden Bq	Number of Individuals	Body Burden Bq	Number of Individuals	Time Post Rehabitation Days	Year
^{60}Co	3.6×10^{1}(A)	-	2.8×10^{1}(A)	-	3.1×10^{1}(A)	-	3926	1965
^{65}Zn	1.4×10^{4}	2(B)	(C)	(C)	(C)	(C)	1039	1957
	1.0×10^{4}	14	5.9×10^{3}	15	7.8×10^{3}	29	1734	1959
^{55}Fe	6.0×10^{3}(A)	-	5.8×10^{3}(A)	-	5.8×10^{3}(A)	-	5721	1970
^{90}Sr	1.8×10^{1}	5	[illegible]	2	1.9×10^{1}	-	1734	1959
	4.0×10^{1}	5	[illegible]	6	3.9×10^{1}	[illegible]	7213	1974
	6.1×10^{0}	17	(C)	(C)	(C)	(C)	8659	1978
	5.6×10^{0}	16	5.0×10^{0}	16	5.4×10^{0}	32	9225	1979
^{137}Cs	1.5×10^{4}	(D)	1.0×10^{4}	(D)	1.2×10^{4}	(D)	1039	1957
	1.1×10^{4}	15	7.4×10^{3}	15	9.3×10^{3}	30	1734	1959
	9.6×10^{3}	9	4.8×10^{3}	13	6.8×10^{3}	22	7213	1974
	4.4×10^{3}	27	2.9×10^{3}	21	[illegible]	48	8309	1977
	2.3×10^{3}	19	1.6×10^{3}	17	2.0×10^{3}	36	9225	1979
	3.7×10^{3}	61	2.5×10^{3}	65	[illegible]	126	9935	1981
^{239}Pu	2.4×10^{1}(A)	-	1.5×10^{2}(A)	-	1.2×10^{2}(A)	-	6848	1973

A ≡ Adapted from Rongelap data, see text.
B ≡ Measured at Argonne National Laboratory.
C ≡ No females measured.
D ≡ Number of individuals not recorded.

burdens for these nuclides in Rongelap residents and the observed atoll-to-atoll ratios of adult average body burden for ^{65}Zn, ^{90}Sr and ^{137}Cs were used in the calculation. Ratios were estimated for the period after the Rongelap adult body burdens reached a maximum value. The Rongelap-to-Utirik ratio, 2.6 ± 0.39, has been relatively constant since 1958.

The initial increase in 1958 in the ^{137}Cs average body burden for Rongelap adults (see Fig. 2) was due to dietary intake of ^{137}Cs and a small intake of ^{137}Cs from the air and water due to above-ground nuclear tests in the Marshall Islands during 1958. The subsequent drop in the 1959 ^{137}Cs body burden may have been due to increased use of imported food and the conclusion of the testing. The reason for an increasing ^{137}Cs body burden at Rongelap during the 1960s was uncertain. Residual contamination from the Hardtack weapons-testing program and subsequent incorporation of ^{137}Cs into diet items was one hypothesis.

The Hardtack Phase I series of tests was conducted during 1958, just before an increase in the exposure rate at Rongelap Atoll (Un59). Small amounts of fallout from the CACTUS, YELLOW WOOD and HICKORY experiments in this series reached Rongelap. However, several observations support the conclusion that ^{137}Cs from this series was insignificant relative to ^{137}Cs from the Castle series. First, the peak ^{137}Cs body burden of a similar population at Utirik occurred three yr after the initiating event (Castle BRAVO in 1954) while the 1965 peak ^{137}Cs body burden at Rongelap followed the Hardtack series by seven yr. Secondly, the peak exposure rate on Rongelap which occurred during the Hardtack series in 1958 was about 10,000 times less than the peak exposure rate following BRAVO. These facts suggest that debris from the Hardtack series was not a major factor influencing the Rongelap ^{137}Cs body-burden pattern during the mid 1960s. In addition to Hardtack series fallout, the adult average body-burden pattern would have also been influenced by (1) worldwide fallout fluctuations, (2) movement of adults in the study population to a clean island or atoll for a month's visit with family or friends and (3) to the initial success and subsequent failure of a food subsidy program which began at Rongelap in 1958 (Co80b).

Table 3. Summary—whole-body counting results—August 1981

Island	Population Sub-Group	Number Counted	Average or Average ± 1 s.d. Age, a	Height, m	Weight, kg	Cs-137, Bq	K, kg	Range Cs-137, Bq	K, kg	Comparison Data Average or Average ± 1 s.d. Cs-137, Bq (A)	K, kg (B)
Rongelap	Adult Males	34	32±14	1.6±.063	67±11	6,700±3,600	.15±.028	220-17,000	.030-.21	85	.13±.020
Rongelap	Adult Females	30	34±16	1.5±.045	59±11	7,000±4,200	.12±.027	1,500-25,000	.048-.17	78	.096±.020
Rongelap	Adoles. Males	14	12±.97	1.4±.083	34±7.5	6,100±2,300	.075±.026	2,800-10,000	.018-.11	52	.083±.011
Rongelap	Adoles. Females	8	13±1.5	1.5±.070	38±6.0	9,400±3,100	.10±.046	3,400-13,000	.033-.18	52	.071±.011
Rongelap	Child Males	18	7.6±1.6	1.2±.080	22±2.8	3,900±2,100	.060±.031	1,300- 8,200	(C)-.089	48	.048±.006
Rongelap	Child Females	13	7.7±1.4	1.2±.087	22±3.0	3,600±1,800	.062±.022	1,800- 7,800	.018-.10	48	.046±.006
Utirik	Adult Males	61	36±17	1.6±.057	66±12	3,700±1,500	.15±.030	340- 7,400	.095-.22	85	.13±.020
Utirik	Adults Females	65	37±18	1.5±.048	62±15	2,500±1,100	.092±.024	370- 5,900	.044-.15	78	.099±.020
Utirik	Adoles. Males	21	12±1.1	1.4±.11	33±8.7	2,500±1,400	.076±.033	560- 5,200	.011-.15	52	.083±.011
Utirik	Adoles. Females	16	13±1.4	1.4±.080	38±7.0	2,300±1,000	.078±.021	850- 4,800	.047-.13	52	.071±.011
Utirik	Child Males	31	8.2±1.6	1.2±.12	23±5.1	1,900± 800	.051±.020	930- 4,300	.024-.096	48	.052±.006
Utirik	Child Females	20	8.3±1.7	1.2±.12	24±6.3	1,700± 800	.044±.019	590- 3,200	.017-.086	48	.049±.006
Rongelap	All Adults	64	33	1.6	63	6,800	.12	220-25,000	.030-.21	81	.12
Rongelap	All People	117	23	1.4	48	6,300	.094	220-25,000	(C)-.21	60	.090
Utirik	All Adults	126	37	1.6	64	3,100	.12	340-7,400	.044-.22	81	.12
Utirik	All People	214	25	1.4	49	2,700	.095	340-7,400	.011-.22	60	.093

A ≡ May 1979 - Comparison population at Majuro, minimum detection limit 37 Bq.
B ≡ Adapted from ICRP23 and adjusted according to mean age.
C ≡ Minimum detection limit, .01 kg.

Derived quantities

The k values calculated for each nuclide in the Rongelap and Utirik adult populations are given in Table 4. In the cases of the Rongelap and Utirik people for whom sequential body-burden data was available, k was found to have a positive value for ^{137}Cs, ^{65}Zn, ^{60}Co, ^{239}Pu and ^{90}Sr. The ^{239}Pu data for urine of three adult males at Rongelap in 1973 and 1976 provided a single tentative estimate of k. The value of k for ^{239}Pu was $7.5 \times 10^{-5} \pm 9.1 \times 10^{-5}$ d^{-1}. For ^{55}Fe, only one bioassay estimate was published as a result of studies by the BNL medical program (Be72; Co75); thus an estimate of k was not possible. For the estimate of cumulated ^{55}Fe intake, k was assumed equal to zero which implies that radioactive decay was the only cause of reduced daily activity intake during residence.

Where data were available for comparison, the values of k for ^{137}Cs and ^{90}Sr were found to be similar for both males and females as well as for residents of both Rongelap and Utirik. The yearly percent decrease in the atom ingestion rate was computed using equation (3) and the derived k value for each nuclide of interest. This intake relationship shows a 9% reduction in dietary ^{137}Cs for each year at Rongelap and Utirik. For dietary ^{90}Sr, an 8% reduction was estimated for each year at Rongelap and Utirik. The ^{60}Co and ^{65}Zn intakes were reduced rapidly during the first few years post-return to Rongelap Atoll. An 80% per yr reduction in dietary ^{65}Zn and a 60% per year reduction in dietary ^{60}Co were observed for adults. Also, for adult males at Rongelap, a tentative value of 3% per yr reduction in dietary ^{239}Pu was estimated from sparse data.

The derived quantity, daily activity ingestion rate on day of return to Rongelap (29 June 1957), was calculated for many individuals for ^{137}Cs and was plotted as a function of age in Fig. 4. An example of the variation in ^{137}Cs values for male and female intake on day of rehabitation is shown in this figure. Differences in the daily ingestion rate of stable elements at the same geographic location have been shown to occur among subgroups of a population (ICRP74). As an example of the dietary variation at Rongelap, it was observed that coconut sap was used both as a major food supplement

Table 4. Summary of estimates of committed effective dose equivalent

Nuclide	Atoll	Ingestion Rate on Day of Rehabitation, λP^o Bq d^{-1} ± s.d.	Decay Constant, λ d^{-1} ± s.d.	Dietary Removal Rate Constant, k d^{-1} ± s.d.	Fifty-Year Cumulated Intake Bq ± s.d.	60 kg Adult Average Committed Effective Dose Equivalent Sv ± s.d.
^{60}Co	Rongelap	$9.5\times10^{1} \pm 3.2\times10^{1}$	$3.6\times10^{-4} \pm 7.5\times10^{-8}$	$2.0\times10^{-3} \pm 1.9\times10^{-3}$	$4.0\times10^{4} \pm 2.8\times10^{4}$	$3.4\times10^{-4} \pm 2.3\times10^{-4}$
^{137}Cs	Rongelap	$3.9\times10^{2} \pm 1.3\times10^{2}$	$6.3\times10^{-5} \pm 7.1\times10^{-8}$	$2.0\times10^{-4} \pm 4.6\times10^{-4}$	$1.5\times10^{6} \pm 2.2\times10^{6}$	$2.2\times10^{-2} \pm 3.2\times10^{-2}$
^{65}Zn	Rongelap	$1.3\times10^{3} \pm 9.4\times10^{2}$	$2.8\times10^{-3} \pm 2.3\times10^{-6}$	$1.3\times10^{-3} \pm 5.3\times10^{-3}$	$3.1\times10^{5} \pm 3.4\times10^{5}$	$1.9\times10^{-3} \pm 2.0\times10^{-3}$
^{90}Sr	Rongelap	$2.1\times10^{0} \pm 1.1\times10^{0}$	$6.6\times10^{-5} \pm 3.2\times10^{-7}$	$1.7\times10^{-4} \pm 1.5\times10^{-3}$	$9.0\times10^{3} \pm 5.5\times10^{4}$	$5.3\times10^{-4} \pm 3.2\times10^{-3}$
^{55}Fe	Rongelap	$1.7\times10^{3} \pm 9.3\times10^{2}$	$7.1\times10^{-4} \pm 2.6\times10^{-6}$	(A)	$2.4\times10^{6} \pm 1.3\times10^{6}$	$4.8\times10^{-4} \pm 2.5\times10^{-4}$
^{60}Co	Utirik	$1.3\times10^{2} \pm 4.4\times10^{1}$	$3.6\times10^{-4} \pm 7.5\times10^{-8}$	$2.0\times10^{-3} \pm 1.9\times10^{-3}$(B)	$5.4\times10^{4} \pm 4.0\times10^{4}$	$4.4\times10^{-4} \pm 3.3\times10^{-4}$
^{137}Cs	Utirik	$2.1\times10^{2} \pm 1.1\times10^{2}$	$6.3\times10^{-5} \pm 7.1\times10^{-8}$	$1.8\times10^{-4} \pm 5.7\times10^{-4}$	$8.6\times10^{5} \pm 1.7\times10^{6}$	$1.3\times10^{-2} \pm 2.5\times10^{-2}$
^{65}Zn	Utirik	$2.1\times10^{4} \pm 1.6\times10^{4}$	$2.8\times10^{-3} \pm 2.3\times10^{-6}$	$1.3\times10^{-3} \pm 5.3\times10^{-3}$(B)	$5.2\times10^{6} \pm 7.5\times10^{6}$	$3.0\times10^{-2} \pm 4.4\times10^{-2}$
^{90}Sr	Utirik	$4.0\times10^{-1} \pm 3.0\times10^{-1}$	$6.6\times10^{-5} \pm 3.2\times10^{-7}$	$1.6\times10^{-4} \pm 2.1\times10^{-4}$	$1.7\times10^{3} \pm 1.7\times10^{3}$	$1.0\times10^{-4} \pm 1.0\times10^{-4}$
^{55}Fe	Utirik	$1.3\times10^{3} \pm 7.1\times10^{2}$	$7.1\times10^{-4} \pm 2.6\times10^{-6}$	(A)	$1.9\times10^{6} \pm 1.0\times10^{6}$	$3.6\times10^{-4} \pm 2.0\times10^{-4}$

(A) Assumed to equal zero.
(B) Rongelap values were used.

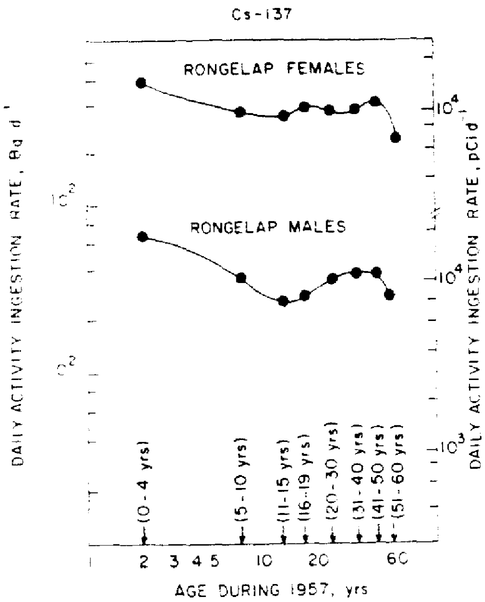

FIG. 4. Daily activity ingestion rate for day of return to Rongelap Atoll.

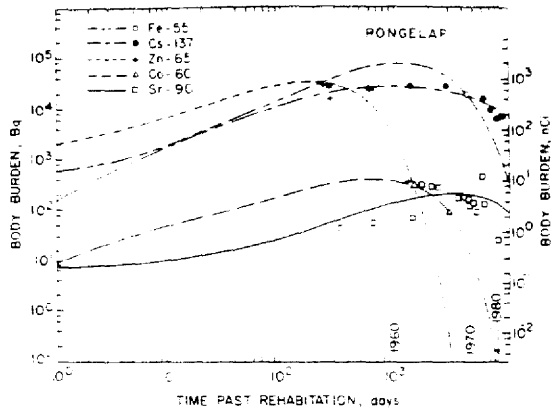

FIG. 5. Body-burden history for Rongelap adults.

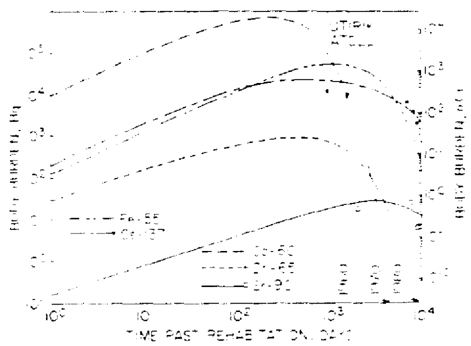

FIG. 6. Body-burden history for Utirik adults.

for infants, and then again (in a fermented form) in adult life by males as a component of daily fluid intake (Na80). Children and adolescents, however, were observed to receive a large portion of their daily fluid intake from two imported meals per day as part of the school lunch program. Studies indicated that coconuts and coconut tree sap provided the major source of ^{137}Cs in the diet (Le80a; Mi80). Thus, the undulating shape of Fig. 4 reflected this variation in the dietary intake of ^{137}Cs-contaminated foods.

Adult average values for activity ingestion rate on day of return were calculated for all nuclides. Results are listed in Table 4. This information, together with the estimate of k for the nuclide of interest, was used in equation (2) to estimate adult body-burden histories based on the assumption of declining continuous intake (see Figs. 5 and 6).

The declining continuous intake equation (3) provided a smooth body-burden function for Rongelap and Utirik adults. The equation was a tool to provide retroactive body-burden estimates during the early years post-return to Utirik. Few direct measurements were made at this time. The data plotted in Fig. 6 for ^{60}Co and ^{55}Fe were derived from Rongelap measurements.

Biological variation and errors in the collection and analysis of urine samples introduced larger errors in body-burden estimates than did direct whole-body counting. These variations can be observed in Fig. 5 where ^{90}Sr data vary widely from the theoretical curve. In contrast, the ^{137}Cs data fit the curve closely.

The method used to generate Figs. 5 and 6 was not chosen to minimize the weighted sum of squares of deviations of the body-burden estimates and measurements from the fitting function (equation (2)). Instead average values of k and P° were selected to represent all the body-burden data. For Rongelap, the ^{137}Cs body burdens varied from the fitted function by a maximum factor of 1.7 and an average factor of 1.4; the ^{90}Sr body burdens varied from the fitted function by a maximum factor of 3 and an average factor of 1.6. These factors reflect the quality of fit for directly measured body burdens and urine-derived body burdens in general.

The integral intake for 50 yr and the committed effective dose equivalent were derived quantities which depended on knowledge of k and P for each population subgroup. The 50-yr interval chosen for integral intake represented the years 1957–2007 for Rongelap residents. For Utirik residents, the 50-yr interval represented the years 1954–2004. The committed effective dose equivalent was based on this cumulated intake and both values can be found in Table 4.

An important result of using the fitting function was that ^{65}Zn and ^{137}Cs were the largest contributors to dose equivalent for each population. The ^{65}Zn dose equivalent was greatest at Utirik because of a three-month interval separating the BRAVO event and day of rehabitation and because of the shorter half-life of ^{65}Zn. The ^{137}Cs dose equivalent is important over the long term. It may be the chief nuclide of concern during an individual's lifetime post-rehabitation of a fallout-contaminated environment.

Statistical analysis of data

In the cases of ^{137}Cs, ^{65}Zn and ^{90}Sr, a large set of individual adult values for k and P° were available in addition to a set of adult average values. The whole-body counting techniques and urine-bioassay techniques employed were similar throughout the program's history. The short-term factors influencing the pattern of an individual's body burden, e.g. sickness, local diet changes, eating imported food, recent travel to uncontaminated areas, were factors which influenced the pattern of adult average body burden throughout the entire residence interval. Therefore the ratio of the standard deviation to the adult average k's and P°'s should have been equal to the same ratio for individual adult values. This was in fact the case for ^{137}Cs, ^{65}Zn and ^{90}Sr. The standard deviations and the adult average k's and P°'s for these nuclides were listed in Table 4. Tables of individual adult values were not reproduced here, however, individual body-burden data obtained in sequence are found in the references given in the introduction. These body burdens may be used with a fitting function to generate individual adult k's and P°'s.

The standard deviations for adult average k's and P°'s were used to estimate the standard deviations for adult average committed effective dose equivalents (see Table 4). Because the ratio of standard deviation to the average k and P° was the same for either adult average or individual adult k and P° data for ^{137}Cs, ^{65}Zn and ^{90}Sr, it was assumed to be true for ^{60}Co and ^{55}Fe. Thus, the standard deviations for the adult average k, P°, 50-yr cumulated intake and committed effective dose equivalent were estimated and given in Table 4 for each of these nuclides as well.

The standard deviation for the 50-yr cumulated intake for each nuclide does not include the deviations due to the variation or uncertainty of biological removal rate constants, radioactive decay constants or the fraction of an element eliminated via the urine pathway. These variations plus the variation of specific absorbed fraction of photon energy would introduce even greater standard deviation than that indicated in Table 4 for the estimate of committed effective dose equivalent.

External radiation exposure

External exposure-rate history curves for periods following resettlement are plotted on Figs. 7 and 8. These exposure rates were many times less than the 1 March 1954 exposure rates 12 hr after detonation of BRAVO. At that time they were estimated to average 2.3×10^6 nC kg^{-1} h^{-1} (8.9 R h^{-1}) for Rongelap Island, Rongelap Atoll and 8.9×10^4 nC kg^{-1} h^{-1} (0.34 R h^{-1}) for Utirik Island, Utirik Atoll (Le80b). These estimates were extrapolated values based on survey measurements made several days after the BRAVO detonation (OC68).

The external exposure at Rongelap and Utirik Atolls since rehabitation varied due to radioactive decay of BRAVO fallout and the addition of low-level contamination from several other nuclear tests (see Figs. 7 and 8). The estimated total 50-yr background subtracted exposure post-rehabitation was 5.9×10^{-4} C kg^{-1} (2.3 R) at Rongelap Island and 1.5×10^{-3} C kg^{-1} (5.6 R) at Utirik Island.

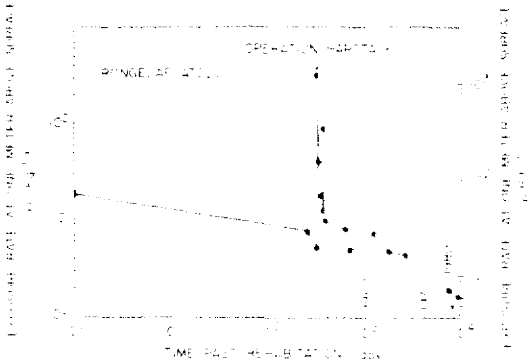

FIG. 7. Exposure-rate history at Rongelap Atoll.

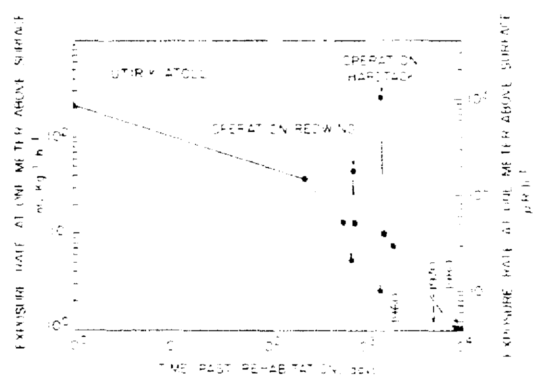

FIG. 8. Exposure-rate history at Utirik Atoll.

These values were based on the exposure-rate history for each island and do not include the exposure contribution before rehabitation or from natural background radiation. The background exposure rate was measured by Miltenberger and Greenhouse (Gr77b) and was 9.6×10^{-1} nC kg^{-1} h^{-1} (3.7×10^{-6} R h^{-1}). The 50-yr external effective dose equivalent was estimated to be 1.7×10^{-2} Sv (1.7 rem) at Rongelap and 4.1×10^{-2} Sv (4.1 rem) at Utirik. The external exposure rate is expected to decline to nearly natural background levels by the year 2072.

The ratio of internal committed effective dose equivalent to 50 yr of net external dose equivalent was 1.1 for Utirik and 1.5 for Rongelap. The internal portion of these dose equivalent ratios does not include the contribution from ^{239}Pu due to the uncertainty in Pu bioassay data.

DISCUSSION

The body-burden and urine data indicated a definite decline with time from the day of return atom ingestion rate for ^{137}Cs, ^{65}Zn, ^{60}Co and ^{90}Sr. The data for ^{239}Pu were uncertain but indicated a decline. These measurements of internal levels of radionuclides used in conjunction with the declining continuous intake equations provided an estimate of the total intake, the committed effective dose equivalent and the rate of decline of radionuclides in the overall diet. The data for directly measured body burdens at Rongelap Atoll were the best quality data for determining derived quantities.

Based on a declining continuous intake due solely to radioactive decay and the 1970 ^{55}Fe adult average body burden for each atoll, an estimate of the daily activity ingestion rate for ^{55}Fe on the day of return was calculated. Based on this ingestion rate, it was estimated that ^{55}Fe contributed a negligible amount to the total committed effective dose equivalent (see Table 4). The assumption that $k = 0$ for ^{55}Fe was made because sequential body-burden data were not available. Assigning $k = 2.0 \times 10^{-3}$ d^{-1}, the value determined for ^{60}Co, leads to an ^{55}Fe committed effective dose equivalent of 2.3×10^{-3} Sv (2.3×10^{-1} rem) for Rongelap adults. This is larger by a factor of 5 than the estimate for committed effective dose equivalent based on $k = 0$.

Use of the body-burden extrapolation equation leads to the conclusion that ^{65}Zn could have been the major contributor to the ingested activity during the first year post-rehabitation of Utirik Atoll (see Table 4). This was supported to some extent by a Japanese report (JCCRRER56) which indicated a rise in the photon count rate at the surface of various types of tuna retrieved from the Marshall Islands' fishing grounds from March to August 1954 (100–10,000 cpm). Fish with count rates greater than 100 cpm at the surface were discarded. Radiochemical techniques indicated the prominence of ^{65}Zn in the tuna's edible flesh. If it was assumed (1) that ^{65}Zn was the principal contributor to the external photon count rate, (2) that a self-sufficient living pattern existed on Utirik in which adults consumed 300 g of fish each day (Na80), and (3) that 1% of the fish eaten was tuna, then the daily activity ingestion rate might have been 7×10^{5} Bq d^{-1} (2×10^{1} μCi d^{-1}) in May and June and 7×10^{4} Bq d^{-1} (2×10^{0} μCi d^{-1}) in July and August of 1954. This method of estimating ^{65}Zn daily activity ingestion rates yields a 10-times greater estimate of total intake than the total intake suggested by body-burden extrapolation techniques (equations (2)). Although the ^{65}Zn total intake estimate indicated for Utirik adults in Table 4 was based on scanty data, it was made with fewer assumptions than was the above estimate using Japanese fishing data.

The validity of the ^{239}Pu data used to estimate the body burden at Rongelap Atoll (see Table 1) in 1973 had been considered by an Energy Research and Development Agency *ad hoc* committee. The committee concluded that because of the possibility of urine-sample contamination these data were uncertain. This may indeed have been a factor since a radiochemical analysis of BRAVO debris indicated Rongelap Atoll was contaminated with ^{239}Pu (Ts55). No special precautions had been taken when the urine samples were collected in the field, therefore not much credence could be given to these data.

In 1976, three male adults at Rongelap Atoll provided urine samples for ^{239}Pu analysis. Two yielded results below the minimum detection limit of 3.7×10^{-4} Bq l.$^{-1}$ (10 fCi l.$^{-1}$) and one yielded 3.3×10^{-3} Bq l.$^{-1}$ (90 fCi l.$^{-1}$). The average of these values along with the 1973 adult average data that was reported by Conard (Co75) were used to derive potential body burdens. The results were listed in Table 1.

The estimates for ^{239}Pu adult body burden were not used to derive values of intake and committed effective dose equivalent since they may have been the result of an erroneous urine collection technique and not the result of internal deposition. The potential for contamination also existed for ^{90}Sr, however the impact of contamination on dose assessment was much greater for Pu.

Questions concerning the ^{239}Pu estimates have led to a study of the sampling and analysis procedures which indicated that some ^{239}Pu in urine may not have been chemically recovered along with the tracer (Ry82). The extent of sample contamination during collection and the fundamental reasons for variation in recovery of ^{239}Pu from urine samples remain unanswered at this time. Several investigations are underway. In August 1981, fecal and urine samples were obtained from Rongelap and Utirik residents and are to be analyzed after complete dissolution followed by a liquid solvent extraction technique used in conjunction with a photon-electron rejecting liquid scintillation spectrometer developed by McDowell for low-level alpha spectroscopy (Mc72). The question of initial sample contamination will be answered following additional analysis of urine collected in 1980 from former Bikini Atoll residents.

CONCLUSION

The principle results of this investigation were that: ^{137}Cs and ^{65}Zn were major contributors to the committed effective dose equivalent; the overall body-burden pattern was one of initial increase followed by continuous decline over a period of years: the daily intake pattern was probably one of continuous decline, this conclusion was based on the fitting of sequential body-burden data to equation (2); the impact of each nuclide on internal committed effective dose equivalent was dependent upon the time between contamination and rehabitation: and the internal committed effective dose equivalent exceeded external dose equivalent during the rehabitation period. The sparse ^{239}Pu data indicated further research was necessary to estimate accurately the activity intake and committed effective dose equivalent from this nuclide.

For committed effective dose equivalent, the impact of nuclides with a short mean residence time in the diet (^{65}Zn, ^{60}Co) was greater at Utirik because the population reinhabited within months of the BRAVO event. The impact of nuclides with a long mean residence time in the diet (^{137}Cs, ^{90}Sr, ^{55}Fe) was greater at Rongelap because of greater initial contamination.

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