^{137}Cs EXPOSURE IN THE MARSHALLESE POPULATIONS: AN ASSESSMENT BASED ON WHOLE-BODY COUNTING MEASUREMENTS (1989–1994)

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Abstract—**The Marshall Islands were the site of numerous tests of nuclear weapons by the United States. From 1946 to 1958, nuclear devices were detonated at Enewetak and Bikini Atolls. Following the inadvertent contamination of the northern islands downwind of the 1954 Bravo Test, Brookhaven National Laboratory became involved in the medical care and the radiological safety of the affected populations. One important technique employed in assessing the internally deposited radionuclides is whole-body counting. To estimate current and future exposures to ^{137}Cs, data from 1989 to 1994 were analyzed and are reported in this paper. During this period, 3,618 measurements were made for the Marshallese. The cesium body contents were assumed to result from a series of chronic intakes. Also, it was assumed that cesium activity in the body reaches a plateau that is maintained over 365 d. We estimated the annual effective dose rate for each population, derived from the recommendations of the International Commission on Radiological Protection. The average ^{137}Cs uptake measured by the whole-body counting method varies from one population to another; it was consistent with measurements of external exposure rate. The analysis, though based on limited data, indicates that there is no statistical support for a seasonal effect on ^{137}Cs uptake. The critical population group for cesium uptake is adult males. Within the 5-y monitoring period, all internal exposures to ^{137}Cs were less than 0.2 mSv y^{-1}. Similarly, a persistent average cesium effective dose rate of 2 μSv y^{-1} was determined for Majuro residents.**
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Key words: Marshall Islands; whole body counting; cesium; dose assessment

INTRODUCTION

The Republic of the Marshall Islands (RMI) is located in the central Pacific Ocean about 3,500 km southwest of Hawaii and 4,500 km east of Manila, Philippine Islands; the islands lie near the intersection of the Equator and the International Dateline (Fig. 1). The RMI consists of 29 coral atolls and 5 coral islands, all just above sea level. The total land area is about 180 km^2 (Patterson 1986). In 1946, the RMI was chosen for nuclear testing because of its remoteness, extremely low population density, and its geological features (DNA 1981).

Between 1946 and 1958, numerous nuclear devices and weapons were tested in the northern RMI at Bikini and Enewetak Atolls. Although these tests were considered vital to the defense of the free world during the cold war, the resulting radiological contamination and clean-up efforts remain as critically important consequences. Environmental contamination still is a health and safety issue for the RMI population (Lane 1989; Kohn 1988, 1989; National Research Council 1982, 1994; Baverstock et al. 1995). Many technical and non-technical reports on environmental, medical, radiological, health, and safety impacts on the Marshallese populations are available (AEC 1956a, 1956b; Committee on Atomic Energy 1957; Conard et al. 1975; ERDA 1977; DOE 1980; Tipton and Meinbaum 1981; U.S. Committee on Interior and Insular Affairs 1989; U.S. Committee on Energy and Natural Resources 1991; Conard 1992; U.S. Committee on Natural Resources 1994).

^{137}Cs, a product of uranium fission, has a considerable public-health impact because of its high yield and relatively long half-life of 30 y in the environment. In recent whole-body counting (WBC) field missions, ^{137}Cs was the only long-lived, gamma-emitting, weapons-related isotope detected in the Marshallese. Even the 5-y half-life of ^{60}Co, a common activation product generated in nuclear tests, is below our determined minimum detectable amount (MDA). Cesium compounds in the environment are water soluble and, therefore, may be transported and widely dispersed. Cesium also adheres to many components of soil from which uptake into the biota occurs. The major exposure pathways of cesium intake in the monitored populations are from inhalation of contaminated dust particles resuspended in the air, and from ingestion of contaminated foods stuffs, drinking water, and soil particulates (NCRP 1977, 1985a; UNSCEAR 1993; IAEA 1988). The effective half-life of ^{137}Cs in humans is about 110 d, which is much shorter than its radiological decay half-life ($T_{½} \approx$ 30-y) (NCRP 1977). Once in the body, cesium is quickly and uni-

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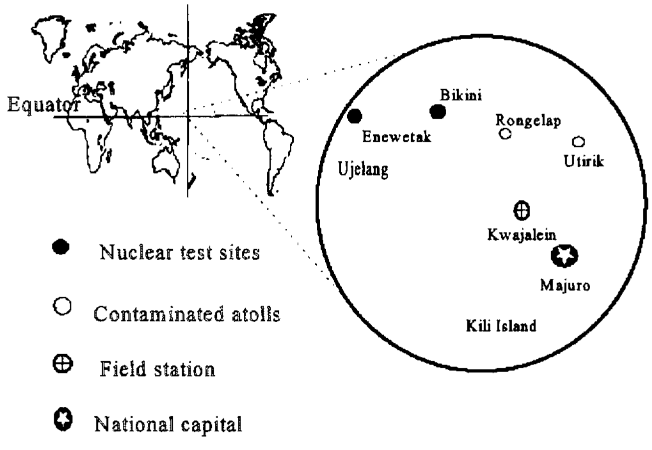

Land and water areas are 180 and 2,500,000 km^2, respectively.

Fig. 1. View of the Republic of the Marshall Islands' global location.

formly distributed.

The U.S. Department of Energy (DOE) Office of Health assigned responsibility for the Marshallese radiological assessments to Brookhaven National Laboratory (BNL) and Lawrence Livermore National Laboratory (LLNL). The two laboratories use independent methodologies for radiological monitoring and dose assessment. BNL has used whole-body counting and radiological analyses of urine samples (Cohn 1956, 1963; Cohn and Gusmano 1965; Greenhouse et al. 1977, 1979, 1980; Miltenberger et al. 1980, 1981; Lessard et al. 1980a, 1980b, 1984; Sun et al. 1991, 1992, 1993, 1995). LLNL researchers have based their assessments upon data from measuring levels of radionuclides in the environment (e.g., soil, water, plants, animals), from assessments of intake and distribution pathways for radionuclides entering the body, and from analyses of dietary patterns (Noshkin et al. 1979, 1988, 1994; Jennings and Mount 1983; Robison 1983; Shingleton et al. 1987; Robison and Stone 1992; Robison et al. 1980, 1982, 1987, 1988; Kercher and Robison 1993). The local foods in the northern atolls of RMI are coconuts, leaves, breadfruit, pandanas, taro, arrow root, birds, and a variety of seafood. Many assessments of cesium doses among Marshallese have considered the correlation between dietary patterns and nuclide concentrations in foodstuffs (Held et al. 1965; Hardy et al. 1964; Naidu et al. 1980; Robison 1983; Simon and Graham 1996). The BNL and LLNL determinations of Marshallese ^{137}Cs uptakes were first presented together during the 19th Annual Meeting of the NCRP (Robison 1983), and later, along with their results on ^{239}Pu uptake, at the Eighth International Radiation Protection Association Congress (Sun et al. 1992). Both ^{137}Cs and ^{239}Pu doses determined by the two laboratories substantially agree.

In March 1990, a six-member Marshall Islands Independent Scientific Advisory Committee (MIISAC)[†] reviewed BNL's quality assurance performance for all components of the bioassay monitoring and internal dose assessment programs for the Marshallese. The Committee stated that the WBC procedures used for estimating Marshallese body contents of ^{137}Cs, ^{60}Co, and ^{40}K conformed to recognized standards for measuring these nuclides *in vivo* (Hall et al. 1990).

In vivo WBC is a simple, accurate, and effective method of determining the quantity of gamma emitters in the body. WBC missions were conducted by BNL for the people of Bikini, Enewetak, Rongelap, and Utirik in 1989, 1991, 1993, and 1994. During this period, ^{137}Cs was the only fission product that was detected in these populations. This paper compiles the WBC results and associated dose analyses for these populations from 1989 to 1994.

On 21 February 1992, the DOE and RMI signed a Memorandum of Understanding (MOU 1992). Two action limits were agreed upon as conditions of the Rongelap Resettlement: (1) Rongelap residents would not receive a calculated annual effective dose of 1 mSv above local natural background, and (2) they would not be exposed to more than 630 Bq kg^{-1} (17 pCi g^{-1}) of transuranium elements in the soil of inhabited areas or food-gathering ones. Where these limits were exceeded, radiological dose-reduction methods would be initiated. For this reason, the cesium effective dose rate (μSv y^{-1}) is reported in this paper.

MATERIALS AND METHODS

Whole-body counting system

Whole-body counting was performed in two shadow-shielded chairs transported within RMI using a contracted vessel. Each WBC unit has a single thallium-doped sodium iodide detector, 29.2 cm (11.5 inch) diameter by 10.2 cm (4 inch) thick, manufactured by Bichron.[‡] The WBC detector is mounted on a pivoted arm allowing it to be centered across the front of the chair during counting and moved out of the way to allow access to the chair (Fig. 2). Since 1989, a Canberra System 100 (S-100)[§] multichannel analyzer (MCA) has been used in conjunction with an IBM[||] personal computer (model Thinkpad 750). The counting signal is registered through the MCA's circuit board and the isotopes identified and their activity assessed with Canberra's GAMMA-AT[§] software.

The WBC system is calibrated with a bottle mannequin absorber (BOMAB) phantom. Energy identifications are based on four distinct photon peaks: 0.662

[†] The members were Roscoe Hall (Chairman, deceased; Savannah River National Laboratory), Norman Cohen (Environmental Measurement Laboratory), Keith Eckerman (Oak Ridge National Laboratory), Henry Kohn (Rongelap Reassessment Laboratory), Leonard Newman (BNL), and Hylton Smith (National Radiological Protection Board).

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[§] Canberra, 800 Research Parkway, Meriden, CT 06450.

[||] IBM, http://www.ibm.com/.

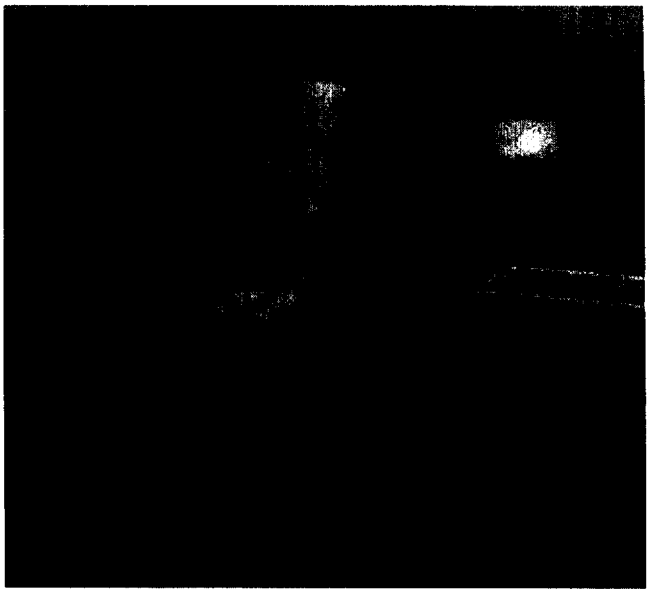

Fig. 2. Brookhaven's WBC system with BOMAB.

(^{137}Cs), 1.17 and 1.13 (^{60}Co), and 1.46 MeV (^{40}K). Counting efficiencies are established for four geometries by selecting whole or partial sets of the BOMAB phantom's segments called large, medium, small, and infant. The counting efficiency obtained with the large geometry is used to analyze spectra from persons weighing 60 kg or more, the medium geometry for people between 40 and 60 kg, and the small geometry for youngsters age 3 y or older who weigh less than 40 kg. The infant geometry is used for children younger than 3 y (weighing 8–15 kg). For an empty chair, the MDA of ^{137}Cs and ^{60}Co at the 95% confidence level for the present WBC system was established at 60 and 52 Bq, respectively (NCRP 1985b), in a 15-min count.

RESULTS AND DISCUSSIONS

Whole-body measurements and cesium activities

During BNL's 1989 field mission (July and August), staff visited Ebeye, Enewetak, Majuro, Mejatto, Utirik, and Bikini Islands. In 1991 and 1993, there were two missions each year, one in January–February (winter season), and the other in June–August (summer season). Weather and sea conditions during the winter season prohibit WBC operations at Mejatto Island, restricting all visits there to summer. The most recent WBC mission was conducted at Bikini, Enewetak, Mejatto, and Ebeye Islands in summer, 1994. All measurements were made on volunteers from among the Marshallese who either were directly exposed to fallout radiation, or resided on the Bikini, Enewetak, Rongelap, and Utirik Atolls. As a quality assurance procedure, 5% of the volunteers at each counting location were re-counted, either in the same chair or in the second chair. Table 1 shows that 3,618 WBC measurements were made on the Marshallese during these 5 y, including 13 Bikinian DOE employees who were working on Bikini Island, Bikini Atoll. The numbers given are for each population group, irrespective of where the whole-body counts were made. No attempt was made to examine the former inhabitants of Bikini located on Kili Island.

Table 1. Distribution of WBC measurements.[a]

Population	1989	1991	1993	1994
Enewetak	228	355	442	283
Rongelap	273	446	216	333
Utirik	423	290	316	—
Bikini[b]	8	—	—	5
Column sum:	932	1,091	974	621

[a] Number of participants is about 10% fewer due to quality assurance by double counting.
[b] These Bikinians were DOE workers and members of their immediate families.

Enewetak

As Fig. 3 shows, in December 1947, before the nuclear testing program began on Enewetak Atoll, all residents of Enewetak were relocated to Ujelang, Ujelang Atoll, RMI (~250 km southwest of Enewetak) (DNA 1981). After the tests ended, a major radiological cleanup and rehabilitation program was conducted during the late 1970's and early 1980's. After the cleanup, the people of Enewetak began to resettle the southeastern part of Enewetak Atoll (Enewetak, Japtan and Medren Islands) in 1980, at which time a routine WBC program was started.

Table 2 summarizes the ^{137}Cs body content of Enewetak volunteers measured during the 1989, 1991,

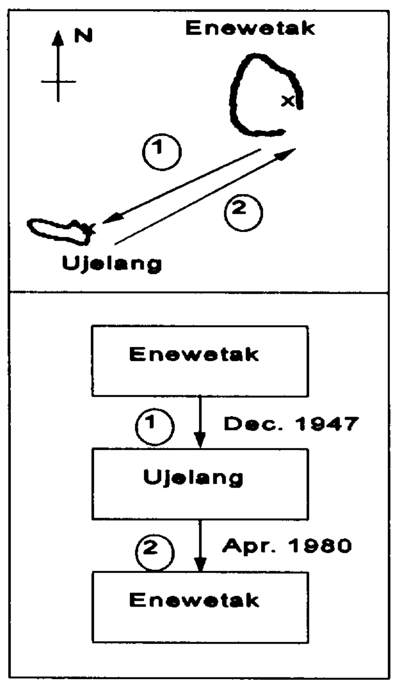

Fig. 3. Relocation timeline for Enewetak population.

Table 2. Comparison of ^{137}Cs measurements of the Enewetak population by age and gender.

	July–August 1989					February 1991					February 1993					August 1994				
Value	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All
	Both sexes					*Both sexes*					*Both sexes*					*Both sexes*				
No. counted	146	58	24	0	228	245	62	47	1	355	261	86	91	4	442	105	35	125	18	283
Mean	615	398	180		514	1,043	575	192	50	846	276	94	63	39	195	365	146	69	40	187
SD[b]	676	564	155		629	1,427	744	222		1,267	351	63	51	13	289	354	96	50	20	261
Maximum	4,059	3,189	715		4,059	14,122	3,589	939		14,122	3,646	283	346	55	3,646	2,463	378	346	113	2,463
Median	419	253	112		318	632	272	62		448	165	74	43	36	87	320	128	54	33	88
	Males					*Males*					*Males*					*Males*				
No. counted	80	24	15	0	119	129	30	23	0	182	131	47	47	2	227	58	26	63	8	155
Mean	829	553	190		693	1,271	884	233		1,076	358	102	69	29	242	457	139	74	45	227
SD	818	798	180		791	1,683	944	224		1,508	439	60	58	2	362	416	91	49	29	315
Maximum	4,059	3,189	715		4,059	14,122	3,589	826		14,122	3,646	260	346	31	3,646	2,463	315	281	113	2,463
Median	579	300	113		454	824	490	169		628	210	87	43		110	389	119	59	35	108
	Females					*Females*					*Females*					*Females*				
No. counted	66	34	9	0	109	116	32	24	1	173	130	39	44	2	215	47	9	62	10	128
Mean	355	289	163		319	790	286	153	50	604	194	85	57	49	145	252	167	64	36	138
SD	285	274	108		275	1,023	281	218		891	201	65	42	10	171	212	111	50	8	163
Maximum	1,582	1,423	293		1,582	5,809	1,067	939		5,809	1,171	283	290	55	1,171	1,145	378	346	52	1,145
Median	303	219	112		253	462	186	46		348	104	61	44		61	173	158	49	33	71

[a] All values are in the unit Bq, exept counts.
[b] Standard deviation.

1993, and 1994 field missions. All these values were based on individual weight and later were classified by age and sex. The Marshallese age groups were defined as follows: adult (A) were individuals 16 y and older; teenage (T) were 11–15 y; juvenile (J) were children of 3–10 y; and infants encompassed birth through 2 y of age.

Table 2 shows that 146 adults were measured in 1989: 80 males and 66 females. The arithmetic mean ($\bar{x}$) and standard deviation (SD) of this group was 615 ($\bar{x}$) ± 676 (1 SD) Bq. The median value was 419 Bq, and the measured values ranged from 4,059 Bq to less than the MDA.

Table 2 also shows that ^{137}Cs body contents in the Enewetak population increased from 1989 to 1991, then decreased in 1993, and rose somewhat in 1994. Since cesium uptake directly reflects dietary intake and is proportional to the environmental concentration of cesium, the changes cannot be completely explained from dietary patterns (discussed later). However, they may reflect the consumption of contaminated food from the northern islands. In 1991, the highest measurement of ^{137}Cs body content obtained during any mission, 14 kBq (about 0.2 mSv effective dose based on a single acute ingestion), was from an adult male on Enewetak. A few weeks earlier he had camped on Enjebi, an uninhabited northern island in the Enewetak Atoll where the median ^{137}Cs activity in the soil was about 100 times greater than in the inhabited southeastern islands (ERDA 1977), and had eaten the local food; this could account for his elevated body content.

Measurement of external exposure rate is the quickest method to assess ground contamination resulting from fallout. For example, Simon and Graham (1995) reported the median external effective dose rate from ^{137}Cs on Enewetak, Medren, Japtan, and Enjebi Islands was about 2, 5, 10, and 200 $\mu Sv\ y^{-1}$, respectively. The low dose rate for Enewetak Island was an overall result of removing the top 30 cm of soil during the DNA cleanup program. Since then, the entire population has relied primarily on imported food from the United States Department of Agriculture. Further, the major local food supplies, such as coconuts, leaves, and vegetables must be collected from neighboring islands (e.g., Medren and Japtan). Hence, the WBC measurements should not be expected to correlate with the reported low exposure rates at Enewetak Island. Our WBC measurements also show that the adult male group encompasses the maximum individual and the highest average cesium body content. Cesium distributes uniformly throughout the body so that a larger body mass retains more cesium (ICRP 1990; 1993). Although ICRP uses the same model for both genders, apparently the average content of adult females is approximately half that of the males for the people of Enewetak. This difference may reflect fishing and other outdoor activities by males and their associated consumption of foods in which cesium has been concentrated from the northern islands of the atoll.

Coefficient of variation (CV = $\bar{x}$/SD) analysis often is used to assess the stability of data. The CV values associated with the 1991 WBC data are larger than all other years. Since cesium uptake is proportionally related to its concentration in food stuffs, the scattered data may be influenced by the consumption of local foods from other islands. Since the CV values in 1994 were less than unity, the corresponding WBC results are expected to better represent cesium body content for the Enewetak population.

Rongelap

The inclusion of the Rongelap and Utirik populations in BNL's WBC program resulted from the 1954

Bravo test when, due to a large unexpected yield and tropospheric transport, radioactive dust was carried eastward (AEC 1956b). Two hundred and ninety people [64 Rongelapese, 18 Ailinginaese, 157 Utirikese, 28 American servicemen (Cronkite and Bond 1956), and 23 Japanese fishermen (Kumatori et al. 1980)] were exposed to this dust. The estimated external radiation was about 1.75 Gy for the Rongelap population and 0.14 Gy for the Utirik population (Sondhaus et al. 1956); each population was evacuated elsewhere in the Marshall Islands, for 3 y and for 3 mo, respectively. Fig. 4 shows their detailed relocation timeline, including the 1985 relocation of the Rongelapese to Mejatto Island, Kwajalein Atoll, because of their concern with the health and environmental effects of residual fallout (U.S. Committee on Interior and Insular Affairs 1989; U.S. Committee on Energy and Natural Resources 1991; Baverstock et al. 1995; Sun et al. 1995).

Table 3 summarizes WBC data for the Rongelapese living on Mejatto Island. The average cesium body content in the population has decreased slightly over the 5-y monitoring period. The latest data show that cesium intake at this location is similar to levels in the southern RMI areas unaffected by Bravo fallout. It is important to note that because of the low level of intake and the relatively short effective half-life of ^{137}Cs in humans (110 d), the measurement could only detect post-intake of cesium within 160 d ($\sim 110 \div 0.693$). Therefore, the current estimates represent peoples' diet and lifestyle on Mejatto Island only. Simon and Graham (1995) reported that the median external effective dose rate from ^{137}Cs on Mejatto Island (N. Kwajalein Atoll) and Rongelap Island were at about 2 and 150 $\mu Sv\ y^{-1}$, respectively. These

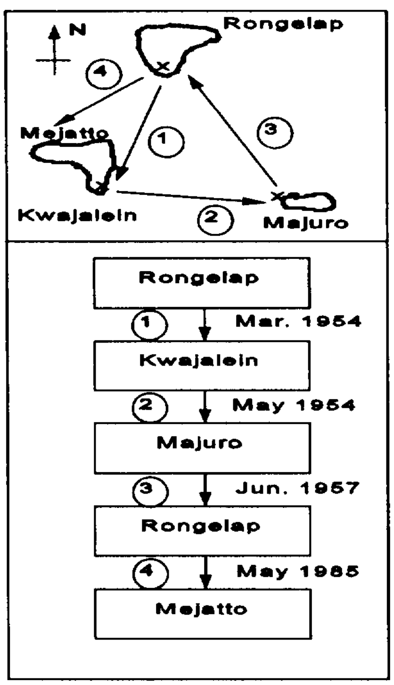

Fig. 4. Relocation timeline for Rongelap population.

values suggest that ^{137}Cs exposure upon return to Rongelap might be significantly greater than the values reported in Table 3.

Utirik

The people of Utirik remained on their island, except for 3 mo after the Bravo detonation (Fig. 5). Table 4 summarizes the statistical WBC parameters from 1989 to 1993 for the Utirikese. The average cesium body content in the Utirik population was 761 ± 778 (1 SD), 904 ± 766 (1 SD), and 310 ± 357 (1 SD) Bq in the 1989, 1991, and 1993 missions, respectively; cesium exposures increased from 1989 to 1991, and decreased from 1991 to 1993. Many Utirikese reside on Majuro and Ebeye Islands for economic and social reasons. Majuro Island is the capital of the RMI and has become the most socioeconomically developed island in the Marshall archipelago. Ebeye Island attracts many Marshallese people because of the job opportunities at the nearby Kwajalein U.S. Military Base. Due to the short retention time of cesium in the body and changes in dietary intake of cesium at each location, it is difficult to obtain a reliable population average value for Utirik inhabitants without separating the population according to each WBC location. Therefore, neither the annual average cesium burdens nor the trend of population uptake, shown in Table 4, is expected to be representative of the people living on Utirik Island.

Table 5 lists the values of WBC measurements obtained from the Utirik population at the location where the WBC was performed: Ebeye, Majuro, and Utirik Islands. These islands are more than 200 km apart, and only Utirik lies in the downwind direction that received fallout from the Bravo test. At each place, the lifestyle and foods vary as do the environmental cesium levels.

The median external effective dose rates from ^{137}Cs on Ebeye Island (S. Kwajalein Atoll), Majuro Island, and Utirik Island were about 1.5, 0.1, and 18 $\mu Sv\ y^{-1}$, respectively (Simon and Graham 1995). Simon and Graham indicate that the ^{137}Cs concentrations in soil on both Majuro and Ebeye Islands are only slightly above those expected from global fallout deposition. Thus, although the values in Table 5 can be used to determine baselines for cesium uptake for inhabitants of Majuro and Ebeye, it is only the values obtained on Utirik Island that reflect cesium intake related to the United States nuclear tests done in the Marshall Islands, and, therefore, are appropriate for dose estimations of Utirik populations.

Bikini

Bikini residents were first relocated to Rongerik Atoll before testing began in 1946, and then moved to Kili Island in 1948. For radiological safety reasons, they have been unable to resettle in their homes, except for a short period in the 1970's when the island was thought to be safe for habitation (Greenhouse and Miltenberger 1977; Greenhouse et al. 1977, 1979, 1980; Robison 1983; Lessard et al. 1980b, 1984). Today, the majority of

Table 3. Comparison of ^{137}Cs measurements of the Rongelap population by age and gender.

	July–August 1989					June 1991					July–August 1993					August 1994				
Value	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All
	Both sexes					*Both sexes*					*Both sexes*					*Both sexes*				
No. counted	193	63	17	0	273	257	113	76	0	446	129	54	33	0	216	199	69	63	2	333
Mean	90	66	66		83	78	56	47		67	70	63	40		64	70	56	42	35	61
SD[b]	51	16	27		45	40	26	16		36	39	34	11		36	45	34	13	2	40
Maximum	435	131	129		435	283	168	146		283	249	194	71		249	370	205	83	37	370
Median	74	68	50		72	63	46	44		57	53	49	36		49	55	42	36		49
	Males					*Males*					*Males*					*Males*				
No. counted	90	29	9	0	128	124	60	41	0	225	63	33	15	0	111	98	34	35	1	168
Mean	95	67	65		87	84	63	47		72	70	68	41		65	75	66	44	37	66
SD	61	16	27		54	45	34	19		41	41	38	13		39	43	35	14		39
Maximum	435	103	119		435	283	168	146		283	249	194	71		249	211	167	83		211
Median	76	68	49		73	65	47	44		58	50	56	36		50	57	52	37		51
	Females					*Females*					*Females*					*Females*				
No. counted	103	34	8	0	145	133	53	35	0	221	66	21	18	0	105	101	35	28	1	165
Mean	85	66	67		80	73	48	46		63	71	55	40		62	65	47	39	34	56
SD	39	16	30		36	34	10	11		30	37	25	10		34	46	31	11		41
Maximum	246	131	129		246	229	80	107		229	222	130	69		222	370	205	69		370
Median	73	68	59		71	63	46	44		56	58	42	35		48	51	37	36		43

[a] All values are in the unit Bq, exept counts.
[b] Standard deviation.

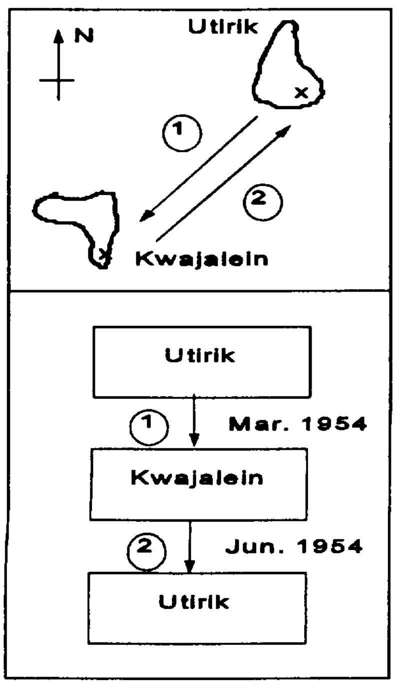

Fig. 5. Relocation timeline for Utirik population.

the Bikini people still live on Kili Island (Ellis 1986). Fig. 6 gives a detailed account of when and where the Bikinians were relocated. The median external effective dose rates from ^{137}Cs on Kili and Bikini Island were measured at about 1 and 600 μSv y^{-1}, respectively (Simon and Graham 1995). Plans are being studied for a radiological cleanup at Bikini.

Table 6 gives the WBC measurements for 13 Bikini residents who either worked for DOE or were family members of DOE workers at Bikini in 1989 and 1994. Their food was imported, and their living conditions were more modern so that measurements of cesium intake are likely to be completely different and inappropriate for estimating uptake of inhabitants with a more traditional style.

Seasonal variations

In both 1991 and 1993, there were two field missions on Majuro Island. The summer seasons' and winter seasons' WBC results were compared to establish whether there are seasonal variations in cesium uptake. Such seasonal variations have been reported and were attributed to increased water consumption during the summer and to seasonal variations in the concentration of the isotope in food stuffs (Hanson et al. 1964; UNSCEAR 1993). Table 7 compares all the measurements obtained on Majuro Island. These data suggest that 1) for adults (male and female), the cesium average in February is higher than in July, and 2) unlike adults, there is no seasonal difference for youths. Statistical comparisons of the 1991 and 1993 seasonal data only partially support these statements because of the high standard deviations. The apparent variations for the adults may reflect more outdoor activity in the summer, such as sailing and fishing, and less in the winter due to ocean waves and strong winds. Possibly because of increased activity in the summer, people may eliminate fluid faster, which lowers cesium concentration. Youths' land-based, outdoor activities are more consistent throughout the year; in fact, the difference in temperature and precipitation between February and July are small in the RMI.

The data in Table 7 were derived from a mixed, exposed population (i.e., Enewetak, Rongelap, and Utirik) whose whole-body counts were taken at Majuro Island. Hence, these WBC data should neither be com-

Table 4. Comparison of ^{137}Cs measurements of the Utirik population by age and gender.

Value[a]	July–August 1989					February 1991					July–August 1993				
	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All
	Both sexes					*Both sexes*					*Both sexes*				
No. counted	316	80	27	0	423	212	47	27	4	290	184	82	50	0	316
Mean	810	673	449		761	997	780	492	185	904	400	263	53		310
SD[b]	784	794	554		778	772	835	317	106	766	405	249	55		357
Maximum	3,537	2,946	1,936		3,537	3,992	3,352	1,342	329	3,992	2,048	1,289	285		2,048
Median	682	193	77		585	904	585	491	158	801	260	191	36		137
	Males					*Males*					*Males*				
No. counted	164	42	11	0	217	106	27	14	3	150	95	39	26	0	160
Mean	975	821	646		929	1,187	929	572	215	1,063	481	286	51		363
SD	876	934	631		878	865	1,037	356	107	882	466	244	42		411
Maximum	3,537	2,946	1,936		3,537	3,992	3,352	1,342	329	3,992	2,048	809	209		2,048
Median	902	167	848		830	1,171	642	674	200	889	390	231	36		179
	Females					*Females*					*Females*				
No. counted	152	38	16	0	206	106	20	13	1	140	89	43	24	0	156
Mean	632	510	314		585	807	580	407	96	732	315	242	55		255
SD	625	574	468		609	614	375	256		574	307	255	67		283
Maximum	2,579	2,322	1,327		2,579	3,544	1,290	806		3,544	1,300	1,289	285		1,300
Median	450	193	67		318	829	523	390		752	162	174	35		95

[a] All values are in the unit Bq, except counts.
[b] Standard deviation.

Table 5. Comparison of ^{137}Cs measurement of the Utirik population obtained on various islands.

Value[a]	July–August 1989					February 1991					July–August 1993				
	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All
	Utirik					*Utirik*					*Utirik*				
No. counted	154	37	10	0	201	156	30	20	4	210	106	47	6	0	159
Mean	1,297	1,316	1,109		1,291	1,268	1,044	583	185	1,150	609	407	171		533
SD[b]	703	721	345		692	698	809	267	106	721	387	229	98		358
Maximum	3,537	2,946	1,936		3,537	3,992	3,352	1,342	329	3,992	2,048	1,289	285		2,048
Median	1,120	1,135	963		1,120	1,109	788	571	158	988	558	397	178		483
	Majuro					*Majuro*					*Majuro*				
No. counted	111	29	9	0	149	56	17	7	0	80	57	26	32	0	115
Mean	456	149	63		373	240	315	233		255	143	81	38		100
SD	616	303	18		566	347	676	322		430	243	113	13		184
Maximum	3,323	1,648	93		3,323	1,996	2,126	705		2,126	1,519	460	109		1,519
Median	146	74	50		82	81	58	46		71	54	40	35		45
	Ebeye										*Ebeye*				
No. counted	51	14	8	0	73						21	9	12	0	42
Mean	111	59	60		95						48	37	35		42
SD	132	9	12		113						14	3	1		12
Maximum	791	71	79		791						106	42	36		106
Median	70	62	59		68						46	35	35		40

[a] All values are in the unit Bq, except counts.
[b] Standard deviation.

pared to those of Table 5, nor used to develop a cesium baseline for Majuro Atoll.

Cesium retention model and dose calculation

For a single acute uptake, the ICRP recommended the following two-exponential-term function to describe the systemic retention of cesium in the body (1978, 1990, 1993):

$$R(t) = ae^{-\left(\frac{\ln 2}{T_1}\right)t} + (1 - a)e^{-\left(\frac{\ln 2}{T_2}\right)t}, \qquad (1)$$

where a and T_1 are the distribution fraction and effective half-time, respectively. The recommended adult values for a, T_1, and T_2 are 0.1, 2 d, and 110 d, respectively.

Because of the short half-time (T_1) and a low distribution fraction, the first component of the cesium retention function is relatively unimportant for assessing dose. However, the long half-time component of cesium for the Marshallese people was investigated in detail by Hardy et al. (1964) and Miltenberger et al. (1981), and the overall results agree with ICRP's (1978, 1990, 1993) suggested value of 110 d. Henrichs et al. (1989) report cesium gut-transfer coefficient (f_1) values for adult males and females of 0.75 ± 0.06 (1 SD) and 0.81 ± 0.06 (1 SD), respectively. However, the f_1 value recommended by the ICRP gives more conservative dose estimates, implying that uptake equals intake. Therefore, the ICRP

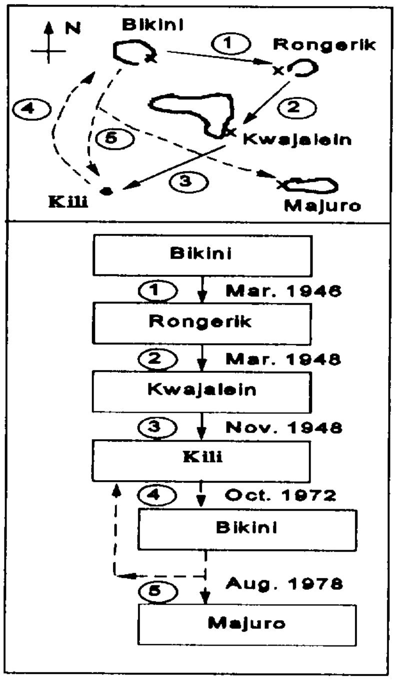

Fig. 6. Relocation timeline for Bikini population.

age-specific dose conversion factors (DCFs) for ingestion intake of cesium can be used directly to interpret the WBC data.

We based the computation of ^{137}Cs doses on ICRP's (1990, 1993) age-specific biokinetic data and retention function for radiation protection purposes. The difference between DCF's derivation in the two publications arises solely from the tissue-weighting factor recommended by ICRP Publications 26 (1977) and 60 (1991). Table 8 compares the two sets of values for each age-specific DCF; the difference between each corresponding value is not large due to the rapid solubility and short retention time of cesium in the body. For conservative reasons, a DCF of 1.4×10^{-2} μSv Bq^{-1} was used for all teenage and adult dose calculations in this study, and a DCF of 2.1×10^{-2} μSv Bq^{-1} was used for infants. The product of DCF and intake yield a committed effective dose from the year of intake to age 70 y.

Cesium body contents in Marshallese must be interpreted on the basis of chronic intakes (Lessard et al. 1980a; Sun et al. 1991, 1992; Kercher and Robison 1993). A computational algorithm was developed to interpret the dose from a chronic exposure pattern using the ICRP DCFs, which had been designed for single, acute intakes. From the biokinetic data for an adult given in Table 8-1 of ICRP Publication 56, an effective dose rate attributable to a steady-state, chronic uptake of cesium in an adult (A) can be estimated by the following equation:

$$\text{Effective dose rate } (\mu\text{Sv y}^{-1})_{\text{Adult}} = 2.55 \times 1.4 \times 10^{-2} \ (\mu\text{Sv Bq}^{-1}) \times \text{Body Content (Bq)}, \quad (2)$$

where 2.55 is a constant factor used to convert from committed effective dose to annual effective dose rate in Reference Man (ICRP 1975). The Appendix describes the computational algorithm. It accounts for the fact that 90% of the committed effective dose will be received within the first year of intake and that the build-up constant for chronic intakes of cesium is the ratio of an integrated transformation in this first year due to a chronic intake to that of a single, recent acute intake. Similarly, the conversion factors for committed effective dose to individuals of 3 mo, 1 y, 5 y, 10 y, and 15 y are, respectively, 15.8, 19.5, 12.3, 6.88, and 3.25. Therefore, the annual effective dose rate attributable to a steady-state and chronic uptake of cesium in teens (T), juveniles (J), and infants (birth to 2 y) can be estimated by the following equations:

$$\text{Effective dose rate } (\mu\text{Sv y}^{-1})_{\text{Teen}} = 3.25 \times 1.4 \times 10^{-2} \ (\mu\text{Sv Bq}^{-1}) \times \text{Body Content (Bq)}, \quad (3)$$

$$\text{Effective dose rate } (\mu\text{Sv y}^{-1})_{\text{Juvenile}} = 6.88 \times 1.4 \times 10^{-2} \ (\mu\text{Sv Bq}^{-1}) \times \text{Body Content (Bq)}, \quad (4)$$

$$\text{Effective dose rate } (\mu\text{Sv y}^{-1})_{\text{Infant}} = 19.5 \times 1.2 \times 10^{-2} \ (\mu\text{Sv Bq}^{-1}) \times \text{Body Content (Bq)}. \quad (5)$$

This annual dose is likely to be conservative because it is assumed that a constant body content will be maintained over 1 y. In addition, based on ICRP's (1990, 1993) retention and excretion model, the younger the individual, the faster the elimination rate.

Dose estimates and discussion

Table 9 shows the mean and standard deviation of cesium effective dose rate (μSv y^{-1}) values for people living on Enewetak, Mejatto (Rongelap population), and Utirik Islands. The effective dose rates were calculated from measurements of the average cesium body content of a whole population. The Enewetak and Rongelap annual effective dose rates are based on the data in Tables 2 and 3, respectively. The Utirik annual effective dose rates are based on information in Table 5 separated by gender (not on Table 4 due to the spread of the population over three major locations).

In general, the environmental decay and dilution of radiocesium will decrease the cesium uptake. The data in Table 9 support this phenomenon, except for the Enewetak 1991 data. There, the increase in two years apparently was the result of people eating more food harvested from contaminated islands. The large coefficient of variation (CV) for the 1991 dose rate suggests that the data are most likely perturbed for the same reason.

Estimates of annual effective dose rates for the Rongelap population at Mejatto remained relatively constant from 1989 to 1994 (unlike the Enewetak and Utirik populations). The lower average dose rates for the people of Rongelap reflect the lower amounts of ^{137}Cs environ-

Table 6. Comparison of ^{137}Cs measurements of the Bikini population by age and gender.[a]

Value[b]	July–August 1989					August 1994				
	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All
	Both sexes					*Both sexes*				
No. counted	8	0	0	0	8	3	1	1	0	5
Mean	880				880	76	33	63		65
SD[c]	1,342				1,342	26				26
Maximum	3,726				3,726	93				93
Median	219				219	90				63
	Males					*Males*				
No. counted	8	0	0	0	8	2	0	0	0	2
Mean	880				880	70				70
SD	1,342				1,342	33				33
Maximum	3,726				3,726	93				93
Median	219				219	70				70
	Females					*Females*				
No. counted	0	0	0	0	0	1	1	1	0	3
Mean						90	33	63		62
SD										28
Maximum										90
Median										63

[a] Limited to DOE workers and immediate family members.
[b] All values are in the unit Bq, except counts.
[c] Standard deviation.

Table 7. Comparison of ^{137}Cs measurement obtained on Majuro Island in two different seasons.

Value[a]	1991					1993				
	Adults	Teens	Juveniles	Infants	All	Adults	Teens	Juveniles	Infants	All
	February					*February*				
No. counted	108	36	25	0	169	112	27	22	0	161
Mean	328	207	97		268	122	41	41		97
SD[b]	583	498	183		529	189	10	11		162
Maximum	4,051	2,126	705		4,051	1,150	87	76		1,150
Median	87	46	45		64	51	39	37		47
	July					*July*				
No. counted	77	29	13	0	119	98	37	48	0	183
Mean	73	52	53		66	106	71	38		82
SD	40	14	23		35	190	96	12		148
Maximum	241	90	107		241	1,519	460	109		1,519
Median	61	47	44		57	50	41	35		45

[a] All values are in the unit Bq, except counts.
[b] Standard deviation.

mental activity on Mejatto Island (approximate background for the RMI environment), similar to the levels on Majuro and Ebeye Islands. Other than global fallout, both islands are recognized as areas which were not contaminated by the Bikini or Enewetak nuclear tests. Therefore, the cesium baseline for the Rongelapese who may return to their homeland could be within the estimated 3 ± 2 μSv y^{-1} value.

Since 1985, the people of Rongelap have resided on Mejatto Island. The estimate of annual effective dose rate in Table 9, therefore, is not representative of the exposure that would result from living on Rongelap Island. By comparing WBC measurements taken at Rongelap to those at Utirik over the 4 y before to the Rongelapese relocation in 1985, Lessard et al. (1984) reported a ratio of 3 (upper bound). Therefore, the 26 ± 15 μSv y^{-1} determined in 1993 for Utirik adult males gives an estimated effective dose rate of 78 ± 45 μSv y^{-1} if they had lived on Rongelap Island during 1993. This predicted average annual effective dose rate agrees with the prediction of 63 μSv y^{-1} by Lessard (Lane 1989).

Table 8. Comparison of ICRP-56 and ICRP-67 age-specific dose conversion factors μSv Bq^{-1}.

Age at intake[a]	ICRP 56 (1990)	ICRP 67 (1993)
0–12 mo	2.0×10^{-2}	2.1×10^{-2}
1–2 y	1.1×10^{-2}	1.2×10^{-2}
3–7 y	9.0×10^{-3}	9.7×10^{-3}
8–12 y	9.8×10^{-3}	1.0×10^{-2}
13–17 y	1.4×10^{-2}	1.3×10^{-2}
Adult (>17 y)[b]	1.3×10^{-2}	1.4×10^{-2}

[a] The age ranges are consistent with ICRP Publication 56 (1990).
[b] The dose conversion factors are derived on the basis of an integrated dose over the 50 y following a single acute intake. For all other age groups, the integration dose period is to age 70 y.

Moreover, doses from cesium estimated for populations living on Bikini, Enewetak, Rongelap, and Utirik

Table 9. Annual effective dose rate (μSv y^{-1}) estimates by location, year, age, and sex using WBC measurements, mean ± 1 standard deviation.

		Adults (A)		Teens (T)		Juveniles (J)	
Location	Year	M	F	M	F	M	F
Enewetak	89	30 ± 29	13 ± 10	25 ± 36	13 ± 12	18 ± 17	16 ± 10
	91	45 ± 60	28 ± 37	40 ± 43	13 ± 13	22 ± 22	15 ± 21
	93	13 ± 16	7 ± 7	5 ± 3	4 ± 3	7 ± 6	5 ± 4
	94	16 ± 15	9 ± 8	6 ± 4	8 ± 5	7 ± 5	6 ± 5
Mejatto[a]	89	3 ± 2	3 ± 1	3 ± 1	3 ± 1	6 ± 3	6 ± 3
	91	3 ± 2	3 ± 1	3 ± 2	2 ± 0	5 ± 2	4 ± 1
	93	2 ± 1	3 ± 1	3 ± 2	3 ± 1	4 ± 1	4 ± 1
	94	3 ± 2	2 ± 2	3 ± 2	2 ± 1	4 ± 1	4 ± 1
Utirik[b]	89	53 ± 27	39 ± 20	75 ± 34	44 ± 24	109 ± 41	104 ± 23
	91	53 ± 27	37 ± 19	65 ± 47	32 ± 13	63 ± 32	50 ± 17
	93	26 ± 15	18 ± 10	19 ± 10	18 ± 11	14 ± 6	20 ± 13

[a] Rongelap population center.
[b] Utirik population living on Utirik Island.

Islands employed dietary intake and food-chain pathway analyses (Robison 1983; Kohn 1988, 1989; Kercher and Robison 1993; Simon and Graham 1995, 1996). Kercher and Robison (1993) report better precision in predicting cesium burden using environmental measurements for individuals with slower metabolism. Furthermore, an overestimate of the dose resulting from using age-specific or age-dependent DCFs for younger age groups was reported by the World Health Organization (WHO 1987); this suggests that the effective dose rates given in Table 9 may overestimate the exposure of teens and younger age groups.

Table 10 compares the external cesium effective dose rates for adults reported by Simon and Graham (1995) and the estimates of internal cesium dose rates using WBC measurements. The table shows the consistencies between external and internal dose rates at Ebeye, Mejatto, and Utirik, but inconsistencies at Enewetak and Bikini. Cesium body contents and the concentrations in the biota are expected to correspond to one another, depending on lifestyle and intake rates of local foods and water. Therefore, the inconsistencies in Table 10 are ascribed primarily to the latter. For example, the internal cesium dose rate for Enewetakese more likely reflects the consumption of local foods from Japtan, Medren, and Enjebi Islands, which have higher reported levels of external cesium dose rates. On the other hand, the external dose rate from cesium at Bikini is far more than the internal dose rate, which indicates that the main diet of the Bikini workers is not local food, but imported foods. This finding may indicate that ingestion intakes are more important in assessing cesium uptake than is inhalation.

During the missions, parents were encouraged to bring infants for whole-body counting. Unfortunately, the water in the lagoon was rough during the missions, and it was unsafe to transport infants to the boat. Therefore, only a few infants were measured (see Tables 2, 3, and 4). In 1991, one infant from Enewetak had a body content below the MDA of 50 Bq, while the contents in four infants from Utirik were approximately 96, 117, 200, and 329 Bq. Assuming that an equilibrium condition would be established in the body due to steady-state intake in the next 365 d, an annual effective dose rate of 77 μSv for the infant with the largest burden was calculated using the ICRP recommended retention function and biokinetic data for a 3-mo-old (shown in the Appendix).

Table 10. Comparison of external and internal cesium effective dose rates (μSv y^{-1}) at various locations.

Atoll	Island	External dose rate[a]	Internal dose rate[b]
Bikini	Bikini	600	3 ± 1 (1 SD)
Enewetak	Enewetak	2	13 ± 13
Enewetak	Enjebi	25	—
Enewetak	Japtan	10	—
Enewetak	Medren	5	—
Kwajalein	Ebeye	2	2 ± 1
Kwajalein	Mejatto	2	2 ± 1
Majuro	Majuro	<1	5 ± 9
Utirik	Utirik	18	22 ± 14

[a] Measurements reported by Simon and Graham (1995).
[b] Based on the 1994 or the latest WBC average from adults (male and female).

CONCLUSIONS

The impact of the nuclear tests conducted at Bikini and Enewetak Atolls from 1946 to 1958 is still being felt because of the continued concern about radiological contamination that affected the Marshallese. This paper presents the WBC results obtained from 1989 through 1994 during field missions. During these five years, ^{137}Cs was the only fallout-related radionuclide detected by whole-body counting. In general, the effective dose rates from cesium are decreasing and are lower than those reported earlier by Conard et al. (1956, 1975), Miltenberger et al. (1980), and Lessard et al. (1980a, 1980b, 1984).

Tables 2, 3, 4, and 5 indicate that median values are often less than mean values. The former are more robust, but, for conservative dose estimates, mean values were used to construct Table 9. Our estimates of the cesium body content are based on data collected over 5 y, and the assessment of annual effective dose rates for all age-specific groups is based on the internationally recommended dosimetric models and parameter values (ICRP 1990, 1993). For example, our latest measurements (Table 9) indicated that male adults who lived in Enewetak, Mejatto, and Utirik in 1993 had received 16, 3, and 26 μSv, respectively, in a year.

Our WBC data suggest the critical group is adult males because of the consistently higher average cesium body contents in comparison with other sex and age groups. Unfortunately, intake of weapons-generated cesium is inevitable, especially for the inhabitants of the northern four atolls (Bikini, Enewetak, Rongelap, and Utirik). The UNSCEAR 1993 Report states that the total effective dose commitment from external and internal ^{137}Cs produced in atmospheric nuclear weapons tests is about 0.5 mSv for the world population, and even more for the north temperate zone populations. Further, Noshkin et al. (1994) indicate that the total annual effective dose rate from natural background in the Marshall Islands is 2.4 mSv y^{-1}, like other areas of the world. Within the 5-y monitoring period, all internal exposures to ^{137}Cs were less than 0.2 mSv y^{-1}.

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APPENDIX

Method for calculating annual dose using ICRP 56 data

During chronic intakes, the body content of cesium may reach a plateau as it comes into equilibrium with the environmental cesium level. It is assumed that the body content measured by a 15-min WBC is maintained at a constant level over a 1-y period. ICRP 56 (1990) biokinetics data for adults can be integrated for any specific time interval. The following integration function gives the total cesium transformations in the body in T days after a single acute intake.

$$Y(T) = \int_0^T (0.1e^{-0.347t} + 0.9e^{-0.0063t})\,dt. \tag{A1}$$

Then, Y (1 y) = 129 and Y (50 y) = 143. Also, the number of transformations due to a chronic intake of one unit per day over 1 y is 365. These values are needed for computing effective dose rates using the ICRP Publication 56 committed effective dose coefficient.

The adult group adjusting factors are $Y(1\ y)/Y(50\ y) = 0.9$ and $365/Y(1\ y) = 2.8$. The former means that 90% of the committed effective dose will be received within the first year of cesium intake. The latter is the ratio of the total transformations within the first year of intake from a steady-state, uniform, chronic intake and from a single, acute intake of cesium. Thus, the conversion factor from committed effective dose to an annual effective dose rate for adults is 2.55 (i.e., 0.9×2.8). Similarly, the conversion factors for children of 3 mo, 1 y, 5 y, 10 y, and 15 y are 15.8, 19.5, 12.3, 6.88, and 3.25, respectively.

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