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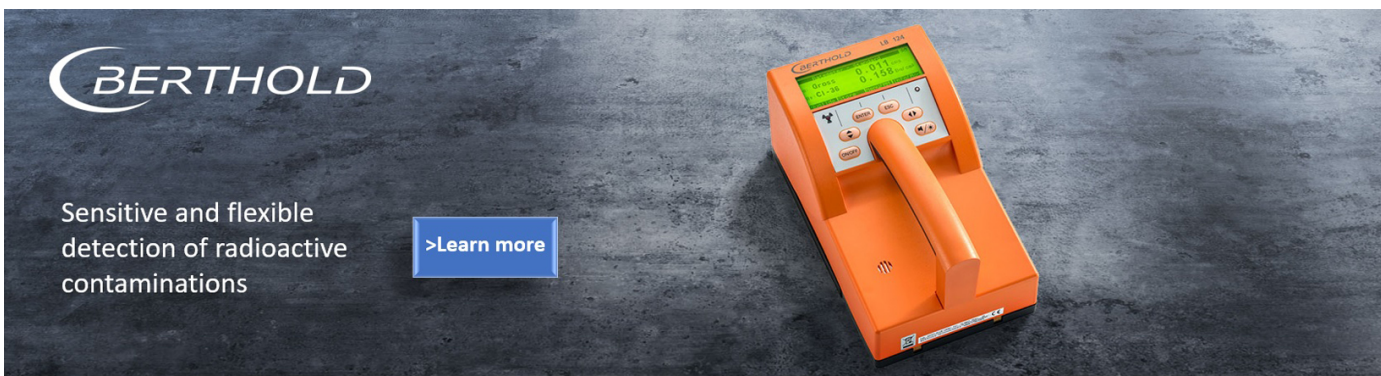
## Japanese population dose from natural radiation

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## Review

# Japanese population dose from natural radiation

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## Abstract

The radiation doses from natural radiation sources in Japan are reviewed using the latest knowledge. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) and the Nuclear Safety Research Association report the annual effective doses from cosmic rays, terrestrial radiation, inhalation, and ingestion as natural sources. In this paper, the total annual effective dose from cosmic-ray exposure is evaluated as 0.29 mSv. The

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arithmetic mean of the annual effective dose from external exposure to terrestrial radiation is 0.33 mSv for the Japanese population using the data of nationwide surveys by the National Institute of Radiological Sciences. Previously in Japan, although three different groups have conducted nationwide indoor radon surveys using passive-type radon monitors, to date only the Japan Chemical Analysis Center (JCAC) has performed a nationwide radon survey using a unified method for radon measurements conducted indoor, outdoor, and in the workplace. Consequently, the JCAC results are used for the annual effective dose from radon and that for radon inhalation is estimated as 0.50 mSv using a current dose conversion factor. In this paper, UNSCEAR values are used for the mean indoor and outdoor thoron-progeny concentrations, and the annual effective dose from thoron is reported as 0.09 mSv. Thus, the annual effective dose from radon and thoron inhalation is 0.59 mSv. From a JCAC large-scale survey of foodstuffs, the committed effective dose from the main radionuclides in dietary intake is 0.99 mSv. Finally, the Japanese population dose from natural radiation is given as 2.2 mSv, which is similar to the reported global average of 2.4 mSv.

Keywords: population dose, effective dose, Japanese adults, natural source

(Some figures may appear in colour only in the online journal)

## 1. Introduction

Nowadays, although radiation benefits people's lives (e.g. medicine, pharmacy, engineering, agriculture), care must be taken to prevent adverse health effects from radiation exposure. Accordingly, the use of radiation is controlled with reference to the 'radiation dose,' which is an index for estimating the level of exposure. In addition to 'artificial' radiation, people are exposed to 'natural' radiation from either the Earth or space. Since its establishment in 1955, the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) has reviewed and evaluated global and regional exposures to radiation (UNSCEAR 2000, 2010), and the review and evaluation results have been summarised in UNSCEAR reports. The last UNSCEAR 2008 report provides data on individual annual average doses and the ranges of radiation from various sources (UNSCEAR 2010). In Japan, many studies have been conducted to derive dose levels from various radiation sources. The Nuclear Safety Research Association (NSRA) in Japan reported on national radiation doses in Japan in 1992 and 2011 (NSRA 1992, 2011), and the National Institute of Radiological Sciences (NIRS) under the auspices of the National Institutes for Quantum and Radiological Science and Technology (QST) has released information entitled 'Dose Scale' that compares the average doses from natural radiation sources in Japan with those from artificial radiation sources (National Institutes for Quantum and Radiological Science and Technology National Institute of Radiological Sciences 2018).

Regarding dosimetry, in its Publication 137, the International Commission on Radiological Protection (ICRP) recently revised the dosimetry data for radon ( $^{222}\text{Rn}$ ) and its decay products (ICRP 2017). In addition, dose coefficients are to be revised for internal dosimetry, and dose conversion coefficients are being summarised to assess the public exposure dose from radionuclides in the environment (ICRP 2018a, 2018b). Accordingly, if newly given or

**Table 1.** Average individual annual effective dose for the public from natural sources [mSv].

Natural source	World (UNSCEAR 2010)	Japan (NSRA 2011)
Terrestrial radiation	0.48	0.33
Cosmic ray	0.39	0.30
Inhalation of radon and thoron	1.26	0.48
Ingestion	0.29	0.99
Total	2.4	2.1

to-be-published data is used to assess radiation doses, then those given in the UNSCEAR 2008 report and the NSRA 2011 report might have to be revised.

In Japan, the general public has been more interested in individual radiation doses from exposure since the Fukushima Dai-ichi Nuclear Power Plant (FDNPP) accident in 2011. Information on radiation doses from natural and artificial sources in Japan before the FDNPP accident helps the general public to understand the influence of the FDNPP accident. Accordingly, the Dose Scale was included in the announcement of monitoring data following the FDNPP accident. However, neither the NSRA 2011 report nor the Dose Scale for exposure from natural radiation sources in Japan give ranges for individual doses. In the present paper, we use the latest knowledge to review the radiation doses from natural radiation sources in Japan.

## 2. Methods for assessing radiation sources and doses

As natural sources of exposure, the UNSCEAR 2008 report and the NSRA 2011 report consider (i) external terrestrial (i.e. terrestrial gamma rays), (ii) cosmic, (iii) inhalation, and (iv) ingestion. For each of the four sources, table 1 summarises the annual individual doses averaged both globally and for Japan at the times of the two reports (UNSCEAR 2010, NSRA 2011). For the average annual effective dose from internal exposure, the contributions differ between Japan and the world, although the total annual dose from natural radiation sources is almost the same.

In table 1, the quantities of radiation dose are the effective dose and the committed effective dose for external and internal exposure, respectively. ICRP mentions the use of effective dose in their publications (ICRP 2007). Although effective dose is used mainly for radiological protection, it is also useful for comparing exposure levels among different radiation sources. Thus, discussions are made with respect to the effective dose for the four natural sources. Specifically, the internal exposure dose is assessed using the committed effective dose, but hereinafter ‘effective dose’ is also used for internal dosimetry. UNSCEAR explained the methodologies for dose assessment in its 2000 report (UNSCEAR 2000) and used them to update exposure levels in its 2008 report (UNSCEAR 2010). For each of the four natural sources, the effective dose is generally assessed using the methods described in the following sections.

## 3. External exposure to cosmic rays

### 3.1. Variable factors in cosmic-ray dose rate

The secondary cosmic rays that cause external exposure comprise mainly neutrons, protons, helium nuclei, muons, electrons (positrons), and photons, and those cosmic rays other than

neutrons are sometimes referred to as an 'ionising component,' conventionally including photons. Exposure to muons dominates at the ground surface, whereas that to neutrons dominates at aviation altitudes (UNSCEAR 2010). There are four variable factors constraining the cosmic-ray dose rate (CRDR).

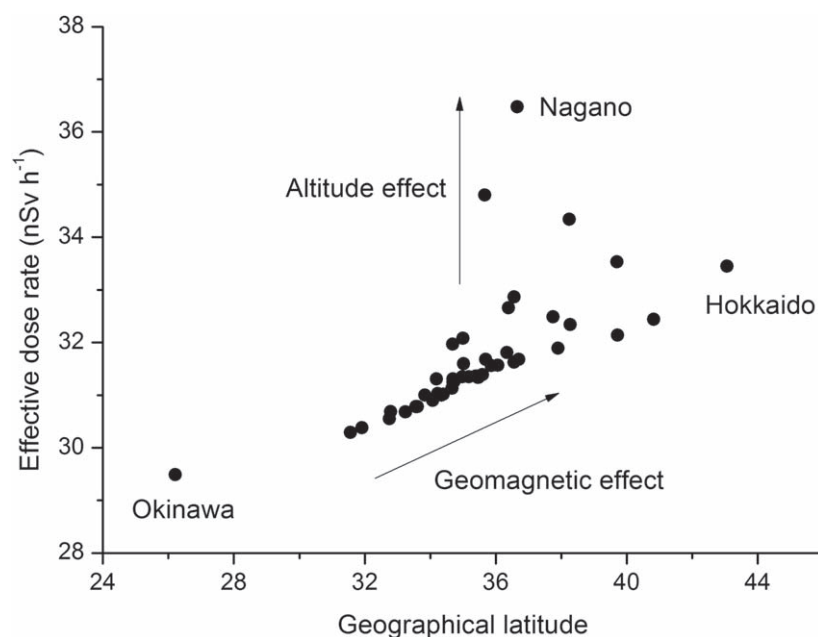
The first factor is the altitude dependence of the CRDR. The thick atmosphere acts as an absorber of penetrating galactic cosmic rays, which subsequently form particles and photons. These components lose their energies by interacting with air molecules. Because atmospheric depth (columnar mass density of the air;  $\text{g cm}^{-2}$ ) increases with decreasing altitude, the CRDR decreases monotonically with decreasing altitude (e.g. Furukawa *et al* 1995, Matsumoto *et al* 1995, Kowatari *et al* 2005, Nagaoka *et al* 2008). In measurements made along a climbing trail on Mount Fuji (elevation: 3,776 m) in Japan, the CRDR of the ionising component at the mountain top was reported as being 3.6 times that at sea level (Furukawa *et al* 1995) and that of the neutron as being ten times (Matsumoto *et al* 1995). In the geomagnetic latitude range for Japan, the rate of change of the CRDR with altitude normalised by the CRDR at sea level is almost constant (Furukawa 1998, Nagaoka *et al* 2008).

The second factor is the geomagnetic field of the Earth, which acts as a barrier against the penetration of galactic cosmic rays. The geomagnetic line along which the charged particles of galactic cosmic rays reach the Earth is parallel to the Earth's surface at the geomagnetic equator and perpendicular to it at the geomagnetic poles. Thus, the CRDR exhibits latitudinal differences, and this factor is parameterised as a vertical cut-off rigidity; the lowest rigidity of a vertically incoming charged particle penetrating through the geomagnetic field. According to Nagaoka *et al* (2008), the CRDRs of the neutron and ionising components at sea level increase at the rates of 2% and 0.3% per degree, respectively, in the geomagnetic latitude range of 14–36 °N.

The third factor is the strength of solar activity. The solar wind induced by solar activity prevents those galactic cosmic rays that originate in outer systems from penetrating into our solar system. Solar activity varies with a cycle of approximately 11 y, with more galactic cosmic rays reaching the Earth at a solar minimum and less at a solar maximum. That is, the CRDR is correlated negatively with solar activity, and this factor is parameterised as a heliocentric potential.

The other factor is the different properties of the ground surface, which is known as the 'albedo effect.' This factor affects the neutron dose rate at ground level. Neutrons are reflected by dry ground but are captured by wet ground because of the presence of water molecules. Thus, wet ground tends to absorb more neutrons than dry ground does. Nagaoka *et al* (2008) found a decreased neutron dose rate due to snow cover but no dependence on either (i) lateral distance from the seashore or (ii) the presence of rainfall. According to a simulation of global population-weighted effective dose conducted by Sato (2016) and discussed in section 3.2.1, its uncertainty due to variable water content of the ground is expected to be less than a few percent because of the minor role played by neutrons in cosmic-ray exposure at ground level.

Figure 1 shows the effective dose rates from cosmic-ray exposure at ground level in the capital cities of 47 Japanese prefectures. These simulation results were produced using version 4.06 of EXPACS (EXcel-based Programme for calculating Atmospheric Cosmic-ray Spectrum) developed by Sato (2015) with input data on locality, elevation, heliocentric potential, and soil moisture. The inputted values of the latter two parameters were the same as those in the simulation of population-weighted effective dose by Sato (2016) discussed in section 3.2.1. The altitude and geomagnetic effects on the CRDRs are evident in their spatial distribution.



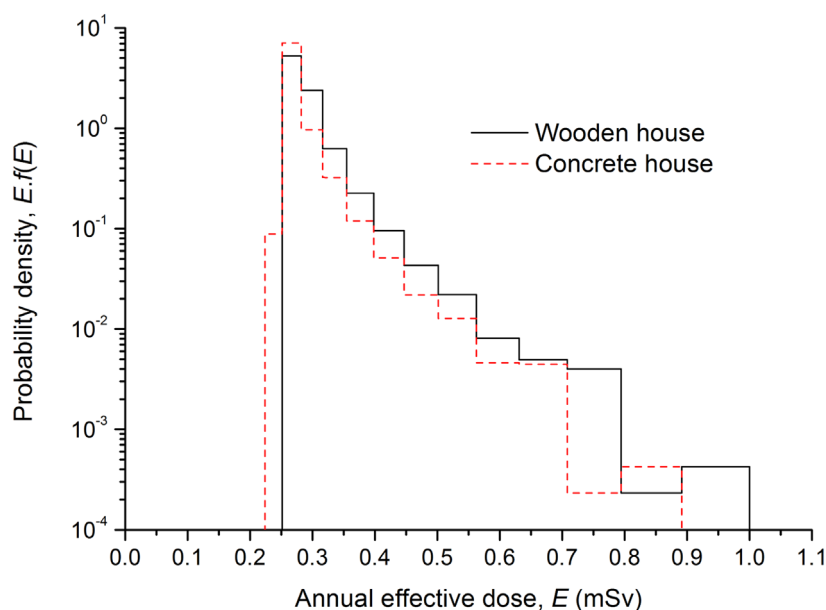
**Figure 1.** Distribution of effective dose rates from cosmic-ray exposure at ground level in the capital cities of the 47 prefectures in Japan calculated using a CRDR calculation programme EXPACS.

### 3.2. Dose from exposure to cosmic rays

To date, two types of dose from exposure to cosmic rays have been evaluated as population doses. One is the dose received in the living environment, namely exposure at the ground surface (e.g. Fujimoto and O'Brien 2002, Fujimoto 2004, Sato 2016), and the other is the dose received during flights, namely exposure at around 11 km altitude (Yasuda and Yajima 2018). In the present paper, the dose received during a flight is not regarded as a component of population dose but is regarded as a variable factor because not all Japanese people take flights in a given year.

**3.2.1. Dose from cosmic-ray exposure in the living environment.** Important studies clarifying the nationwide distribution of annual effective doses from cosmic-ray exposure at ground level in Japan are those by Fujimoto and O'Brien (2002), Fujimoto (2004), and Sato (2016). Fujimoto and O'Brien (2002) and Fujimoto (2004) simulated the CRDR distribution in local communities smaller than prefectures (cities, towns, and villages) and evaluated the annual effective doses based on ICRP Publication 60 (ICRP 1991a), whereas Sato (2016) did so with a geographical resolution of 2.5 arcmin based on ICRP Publications 116 and 123 (ICRP 2010, 2013). However, direct comparison among these studies cannot be made because the annual effective doses were evaluated using different radiation weighting factors. Thus, the present paper reviews the study by Sato (2016) in detail.

Sato (2016) simulated the population-weighted effective doses due to cosmic-ray exposure for the Japanese population as well as for global populations. Version 3.0 of PARMA (PHITS-based Analytical Radiation Model in the Atmosphere, where PHITS is the Particle and Heavy Ion Transport code System) was used for the simulation, as developed



**Figure 2.** Probability density of annual effective dose from cosmic-ray exposure in the case of wooden and concrete houses. The integral of  $f(E)$  with respect to  $E$  is normalised to 1.0. The data are from the study by Sato (2016).

originally and improved by Sato and Niita (2006), Sato *et al* (2008), and Sato (2015). This model predicts cosmic-ray flux energy spectra at any time and location in the atmosphere by specifying values for the aforementioned variable factors such as elevation (atmospheric depth), vertical cut-off rigidity, heliocentric potential, and soil moisture. The model was also mounted on the aforementioned EXPACS. The accuracy of the model was verified by comparisons with direct measurements of cosmic-ray fluxes, radiation dose, and count rates of ground-level neutron monitors (Sato 2015). The cosmic-ray exposure at ground level was simulated considering the population density in 2.5 arcmin grid cells, the variable factors of the elevation in 30 arcsec grid cells and the vertical cut-off rigidity  $1^\circ$  grid cells, and the mean value of the heliocentric potential from the year 2000 onward. The dose evaluation using the cosmic-ray spectra adopted conversion coefficients from flux to effective dose for the isotropic irradiation geometry given by ICRP Publications 116 and 123 (ICRP 2010, 2013).

Sato (2016) evaluated the annual effective doses from cosmic-ray exposure with and without the effects of building shielding. The latter evaluation can be regarded as the annual effective dose for people living in wooden houses with effectively no shielding against cosmic rays, whereas the former is equivalent to the annual effective dose for people living in concrete houses. Sato calculated a shielding factor of 0.91 for a typical Japanese concrete house with a surface density of the construction parts (wall, roof, and ceiling) of  $30 \text{ g cm}^{-2}$  and took the indoor occupancy factor as 0.8, as did UNSCEAR (2000).

Figure 2 shows the probability densities of the annual effective dose from cosmic-ray exposure at ground level for wooden and concrete houses as given by Sato (2016). The Japanese population-weighted annual effective dose was evaluated as 0.29 mSv for a wooden house with a minimum, maximum, and standard deviation of 0.26, 0.97, and 0.04 mSv, respectively. The same for a concrete house was 0.27 mSv with the corresponding statistical values of 0.24, 0.86, and 0.03 mSv. The minimum and maximum values were simulated at

**Table 2.** Japanese population-weighted annual effective doses from cosmic-ray exposure estimated for two types of dwelling. The data are from the study by Sato (2016).

Dwelling type	Annual effective dose (mSv)			
	Average	Standard deviation	Minimum	Maximum
Wooden	0.29	0.04	0.26	0.97
Concrete	0.27	0.03	0.24	0.86

Hateruma-jima Island, Okinawa Prefecture (low-latitude location) and Mount Fuji (high-elevation location), respectively, and the results are summarised in table 2. According to UNSCEAR (2010), a representative value for the annual effective dose for the global population is 0.38 mSv, but direct comparison with that for the Japanese population cannot be made because of different radiation weighting factors having been used.

The present paper takes 0.29 mSv as a representative value for the annual effective dose from cosmic-ray exposure in the living environment. This is because dwellings without effective shielding against cosmic rays account for a large fraction of Japanese dwellings. According to the 2018 Housing and Land Survey of Japan (Statistics Bureau of Japan 2019), the fractions of wooden, concrete, and steel-framed houses are 56.9%, 34.0%, and 8.8%, respectively. Assuming that wooden and steel-framed houses have the same shielding power, then 66% of all dwellings in Japan lack effective shielding.

The present paper does not calculate a representative value for the annual effective dose weighted by house fraction. Dwelling fraction varies considerably from prefecture to prefecture; for example, wooden houses account for 89% in Akita Prefecture but only 3% in Okinawa Prefecture. In addition, Sato (2016) did not provide prefecture averages for the CRDR. However, even if dwelling fractions are considered in dose evaluation, the representative value does not change greatly because the difference in population-weighted annual effective dose between wooden and concrete houses is small (0.02 mSv) based on the evaluation by Sato (2016).

At present, a person can easily calculate her/his effective dose from cosmic-ray exposure at ground level by running the publicly available EXPACS programme and inputting geographical information. Compared to natural radiation sources such as terrestrial gamma radiation and radon, with cosmic-ray exposure at ground level it is more important to evaluate the effective doses for individuals. The effect of building structure on CRDR is fundamental for making such dose evaluation. Sato (2016) evaluated the shielding factor as being 0.91 for a typical Japanese concrete house, but it must be evaluated precisely for individual dwellings. In urban areas such as Tokyo, the surrounding buildings act as absorbers of cosmic rays (e.g. Nagaoka *et al* 2009). For a tall building, the floor dependence of the CRDR is not negligible (e.g. Nagaoka 1987). These factors are yet to be considered in dose evaluation. In addition, a CRDR database is yet to be established. Currently, only the nationwide distribution of the neutron dose rate is available (Nagaoka *et al* 2008). It is particularly important to construct a database on the dose rates of the ionising component because this would contribute to validating model calculations as well as to population-dose evaluation.

**3.2.2. Dose from cosmic-ray exposure during flights.** The first study to evaluate aviation doses for international and domestic flyers as a Japanese population dose was that by



**Table 3.** Number of flyers and annual collective effective doses for international and domestic flights in Japan. The data are from the study by Yasuda and Yajima (2018).

Year	International flight		Domestic flight	
	Number of flyers ( $\times 1,000$ )	Collective effective dose (man Sv)	Number of flyers ( $\times 1,000$ )	Collective effective dose (man Sv)
2009	15,446	843	83,948	137.6
2010	16,637	906	84,367	138.6
2011	16,994	871	77,589	127.5
2012	18,491	927	84,939	140.5
2013	17,473	975	90,942	151.4
2014	16,903	959	94,505	157.7
Average	16,991	913	86,048	142.2

Yasuda and Yajima (2018). The present paper treats this component as a variable factor because not all Japanese people take flights in a given year.

For dose assessment, Yasuda and Yajima (2018) selected the international routes between Tokyo and nine cities (Seoul, Beijing, Hong Kong, Singapore, Delhi, Sydney, Honolulu, Chicago, and Frankfurt) representative of the nine regions visited by 95% of the international flyers, whereas for domestic flyers, they selected the domestic routes between Tokyo and three cities (Sapporo, Fukuoka, and Naha). The targeted period was approximately a half-cycle of the solar activity from 2009 to 2014, covering the solar minimum in 2009 and maximum in 2014. The dose evaluation was done using the publicly available programme JISCARD EX developed by Yasuda *et al* (2010), which calculates an aviation-route dose based on the PARMA cosmic-ray spectrum prediction model, the same methodology used by Sato (2016). The accuracy was confirmed by intercomparison with a European programme EPCARD.NET (Mares and Yasuda 2010) and with direct measurements of neutron dose rates at aviation altitudes (Yasuda *et al* 2009, 2011).

Table 3 summarises the number of Japanese international flyers and collective effective doses for international flights to the nine regions during the targeted period as given by Yasuda and Yajima (2018). The annual collective effective doses ranged from 843 man Sv in 2009 to 975 man Sv in 2013. Most fractions of collective effective dose were attributed to exposure during long, high-latitude flights to Chicago and Frankfurt. In detail, the round-trip route doses for the flights to/from Seoul were evaluated as 5.6  $\mu\text{Sv}$  in 2009 and 5.9  $\mu\text{Sv}$  in 2014, whereas the corresponding values for the flights to and from Frankfurt were 103.4 and 89.8  $\mu\text{Sv}$ , respectively. The average dose per flyer ranged from 50  $\mu\text{Sv}$  in 2012 to 57  $\mu\text{Sv}$  in 2014. Comparing the solar minimum (2009) and maximum (2014), there is no apparent evidence that solar activity influences the collective effective doses or the average dose per flyer. Based on the breakdown of the number of flyers in 2009 and 2014 (table 2 in Yasuda and Yajima (2018)), this is attributed to a shift in the visited regions from northern Asian cities (Seoul and Beijing) on short, low-latitude flights to southern Asian (Singapore), American (Honolulu and Chicago), and European (Frankfurt) cities on long, high-latitude flights. Table 3 also summarises the number of Japanese domestic flyers and the collective effective doses during the targeted period. To evaluate the collective effective doses, the 'unit-distance route dose' (Sv per  $10^3$  man km) was calculated from the aviation doses and the number of flyers for the three domestic routes and was then multiplied by collective flight distance (man km) for all the domestic

**Table 4.** Flyer-average *per capita* effective doses for international and domestic flights. The data are from the study by Yasuda and Yajima (2018).

Year	Flyer-average <i>per capita</i> effective dose ( $\mu\text{Sv}$ )	
	International flight	Domestic flight
2009	54.6	1.6
2010	54.4	1.6
2011	51.2	1.6
2012	50.1	1.7
2013	55.8	1.7
2014	56.7	1.7
Average	53.8	1.7

flights. The collective effective dose increased monotonically from 138 man Sv in 2009 to 158 man Sv in 2014 except for the 128 man Sv in 2011 when the Great East Japan Earthquake occurred. Table 4 summarises the average doses per flyer from cosmic-ray exposure during international and domestic flights.

Although the *per capita* doses were not included in a population dose, they would not be negligible in effective doses from cosmic-ray exposure for frequent flyers. In addition, note that these estimates contain large uncertainty. The CRDR in an aircraft varies depending on aircraft body size, the number of passengers and their seat positions, and the amounts of luggage and fuel (e.g. Ferrari *et al* 2004, Yajima *et al* 2010), but the aircraft shielding effect was not considered in the dose evaluation.

## 4. External exposure to terrestrial radiation

### 4.1. Sources of terrestrial radiation

External exposure to terrestrial radiation is caused by the presence of radionuclides (e.g.  $^{235,238}\text{U}$ ,  $^{232}\text{Th}$ ,  $^{40}\text{K}$ ,  $^{87}\text{Rb}$ ) that originated in the primordial Earth (Plant and Saunders 1996, Mc Laughlin 2015) and still exist because of their long half-lives ( $10^8$ – $10^{10}$  y). Among these elements,  $^{238}\text{U}$ ,  $^{232}\text{Th}$ , and  $^{40}\text{K}$  play major roles in contributing to external radiation exposure.  $^{238}\text{U}$  and  $^{232}\text{Th}$  form decay series that end in stable lead isotopes, and the gamma-emitter radionuclides in those decay series cause external radiation exposure.

Primordial radionuclides are distributed ubiquitously throughout the ground, which comprises igneous, sedimentary, and metamorphic rocks and soils developed from those rocks (Plant and Saunders 1996). Gamma rays that originate in the ground come from the top 30 cm layer (Beck 1972, Kocher and Sjoeren 1985); hardly any of those that originate from deeper than 30 cm reach the surface because of attenuation in the upper layer (i.e. self-absorption).

Terrestrial radiation also includes gamma rays emitted by suspended particles in the air, some of which are radioactive aerosols formed by the progenies of radon isotopes  $^{222}\text{Rn}$  and  $^{220}\text{Rn}$ , which belong to the  $^{238}\text{U}$  and  $^{232}\text{Th}$  decay series, respectively (Porstendörfer 1994).  $^{222}\text{Rn}$  and  $^{220}\text{Rn}$  are gaseous; therefore, those that originate in the ground through the decay of radium migrate through pores and fractures and are finally exhaled from the ground surface. These suspended particles temporally elevate the gamma dose rate in the air because

of their wet deposition onto the ground surface by rain (e.g. Fujitaka *et al* 1992, Inomata *et al* 2007, Melintescu *et al* 2018).

Building materials such as bricks and concrete made from earth materials (e.g. sand, gravel) become sources of terrestrial radiation (UNSCEAR 2010). The gamma emission intensity depends primarily on the activity concentrations of primordial radionuclides. Some researchers have reported the activity concentrations in building materials such as concrete, tiles, gypsum boards, and wallpaper collected in Japan (e.g. Ito and Asano 1998, Suzuki *et al* 1998, Furuta *et al* 2007). In the case of concrete, Suzuki *et al* (1998) measured the natural radionuclides ( $^{238}\text{U}$ ,  $^{228}\text{Ra}$  ( $^{232}\text{Th}$ ), and  $^{40}\text{K}$ ) of its composite materials (e.g. cement, aggregate) and evaluated those of a concrete supposed to have a mixing ratio representative of compositions in Japan. Suzuki *et al* (1998) found that the  $^{238}\text{U}$ ,  $^{228}\text{Ra}$  ( $^{232}\text{Th}$ ), and  $^{40}\text{K}$  concentrations were 7–77 (average: 39), 3–34 (average: 24), and 17–912  $\text{Bq kg}^{-1}$  (average: 517  $\text{Bq kg}^{-1}$ ), respectively, and that they tended to be higher in western parts of Japan (the districts west of Kanto district). The gamma emission intensity of building materials also depends on their density and thickness (Koblinger 1978, Croymans *et al* 2018, Omori *et al* 2019). Similar to the case of gamma rays from the ground, building materials act as both sources and absorbers, namely the gamma emission intensity saturates at a certain thickness (Koblinger 1978, Croymans *et al* 2018).

#### 4.2. Outdoor terrestrial radiation

Yamagata and Iwashima (1967), Abe *et al* (1981), Minato (2006), and Imai and Okai (2014) conducted nationwide surveys to clarify how the terrestrial gamma dose rate is distributed in Japan. Yamagata and Iwashima (1967) evaluated outdoor exposure rates based on natural radionuclide concentrations ( $^{238}\text{U}$ -,  $^{232}\text{Th}$ -series elements, and  $^{40}\text{K}$ ) sampled at 230 locations along main roads, whereas Imai and Okai (2014) did absorbed dose rates in air based on a geochemical map of Japan in 2004 drawn by analysing around 3,000 samples of river sediment. In contrast, Abe *et al* (1981) and Minato (2006) conducted *in situ* measurements using gamma-ray detectors (e.g. ionisation chamber, NaI(Tl) scintillation survey metre) at thousands of locations. Among the aforementioned studies, the latter two are important for assessing the Japanese population dose. Hosoda *et al* (2012) reviewed those studies, and the present paper also reviews studies published subsequently.

The NIRS conducted a nationwide survey of the outdoor ambient gamma dose rate from 1967 to 1977 (Abe *et al* 1981, Abe 1982). The authors divided the islands of Japan into 20 km mesh areas and selected at least one location (the actual number depended on the local population and geological setting) as a measurement point in each area. Consequently, between 8 and 89 locations were surveyed in each of the 47 prefectures. The selected locations were bare ground such as playgrounds or parks (mainly schoolyards) with an area exceeding 1,000  $\text{m}^2$ . Measurements were made five times at 1 m above the ground around a location using an NaI(Tl) scintillation survey metre. The readings were converted into exposure rates equivalent to the readings of a calibrated ionisation chamber. After the first survey (1967–1977) was completed, a second survey focusing on unsurveyed areas was continued in the same manner until 1991 (Furukawa 1993). Consequently, outdoor ambient gamma dose rates were obtained at more than 1,000 locations in the 47 prefectures of Japan. The absorbed dose rates in air after conversion from exposure rates and excluding the contribution of cosmic rays ( $30 \text{ nGy h}^{-1}$ ) are summarised in table 5. The average absorbed dose rate in a prefecture ranged from 19  $\text{nGy h}^{-1}$  (Kanagawa Prefecture) to 80  $\text{nGy h}^{-1}$  (Gifu Prefecture).

**Table 5.** Outdoor ambient gamma dose rates and doses from external exposure to terrestrial radiation in Japan.

District	Prefecture	Population <sup>a</sup> (×1,000)	NIRS		Minato		Annual effective dose-NIRS (mSv)
			Number of data	Dose (nGy h <sup>-1</sup> )	Number of data <sup>b</sup>	Dose (nGy h <sup>-1</sup> )	
Hokkaido	Hokkaido	5,382	89	39.5	212	42.9	0.26
Tohoku	Aomori	1,308	23	26.4	67	27.2	0.17
	Iwate	1,280	18	37.2	32	35.0	0.24
	Miyagi	2,334	17	47.2	74	32.4	0.31
	Akita	1,023	15	47.8	60	34.3	0.31
	Yamagata	1,124	22	45.2	22	41.9	0.30
	Fukushima	1,914	28	52.7	39	34.4	0.35
Kanto	Ibaraki	2,917	22	52.9	63	36.6	0.35
	Tochigi	1,974	14	55.4	22	36.6	0.36
	Gunma	1,973	16	37.1	23	28.0	0.24
	Saitama	7,267	26	34.5	19	39.3	0.23
	Chiba	6,223	44	27.9	25	29.7	0.18
	Tokyo	13,515	69	32.5	25	24.0	0.21
	Kanagawa	9,126	37	18.5	23	22.4	0.12
Chubu	Niigata	2,304	53	59.7	47	52.3	0.39
	Toyama	1,066	14	54.8	22	54.6	0.36
	Ishikawa	1,154	17	57.5	41	40.7	0.38
	Fukui	787	14	69.7	35	49.8	0.46
	Yamanashi	835	10	33.5	59	28.2	0.22
	Nagano	2,099	25	50.1	23	37.5	0.33
	Gifu	2,032	24	79.5	92	56.2	0.52
	Shizuoka	3,700	35	45.4	47	34.2	0.30
	Aichi	7,483	61	64.6	88	43.9	0.42
	Kinki	Mie	1,816	20	57.6	75	54.2
Shiga		1,413	8	73.8	51	56.4	0.48
Kyoto		2,610	20	52.6	22	50.3	0.34
Osaka		8,839	35	60.8	41	50.8	0.40
Hyogo		5,535	43	59.2	49	69.2	0.39
Nara		1,364	11	52.5	43	47.4	0.34
Wakayama		964	11	59.5	21	49.7	0.39
Chugoku	Tottori	573	11	58.3	18	59.7	0.38
	Shimane	694	22	36.0	57	46.5	0.24
	Okayama	1,922	17	59.4	55	67.2	0.39
	Hiroshima	2,844	25	58.9	23	73.8	0.39
	Yamaguchi	1,405	24	51.5	28	60.6	0.34
Shikoku	Tokushima	756	17	53.1	21	51.4	0.35
	Kagawa	976	12	70.3	20	59.9	0.46
	Ehime	1,385	32	52.3	70	68.0	0.34
	Kochi	728	10	62.5	53	62.2	0.41
Kyushu	Fukuoka	5,102	22	61.7	97	40.8	0.40
	Saga	833	9	60.2	49	47.5	0.39
	Nagasaki	1,377	19	45.8	71	41.9	0.30
	Kumamoto	1,786	19	45.4	22	40.6	0.30
	Oita	1,166	17	52.8	57	36.5	0.35
	Miyazaki	1,104	20	54.2	20	44.1	0.36

Table 5. (Continued.)

District	Prefecture	Population <sup>a</sup> ( $\times 1,000$ )	NIRS		Minato		Annual effective dose- NIRS (mSv)
			Number of data	Dose (nGy h <sup>-1</sup> )	Number of data <sup>b</sup>	Dose (nGy h <sup>-1</sup> )	
	Kagoshima	1,648	48	42.8	80	45.9	0.28
	Okinawa	1,434	45	43.0			0.28

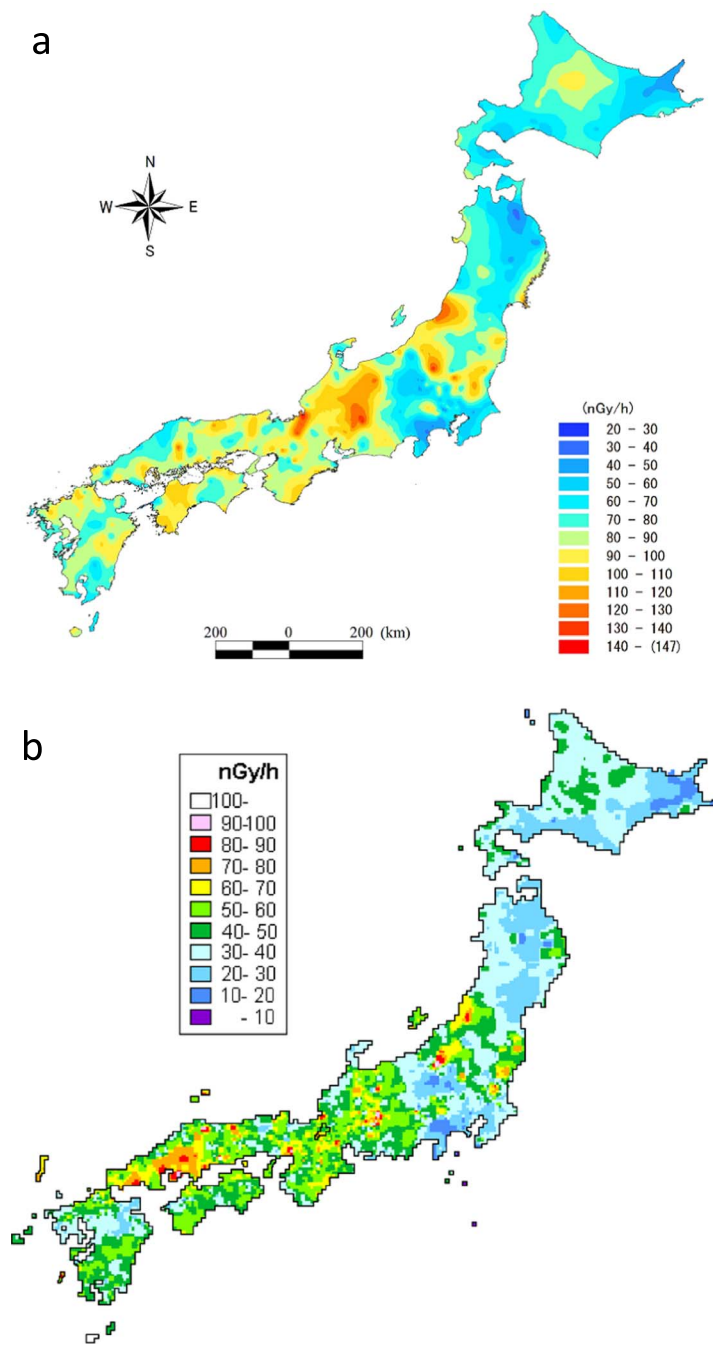
<sup>a</sup> Population as of 2015 was cited from Statistics Bureau of Japan (2016).

<sup>b</sup> The data are from the study by Minato (2015). The number of data is not equal to 4372 because the author sorted some data points for analysis.

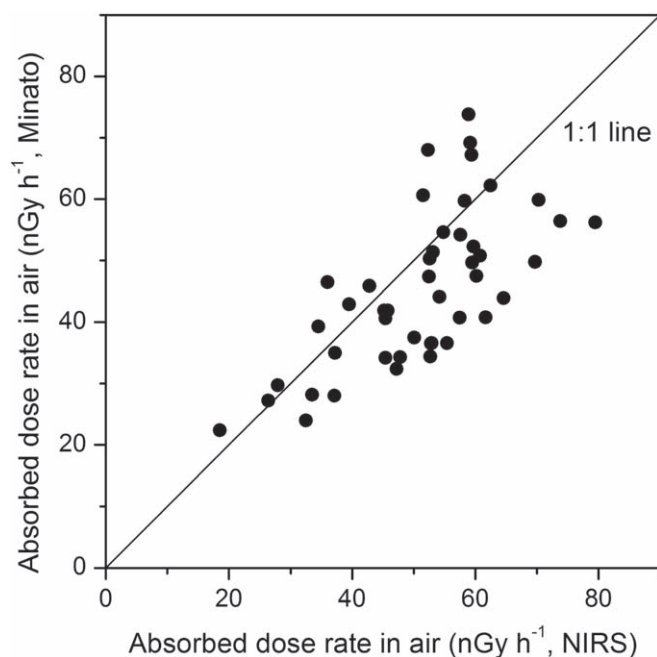
Minato (2006) collected 4372 data points on outdoor ambient gamma dose rates obtained from the *in situ* measurements that Minato, universities, and public institutes each conducted in prefecture-wide surveys until 2005 after the first nationwide survey by NIRS was completed. Unlike the NIRS surveys, the measurement methods in these surveys were not unified; the radiation detectors that were used were NaI(Tl) scintillation survey metres, Ge semiconductor detectors, glass dosimeters, and thermoluminescence dosimeters, and the number of measurement locations in each prefecture (except for the Okinawa Prefecture) depended on the researcher and organisations. Minato (2015) improved his datasets by using the natural radionuclide concentrations of river sediments based on a finding that the ambient gamma dose rates of 23 prefectures all obtained with a unified method by the same institute were strongly correlated (correlation coefficient: 0.96) with those deduced from the radionuclide concentrations given in a geochemical map of Japan, as done by Imai and Okai (2014). The prefecture averages of the absorbed dose rate in air improved by Minato (2015) are summarised in table 5. Similar to the NIRS results, the average absorbed dose rate in air for a prefecture ranged from 22 nGy h<sup>-1</sup> (Kanagawa Prefecture) to 74 nGy h<sup>-1</sup> (Hiroshima Prefecture).

Figure 3 shows distribution maps of terrestrial radiation in Japan depicted using the datasets of NIRS and Minato (2015). The distribution pattern is quite similar between these maps; the outdoor gamma dose rates are higher in western parts of Japan. The distribution of gamma dose rate is governed primarily by Japan's geological settings. Furukawa (1993) compared the NIRS dose rate map with surface geology and soil maps and found that while points of high dose rate tended to be concentrated in granite areas, those of low dose rate were concentrated in areas with exposed volcanic rocks (mainly andesite) or andosol, which is a type of soil developed from volcanic ash. These characteristics were confirmed by the former dataset of Minato (2006), which recompiled the gamma dose rate data with respect to exposed igneous rocks (table 6). Among igneous rocks, granitic ones have the highest content of natural radionuclides (UNSCEAR 1977, Plant and Saunders 1996).

Figure 4 shows scatter plots of ambient gamma dose rates from the datasets of NIRS and Minato (2015). As inferred from the similarity between the dose-rate distribution maps, good correlation was seen, which means that these datasets can be used to evaluate the external exposure to terrestrial radiation in Japan. The difference between these two datasets reaches 23 nGy h<sup>-1</sup> in Gifu Prefecture, and the wide scattering around the 1:1 line may be attributed to the use of different methodologies such as radiation detectors and locality conditions (locations and number of points).



**Figure 3.** Distribution maps of outdoor absorbed dose rates in air in Japan. Map (a) was drawn by Furukawa and Shingaki (2012) using the NIRS datasets, and map (b) was drawn by Minato (2015) using his datasets.



**Figure 4.** Comparison of prefecture averages in outdoor absorbed dose rates in air between the datasets of the studies by NIRS and Minato (2015).

**Table 6.** Outdoor gamma dose rates with respect to types of igneous rock exposed at the surface. This table is after the study by Minato (2006).

Rock type	Number of data	Absorbed dose rate in air (nGy h <sup>-1</sup> )
Granite	143	79.4 ± 24.8
Granite-granodiorite	36	67.5 ± 17.9
Quartz monzonite	52	62.1 ± 16.7
Granodiorite	201	58.3 ± 15.1
Tonalite-granodiorite-quartz diorite	39	49.4 ± 17.8
Quartz diorite	25	33.4 ± 9.5
Gabbro-diorite	12	40.5 ± 10.8
Gabbro	7	22.1 ± 4.7
Rhyolite	13	70.5 ± 19.4
Ryholite-dacite	96	63.9 ± 21.0
Dacite-ryolite	175	42.5 ± 11.9
Andesite	144	36.9 ± 10.1
Hornblende andesite	28	37.4 ± 6.4
Pyroxene andesite	33	26.4 ± 8.3
Andesite-basalt	188	34.8 ± 13.5
Basalt	49	20.4 ± 12.0

The present study further reviewed the results of prefecture-wide surveys not cited in the aforementioned datasets. The collected studies were those by Shimo *et al* (1999), Sugino and Shimo (2002), Sugino *et al* (2007), Minami *et al* (2008), Furukawa *et al* (2015),

**Table 7.** Comparisons of outdoor gamma dose rates in nationwide surveys and another surveys.

Prefecture	Absorbed dose rate in air (nGy h <sup>-1</sup> )				
	Nationwide survey		Prefecture-wide survey		
	NIRS	Minato	Value <sup>a</sup>	Reference	Remarks
Aomori	26.4	27.2	22	Hosoda <i>et al</i> (2016)	Car-borne measurement
Miyagi	47.2	32.4	27	Andoh <i>et al</i> (2017)	Car-borne measurement
Fukushima	52.7	34.4	45	Andoh <i>et al</i> (2017)	Car-borne measurement
Ibaraki	52.9	36.6	33	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
Tochigi	55.4	36.6	49	Andoh <i>et al</i> (2017)	Car-borne measurement
			37	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
Gunma	37.1	28.0	43	Andoh <i>et al</i> (2017)	Car-borne measurement
			26	Sugino and Shimo (2002)	Car-borne measurement
			29	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
Saitama	34.5	39.3	34	Andoh <i>et al</i> (2017)	Car-borne measurement
			41	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
			43	Andoh <i>et al</i> (2017)	Car-borne measurement
Chiba	27.9	29.7	30	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
Tokyo	32.5	24.0	29	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
			49	Inoue <i>et al</i> (2015)	Car-borne measurement
			50	Andoh <i>et al</i> (2017)	Car-borne measurement
Kanagawa	18.5	22.4	22	Sugino <i>et al</i> (2007)	<i>In situ</i> measurement
Gifu	79.5	56.2	53	Shimo <i>et al</i> (1999)	Car-borne measurement
Aichi	64.6	43.9	39	Shimo <i>et al</i> (1999)	Car-borne measurement
Mie	57.6	54.2	45	Shimo <i>et al</i> (1999)	Car-borne measurement
Shiga	73.8	56.4	52	Minami <i>et al</i> (2008)	Car-borne measurement
Kyoto	52.6	50.3	52	Minami <i>et al</i> (2008)	Car-borne measurement
Okinawa	43.0		47	Furukawa <i>et al</i> (2015)	<i>In situ</i> measurement

<sup>a</sup> Andoh *et al* (2017) reported ambient dose equivalent rates (Sv h<sup>-1</sup>) as gamma dose rates. The ambient dose equivalent rates were divided by a conversion coefficient 1.23 (Sv Gy<sup>-1</sup>) given by ICRP (2010) to convert to absorbed dose rates in air.

Inoue *et al* (2015), Hosoda *et al* (2016), and Andoh *et al* (2017). These studies all presented outdoor gamma dose rates obtained by car-borne measurements on roads, except for Sugino *et al* (2007) and Furukawa *et al* (2015), who conducted *in situ* measurements on bare ground. The surveys by Hosoda *et al* (2016) and Andoh *et al* (2017) were conducted after the occurrence of the FDNPP accident in 2011, but the measured (analysed) values in these studies were not affected by the accident.

Comparisons of prefecture averages of outdoor gamma dose rates among the surveys of NIRS, Minato (2015), and the collected studies are shown in table 7. For the prefectures except for the Tokyo metropolitan area (Chiba, Kanagawa, Tokyo, and Saitama Prefectures) and Okinawa Prefecture, the gamma dose rates in the NIRS surveys appeared to have higher values, but it is unclear at present whether this characteristic can be explained by the survey methodologies being different.

An interesting characteristic is seen in the outdoor gamma dose rates in Tokyo (table 7); the ambient gamma dose rates obtained by *in situ* measurements (Abe *et al* 1981, Minato 2015, Sugino *et al* 2007) were lower than those by car-borne measurements



(Inoue *et al* 2015, Andoh *et al* 2017). The former three studies were made above bare ground, whereas the latter two were done on roads. According to Saito *et al* (1997), the outdoor gamma dose rates were  $19 \text{ nGy h}^{-1}$  above bare ground at 22 locations, whereas they were  $44 \text{ nGy h}^{-1}$  in highly residential areas near the bare-ground locations. In highly urbanised areas, where concrete buildings are densely located and the main surface coverage is pavement, the gamma dose rates are possibly elevated.

#### 4.3. Indoor terrestrial radiation

The indoor ambient gamma dose rate is a large part of the dose from external exposure to terrestrial radiation because people spend most of their time indoors. However, to date there has been no nationwide survey to investigate the indoor ambient gamma dose rate in Japan. Therefore, instead of the indoor ambient gamma dose rate, the outdoor gamma dose rate combined with an indoor to outdoor ratio of the ambient gamma dose rate may be useful practically for assessing indoor external radiation exposure. This section focuses on the results of some regional surveys demonstrating the indoor ambient gamma dose rate in relation to the outdoor gamma dose rate.

Matsuda *et al* (1990) surveyed for 94 model houses in Nagoya, central Japan, comprising of seven ferro-concrete houses, 47 fireproof wooden houses and 40 lightweight steel-framed (or light-gauge steel (LGS)) houses. The average indoor gamma dose rates for the corresponding dwelling types were  $77 \pm 11$ ,  $54 \pm 9$  and  $49 \pm 8 \text{ nGy h}^{-1}$ , respectively. Matsuda *et al* also investigated the distribution of gamma dose rate with respect to floor number for the two- and three-storey houses. No clear dependence on floor number was found for the ferro-concrete houses, whereas for the other dwellings the indoor gamma dose rate decreased with increasing floor number by a factor of 0.7–0.8. A reasonable explanation for this is the different locations of terrestrial radiation sources. That is, natural radionuclides in building materials contribute to indoor terrestrial radiation for ferro-concrete houses, whereas those in the ground are the main contributors for wooden and LGS houses. The indoor-to-outdoor dose-rate ratios were  $0.95 \pm 0.15$ ,  $0.77 \pm 0.10$  and  $0.72 \pm 0.13$  for the ferro-concrete, wooden, and LGS houses, respectively. For the wooden and LGS houses, the indoor gamma dose rates were lower than those outdoors because the terrestrial radiation came primarily from natural radionuclides in the ground and furthermore was attenuated partly by the building structure.

Iyogi *et al* (2002) monitored the cumulative indoor gamma dose rates in every quarter using glass dosimeters for 81 wooden houses, including a steel-framed one in Aomori Prefecture, northern Japan. They reported  $41 \pm 8 \text{ nGy h}^{-1}$  as the annual mean of the indoor gamma dose rates. The indoor gamma dose rates were almost constant throughout the seasons, which was different from the outdoor gamma dose rates having a seasonal variation, with the minimum in winter being due to the shielding effect of snow cover. The indoor gamma dose rates were generally higher than the outdoor ones by a factor of  $1.42 \pm 0.26$ . Iyogi *et al* found that stucco-covered walls increased the indoor gamma dose rate. The dose-rate ratio being greater than unity was possibly related to the location of the glass dosimeters close to a wall.

Abe *et al* (1984) reported the indoor-to-outdoor exposure-rate ratio for 135 wooden houses using ionisation chambers and thermoluminescence dosimeters in four towns and cities in Yamanashi, Shimane, Kagoshima and Okinawa Prefectures. The indoor and outdoor exposure rates differed from each other by only  $\pm 15\%$ . The indoor-to-outdoor exposure-rate ratio is 1.02 on average, whereas it is 1.01 and 0.97 for the data obtained using the ionisation chambers and 1.08 and 1.10 using the thermoluminescence dosimeters. As discussed by

Iyogi *et al* (2002), the latter ratios being higher than the others might have been because of the locations of the dosimeters. Note that the exposure rates included the contribution of cosmic rays. However, by comparing indoor and outdoor measurements using ionisation chambers and NaI(Tl) scintillation survey metres, Abe *et al* (1984) noted that wooden houses may not provide significant shielding against cosmic rays. If the exposure rate from cosmic rays remains constant throughout the areas, then an exposure-rate ratio of approximately unity can be regarded as an indoor-to-outdoor dose-rate ratio for terrestrial radiation for wooden houses.

It may be valuable to summarise the work by Saito *et al* (1997) reporting indoor ambient gamma dose rates in the Tokyo metropolitan area. They investigated 246 houses in Tokyo and surrounding prefectures (e.g. Kanagawa, Chiba, Saitama), comprising 103 concrete houses, 119 wooden houses and 24 LGS houses. The indoor gamma dose rates were 22–83 (average:  $54 \pm 14$ ), 15–71 (average:  $39 \pm 11$ ) and 13–54  $\text{nGy h}^{-1}$  (average:  $33 \pm 10 \text{ nGy h}^{-1}$ ) for the concrete, wooden, and LGS houses, respectively. Saito *et al* also investigated 58 concrete office buildings in Tokyo, and the average indoor dose rate was  $47 \pm 9 \text{ nGy h}^{-1}$ . The average dose rate was highest in the concrete houses. Although there was a statistically significant difference in the gamma dose rate between the wooden and LGS houses, the difference became insignificant if those wooden houses that were built mostly from wood and contained no concrete, soil, or gypsum were selected for comparison. Saito *et al* pointed out that (i) the building materials of the LGS houses contained few radiation sources and (ii) the shielding effect against terrestrial radiation was similar between the LGS and wooden houses. Although Saito *et al* did not measure the outdoor ambient gamma dose rate around the houses, they did examine the indoor-to-outdoor dose-rate ratio based on the standard outdoor gamma dose rate ( $19 \text{ nGy h}^{-1}$ ) measured at 22 bare-ground locations in Tokyo. Consequently, the dose-rate ratios were roughly 2.8, 2.0 and 1.7 for the concrete, wooden, and LGS houses, respectively. The Tokyo metropolitan area is highly urbanised and the buildings there are densely located; thus, the higher ratios were attributed to the contribution of gamma radiation from artificial structures rather than terrestrial radiation from the ground. In fact, the average outdoor gamma dose rate was  $44 \text{ nGy h}^{-1}$  at 56 points in urbanised residential areas near the 22 bare-ground locations. When this dose-rate value is used, the dose-rate ratios decrease to 0.7–1.2. Saito *et al* also examined the vertical profile of the gamma dose rates with respect to floor number but found no obvious trends for the concrete and wooden houses.

Table 8 summarises the main results in the four case studies reviewed in the present paper. The following general tendencies are evident for characterising the investigated houses (buildings).

1. The indoor gamma dose rate in the concrete houses was higher than those in the wooden and LGS houses.
2. The indoor gamma dose rate in the wooden houses was slightly higher than that in the LGS houses.
3. The indoor-to-outdoor gamma dose-rate ratio was around unity if the outdoor gamma dose rate was measured at appropriate locations.

#### 4.4. Dose from exposure to terrestrial radiation

The dose from external exposure to terrestrial radiation depends on the fractions of time that people spend indoors and outdoors and on the ambient gamma dose rates there. In this context, the annual effective dose ( $E_t$  [mSv]) can be evaluated using

$$E_t = [D_i \times OF + D_o \times (1-OF)] \times T_a \times DCF \times 10^{-6}, \quad (1)$$

**Table 8.** Indoor gamma dose rates and indoor-to-outdoor gamma dose-rate ratios obtained in regional surveys in Japan.

Reference	Indoor gamma dose rate (nGy h <sup>-1</sup> )			Indoor-to-outdoor dose-rate ratio			Number of data	Place
	Concrete	Wooden	LGS	Concrete	Wooden	LGS		
Abe <i>et al</i> (1984)					0.97 <sup>a</sup> , 1.01 <sup>a</sup> 1.08 <sup>b</sup> , 1.10 <sup>b</sup>		77 58	Shimane, Yamanashi Kagoshima, Okinawa
Matsuda <i>et al</i> (1990)	77 ± 11	54 ± 9	49 ± 8	0.95 ± 0.15	0.77 ± 0.10	0.72 ± 0.13	C: 7 W: 47 L: 40	Aichi
Saito <i>et al</i> (1997)	54 ± 14	39 ± 11	33 ± 10	2.8 <sup>c</sup> (1.2) <sup>d</sup>	2.0 <sup>c</sup> (0.89) <sup>d</sup>	1.7 <sup>c</sup> (0.68) <sup>d</sup>	C: 103 W: 119 L: 24	Tokyo metropolitan area
Iyogi <i>et al</i> (2002)		41 ± 8 <sup>e</sup>			1.42 ± 0.26 <sup>e</sup>		81 <sup>e</sup>	Aomori

<sup>a</sup> Measurements were made using NaI(Tl) scintillation survey metres.

<sup>b</sup> Measurements were made using thermoluminescent dosimeters.

<sup>c</sup> Outdoor gamma dose rate (19 nGy h<sup>-1</sup>) measured above bare ground was used.

<sup>d</sup> Outdoor gamma dose rate (44 nGy h<sup>-1</sup>) measured in urbanised residential areas was used.

<sup>e</sup> One house with LGS structure is included.

where  $D_i$  and  $D_o$  [ $\text{nGy h}^{-1}$ ] are the absorbed dose rate in air indoors and outdoors, respectively,  $OF$  is the occupancy factor,  $T_a$  is the annual time 8760 h ( $24 \text{ h} \times 365 \text{ d}$ ), and  $DCF$  [ $\text{Sv Gy}^{-1}$ ] is a factor for converting the absorbed dose in air into the effective dose. No seasonal variation is considered in the absorbed dose rate in air because the dose rate indoors is constant throughout year (Iyogi *et al* 2002). Compared with the absorbed dose rate in air outdoors, there is insufficient data on the absorbed dose rate in air indoors to evaluate the population dose. Instead, the indoor-to-outdoor dose-rate ratio ( $R$ ) was introduced, whereby equation (1) was transformed into

$$E_t = [1 + (R-1) \times OF] \times D_o \times T_a \times DCF \times 10^{-6}, \quad (2)$$

which was used to evaluate the annual effective doses from external exposure to terrestrial radiation for the Japanese population. As discussed in section 4.3, the indoor-to-outdoor dose-rate ratio was taken as unity, namely the evaluated doses were treated as being independent of the occupancy factor.

The ICRP has summarised the values of the dose conversion factor  $DCF$ . In the natural environment, gamma rays with energies from several tens to less than 3000 keV cause external radiation exposure. According to ICRP Publication 116 (ICRP 2010), the latest  $DCF$  values range from 0.7 to 0.8 [ $\text{Sv Gy}^{-1}$ ] for monoenergetic gamma rays in the aforementioned energy range under isotropic irradiation geometry. The UNSCEAR 2008 report (UNSCEAR 2010) used  $DCF = 0.7$  [ $\text{Sv Gy}^{-1}$ ] for dose evaluation, whereas Moriuchi *et al* (1990) evaluated a factor of 0.748 [ $\text{Sv Gy}^{-1}$ ] for converting the absorbed dose rate in air into the effective dose equivalent for terrestrial gamma radiation. The latter value can be regarded as that of  $DCF$  because the effective dose and the effective dose equivalent differ by only a few percent based on data in ICRP Publications 51 and 116 (ICRP 1987, 2010). Accordingly, the present paper used  $DCF = 0.748$  [ $\text{Sv Gy}^{-1}$ ] when evaluating the annual effective dose.

Table 5 shows the annual effective doses from external exposure to terrestrial radiation for the Japanese population using the NIRS dataset. The annual effective dose ranges from 0.12 mSv (Kanagawa Prefecture) to 0.52 mSv (Gifu Prefecture), reflecting the geological and geographical settings of the living areas as shown in section 4.2, and their arithmetic mean was calculated as 0.33 mSv. The UNSCEAR 2008 report (UNSCEAR 2010) evaluated a representative annual effective dose from terrestrial radiation for the global population as being 0.48 mSv (typical range: 0.3–1.0 mSv), which is greater than the annual effective doses for the Japanese population.

Note that the annual effective doses ranging from 0.12 to 0.52 mSv were evaluated using prefecture averages of the outdoor absorbed dose rates in air with several assumptions. The outdoor gamma dose rate varies among the prefectures; if gamma dose rates at scales smaller than prefecture level were used, then the range of annual effective dose would widen. For example, an outdoor gamma dose rate of 165  $\text{nGy h}^{-1}$  was obtained in Miyako-jima Island, which is 3.5 times the prefecture average (47  $\text{nGy h}^{-1}$ ) of Okinawa Prefecture (Furukawa *et al* 2015). In the case of the NIRS survey, the maximum outdoor gamma dose rate of 139  $\text{nGy h}^{-1}$  was obtained in Tsuruga City, Fukui Prefecture, which is equivalent to the annual effective dose of 0.91 mSv.

Much work is required if future dose evaluations are to be precise. In the present study, the NIRS dataset was used for evaluating the population dose. As noted in section 4.2, the outdoor gamma dose rates given by NIRS appear to be slightly higher than those in recent publications and perhaps also that by Minato (2015). This can probably be attributed to the different methods that different institutes use to measure the gamma dose rate. To minimise the effect, a nationwide survey is required of outdoor gamma dose rates conducted with a unified methodology and by just one institute. A similar issue exists regarding measurements

of indoor gamma dose rates, and the nationwide distribution of indoor gamma dose rates yet to be clarified. In the present study, the indoor-to-outdoor dose-rate ratio was taken as unity based on four regional surveys. However, as Saito *et al* (1997) pointed out, the dose-rate ratio depended on the choice of outdoor measurement locations (i.e. bare ground versus pavement) in the Tokyo metropolitan areas. In addition, the vertical profile of the indoor gamma dose rate varies from dwelling to dwelling. Matsuda *et al* (1990) reported almost-constant vertical profiles for the ferro-concrete houses, whereas Nagaoka *et al* (2009) and Omori *et al* (2019) reported changes of the indoor gamma dose rate depending on floor number by a factor of two for concrete buildings. These findings indicate large uncertainty in the indoor gamma dose rates predicted using measured outdoor gamma dose rates and a unit dose-rate ratio. If the latter is not applicable, then the occupancy factor must be considered for dose evaluation. Because people spend most of their time indoors, a nationwide survey of indoor gamma dose rates is required for precise dose evaluation.

## 5. Internal exposure due to radon, thoron and their progenies

$^{222}\text{Rn}$  (radon) and  $^{220}\text{Rn}$  (thoron) are normally generated by alpha decay from  $^{226}\text{Ra}$  and  $^{224}\text{Ra}$  in soil, rocks, building materials, and water. Radon inhalation is believed to increase the risk of lung cancer and is second only to tobacco smoking as a risk factor. Darby *et al* (2005) reported that the risk of lung cancer increases by 16% per  $100 \text{ Bq m}^{-3}$ , after which the World Health Organization (WHO) proposed a reference level of  $100 \text{ Bq m}^{-3}$  to minimise health hazards due to indoor radon exposure. If this level cannot be reached because of the specific conditions of particular country, then the chosen reference level should not exceed  $300 \text{ Bq m}^{-3}$ , which corresponds to 10 mSv according to ICRP recommendations (WHO 2009).

It is well known that the most significant contributor to the internal dose to the general public is inhalation of radon and thoron progenies (Shimo 1984, 1987, 1990, Tokonami 1999, 2000). However, the methodology for radon and thoron gases is much simpler than that for their progenies. Therefore, many researchers have reported the results of radon and thoron gas measurements for estimating inhalation doses.

### 5.1. Indoor radon survey

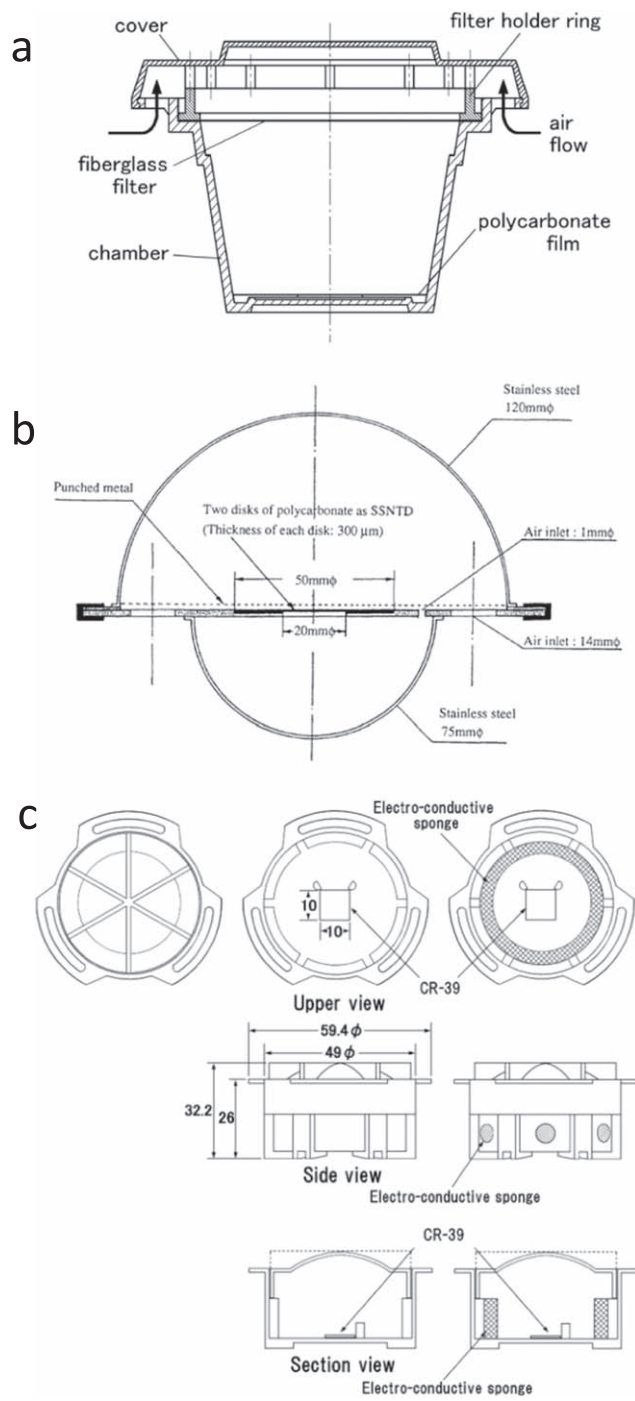
Many countries have reported the results of the nationwide indoor radon survey (McLaughlin and Wasiolek 1988, Langroo *et al* 1991, Marcinowski 1992, Bochicchio *et al* 1996, Friedmann 2005, Ivanova *et al* 2013, Smetsers *et al* 2016, Dowdall *et al* 2017). In Japan, three different groups have conducted nationwide indoor radon surveys using passive radon monitors, the results of which are summarised in table 9.

**5.1.1. The first nationwide survey.** The first survey was conducted at more than 7000 dwellings from 1985 to 1991 by the NIRS using a passive radon monitor developed by the Karlsruhe Nuclear Research Centre (a KfK-type radon monitor) (Fujimoto *et al* 1997). A schematic of a prototype of this KfK-type radon monitor is shown in figure 5. The monitor comprised a polycarbonate alpha-track etch detector inside a diffusion chamber (Urban and Piesch 1981). However, the radon measurements were found to contain significant thoron interference due to the high air exchange rate of the monitor. In this first survey, the annual average indoor radon concentrations in 5717 dwellings were used for dose estimation after scrutinising the raw data. The indoor radon concentrations that were obtained were discussed taking into account the sensitivity of thoron to the KfK-type radon monitor, which was

**Table 9.** Results of nationwide indoor radon surveys conducted by various institutes.

Survey period	Number of dwellings	AM (SD) Bq m <sup>-3</sup>	GM (GSD) Bq m <sup>-3</sup>	Median Bq m <sup>-3</sup>	Maximum Bq m <sup>-3</sup>	Institution
1985–1991	5,717	20.8 (18.8)	16.9 (1.81)	16.0	313	NIRS (Fujimoto <i>et al</i> 1997)
1994–1996	899	15.5 (13.5)	12.7 (1.78)	11.7	208	JCAC (Sanada <i>et al</i> 1999)
2007–2010	3461 Population-weighted average	14.3 (14.7)	10.8 (2.1)			NIPH (Suzuki <i>et al</i> 2010)
		13.7 (12.3)	10.4 (2.0)			

AM: arithmetic mean, SD: standard deviation, GM: geometric mean, GSD: geometric standard deviation.



**Figure 5.** Several passive radon monitors used for nationwide indoor radon surveys. (A) KfK-type radon monitor (Urban and Piesch 1981). (B) radon–thoron discriminative monitor developed by Doi *et al* (1994). (C) radon–thoron discriminative monitor developed by Tokonami *et al* (2005).

evaluated as 35%, and finally the arithmetic mean with standard deviation was evaluated as  $20.8 \pm 18.8 \text{ Bq m}^{-3}$  as shown in table 9 (Fujimoto *et al* 1997).

However, from experiments using a thoron exposure chamber, Tokonami *et al* (2001) estimated the relative thoron sensitivity of the KfK-type radon monitor as 78%. That is, the thoron sensitivity of the monitor evaluated in the first survey was underestimated comparing with the experimental result using the thoron exposure chamber. Therefore, the indoor radon concentration obtained by the first nationwide survey is not suitable for estimating the national effective dose. Tokonami *et al* (2001, 2004) and Tokonami (2010) listed the thoron sensitivities of various passive radon monitors that were used for nationwide radon surveys in various countries.

*5.1.2. The second nationwide survey.* The second survey (Sanada *et al* 1999) was conducted at 940 dwellings from 1994 to 1996 by JCAC using radon–thoron discriminative monitors developed by Doi and Kobayashi (1994). The radon monitors were calibrated by experiments in a radon exposure chamber of the National Radiological Protection Board in the UK. In this second survey, indoor radon concentrations were measured at 20 dwellings in each prefecture for four successive three-month periods to cover an entire year. The radon monitor was placed in either a bedroom or a living room, where residents spend most of their time.

The monitor developed by Doi and Kobayashi (1994) comprised two electroconductive hemispheres as shown in figure 5. Two polycarbonate films were installed in the center of the two hemispheres, and a glass-fibre filter was placed at the open mouth of the first hemisphere to prevent the intrusion of radon progeny. Consequently, only gaseous radon and thoron could penetrate the filter and enter the first hemisphere. The film in the first hemisphere was exposed to alpha particles from radon, thoron, and their progenies. The film in the second hemisphere was exposed to alpha particles from only radon and its progeny because a small hole between the two hemispheres hindered the diffusion of short-lived thoron into the second hemisphere.

The radon concentration in 899 dwellings ranged from 3.1 to 208  $\text{Bq m}^{-3}$ , and the arithmetic mean with standard deviation was evaluated as  $15.5 \pm 13.5 \text{ Bq m}^{-3}$  as shown in table 9. The arithmetic mean of the first nationwide survey, namely  $20.8 \text{ Bq m}^{-3}$ , was approximately  $5 \text{ Bq m}^{-3}$  higher than that of the second one because of the thoron interference in the radon measurements. Evaluated using the data of concrete houses with a log-normal distribution, approximately 1% of the house types exceed WHO's reference level ( $100 \text{ Bq m}^{-3}$ ). According to the Ministry of Land, Infrastructure and Transport (Ministry of Land, Infrastructure and Transport 2018), the total number of dwellings in Japan is approximately 53 700 000 dwellings and the number of concrete houses is 18 200 000. That is, the radon concentrations of more than 182 000 concrete houses will exceed WHO's reference level. Fujimoto and Sanada (1999) reported that the indoor radon concentration in wooden houses was relatively constant with the year of house construction until 1960 and then decreased, whereas the radon concentration in concrete houses increased sharply in houses constructed after 1970. Because after the oil shock energy conservation became a general concern and air tight buildings were introduced for energy saving for heating in winter season as well as for cooling in summer. For example, iron frame windows replaced aluminium frame sashes, making the house more airtight. Until 1980, gypsum board made from phosphate gypsum with a high radium content was widely used in Japan. The indoor radon concentration in concrete houses built before 1975 was almost the same as that in contemporary wooden houses. However, the concentration in contemporary concrete houses was around twice that in wooden houses.



**Table 10.** Indoor radon concentrations in seven regions (Sanada *et al* 1999).

Region	Number of dwellings	Arithmetic mean (Bq m <sup>-3</sup> )	Standard deviation (Bq m <sup>-3</sup> )	Median (Bq m <sup>-3</sup> )	Maximum (Bq m <sup>-3</sup> )
Hokkaido and Tohoku	138	16.0	12.9	12.4	85
Kanto	134	12.4	9.5	9.7	70
Chubu	174	14.1	9.4	11.5	63
Kinki	132	17.1	16.2	12.7	143
Chugoku	95	16.7	9.8	14.4	55
Shikoku	78	14.4	8.7	12.2	61
Kyusyu and Okinawa	148	17.6	20.4	12.7	20
Total	899	15.5	13.5	11.7	208

**Table 11.** Indoor radon concentrations in each dwelling structure (Sanada *et al* 1999).

Structure type	Number of dwellings	Arithmetic mean (Bq m <sup>-3</sup> )	Standard deviation (Bq m <sup>-3</sup> )	Median (Bq m <sup>-3</sup> )	Maximum (Bq m <sup>-3</sup> )
Wooden	597	12.9	8.1	10.9	78
Concrete	182	23.1	15.5	18.7	94
Steel frame	90	12.8	9.5	11.0	77
Concrete block	16	42.5	55.4	22.6	208
Prefabricated	6	10.0	3.8	9.5	17

Sanada *et al* (1999) divided the Japanese islands into seven regions to compare the radon concentrations in each region (table 10). From this, the radon concentrations in the Kinki and Chugoku regions were slightly higher than those in the Kanto region, a possible reason being the difference in bedrock. Granite and rhyolite, which have high <sup>226</sup>Ra concentrations, are distributed mainly in the southwest of Japan, whereas volcanic ash (Kanto loam), which has a low <sup>226</sup>Ra concentration, is distributed in the Kanto region. Hosoda *et al* (2004, 2008, 2010) reported that the radon exhalation rate from soil, which affects both indoor and outdoor radon concentrations, was higher in regions of chemically weathered rock than in those of loam. Therefore, the radon concentration depends on the type of bedrock.

Furthermore, the indoor radon concentration is also affected by the dwelling structure (Sanada *et al* 1999). Table 11 gives the indoor radon concentration for each dwelling structure. As a traditional Japanese dwelling structure, wooden houses accounted for 66% of all dwellings in the second nationwide survey. The arithmetic mean of the radon concentrations in wooden houses was reported as 12.9 Bq m<sup>-3</sup>, and the mean value, which was 56% of that in concrete house and 30% of that in concrete-block houses.

The indoor radon concentration is also affected by the types of building material. Iwaoka *et al* (2013a) investigated the natural radionuclide concentrations and exhalation rates of radon and thoron of approximately 140 samples of rock used as decorative wall coverings, and the radon exhalation rate from the granite samples was found to be higher than that from the other samples. Iwaoka *et al* also found the maximum effective dose for granite to be 0.34 mSv, which included the dose from thoron.

**5.1.3. The third nationwide survey.** The third survey was carried out by the NIPH to confirm the increasing radon concentration. This NIPH survey (Suzuki *et al* 2010) was conducted at 3461 dwellings from 2007 to 2010 using a different type of radon–thoron discriminative monitor (RADUET) developed by Tokonami *et al* (2005), in which radon, which has a half-life of 3.82 d in air, entered the chamber by molecular diffusion through an invisible air gap between its lid and bottom. However, because this air gap functioned as a high diffusion barrier, hardly any thoron could enter the chamber with such a small pathway because of its short half-life of 55.6 s. Instead, to detect thoron effectively, six holes of 6 mm in diameter were made in the chamber side wall and were covered with an electroconductive sponge. In this way, the RADUET monitor could discriminate between radon and thoron.

After adjusting for seasonal fluctuation, the arithmetic mean with standard deviation of the indoor radon concentration was  $14.3 \pm 14.7 \text{ Bq m}^{-3}$ , as shown in table 9. The arithmetic mean obtained in the third survey was similar to that in the second survey. Furthermore, Suzuki *et al* (2010) also reported a population-weighted average value of  $13.7 \pm 12.3 \text{ Bq m}^{-3}$ . This third survey found that the radon concentrations were the highest in houses constructed in the mid 1980s and decreased thereafter.

### 5.2. Indoor workplace and outdoor radon surveys

JCAC also conducted a nationwide indoor workplace radon survey from 2000 to 2003 at 705 sites in offices, factories, schools, and hospitals (Oikawa *et al* 2006). The radon concentrations were measured using the same radon monitors as those used in the indoor and outdoor surveys. The radon monitors were replaced every quarter to observe seasonal variations. The indoor workplace radon concentrations ranged from 1.4 to  $182 \text{ Bq m}^{-3}$ , and the arithmetic mean with standard deviation was  $20.8 \pm 19.5 \text{ Bq m}^{-3}$ . Additionally, the geometric mean was reported as  $15.5 \text{ Bq m}^{-3}$ , assuming a log-normal distribution of radon concentration at indoor workplaces. The arithmetic mean radon concentrations evaluated at offices, factories, schools, and hospitals were 22.6, 10.1, 28.4, and  $19.8 \text{ Bq m}^{-3}$ , respectively.

JCAC also conducted a nationwide outdoor radon survey at 696 points from 1997 to 1999 using the same radon monitors as those used in their indoor survey (Oikawa *et al* 2003). More than 70% of all the radon monitors were installed at bare ground, and the radon monitors were replaced every quarter to observe seasonal variations. Because of the geological characteristics, the outdoor radon concentrations ranged from  $3.3 \text{ Bq m}^{-3}$  in the Okinawa region to  $9.8 \text{ Bq m}^{-3}$  in the Chugoku region. The arithmetic mean with standard deviation and the geometric mean were evaluated as  $6.1 \pm 1.9$  and  $5.9 \text{ Bq m}^{-3}$ , respectively.

### 5.3. Dose estimation from inhaled radon and thoron

**5.3.1. Dose estimation for radon inhalation using typical factors.** As reported by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR (2000)), the annual effective dose ( $E_r$ ) to the general public from radon can be calculated using

$$E_r = E_i + E_w + E_o = \{(Q \cdot F \cdot T \cdot K)_i + (Q \cdot F \cdot T \cdot K)_w + (Q \cdot F \cdot T \cdot K)_o\}, \quad (3)$$

where  $E_i$ ,  $E_w$ , and  $E_o$  are the annual effective doses for indoors, workplaces, and outdoors, respectively;  $Q$  is the arithmetic mean of the radon concentrations at each measurement site;  $F$  is the equilibrium factor for radon, which has been reported by UNSCEAR (2000);  $T$  is the annual occupancy time, which was calculated using the database of the Ministry of Internal Affairs and Communications in 2006 (Ministry of Internal Affairs and Communications 2016); and  $K$  is the dose conversion factor, for which in this estimation the value  $9 \times 10^{-6} \text{ mSv per Bq h m}^{-3}$  was used according to UNSCEAR (2000).

**Table 12.** Radon concentrations, annual occupancy time, and annual effective dose.

Radon concentration ( $\text{Bq m}^{-3}$ )			
Indoor	Workplace	Outdoor	Annual effective dose (mSv)
15.5 ( $F = 0.4$ )	20.8 ( $F = 0.4$ )	6.1 ( $F = 0.6$ )	0.50
Annual occupancy time (hours)			
Indoor	Workplace	Outdoor	Total (hours)
6,059 (69.2%)	1,783 (20.4%)	913 (10.4%)	8,755

$F$ : equilibrium factor.

Note that only JCAC has measured nationwide radon concentrations in various environments using the same type of radon monitor. Therefore, the arithmetic mean radon concentrations in various environments obtained by JCAC were used for the present dose estimation. The arithmetic mean radon concentrations, the equilibrium factor for radon, and the annual occupancy time are summarised in table 12. In the UNSCEAR report (UNSCEAR 2000), the arithmetic mean indoor and outdoor radon concentrations were reported as 40 and 10  $\text{Bq m}^{-3}$ , respectively. That is, the Japanese indoor and outdoor radon concentrations were 39% and 61% lower, respectively, than those of the global average.

The annual effective doses for indoors, workplaces, and outdoors were estimated as 0.34, 0.13, and 0.03 mSv, respectively. Thus, the total annual effective dose to the general public is 0.50 mSv, which is approximately 43% of the global average for radon inhalation (1.1 mSv) reported by UNSCEAR (2000).

**5.3.2. Variation of equilibrium factor.** The equilibrium factor is important for estimating the effective dose, and UNSCEAR (2000) reported typical values. The values of the equilibrium factor in several environments are summarised in table 13.

Some papers have been published on actively obtained values of the equilibrium factor for Japanese dwellings. Regarding the indoor environment, Iimoto (2000) reported how the equilibrium factor varied with time in a typical Japanese apartment made of reinforced concrete. From year-long measurements, the equilibrium factor was evaluated as  $0.43 \pm 0.09$ . Yasuoka *et al* (2009) reported a similar equilibrium factor of 0.42 in a typical Japanese apartment. Those results obtained in typical Japanese dwellings agree well with the factor reported by UNSCEAR (2000).

Meanwhile, Németh *et al* (2005) reported an equilibrium factor of  $0.33 \pm 0.12$  in a Japanese hot spa, which is slightly lower than the UNSCEAR value. Kranrod *et al* (2009) evaluated the equilibrium factor for dwelling in Japan's subtropical region; the value obtained from short-term measurements was  $0.14 \pm 0.01$ , which is 35% of the UNSCEAR value. Furthermore, several papers have noted reduced concentrations of radon progeny when air cleaners and air conditioners are operated (Tokonami *et al* 1996, 2003, Kranrod *et al* 2009, Iwaoka *et al* 2013b), in which case the equilibrium factor will be lower.

Some researchers have reported the equilibrium factor for Japanese workplaces, with values ranging from 0.25 to 0.51 (Yamasaki 1990, Hattori *et al* 1995a, Tokonami *et al* 1996). Thus, it may be possible to use the value reported by UNSCEAR for dose estimation in Japanese workplaces. Furthermore, Tokonami *et al* (1996) reported that the equilibrium factor

**Table 13.** Equilibrium factors in several environments.

	Place	Value		Reference
Indoor	Typical apartment of reinforced concrete	$0.43 \pm 0.09$	12 months	Iimoto (2000)
	Typical apartment	0.42	2 weeks	Yasuoka <i>et al</i> (2009)
	Room of Japanese hotspa	$0.33 \pm 0.12$	17 hours	Németh <i>et al</i> (2005)
	Dwelling at the subtropical region: without air cleaner with air cleaner	$0.14 \pm 0.01$	24 hours	Kranrod <i>et al</i> (2009)
Workplace		$0.04 \pm 0.01$		
	Laboratory	$0.51 \pm 0.05$	4 months	Yamasaki (1990)
	Office building (A)	0.41–0.47	6 months	Hattori <i>et al</i> (1995a)
	Office building (B)	0.24–0.44 (from the figure)	7 months	
	Laboratory: stable condition	0.6	24 hours	Tokonami <i>et al</i> (1996)
	Laboratory: with air circulating system	0.25		
	Computer room (air conditioner: on, ventilation: off)	0.21	31 hours	Tokonami <i>et al</i> (2003)
	Computer room (air conditioner: off, ventilation: off)	0.54	18 hours	
	Computer room (air conditioner: on, ventilation: on)	0.39	5.5 hours	
Computer room (air cleaner only)	0.22	18 hours		
Outdoor	Tokyo (1.5 m height above the ground)	$0.69 \pm 0.25$	16 months	Hattori <i>et al</i> (1995b)
	Chiba	$0.51 \pm 0.12$	12 months	Kojima (1996)
	Waste rock pile sites: in site	0.15–0.44	3 months	Ishimori <i>et al</i> (2000)
	Waste rock pile: around the site	0.36–0.64	3 months	
	Control area (Okayama, Tottori)	0.59–0.78	3 months	

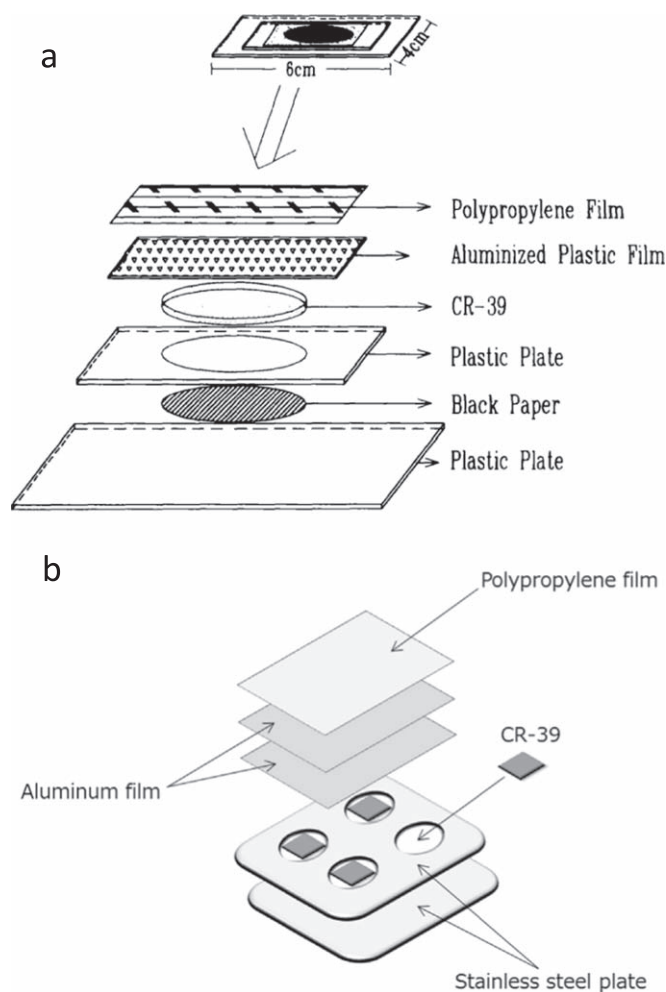
under stable conditions (i.e. without the operation of air circulation systems) increased from 0.25 to 0.6.

The equilibrium factors in outdoor environments were evaluated by year-long measurements of radon and its progeny in Tokyo and Chiba (Hattori *et al* 1995b; Kojima 1996). The arithmetic means with standard deviations observed in Tokyo and Chiba were reported as  $0.69 \pm 0.25$  and  $0.51 \pm 0.12$ , respectively. Thus, the UNSCEAR value of 0.6 is considered to be suitable for dose estimation in outdoor environments. Additionally, Ishimori *et al* (2000) reported outdoor equilibrium factors using three months of monitoring data obtained in and around a waste rock pile site and control area. The equilibrium factor in the site ranged from 0.15 to 0.44, that around the site ranged from 0.36 to 0.64, and that in the control area ranged from 0.58 to 0.78.

**5.3.3. New dose conversion factor recommended by ICRP.** Recently, in its Publication 137, the International Commission on Radiological Protection published new dose conversion factors for radon (ICRP 2017). It also summarised its recommendations on dose estimation from radon exposure (ICRP 2018c). Furthermore, Tokonami (2018) explained the details of the new dose conversion factors for radon. The dose coefficients for miners and sedentary office workers are reported as 3.3 and 4 mSv per mJ h m<sup>-3</sup>, respectively, using the dosimetric models (ICRP 2018c, Tokonami 2018). Using the same methodology, the dose conversion factor for radon exposure in houses is given as 3.7 mSv per mJ h m<sup>-3</sup> (ICRP 2018c, Tokonami 2018). In its Publication 137, in most circumstances the recommended dose conversion factor is 3 mSv per mJ h m<sup>-3</sup> (=10 mSv per WLM) for buildings and underground mines (ICRP 2017). Here an equilibrium equivalent radon concentration of 1 Bq m<sup>-3</sup> corresponds to a potential alpha energy concentration of  $5.6 \times 10^{-6}$  mJ m<sup>-3</sup> (Tokonami 2018). Therefore, the dose conversion factor for buildings and underground mines, including the dwellings and outdoors, can be given as  $17 \times 10^{-6}$  mSv per Bq h m<sup>-3</sup> which is almost twice the current value of  $9 \times 10^{-6}$  mSv per Bq h m<sup>-3</sup>. If we apply these conversion factors to the radon concentrations of the indoor environment, workplaces, and the outdoor environment, then the annual effective doses in each environment increase to 0.64, 0.25, and 0.06 mSv, respectively, and the total annual effective dose from radon inhalation increases from 0.50 to 0.95 mSv. Therefore, it is important to keep a careful watch on the future trends of international organisations such as the International Atomic Energy Agency.

**5.3.4. Dose estimation for thoron inhalation.** High indoor thoron concentrations have been found in traditional Japanese dwellings with clay walls (Shimo 1992, Yonehara *et al* 2005). Therefore, measurements of indoor thoron concentrations are important for dose estimation. Indoor thoron concentrations were also obtained in the second and third nationwide radon surveys because a radon–thoron discriminative monitor was used in those surveys. It is well known that the thoron concentration in dwellings decreases exponentially with distance from the wall surface (Doi *et al* 1994, Tokonami 2010, Hosoda *et al* 2017). This suggests that with a passive monitor, it is difficult to obtain a representative value for the indoor/outdoor thoron concentration. Because the passive monitors used in the nationwide surveys might have been installed at different distances from the wall surface, the annual effective dose for thoron inhalation cannot be estimated using the results from those surveys. Note that the main purpose of a radon–thoron discriminative monitor is to reduce the influence of thoron on the radon measurements.

A recent report on long-term measurements of thoron and its progeny concentrations found that the equilibrium factor has a wide range of 0.0003–0.29 (Hosoda *et al* 2017). If thoron concentration and the equilibrium factor are used for dose estimation, the uncertainty



**Figure 6.** Schematics of passive thoron-progeny monitors: (a) the monitor developed by Zhuo and Iida (2000); (b) the monitor improved by Tokonami (2010).

of effective dose will become large. Thus, thoron concentration should not be used for radiation protection purposes due to the wide range of the equilibrium factors. However, Yamasaki *et al* (1995a) reported that the  $^{212}\text{Pb}$  concentration, which is an important thoron-progeny nuclide for the bronchial dose, is distributed homogeneously in dwellings. Therefore, nationwide thoron-progeny monitoring using passive thoron-progeny monitors is required for national effective dose estimation from thoron inhalation.

Furthermore, Zhuo and Iida (2000) developed a prototype of a thoron-progeny monitor, and the technique was improved by Tokonami (2010). Recently, Japanese research groups estimated the effective doses from thoron inhalation in areas of China and India with high natural background radiation using the improved passive thoron-progeny monitor shown in figure 6 (Kudo *et al* 2015, Omori *et al* 2016, 2017).

In the present paper, the indoor and outdoor arithmetic mean thoron-progeny concentrations are taken from the UNSCEAR 2000 report (UNSCEAR 2000) for dose estimation from thoron inhalation. The indoor and outdoor thoron-progeny concentrations

have been reported as 0.3 and 0.1 Bq m<sup>-3</sup>, respectively. If the dose conversion factor for thoron is taken as  $40 \times 10^{-6}$  mSv per Bq h m<sup>-3</sup> according to the UNSCEAR 2000 report, then the annual effective dose from thoron is estimated as 0.09 mSv. The assumed value of outdoor thoron progeny concentration may be acceptable based on only a few studies reporting outdoor thoron progeny concentrations in the order of 0.1 Bq m<sup>-3</sup> obtained in at least one-year measurements in Aichi Prefecture (Yamasaki and Iida 1995, Yamasaki *et al* 1995b) and Tottori Prefecture (Ishimori *et al* 2010). However, that of indoor thoron progeny concentration may be lower because one order of magnitude higher values ( $\sim 3$  Bq m<sup>-3</sup>) were reported for wooden houses having soil/clay walls in surveys targeting a small number ( $\sim 10$ ) of dwellings (Guo *et al* 1995, Yamasaki and Iida 1995, Zhuo and Iida 2000, Yonehara *et al* 2005). These studies suggest that thoron inhalation is not negligible enough in total effective dose and nationwide surveys to estimate population dose are required.

**5.3.5. Total annual effective dose for radon and thoron inhalation.** The total annual effective dose from radon and thoron inhalation is 0.59 mSv when the current dose conversion factors for radon and thoron are used. According to NSRA (2011), the population dose in Japan from radon, thoron, and their progenies is 0.48 mSv. It is well known that the radon concentrations obtained from nationwide surveys have an approximately log-normal distribution (Sanada *et al* 1999). Therefore, NSRA used the median radon concentration to estimate the annual effective dose. However, in the present paper the arithmetic mean is used for dose estimation according to the UNSCEAR 2000 report (UNSCEAR 2000).

Furthermore, the dose conversion factor, namely the effective dose per unit exposure to radon or thoron, is needed to estimate the effective dose from radon and thoron. The main parameter in the dose conversion factor is the particle size distribution of radon and thoron progenies (Ishikawa *et al* 2001, 2007). Ishikawa *et al* (2001, 2007) calculated the dose conversion factors for radon and thoron progenies with wide ranges of particle diameters, namely 0.5–20 nm for activity median thermodynamic diameter and 20–5000 nm for activity median aerodynamic diameter, using a computer programme (LUDEP) that implements a respiratory-tract model from ICRP Publication 66. The dose conversion factors for radon and thoron progenies were found to change widely depending on the particle diameters; therefore, the aerosol particle size at the measurement location is required for more-refined dose estimation for radon and thoron inhalation.

## 6. Internal exposure due to intake of foodstuff

### 6.1. <sup>232</sup>Th and <sup>238</sup>U

Shiraishi *et al* (1992) reported the daily intakes of <sup>232</sup>Th and <sup>238</sup>U in Japanese men. The diet samples were collected using a duplicate-portion method from 31 prefectures from the northernmost to the southernmost point of Japan. <sup>232</sup>Th and <sup>238</sup>U in the samples were analysed using inductively coupled plasma mass spectrometry (ICP-MS). In that report, the average daily intakes of <sup>232</sup>Th and <sup>238</sup>U for adult men were evaluated as 1.7 and 8.8 mBq, respectively. The effective dose ‘equivalent’ per unit activity in the target organs (kidneys, red bone marrow, and bone surface) of Japanese adults for <sup>232</sup>Th and <sup>238</sup>U intake was calculated using an internal dose estimation system developed by NIRS. The effective dose equivalents per unit activity for <sup>232</sup>Th and <sup>238</sup>U were calculated to be  $7.0 \times 10^{-7}$  and  $8.4 \times 10^{-8}$  Sv Bq<sup>-1</sup>, respectively. The effective dose equivalent was estimated by applying the dose conversion factor for Japanese adults, and the doses were determined as 0.43  $\mu$ Sv for <sup>232</sup>Th and 0.27  $\mu$ Sv for <sup>238</sup>U.

Compared with other methods, duplicate-portion studies give information about more-realistic daily intake, but market basket studies are advantageous for identifying the main foods for intake (Shiraishi and Yamamoto 1995, Shiraishi *et al* 2000). Shiraishi and Yamamoto (1995) conducted a market basket study to estimate the daily intakes of  $^{232}\text{Th}$  and  $^{238}\text{U}$  using ICP-MS, and from the results they also commented on how imported foods affect the internal dose. In this market basket study, 174 types of foodstuff were collected from a supermarket in Mito City, Ibaraki Prefecture, Japan. The daily intakes of  $^{232}\text{Th}$  and  $^{238}\text{U}$  for Japanese people were reported as 2.2 and 15.5 mBq, respectively, which are slightly larger than the previous values obtained by duplicate-portion studies for Japanese men. Shiraishi and Yamamoto mentioned that the internal doses for Japanese people might be increased for the following reasons: (i) the intake of imported foods that have higher concentrations of natural radionuclides compared with Japanese foods and (ii) the intake of artificial radionuclides in foodstuffs due to the 1986 Chernobyl accident. Shiraishi *et al* (2000) also used ICP-MS to evaluate the intake of  $^{232}\text{Th}$  and  $^{238}\text{U}$ . In that report, the food samples were divided into 18 categories according to the results of national nutrition surveys in 1989 to 1991 by the Ministry of Health, Labor and Welfare, namely rice; cereals excluding rice, nuts, and seeds; potatoes; sugars and confectioneries; fats and oils; bean products; fruit; green vegetables; other vegetables; mushrooms; seaweed; seasonings and beverages; fish and shellfish; meat; eggs; milk and milk products; and cooked meals. The total daily intakes for  $^{232}\text{Th}$  and  $^{238}\text{U}$  were evaluated as 2.7 and 13.8 mBq per day per person, respectively, similar to those reported by the same authors in 1995. The annual effective doses for the intake of  $^{232}\text{Th}$  and  $^{238}\text{U}$  were reported using the dose conversion factor from ICRP Publication 61 (ICRP 1991b), and the values were estimated to be 0.22  $\mu\text{Sv}$  for  $^{232}\text{Th}$  and 0.22  $\mu\text{Sv}$  for  $^{238}\text{U}$ . The application of the different dose conversion factor caused the lower estimated values of internal doses than those in Shiraishi *et al* (1992). Additionally, the dominant food groups for total intake were found to be fish and shellfish (44.3%) for  $^{232}\text{Th}$  and seaweed (49.6%) for  $^{238}\text{U}$ , followed by green vegetables (11.1%) for  $^{232}\text{Th}$  and fish and shellfish (25.8%) for  $^{238}\text{U}$ .

#### 6.2. $^{210}\text{Po}$ and $^{40}\text{K}$ (Investigating based on foodstuffs purchased at supermarkets)

Sugiyama *et al* (2009) analysed samples of the everyday Japanese diet cooked with foodstuffs purchased at supermarkets in seven major domestic cities (population: 0.32–3.75 million) from 2007 to 2008 to estimate the committed effective doses for  $^{210}\text{Po}$  and  $^{40}\text{K}$ . The food samples were divided into 13 categories according to the results of national health and nutrition surveys by the Ministry of Health, Labor and Welfare from 2002 to 2004, namely (i) rice; (ii) grains, potatoes, seeds, and nuts; (iii) sugar, preserves, and sweets; (iv) fats and oils; (v) legumes; (vi) fruit; (vii) green and yellow vegetables; (viii) other vegetables, mushrooms, and seaweeds; (ix) beverages; (x) fish and shellfish, (xi) meat, poultry, and eggs; (xii) milk and dairy products; and (xiii) seasonings and spices. The  $^{210}\text{Po}$  and  $^{40}\text{K}$  activities were analysed using alpha spectrometry and gamma-ray spectrometry, respectively, and the daily intakes of  $^{210}\text{Po}$  and  $^{40}\text{K}$  for Japanese adults were found to be in the ranges of 0.34–1.84 and 68.5–94.2 Bq, respectively. The arithmetic means with standard deviations for  $^{210}\text{Po}$  and  $^{40}\text{K}$  were reported as  $0.66 \pm 0.53$  and  $81.5 \pm 8.5$  Bq, respectively, and the committed effective doses were estimated as  $0.29 \pm 0.24$  mSv for  $^{210}\text{Po}$  and  $0.18 \pm 0.02$  mSv for  $^{40}\text{K}$ . The dose from  $^{40}\text{K}$  reported by Sugiyama *et al* (2009) agrees well with the global average of 0.17 mSv reported by UNSCEAR (2000). According to the UNSCEAR (2000), internal dose from  $^{40}\text{K}$  was estimated using the potassium content in the standard human body. The dose from  $^{40}\text{K}$  reported by Sugiyama *et al* (2009) is in good agreement with the world average value of



**Table 14.** Annual intake, annual activity, and committed effective dose for Japanese adults.

Categories (dominant products)	Annual intake (kg) <sup>a</sup>	Annual activity (Bq)	Committed effective dose (mSv)
Grains	168	36	0.017
Algae	5.33	42	0.014
Fishes and shellfishes	32.2	550	0.64
Seasonings and beverages	60.8	110	0.087
Spices	31.9	33	0.016
Total	609	870	0.80

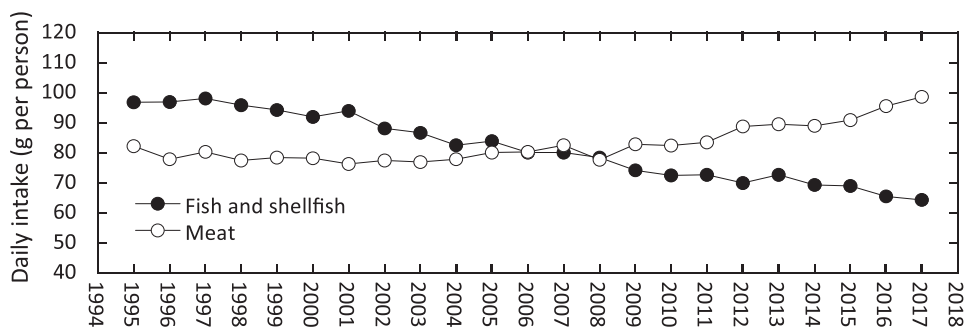
<sup>a</sup> Intake of dietary foods in 2002 was calculated using the database of the Ministry of Health, Labor and Welfare. This table is summarised using the results reported by Ota *et al* (2009).

0.17 mSv reported by UNSCEAR (2000). Thus, internal dose from ingestion of <sup>40</sup>K in foodstuffs is independent of the annual intake due to the homeostasis (Stockigt 1977).

### 6.3. <sup>90</sup>Sr, <sup>137</sup>Cs, <sup>210</sup>Po, <sup>210</sup>Pb, <sup>226</sup>Ra, <sup>232</sup>Th, <sup>238</sup>U, and <sup>239+240</sup>Pu (Investigating based on monitoring of activity concentrations in dietary food)

From 1989 to 2004, JCAC evaluated the activity concentrations of <sup>90</sup>Sr, <sup>137</sup>Cs, <sup>210</sup>Po, <sup>210</sup>Pb, <sup>226</sup>Ra, <sup>232</sup>Th, <sup>238</sup>U, and <sup>239+240</sup>Pu in 137 dietary foods to estimate the committed effective dose for intake (Ota *et al* 2009). Some artificial nuclides (<sup>90</sup>Sr, <sup>137</sup>Cs and <sup>239+240</sup>Pu) originate in past nuclear tests. Although they are categorised as ‘artificial’, people can intake them due to meals in daily life nowadays. The food samples were divided into 17 categories according to a national nutrition survey in 2002, namely cereals, potatoes, sugar, beans, nuts and seeds, vegetables, fruit, mushrooms, algae, fish and shellfish, meat, eggs, milk and milk products, oil and fats, confectioneries, seasonings and beverages, and spices. In that investigation, the activity concentrations of those radionuclides were measured according to radioanalytical methods issued by the Japanese Ministry of Education, Culture, Sports, Science and Technology (Ota *et al* 2009). The annual intakes and activities for the dominant dietary foods are summarised in table 14, as is the committed effective dose for Japanese adults. The dose coefficients (mSv Bq<sup>-1</sup>) for adults given in table 15 were applied according to ICRP Publication 72 (ICRP 1995), and the total committed effective dose from dietary foods was estimated to be 0.80 mSv. Fish and shellfish were found to be the dominant dietary foods, with a reported committed effective dose of 0.64 mSv that accounts for approximately 80% of the total dose. This contribution rate is much higher than the global average of 35% reported by UNSCEAR (2000).

The committed effective doses from dietary intake for each nuclide are given in table 15. The reported dominant nuclide is <sup>210</sup>Po, with an estimated dose of 0.73 mSv that accounts for more than 90% of the total dose. In the UNSCEAR 2000 report (UNSCEAR 2000), the global average committed effective dose was reported as 0.07 mSv, which is much smaller than the Japanese average. However, <sup>210</sup>Po is the largest contributor to the global average (~70%). It is known that Japanese people consume more fish and shellfish than European and American people (Ota *et al* 2009). Furthermore, <sup>210</sup>Po can accumulate in the internal organs of fish and shellfish. As an example, Durand *et al* (1999) investigated <sup>210</sup>Po in the livers of Atlantic mackerel (*Scomber scombrus*), and of the total amount of <sup>210</sup>Po, 80% was found in the soluble fraction of the liver cells.



**Figure 7.** Consumption trend of meat and fish products (fish and shellfish) reported by the Ministry of Health, Labor and Welfare (2018).

**Table 15.** Committed effective doses from dietary intake for each nuclide.

Nuclides	Annual activity (Bq)	Effective dose coefficient (mSv Bq <sup>-1</sup> )	Committed effective dose (mSv)
<sup>90</sup> Sr	59	$2.8 \times 10^{-5}$	0.0017
<sup>137</sup> Cs	60	$1.3 \times 10^{-5}$	0.000 78
<sup>210</sup> Po	610	$6.9 \times 10^{-4}$	0.73
<sup>210</sup> Pb	85	$1.2 \times 10^{-3}$	0.058
<sup>226</sup> Ra	43	$2.8 \times 10^{-4}$	0.012
<sup>232</sup> Th	1.7	$2.3 \times 10^{-4}$	0.000 39
<sup>238</sup> U	15	$4.5 \times 10^{-5}$	0.000 67
<sup>239+240</sup> Pu	0.039	$2.5 \times 10^{-4}$	0.000 0097
Total	870	—	0.80

#### 6.4. Dose estimation from dietary intake

Only Sugiyama *et al* (2009) reported the committed effective dose from <sup>40</sup>K intake. Therefore, the dose estimated as 0.18 mSv can be used to estimate the total dose due to dietary intake. Furthermore, the data on other radionuclides reported by Ota *et al* (2009) are useful for estimating the population dose because the diet samples consumed most by Japanese people were selected for the activity analysis. Therefore, the committed effective dose from the main radionuclides due to dietary intake is given as 0.98 mSv. However, note that the present dose estimation does not account for (i) the doses due to intake of <sup>3</sup>H and <sup>14</sup>C in food and (ii) the ingestion intake of radionuclides in drinking water. Nuclear Safety Research Association (NSRA) (2011) estimated the combined dose due to intake of <sup>3</sup>H and <sup>14</sup>C as 0.01 mSv; thus, the total effective dose from dietary intake is given as 0.99 mSv.

The consumption trend for meat and fish products is shown in figure 7 using data reported by the Ministry of Health, Labor and Welfare (Ministry of Health, Labor and Welfare 2018). Ota *et al* (2009) used the database reported by the MHLW in 2002 for their dose estimation, but according to the MHLW database the total intake of fish and shellfish, which are the dominant products for the dose, decreased by approximately 30% from 2002 to 2017, whereas that of meat products increased by approximately 30% during the same period. The ratio of fish/shellfish intake to meat intake is calculated as being 0.65, which is 57% of the ratio in 2002 (~1.14).

In this paper, the annual effective dose was evaluated using survey data of more than 2,000 foods over 16 years by Ota *et al* (2009). How the annual effective dose varies with the intake of fish and shellfish is given in table 16. The annual effective dose in 2017 due to dietary intake was estimated as being 0.64 mSv, which was obtained using the same methodology as that used by Ota *et al* (2009). Because of the decreased intake of fish and shellfish, the dose estimated in the present paper is 20% lower than that in 2002 (0.80 mSv). However, the contribution rate for the intake of fish and shellfish to the dose is still relatively large at 73%; therefore, the intake of those products contributes greatly to the committed effective dose.

Uddin *et al* (2019) reported that fish samples were compared and significant differences in  $^{210}\text{Po}$  concentration in uncooked samples were observed between species. The effect of the treatment (uncooked, grilled, boiled and stock) was compared for each species and it was found that cooking led to a significant decrease in  $^{210}\text{Po}$  concentration compared to the uncooked samples, with no difference between grilled or boiled treatments. Sugiyama *et al* (2009) states that differences in dietary habits affect exposure level from  $^{210}\text{Po}$  and  $^{40}\text{K}$ .

The dietary habit including cooking methods differs depending on the region in Japan. Therefore, a nationwide survey for dose estimation due to foodstuffs should be done taken into account of the recent Japanese preferences.

## 7. Conclusions

- (1) Japanese population dose from natural radiation: the arithmetic mean annual effective doses from cosmic rays, terrestrial radiation, radon, and foodstuffs were estimated as 0.29, 0.33, 0.59, and 0.99 mSv, respectively. Thus, the Japanese population dose from natural radiation is 2.2 mSv, which is similar to the reported global average of 2.4 mSv.
- (2) External exposure to cosmic rays: the population dose from cosmic-ray exposure is evaluated largely with model calculations. To evaluate it precisely, the effects of building structure and the arrangement of the surrounding buildings must be considered. In addition, the only available database is that on the nationwide distribution of neutron dose rates. The ionising component plays a major role in cosmic-ray exposure at ground level. To validate the model calculations and the population dose evaluation, a database should be constructed on the nationwide distribution of the dose rates of the ionising component.
- (3) External exposure to terrestrial radiation: at present, there is only one dataset on nationwide outdoor ambient gamma dose rates that was obtained using a unified methodology to minimise the uncertainty in the measured values. The outdoor gamma dose rates differed from those in recent publications, presumably because of the different methodologies used in those studies by different institutes to measure the gamma dose rate. In addition, no dataset is yet available on nationwide indoor ambient gamma dose rates. There is a need for nationwide surveys of outdoor and indoor gamma dose rates measured using a unified methodology and conducted by one institute.
- (4) Internal exposure due to radon, thoron, and their progenies: three representative nationwide radon surveys have reported indoor radon concentrations. The results showed good agreement between the arithmetic means of the second and third studies. The ICRP recently published a new conversion factor for dose assessment that is almost twice the previous value. Consequently, it is important to keep a careful watch on the future trends of international organisations and countries around the world. The equilibrium factor for thoron has a wide range. If thoron concentration and the equilibrium factor are used for dose estimation of thoron, the uncertainty of effective dose will become large. Thus,

**Table 16.** Variation of annual effective dose corresponding to the intake of fish and shellfish.

Year	2002	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Intakes of fishes and shellfishes (kg)	32.2	28.7	27.1	26.5	26.5	25.6	26.6	25.3	25.2	23.9	23.5
Annual effective dose (mSv)	0.80	0.74	0.71	0.69	0.70	0.68	0.70	0.67	0.70	0.64	0.64

thoron concentration should not be used for radiation protection purpose. Therefore, a nationwide survey of thoron-progeny as well as radon using passive-type monitors is required for national effective dose estimation from radon and thoron inhalation. Furthermore, the aerosol particle size at the measurement location, which affects the dose conversion factor, is required for a more refined dose estimation for radon and thoron inhalation.

- (5) Internal exposure due to intake of foodstuffs: according to a large-scale survey, the largest contribution to the annual effective dose from Japanese foods is that of  $^{210}\text{Po}$  from seafood. However, the eating habits of Japanese people have changed recently, with the intake of fish products decreasing year by year and that of meat increasing. Therefore, when evaluating the effective dose due to the intake of foodstuffs, it is important to take eating habits into account, which depend on age, region, and temporal trends.

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## Conflicts of interest

The authors declare no conflict of interest.

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