Validation of a Model for Estimating Individual External Dose Based on Ambient Dose Equivalent and Life Patterns

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Background: After the Fukushima Daiichi Nuclear Power Station (FDNPS) accident, a model was developed to estimate the external exposure doses for residents who were expected to return to their homes after evacuation orders were lifted. However, the model's accuracy and uncertainties in parameters used to estimate external doses have not been evaluated.

Materials and Methods: The model estimates effective doses based on the integrated ambient dose equivalent ($H^*(10)$) and life patterns, considering a dose reduction factor to estimate the indoor $H^*(10)$ and a conversion factor from $H^*(10)$ to the effective dose. Because personal dose equivalent ($H_p(10)$) has been reported to agree well with the effective dose after the FDNPS accident, this study validates the model's accuracy by comparing the estimated effective doses with $H_p(10)$. The $H_p(10)$ and life pattern data were collected for 36 adult participants who lived or worked near the FDNPS in 2019.

Results and Discussion: The estimated effective doses correlated significantly with $H_p(10)$; however, the estimated effective doses were lower than $H_p(10)$ for indoor sites. A comparison with the measured indoor $H^*(10)$ showed that the estimated indoor $H^*(10)$ was not underestimated. However, the $H_p(10)$ to $H^*(10)$ ratio indoors, which corresponds to the practical conversion factor from $H^*(10)$ to the effective dose, was significantly larger than the same ratio outdoors, meaning that the conversion factor of 0.6 is not appropriate for indoors due to the changes in irradiation geometry and gamma spectra. This could have led to a lower effective dose than $H_p(10)$.

Conclusion: The estimated effective doses correlated significantly with $H_p(10)$, demonstrating the model's applicability for effective dose estimation. However, the lower value of the effective dose indoors could be because the conversion factor did not reflect the actual environment.

Keywords: Fukushima Daiichi Nuclear Power Station Accident, External Exposure, Effective Dose, Personal Dose Equivalent

Introduction

External exposure from radionuclides deposited on the ground is the primary exposure pathway in areas affected by the Fukushima Daiichi Nuclear Power Station (FDNPS) accident [1]. Immediately after the accident, the Japanese government used a simple model to evaluate the external exposure dose as an integrated ambient dose equivalent ($H^*(10)$) in habitats for decision-making purposes; the model assumes that
Residents spend 8 hours outdoors and 16 hours indoors and that the reduction factor of \(H^*(10)\) from outdoors to indoors is 0.4 [2]. This model conservatively evaluates the external exposure dose relative to the effective dose or the personal dose equivalent \(\left(H_p(10)\right)\) [3] because of an overestimation of the time spent outdoors [4–7] and the lack of consideration of the conversion factor from \(H^*(10)\) to the effective dose or \(H_p(10)\) [8–12]. Although \(H_p(10)\) and the effective dose agree well at energies above approximately 40 keV in a rotational geometry, in which the exposure is comparable to the exposure in an environment where nuclides are uniformly distributed [8], the \(H^*(10)\) exceeds these doses at all energies up to approximately 3 MeV [9], contributing to the simple model’s overestimation.

In 2013, the Nuclear Regulation Authority of Japan stated that a realistic exposure dose for individuals was important for controlling exposure when residents returned to their homes after evacuation orders were lifted [13]. Personal dosimeters are typically used to measure realistic exposure doses; however, their use is challenging. For example, personal dosimeters cannot be used where entrances are restricted, and wearing a personal dosimeter for extended periods can be burdensome. Thus, a model was developed to estimate exposure doses in restricted areas for all residents and assumed life patterns [14]. The model estimates effective doses on the basis of life patterns, which is obtained verbally or via logged global positioning system (GPS) data, and \(H^*(10)\) at each study location, considering dose reduction factors according to the location type (e.g., indoors and in a car) and the age-dependent conversion factor from \(H^*(10)\) to the effective dose. However, the model’s accuracy and uncertainties in parameters for estimating effective doses have not yet been evaluated.

This study was conducted to evaluate the effective dose estimated using the model. Although the effective dose cannot be measured directly, \(H_p(10)\) can be used for validation because \(H_p(10)\) has the same value as the effective dose under rotational geometry [8] or in environments, where \(^{134}\text{Cs}\) or \(^{137}\text{Cs}\) are uniformly distributed on the ground [15]. Therefore, this study compares the effective doses estimated by the model with \(H_p(10)\) measured by personal dosimeters. Furthermore, factors contributing to the validation of results were examined.

Materials and Methods

1. Collection of Personal Dose Equivalent and Life Patterns

Fig. 1 shows a schematic chart of the methods used in this study. This study investigated \(H_p(10)\) and life patterns that were continuously collected for 24 hours from 36 adult participants who lived or worked near the FDNPS. Data were individually collected for 2–15 days between September 2019 and November 2019. In total, 148 days of data were obtained. Primarily, decontamination work employees, office workers, and local government officers around FDNPS were targeted.

![Fig. 1. Schematic chart of the method used in this study. The life patterns were recorded using a smartphone application that records the location and time using global positioning system (GPS). Daily effective doses were estimated based on the life patterns and \(H^*(10)\) map data, considering background \(H^*(10)\), the reduction factor to estimate the indoor \(H^*(10)\), and the conversion factor from \(H^*(10)\) to the effective dose. The hourly \(H_p(10)\) was measured using personal dosimeters and integrated for 24 hours to calculate the daily \(H_p(10)\). The daily effective doses and \(H_p(10)\) were compared. The hourly \(H^*(10)\) was measured using NaI survey meters to know the actual field.](image-url)
The personal dosimeter D-Shuttle (Chiyoda Technol Corporation, Tokyo, Japan) was used in this study. D-Shuttle can measure the hourly cumulative $H_p(10)$; details are given in Naito et al. [16]. The dosimeters were worn during the measurement period, except when doing so was difficult, such as during showers and at bedtime. A previous study reported that the non-use of personal dosimeters at home or bedtime did not generate significant errors in $H_p(10)$ values [17]. Thus, all data during the measurement period were used in our analysis.

The subjects’ life patterns were recorded using a smartphone application [7] developed in a previous study [18]. It can record the location and time information of sites and routes visited by individuals using GPS. Each smartphone was held in the same place as the personal dosimeter. In addition, the participants recorded their life patterns, namely, time, location type (e.g., outdoors, wooden building, concrete building, and transportation), and an outline of their behavior on recording sheets.

2. Data Analysis

Because an effective dose is used as an indicator of radiation protection, this model estimates individual external exposure as an effective dose. An effective dose is the tissue-weighted sum of equivalent doses in all specified organs and tissues of the body [9]; therefore, it cannot be measured directly. Previous studies have demonstrated the quantitative relationship between the effective dose and kerma in free air or $H_p(10)$ (i.e., the conversion factor from kerma in free air or $H_p(10)$ to the effective dose). Golikov et al. [19] evaluated the conversion factor from kerma in free air to the effective dose via measurements using phantoms. In addition, Saito et al. [10] and Saito and Petoussi-Henss [11] simulated the conversion factor from $H_p(10)$ to the effective dose using Monte Carlo simulations. The World Health Organization and the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) used a model for calculating the effective dose from kerma in free using the conversion factor [1, 4]. The method for calculating the effective dose from $H_p(10)$ using the conversion factor is also widely used [3, 14, 19]. Therefore, this study estimated the effective dose from $H_p(10)$ using the conversion factor and considering an individual’s life pattern as

$$E = (\sum_{i:location} (D_i \times T_i)) \times CF,$$

where $D_i$ is the hourly $H_p(10)$ at each location $i$, $T_i$ is the time spent at the location, and $CF$ is an age-dependent conversion factor from $H_p(10)$ to the effective dose. Each location and $T_i$ were obtained from the smartphone application, which reflects an individual’s life pattern. The total $T_i$ was adjusted to be 24 hours; therefore, the effective dose estimated in this study was a daily value. The $CF$ was set to 0.6 for adults, assuming that radiocesium is exponentially distributed in the ground [10, 11].

The outdoor and indoor $H_p(10)$ values are included in $D$. The outdoor $D(D_{out})$ was obtained from $H_p(10)$ map data created via multiscale Bayesian data integration [21] using multiple survey results (walk, carborne, and airborne surveys) performed in 2018 [22]. Because the map is created to be equivalent to the result of the walk survey, which reflects $H_p(10)$ in the living environment, it is suitable for estimating exposure doses. The spatial resolution of the map was 100 m. The map covered the entire region within 80 km of the FDNPS and the entire Fukushima Prefecture. Three participants temporarily stayed outside this area, and $D_i$ in their area was obtained from airborne survey results [23]. The indoor $D(D_{in})$ was calculated using Eq. (2).

$$D_{in} = (D_{out} - BG) \times RF \times CF$$

where $BG$ is the background $H_p(10)$ derived from the natural radionuclides, and $RF$ is the dose reduction factor (the ratio between the indoor and outdoor $H_p(10)$). The $BG$ estimated from car-borne surveys was averaged for each municipality [24]. Because 76% of houses in Fukushima Prefecture are made of wood [25], it was assumed that the participants’ houses were also made of wood. Thus, $RF$ was set to 0.4, a representative reduction factor for one and two story wood-frame houses [26].

Individuals’ hourly $H_p(10)$ values were integrated for 24 hours to calculate their daily $H_p(10)$ values. Daily $H_p(10)$ and estimated effective doses were collected by each participant several times and then averaged. To compare $H_p(10)$ and effective doses indoors and outdoors individually, the hourly $H_p(10)$ and effective doses at sites where participants stayed continuously for 1 hour or more were extracted from the datasets.

3. Measurement of Actual Field

The indoor $H_p(10)$ was estimated by multiplying the outdoor $H_p(10)$ by $RF$. The estimated and measured $H_p(10)$ were compared to evaluate the effect of $RF$ uncertainty on the estimation. In addition, this study compared $H_p(10)$ with the
measured $H^*(10)$ to evaluate the quantitative relationship between them. The $H^*(10)$ was measured using NaI survey meters (TCS-171B; Hitachi Ltd., Tokyo, Japan) at 1 m above the ground or floor at sites, where measurements were possible. The $H^*(10)$ was measured at sites where participants stayed continuously for 1 hour or more. In total, $H^*(10)$ was obtained at 15 outdoor and 99 indoor sites.

**Results and Discussion**

1. **Time Proportion in Life Patterns**

   Fig. 2 shows the average time spent at each location type to understand the participants’ characteristics. The participants spent 87% of their time (approximately 20.8 hours) indoors. In this study, the participants stayed indoors longer than 16 hours assumed by the simple model used by the Japanese government. The target group’s indoor occupancy factor, the proportion of their time spent indoors, was comparable to the value (0.90) for indoor workers reported by UNSCEAR [4].

2. **Relationship between the Estimated Effective Doses and Personal Dose Equivalents**

   Fig. 3 shows the quantitative relationship between daily doses for 36 adult participants. Although the data acquisition period varies among participants, it is normalized to the average daily value. From Fig. 3A, daily $H^*(10)$ and effective doses correlated significantly ($R = 0.94$, $p < 0.01$), showing the model’s applicability for effective dose estimation. However, several small data points in the low $H^*(10)$ range (approximately 2 $\mu$Sv/d) were observed. Fig. 3B shows the ratio of the daily effective doses to $H^*(10)$. The median value of the ratio is 0.68; therefore, the effective doses in this study were lower than the observed $H^*(10)$ (approximately 32% at the median).

   Fig. 4 shows the quantitative relationships between the effective doses and $H^*(10)$ indoors and outdoors. This comparison was performed by extracting the data at sites where participants stayed continuously for 1 hour or more from the entire 148 days of data. The effective doses at the outdoor sites were similar to $H^*(10)$ at the median (Fig. 4A). However, most effective doses at the indoor sites were lower; their value was approximately 39% less than that of $H^*(10)$ at the median (Fig. 4B). Given that the participants spent 87% of their time indoors on average (Fig. 2), the lower indoor effective doses significantly contributed to the lower daily effective doses relative to $H^*(10)$. 

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**Fig. 2.** Average time spent at each location type.

**Fig. 3.** Quantitative relationship between daily doses. (A) Scatter diagram of daily doses. (B) Histogram of the ratio of daily effective doses to $H^*(10)$. The n means number of data and the dashed line in (B) is a reference line showing 1.0.
The following hypotheses were considered factors for the observed lower indoor effective doses than $H_{E}(10)$.

1. Underestimation of the indoor $H^*(10)$: The reduction factor enlarges at low $H^*(10)$ ranges and after decontamination [27]. Therefore, the reduction factor of 0.4 used in this study might be small, resulting in an underestimation of the indoor $H^*(10)$.

2. Underestimation of the conversion factor indoors: The conversion factor varies depending on the irradiation geometry and energy [8]. The conversion factor was set to 0.6, assuming at the 1 m above the ground on which radiocesium is distributed exponentially [10, 11]; however, the indoor irradiation geometry and gamma spectra might differ from those outdoors due to posture of person, height from the ground, and shielding by buildings.

3. Uncertainty in the Indoor Ambient Dose Equivalent

The $H^*(10)$ estimated by the model was compared with that measured by the NaI survey meter to verify whether the reduction factor of 0.4 was valid (Fig. 5). Overall, the estimated indoor $H^*(10)$ tended to be larger than the measured $H^*(10)$ (Fig. 5A). The median ratio of the estimated to measured indoor $H^*(10)$ was 1.14, and more than 75% of the estimated indoor $H^*(10)$ were larger than the measured $H^*(10)$ (Fig. 5B). Therefore, using 0.4 as the reduction factor in this study did not underestimate the indoor $H^*(10)$.

However, the reduction factor of 0.4 can cause underesti-
mation of the indoor $H'_{p}(10)$ in general. The reduction factor increases in low $H'_{p}(10)$ ranges and after decontamination. The reduction factor was increased from 0.43 to 0.63 through decontamination activities after the FDNPS accident [27]. A previous study has reported an average reduction factor of 0.65 for 323 buildings [7]; these figures are all larger than the value of 0.4 used in this study. Most indoor $H'_{p}(10)$ values measured in this study were low (the median was 0.06 µSv, with a range of 0.03–0.27 µSv) (Fig. 5A); therefore, the reduction factor for wooden buildings in this study could also be greater than 0.4. This discrepancy can be attributed to different building types. As shown in Fig. 2, the participants spent approximately 20% of their indoor time in concrete buildings, whose reduction factor is less than 0.4 [26]; hence, $H'_{p}(10)$ in concrete buildings must have been overestimated in this study. This overestimation can offset the underestimation of the indoor $H'_{p}(10)$ estimated using 0.4 as the reduction factor for wooden buildings. Therefore, the indoor $H'_{p}(10)$ estimated in this study did not contribute to the lower estimated effective doses, even though the reduction factor of 0.4 is generally the cause of the underestimation of the indoor $H'_{p}(10)$ in low-radiation fields.

Furthermore, the reduction factor is a highly variable value. Even in the same building, different values have been reported for different rooms and floors. Matsuda et al. [28] indicated that the average reduction factor on the second floor was greater than that on the first floor in wooden and lightweight steel houses because of the uncontaminated effect and roof contamination. Yoshida-Ohuchi et al. [29] reported that the median reduction factor of a room facing the backyard was greater than that of the living room on the same first floor. Although the reduction factor is essential information to evaluate the exposure dose because of the high percentage of time spent indoors, it should be noted that the value can vary significantly depending on various factors, such as the type and size of buildings, floor and location of the room, and the environment outside the buildings.

4. Uncertainty in Conversion Factor Indoors

The ratio of $H_{p}(10)$ to $H'_{p}(10)$ measured by the NaI survey meter corresponded to the practical conversion factor from $H'_{p}(10)$ to the effective dose based on in situ measurements. Fig. 6 shows quantitative relationships between $H_{p}(10)$ and $H'_{p}(10)$. The relationship between $H_{p}(10)$ and $H'_{p}(10)$ differed for indoors and outdoors (Fig. 6A). The median $H_{p}(10)$ to $H'_{p}(10)$ ratio was 0.61 for outdoors (Fig. 6B) and 1.21 for indoors.

![Fig. 6. Quantitative relationships between the measured $H_{p}(10)$ and $H'_{p}(10)$. (A) Scatter diagram and (B, C) histograms showing the $H_{p}(10)$-to-$H'_{p}(10)$ ratio at outdoor and indoor sites. The n means number of data. The solid and dashed lines in (A) are linear regression lines with no intercept for the outdoor and indoor data, respectively. The dashed lines in (B) and (C) are reference lines showing 0.6.](https://doi.org/10.14407/jrpr.2021.00290)
doors (Fig. 6C). A previous study has reported that the $H_p(10)$-to-$H^*(10)$ ratio was 0.6–0.7 [12]. The calculated outdoor ratio was similar to the previous study’s value and the conversion factor (0.6) applied in this study; however, the indoor ratio was significantly larger. This large indoor $H_p(10)$-to-$H^*(10)$ ratio implies that the conversion factor of 0.6 applied in this study is not appropriate for indoors, and it is a factor in the small ratio of the estimated effective dose to $H_p(10)$ indoors.

The conversion factor varies depending on the irradiation geometry and gamma spectra. As hypothesized above, the conversion factor of 0.6 was set in this study, assuming at the 1 m height above ground on which radiocesium distributed exponentially, but the irradiation geometry and gamma spectra indoors might differ from those outdoors due to the posture of a person, height from the ground, and shielding by the buildings. Therefore, the difference in the irradiation geometry and gamma spectra might lead to a small effective dose to $H_p(10)$ ratio.

Furthermore, there is uncertainty in personal dosimetry, including personal dosimeter characteristics and measurement methods, which could also result in the large indoor $H_p(10)$-to-$H^*(10)$ ratio. The response of D-Shuttle is smaller in low-energy ranges, particularly near 200 keV [30]. Most of the indoor $H^*(10)$ values measured in this study were low and at approximately background levels (median, 0.06 µSv; range, 0.03–0.27 µSv) (Fig. 6A). Therefore, the high-energy gamma rays derived from the natural radionuclides contributed more to the entire gamma rays than they do in the contaminated area, gamma rays are mostly the emission from the radiocesium and its scattering in low-energy ranges. Accordingly, the response of D-Shuttle should increase under low $H^*(10)$ range whose gamma spectra have a smaller proportion of low-energy gamma rays. Furthermore, in terms of measurements using personal dosimeters, the participants tended not to wear the personal dosimeters indoors for personal reasons, which could also result in large $H_p(10)$ values. The analysis was conducted assuming that the evaluation of radiation exposure is possible with some accuracy when personal dosimeters are in the same environment as participants, even if the dosimeters are not properly worn [17]. However, the release of the personal dosimeter reduces the shielding effect of the body, possibly increasing the $H_p(10)$. A more quantitative analysis is a future issue.

As described above, the realistic indoor conversion factor might differ from 0.6 because of changes in the irradiation geometry and gamma spectra. In the decade following the FDNPS accident, environmental radiation has been significantly changed by factors, such as the decay of nuclides, decontamination work, and weathering effects, possibly changing the relationships between the indicators of exposure (e.g., $H^*(10)$, $H_p(10)$, and effective dose). Therefore, further studies on the quantitative validation of these effects on doses will improve the evaluation of effective doses.

**Conclusion**

This study investigated the quantitative relationship between effective doses estimated using a model and $H_p(10)$ measured using personal dosimeters. The estimated effective doses significantly correlated with $H_p(10)$, showing the model’s applicability for the effective dose estimation. However, the effective doses estimated from the model were lower than $H_p(10)$ at indoor sites. The following factors were examined for this result: an underestimation of the indoor $H^*(10)$ used in the model and an underestimation of the conversion factor indoors. The $H^*(10)$ measured *in situ* and estimated using the reduction factor were similar, indicating that the reduction factor did not affect the lower effective dose than $H_p(10)$, even though a reduction factor of 0.4 can lead to an underestimation of the indoor $H^*(10)$. Conversely, the indoor ratio of $H_p(10)$ to $H^*(10)$, which corresponds to the practical conversion factor, was significantly larger than the result reported by a previous study and the conversion factor (0.6) applied in this study. Therefore, the realistic indoor conversion factor might differ because of changes in the irradiation geometry and gamma spectra, resulting in lower effective doses relative to $H_p(10)$. This uncertainty in environmental radiation should be quantitatively validated for an accurate estimation of effective doses.

**Conflict of Interest**

No potential conflict of interest relevant to this article was reported.

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Author Contribution

Conceptualization: Sato R, Yoshimura K, Sanada Y, Sato T. Data curation: Sato R, Sato T. Formal analysis: Sato R, Yoshimura K. Methodology: Sato R, Yoshimura K, Sanada Y, Sato T. Project administration: Yoshimura K, Sanada Y. Visualization: Sato R. Writing - original draft: Sato R. Writing - review & editing: Sato R, Yoshimura K. Investigation: Sato T. Software: Sato T. Supervision: Yoshimura K, Sanada Y, Sato T. Validation: Sato R, Yoshimura K, Sato T.

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