The assessment of the annual effective dose due to ingestion of radionuclides from drinking water consumption: calculation methods

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Abstract
In the present paper the different ways of assessing the annual effective dose due to ingestion of radionuclides by drinking water consumption were examined and exemplified. On a set of 10 samples the gross alpha activity, the gross beta activity, the concentration of $^{210}$Po, $^{210}$Pb, $^{238}$U, $^{232}$Th and, $^{226}$Ra were measured. The highest annual effective dose values assessed by relying on the investigated sample set were found by using the rationale according to which all the gross alpha and beta activity is due to the alpha and beta radionuclide, with the highest effective dose coefficient, namely $^{210}$Po and $^{210}$Pb/$^{228}$Ra, respectively.

Keywords Annual effective dose · Effective dose coefficient · Natural radionuclides

Introduction
The main processes contributing to the internal exposure of the human body to ionizing radiation are represented by air inhalation and by water and food ingestion.

The natural radionuclides present in the atmosphere, contributing to the exposure of the human body to ionizing radiation by inhalation, are in fact radioactive decay products resulting from cosmic radiation, gaseous emissions from the soil, emissions from the soil surface matter or from building materials. Radon and its decay products have the most important contribution to the effective dose due to indoor exposure of the population to ionizing radiation by inhalation [1].

Occasionally, artificial radionuclides may be released into the atmosphere, in which case they contribute to internal exposure by inhalation and not only.

UNSCEAR estimates the contribution of natural sources to the population’s effective dose at 2.4 mSv year$^{-1}$, this dose comprising the amount of 0.29 due to food and water consumption [2]. The presence of natural radionuclides in water and food is due to the radioactive decay products from the three natural radioactive series existing on Earth (Uranium-238, Uranium-235, Thorium-232 series) and Potassium-40. It is estimated that thorium, with the highest natural isotopic abundance corresponding to the $^{232}$Th isotope, is three times more abundant than uranium in the Earth’s crust, which is found in the largest proportion in the form of the $^{238}$U isotope [3]. The specific activity of the isotope $^{238}$U ($T_{1/2} = 4.46 \times 10^9$ years, $\Lambda_{\text{specific}} = 12.432 \times 10^3$ Bq g$^{-1}$) is three orders of magnitude greater than the specific activity of the $^{232}$Th isotope ($T_{1/2} = 1.4 \times 10^{10}$ years, $\Lambda_{\text{specific}} = 4.1 \times 10^3$ Bq g$^{-1}$), therefore, in the Earth’s crust, the uranium activity concentration is higher than the thorium activity concentration.

The Uranium-238 and Thorium-232 series isotope content analysis in the human body requires tissue measurements.
In the case of $^{40}$K, regardless of its intake, its concentration in the body is kept under control by hemostatic balance. This biological mechanism does not apply to the radionuclide penetration from the three natural radioactive series whose parents are $^{238}$U, $^{232}$Th, $^{235}$U. Determining the concentration of radionuclides in water and food is a much more useful alternative in assessing the effective dose due to internal exposure of the body to ionizing radiation by drinking water and food consumption. The assessment of the effective dose due to water and food consumption in the world population varies according to geographic specificity, seasonal cultural and food habits, in the range 0.2–0.8 mSv year$^{-1}$ [1]. There are many studies to assess the annual effective dose due to ingestion of radionuclides by food consumption [4–23], and by water consumption [24–57], which use different calculation methods of this physical magnitude.

**Calculation methods**

The quantification of the ionizing radiation effects on the human body due to the consumption of drinking water is performed by calculating the physical magnitude called the annual effective dose—expressed in Sv year$^{-1}$, defined in Council Directive 2013/51/EURATOM as the committed effective dose for 1 year of ingestion resulting from all the radionuclides whose presence has been detected in a supply of water intended for human consumption, of natural and artificial origin, but excluding tritium, potassium-40, radon and short-lived radon decay products.

The European legislation [58] transposed into Romanian legislation [59] recommends that this value parameter should not exceed 0.1 mSv year$^{-1}$.

The specific concentration of natural radionuclides in drinking water, multiplied by the annual intake rate and the effective dose coefficient, for each radionuclide type and age

\[
D_{er} = \frac{C_{i(der.)}}{CF_{n} \times 730(L)}
\]

\[
C_{i(der.)}/(Bq \cdot L^{-1}) = \frac{10^{-4}}{Sv \cdot Bq^{-1}} \times 730(L)
\]
category, serve to determine the annual effective dose due to ingestion of radionuclides by drinking water consumption.

Using the measured concentrations of radionuclides in water, the effective dose coefficients, the fractional absorption in the gastrointestinal tract [62, 63], and the annual water consumption rate [60], the annual effective dose due to radionuclide ingestion through drinking water consumption ($D_{ef}$) according to the following formula is calculated [24–44]:

$$D_{ef} = \sum \left( C_X \times f \times CF \times R \right) \left( \text{Sv year}^{-1} \right)$$  \hspace{1cm} (3)

where $C_X$—is the measured concentration of radionuclide $X$ (example: $X = 210^{\text{Po}}, 210^{\text{Pb}}, 238^{\text{U}}, 232^{\text{Th}}, 40^{\text{K}}, 226^{\text{Ra}}$) (Bq L$^{-1}$), $R$—is the annual consumption rate of drinking water (L year$^{-1}$), assumed to be 730 L year$^{-1}$ for adults; 350 L year$^{-1}$ for children, 250 L year$^{-1}$ for lactation age, [60]; or $R$—is the annual consumption rate of drinking water and beverages (L year$^{-1}$) assumed to be 150 L year$^{-1}$ for infants, 350 L year$^{-1}$ for children, and 500 L year$^{-1}$ for adults [65]. $F$—is the fractional absorption in the gastrointestinal tract. Often the factor of fractional absorption in the gastrointestinal tract is neglected. $CF$—is the effective dose coefficient (the committed effective dose per unit intake for each radionuclide type and age category (Sv Bq$^{-1}$) [63].

Based on this rationale, the following calculation formula is used for assessing of the annual effective dose due to radionuclide intake from drinking water:

$$D_{ef} = \sum \left( (\Lambda_\alpha \times CF_{210^{\text{Po}}} + \Lambda_\beta \times CF_{228^{\text{Ra}}}) \right) \times R \left( \text{Sv year}^{-1} \right)$$  \hspace{1cm} (5)

where $\Lambda_\alpha$—is the gross alpha activity (Bq L$^{-1}$), $\Lambda_\beta$—is the gross beta activity (Bq L$^{-1}$), $R$—is the annual consumption rate of drinking water (L year$^{-1}$) assumed to be 730 L year$^{-1}$ for adults [60], $CF_{210^{\text{Po}}}, CF_{228^{\text{Ra}}}$—is the effective dose coefficient for $210^{\text{Po}}$ and $228^{\text{Ra}}$, respectively (Sv Bq$^{-1}$).

This calculation method does not take into account the following very important aspect, which was reported in WHO, 2017 [60]: the gross beta activity measured in a sample includes a contribution from the primary radionuclide $40^{\text{K}}$, which occurs naturally with K stable in a stable ratio: the specific activity of $40^{\text{K}}$ is 27.6 Bq g$^{-1}$ of K stable. Unlike other radionuclides, $40^{\text{K}}$ introduced into the body it does not accumulate due to the hemostatic balance [65]. Moreover, K is an essential element for the good functioning of the human body. Therefore, the $40^{\text{K}}$ contribution should be subtracted from the gross beta activity, thus remaining the so-called residual beta activity and only this should be assumed for the $228^{\text{Ra}}$ radionuclide. In the light of these considerations, in the annual effective dose calculation formula (2), $\Lambda_\beta$ should represent the residual beta activity in order to avoid dose overestimation.

The assessment of the annual effective dose using the gross alpha and the gross beta activities

Another way of estimating the annual effective dose due to the intake of radionuclides from drinking water is based on the gross alpha and the gross beta activities [45–50].

This method of calculation is based on the following rationale: the specific activities of natural radionuclides from drinking water are due to the specific activities of decay products from the three natural radioactive series, whose parents (series heads) are $238^{\text{U}}, 232^{\text{Th}}, 235^{\text{U}}$. The committed effective dose per unit intake for adults, for the most important radionuclides, increases in the following order:

$$238^{\text{U}} < 235^{\text{U}} < 234^{\text{U}} < 232^{\text{Th}} < 224^{\text{Ra}} < 226^{\text{Ra}} < 221^{\text{Po}}$$  \hspace{1cm} (4)

Considering this series, in order to calculate the annual effective dose due to drinking water intake, was take into account the worst prediction in which were assume that the gross alpha and beta activities are derived from the $210^{\text{Po}}$ and $226^{\text{Ra}}$ radionuclides, these having the highest effective dose coefficients.

The assessment of the annual effective dose using the gross alpha activity

Following the procedure of Fernandez et al. [66], according to which more than 50% of the annual effective dose is due to the contribution of the radium, some authors [51–54] estimate the annual effective dose due to the ingestion of radionuclides through drinking water consumption by using the following formula [67]:

$$D_{ef} = \Lambda \times CF \times 730 \times 2 \left( \text{Sv year}^{-1} \right)$$  \hspace{1cm} (6)

where $\Lambda$—is the gross alpha activity (Bq L$^{-1}$), $CF$—the effective dose coefficient, corresponding to the isotope $226^{\text{Ra}}$, (Sv Bq$^{-1}$), 730 L—is the annual consumption rate of drinking water (adults) [60].

The value of $3.58 \times 10^{-7}$ Sv Bq$^{-1}$ for members of the public committed effective dose per unit intake via ingestion corresponding to the $226^{\text{Ra}}$ isotope, given by the USA-EPA, 1998, Table 2.2, [61] should be updated to the value of $2.8 \times 10^{-7}$ Sv Bq$^{-1}$ given by the IAEA, 2014, Table III. 2D [63].
The assessment of the annual effective dose by associating the gross alpha activity and the gross beta activity with the alpha emitters (ex 210Po, 226Ra, 232Th, 238U), and with the beta emitters, respectively (228Ra, 210Pb).

Some authors [37, 45, 55–57] do not assess the total effective reference dose, but the assumed contribution of each radionuclide:

\[ D_{\text{ef}} = \Lambda \times R \times CF \left( \text{Sv year}^{-1} \right) \]  

(7)

where \( \Lambda \)—is the gross alpha or gross beta activity (Bq L\(^{-1}\)), CF—is the effective dose coefficient (Sv Bq\(^{-1}\)). \( R \)—is the annual consumption rate of drinking water.

Without direct data on the concentrations of the assumed radionuclides, only the gross alpha and beta activity is used. In this type of rationale, the annual effective dose generated by each individual radionuclide is assessed.

The application of the calculation methods: experimental

In order to apply the calculation methods described above, a set of 10 drinking water samples was taken into consideration, on which the following measurements were performed: gross alpha activity, gross beta activity [57] and the concentration of 210Po, 210Pb, 226Ra, 232Th, 238U. The physicochemical and radiological parameters of drinking water depend on the water source used. In the studied area one of the water sources used is the Danube river, characterized in many scientific works [68–76].

The gross alpha and the gross beta activity was determined in accordance with ISO 9696 and ISO 9697, respectively [77, 78]. The 210Po, 210Pb measurements were performed by the spontaneous deposition onto Ni disc and the gross alpha activity measurement [79, 80]. The U-natural and Th-natural measurements were performed through radiocchemical separation on a Dowex ion exchange resin followed by Arsenazo III complexation and spectrophotometric measurement [81]. Based on the natural isotopic abundance of 238U (99.27%) and 232Th (100%) from U-natural and Th-natural, respectively, the specific activities of 238U and 232Th were calculated. The measurement of the 226Ra concentration was performed by storing the samples for 30 days, which were required in order to reach the 226Ra/222Rn equilibrium, and then measuring it by using the Sarad RTM instrument.

The methods for determining the gross alpha activity, gross beta activity, the concentration of 210Po, 210Pb, 226Ra were described in detail on [34, 57]. The measurements were performing in the Ionizing Radiations Laboratory from Galati—the accredited laboratory (Tables 2, 3).

The methods described above for estimating the annual effective dose due to ingestion of radionuclides by drinking water were used.

Due to the fact that the effective dose coefficients, FC for 210Pb and for 228Ra are equal, the annual effective doses for both radionuclides are equal.

It may be noticed that, for all the investigated samples, relation (1) is complied with. The domain of the annual effective dose value variation in a range (one string) of investigated samples, calculated through the same method, is not relevant. Is relevant the variation of the annual effective dose value in the same sample, calculated through the various previously described methods.

For all the investigated samples it may be noticed that the values of the annual effective doses due to the ingestion of radionuclides by drinking water consumption, assessed by the measurement of radionuclide concentration method, Eq. 3, in this case 210Po, 210Pb, 226Ra, 238U, 232Th, are lower as compared to the effective dose values, calculated by assuming the gross alpha and beta activities.

| Sample code | \( \Lambda_{210Pb}^\alpha \) mBq L\(^{-1}\) | \( \Lambda_{210Pb}^\beta \) | \( \Lambda_{210Po} \) | \( \Lambda_{226Ra} \) | \( \Lambda_{238U} \) | \( \Lambda_{232Th} \) |
|-------------|-----------------|-----------------|-----------------|-----------------|-----------------|-----------------|
| DW1         | 6.0 ± 1.8       | 25.0 ± 6.2      | 2.8 ± 0.7       | 3.1 ± 0.8       | 36.0 ± 4.8      | 41.0 ± 4.5      |
| DW2         | 18.5 ± 5.5      | 25.0 ± 6.2      | 1.8 ± 0.5       | 4.2 ± 1.1       | 3.0 ± 0.3       | 10.0 ± 1.1      |
| DW3         | 6.0 ± 1.8       | 25.0 ± 6.2      | 2.9 ± 0.7       | 3.6 ± 0.9       | 31.0 ± 3.4      | 38.0 ± 8.4      |
| DW4         | 6.0 ± 1.8       | 62.4 ± 16.6     | 1.6 ± 0.4       | 5.2 ± 1.3       | 3.0 ± 0.3       | 11.4 ± 1.3      |
| DW5         | 43.3 ± 13.0     | 96.5 ± 24.1     | 11.4 ± 2.9      | 14.7 ± 3.7      | 18.5 ± 2.0      | 24.5 ± 2.7      |
| DW6         | 22.8 ± 6.8      | 25.0 ± 6.2      | 12.5 ± 3.1      | 14.2 ± 3.6      | 21.4 ± 2.4      | 31.4 ± 3.5      |
| DW7         | 22.0 ± 6.6      | 88.9 ± 22.2     | 2.3 ± 0.6       | 6.4 ± 1.6       | 14.5 ± 1.6      | 21.7 ± 2.4      |
| DW8         | 7.7 ± 2.3       | 25.0 ± 6.2      | 11.5 ± 2.9      | 14.9 ± 3.7      | 18.3 ± 2.0      | 21.6 ± 2.4      |
| DW110       | 12.1 ± 3.6      | 25.0 ± 6.2      | 1.8 ± 0.5       | 4.4 ± 1.1       | 6.2 ± 0.7       | 14.2 ± 1.6      |
| DW130       | 6.0 ± 1.8       | 25.0 ± 6.2      | 7.2 ± 1.8       | 10.5 ± 2.6      | 7.5 ± 0.8       | 18.4 ± 2.0      |

\( \Lambda_{210Pb}^\alpha, \Lambda_{210Pb}^\beta \) data published [57]
n excess of 210Po is already volatilized. Therefore, in order to determine the annual effective dose due to the ingestion of the 210Po radionuclide by drinking water consumption.

The evaluation of the annual effective dose due to the ingestion of natural radionuclides by drinking water consumption.

Table 3 The evaluation of the annual effective dose due to the radionuclide ingestion by drinking water—samples taken from Galați during the year 2014, using the Eqs. (3), (5), (6), (7)

| Sample code | $\sum_{i=1}^{6} C_{i}$ | $D_{\text{ef}}$ (μSv year$^{-1}$), calculated by relying on |
|-------------|-------------------------|----------------------------------------------------------|
|             | $C_{\text{i(der.)}}$ | $C_{\text{assum.210Po}}^{\text{assum.228Ra}}$ and $C_{\text{assum.226Ra}}^{\text{assum.210Po}}$ | $\Lambda_{\alpha}$ | $C_{\text{assum.210Po}}^{\text{assum.226Ra}}$ | $C_{\text{assum.226Ra}}^{\text{assum.210Po}}$ |
| DW1         | 0.23                    | 9.78 ± 1.97 | 17.85 ± 4.72 | 2.45 ± 0.74 | 5.26 ± 1.58 | 1.23 ± 0.37 | 12.59 ± 3.15 |
| DW2         | 0.22                    | 5.56 ± 1.11 | 28.80 ± 8.01 | 7.56 ± 2.27 | 16.21 ± 4.86 | 3.78 ± 1.13 | 12.59 ± 3.15 |
| DW3         | 0.23                    | 9.70 ± 1.95 | 17.85 ± 4.72 | 2.45 ± 0.74 | 5.26 ± 1.58 | 1.23 ± 0.37 | 12.59 ± 3.15 |
| DW4         | 0.22                    | 5.59 ± 1.24 | 36.68 ± 9.43 | 2.45 ± 0.74 | 5.26 ± 1.58 | 1.23 ± 0.37 | 31.42 ± 7.85 |
| DW5         | 0.22                    | 14.28 ± 1.17 | 86.59 ± 2.54 | 17.72 ± 5.31 | 37.96 ± 11.39 | 8.86 ± 2.66 | 48.62 ± 12.16 |
| DW6         | 0.23                    | 17.18 ± 1.78 | 32.56 ± 9.14 | 9.32 ± 2.80 | 19.97 ± 5.99 | 4.66 ± 1.40 | 12.59 ± 3.15 |
| DW7         | 0.21                    | 5.36 ± 0.91 | 64.02 ± 16.97 | 8.99 ± 2.70 | 19.26 ± 5.78 | 4.49 ± 1.35 | 44.76 ± 11.19 |
| DW8         | 0.22                    | 14.36 ± 1.17 | 19.33 ± 5.17 | 3.15 ± 0.94 | 6.74 ± 2.02 | 1.57 ± 0.47 | 12.59 ± 3.15 |
| D110        | 0.22                    | 6.49 ± 1.43 | 23.18 ± 6.32 | 4.94 ± 1.48 | 10.58 ± 3.17 | 2.47 ± 0.74 | 12.59 ± 3.15 |
| DW130       | 0.21                    | 8.19 ± 0.52 | 17.85 ± 4.72 | 2.45 ± 0.74 | 5.26 ± 1.58 | 1.23 ± 0.37 | 12.59 ± 3.15 |

$C_{\text{i(der.)}}$—measured concentrations of the following radionuclides: $^{210}\text{Po}$, $^{210}\text{Pb}$, $^{226}\text{Ra}$, $^{238}\text{U}$, $^{232}\text{Th}$, $C_{\text{assum.210Po}}^{\text{assum.228Ra}}$—assumed concentration for the specific activity of $^{210}\text{Po}$, as derived from the entire measured gross alpha activity, $C_{\text{assum.226Ra}}^{\text{assum.210Po}}$—assumed concentration for the specific activity of $^{226}\text{Ra}$, as derived from the entire measured gross alpha activity, $C_{\text{assum.226Ra}}^{\text{assum.210Po}}$—assumed concentration for the specific activity of $^{210}\text{Pb}$, as derived from the entire measured gross beta activity.

Table 4 The Pearson correlation between the alpha radioactivity parameters in the analysed water samples

| (mBq L$^{-1}$) | $\Lambda_{\alpha}$ | $\Lambda_{210\text{Po}}$ | $\Lambda_{238\text{U}}$ | $\Lambda_{232\text{Th}}$ | $\Lambda_{226\text{Ra}}$ |
|---------------|-------------------|------------------------|------------------------|------------------------|------------------------|
| $\Lambda_{\alpha}$ | 1                  |                        |                        |                        |                        |
| $\Lambda_{210\text{Po}}$ | 0.44               | 1                      |                        |                        |                        |
| $\Lambda_{238\text{U}}$ | -0.04              | 0.21                   | 1                      |                        |                        |
| $\Lambda_{232\text{Th}}$ | -0.05              | 0.21                   | 0.98                   | 1                      |                        |
| $\Lambda_{226\text{Ra}}$ | -0.15              | -0.11                  | 0.67                   | 0.66                   | 1                      |

as specific concentrations of $^{210}\text{Po}$ and $^{228}\text{Ra}$ radionuclides, as expected.

This can be explained as follows: in the present study and in general, the measurement of the gross alpha activity [77, 78] was performed using the usual method of residue measurement according to which, obtaining the desired residue, which could be measured from a gross alpha activity point of view, is done at a temperature of up to 350 °C, at which $^{210}\text{Po}$ is already volatilized. Therefore, the measured gross alpha activity does not represent an exact measure of the specific activity of $^{210}\text{Po}$, as one may notice a weak correlation between these two radioactivity parameters assessed by means of the Pearson correlation shown in Table 4. There is not a correlation between the gross alpha activity and the concentration of alpha emitter radionuclides, for these reasons: when this method is used, the different radioisotopes with the different energy are present in standard and samples, the pretreatment might lead to release the radioisotopes from the sample, the very low concentration of isotopes requires their preconcentration and purification to become detectable by radiochemical separation.

It is noteworthy that, for the investigated samples, a very good correlation was found between $\Lambda_{232\text{Th}}$ and $\Lambda_{238\text{U}}$, between $\Lambda_{226\text{Ra}}$ and $\Lambda_{238\text{U}}$, and $\Lambda_{226\text{Ra}}$ and $\Lambda_{232\text{Th}}$.

The values of the annual effective dose ($D_{\text{ef}}$) due to the ingestion of the $^{210}\text{Po}$ radionuclide by drinking water consumption, for the investigated samples, based on the specific measured activity of $^{210}\text{Po}$ ($\Lambda_{\alpha210\text{Po}}$) are lower than the values of the annual effective dose assessed on the basis of the specific assumed activity of $^{210}\text{Po}$ ($C_{\text{assum.210Po}}$), except in the case of the samples DW8, DW110.

In the case of relation (5), this leads to higher $D_{\text{ef}}$ values as compared to relation (3). Moreover, it may be noticed that, in the case of this study, relation (5) leads to the highest values for the annual effective dose. This calculation method is based on the maximum associated risk for estimating the annual effective dose due to the ingestion of natural radionuclides by drinking water consumption.

The $D_{\text{ef}}$ values calculated with relation (3) are lower than the annual effective dose values calculated by assuming the measured beta activity as specific activity of $^{228}\text{Ra}$/$^{210}\text{Pb}$, except for samples DW6 and DW8. In this case, too, it can be said that the measurement of the gross beta activity is not an exact measure of the specific activity of beta radionuclides, in this case $^{228}\text{Ra}$/$^{210}\text{Pb}$, for which radiochemical separations are necessary.
The $D_{ef}$ calculated based on the measured concentration of $^{238}$U has the same order of magnitude as that calculated based on the assumed concentration as derived from the measured gross alpha activity. The same happens when using the measured and the assumed concentration of $^{226}$Ra. However, this is not the case with the $^{232}$Th radionuclide, in which case the $D_{ef}$ calculated on the basis of the assumed concentration of $^{212}$Th is derived from the measured gross alpha activity.

The $D_{ef}$ calculated from a single radionuclide based on the measured concentrations and on the assumed concentrations differ very little in the case of $^{238}$U and very much in the case of $^{232}$Th.

**Conclusions**

In the present study, the calculation methods of the annual effective dose due to the ingestion of natural radionuclides through drinking water consumption were analysed, exemplified and discussed. This was done on the basis of the measured radioactivity parameters, on the one hand, and on the basis of the assumed radioactivity parameters, on the other hand, using the calculation rationale found in the domain-specific literature. The advantages and disadvantages of the calculation methods are are shown in the Table 5.

Two algorithms of assessment of the annual effective dose due to radionuclide intake through of drinking water consumption were identified: based on the measured radionuclide concentrations and on the basis of the assumed radionuclide concentrations. There are major differences between the two algorithms.

The use of relation (3) in order to calculate the annual effective dose leads to an appreciation characterized by high accuracy of the result. The annual effective dose, calculated in this way, should be accompanied by the specification of the radionuclides whose measured concentration was used. It is worth mentioning here that it is advisable to determine the concentrations of radionuclides with the highest effective dose coefficient. Also, when using this calculation method, one should take into account the recommendations of the EU Directive—according to which the contribution of tritium, of potassium-40, of radon and of short-life products resulting from the decay of radon mustn’t be used.

The use of relation (5) in which the entire gross alpha and beta activity is attributed to the $^{210}$Po radionuclide and the $^{210}$Pb/$^{228}$Ra radionuclide, respectively, leads to an over-assessment of the annual effective dose. This calculation method covers the maximum risk due to radiation effects and is taken into account only when gross alpha or beta activity measurements are available and when there isn’t any data on radionuclide concentrations.
Relation (6) generally leads to lower \( D_{ef} \) values than relation (5), but closer to the values determined based on relation (3). Relation (6) represents a rapid method of \( D_{ef} \) assessment, generating a minimal risk of over assessment, often used only when the gross alpha and beta activities for the investigated water samples are available and a quick response is required.

Relation (7) uses assumed alpha-emitting radionuclide concentrations as derived from the measured gross alpha activity and assumed beta-emitting radionuclide concentrations as derived from the measured gross beta activity. This calculation method is useful when it is desirable to assess \( D_{ef} \) on radionuclide type and cannot be compared to the other calculation methods (relations 3, 5, 6) that include the contribution of several radionuclides whose concentration is either measured or assumed.

Depending on the laboratory (both from a technical and from a specialized human resource point of view), on the available data, on the response time available, on the aim pursued, the most adequate method of \( D_{ef} \) assessment is selected, each of the above-mentioned methods providing valuable information on quantifying the exposure of the population to ionizing radiation through drinking water consumption.

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