Journal of Physical Science, Vol. 26(1), 83–98, 2015

Determination of Radiological Hazards and the Transfer Factors of Radionulides from Soil to Vegetables in the Southwestern District of Bangladesh

Md Altaf Hossen[1*] and Nasima Ferdous[2]

[1]Reactor Physics and Engineering Division, Institute of Nuclear Science and Technology, Bangladesh Atomic Energy Commission, Agargaon 1207, Bangladesh
[2]Department of Physics, University of Dhaka, Dhaka 1000, Bangladesh

[*]Corresponding author: altaf335@yahoo.com

Abstract: *The activity concentrations, gamma absorbed dose rates (D), radium equivalent activities (Ra_{eq}), external hazard index (H_{ex}), internal hazard index (H_{in}) and transfer factors (TFs) of the naturally occurring radionuclides ^{226}Ra, ^{232}Th, and ^{40}K, as well as the artificial radionuclide ^{137}Cs, in soil and vegetables (red amaranth) samples were determined from samples collected from Satkhira, the southwestern district of Bangladesh, using a high-purity germanium (HPGe) detector. The activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K in soil samples ranged from 26.05 ± 3.49 to 48.45 ± 4.45 Bq kg^{-1} (average = 35.71 ± 3.92 Bq kg^{-1}), 46.78 ± 3.24 to 62.63 ± 3.68 Bq kg^{-1} (average = 54.06 ± 3.45 Bq kg^{-1}) and 411 ± 67 to 775 ± 77 Bq kg^{-1} (average = 580.6 ± 71.63 Bq kg^{-1}), respectively, and the radioactivity concentrations of ^{226}Ra, ^{232}Th and ^{40}K in vegetable samples from the same locations were 4.04 ± 2 to 12.07 ± 2.36 Bq kg^{-1} (average = 6.86 ± 2.12 Bq kg^{-1}), 9.01 ± 1.54 to 23.62 ± 1.94 Bq kg^{-1} (average = 14.89 ± 1.70 Bq kg^{-1}) and 1124 ± 73 to 1453 ± 76 Bq kg^{-1} (average = 1288.7 ± 74.41 Bq kg^{-1}), respectively. No ^{137}Cs activity concentration was detected in any of the samples from this district. The measured values of D, Ra_{eq}, H_{ex} and H_{in} are comparable to the global averages. With the exception of ^{40}K, the transfer factors of all radionuclides were below unity.*

Keywords: Radionuclides, HPGe, activity concentrations, radiological hazards, transfer factor

1. INTRODUCTION

From the beginning of time, all living creatures have been exposed to radiation. Although man and the animal kingdom have adjusted to this natural radiation, the advent of man-made sources, their widespread application and sometimes accidental uncontrolled release of radioactivity into the environment have altered the balance. Estimates of the total radiation dose to the world population have shown that approximately 96% can be attributed to natural sources while the remaining 4% comes from artificial sources. The concentrations of natural radionuclides ^{238}U, ^{232}Th, their daughter products and ^{40}K, in the soil and rocks, which depend upon the local geology of each region,

cause dosage variation.[1] The most vital sources to the environment are the shipment of radioactive materials, as well as the residual fallout from nuclear weapon testing and nuclear accidents that continually spread a huge amount of radioactivity in the environment. Consequently, radionuclides in soil and plants have been monitored since their discovery and have increased as a result of the raise in nuclear power plant accidents.

In Satkhira, the southwestern district of Bangladesh just beside the Bay of Bengal, red amaranth (*Amaranthus gangeticus*) is one of the most popular vegetables. Most long-lived radioactive elements in vegetables are from the soil. Additionally, farmers in this region use an excess of potassium for its commercial production, increasing potassium uptake. Moreover, the targeted area is approximately 150 km away from the proposed Bangladesh nuclear power plant and some hundreds of km away from Indian nuclear power plants, most of which are across the coastal area. Any accident may raise the radionuclide concentration in soil and plants, increasing health risk. To assess radiation hazards, it is necessary to know the rate at which radiation is received; consequently, it is necessary to study the natural and artificial radioactivity in soil and vegetables to assess the dose to the population in order to know the health risks and have baseline data for the proposed nuclear power plant.

2. EXPERIMENTAL

2.1 Gamma-ray Spectroscopy

A high-purity germanium (HPGe) semiconductor detector was used for the present work.[2] The outstanding advantage of the HPGe detector is its ability to measure gamma radiation directly from the original sample without the need for chemical separation and high resolving power. The resolving power of a detector, called the energy resolution, is used to separate two adjacent peaks in a γ-ray spectrum. A germanium detector gives excellent energy resolution when applied to gamma-ray spectra. The great superiority of the germanium system in energy resolution permits the separation of many closely spaced gamma-ray energies that are unresolved in by NaI(Ti) spectra. Consequently, virtually all gamma-ray spectroscopy involving complex energy spectra now utilises germanium detectors. Sample measurements were obtained using a γ-ray spectrometer with a p-type coaxial intrinsic HPGe detector supplied by Oxford Instruments Inc. Nuclear Measurement Group (Model No. CPVDS 30-30185, Serial No. 2604) with a relative efficiency of 30% and 1.83 keV resolution (FWHM) for the ^{60}Co 1332 keV γ-ray energy.

The detector was coupled to a PC-based multi-channel analyser (PCMCA Card: TRUMP-PCI-8K) and the gamma-ray spectral analysis was performed using Windows-based software (MAESTRO-32, ver. 5.30 [A65-B32], ORTEC, U.S.) that matched the gamma energies at various energy levels to a library of possible isotopes. The detector was shielded by lead on all sides to reduce system background. Detector efficiency is measured by comparing the count rates in each peak of spectra from a source of known radioactivity to the count rates expected from the known intensities of each gamma ray.

2.2 Area Geology

Figure 1 shows the sampling area and the geographic location of the district. The district lies between 21°36′ and 21°54′ north latitudes, and 88°54′ and 89°20′ east longitudes. The district is surrounded by Jessore at the north, Khulna at the east, 24 Parganas of India at the west and the Sundarban, the largest single block of tidal halophytic mangrove forest in the world, as well as the Bay of Bengal, at the south. There are 27 rivers passing over this zone. The sampling areas are Satkhira Sadar, Kalaroa, Shymnagar, Patkelghata and Debhata.

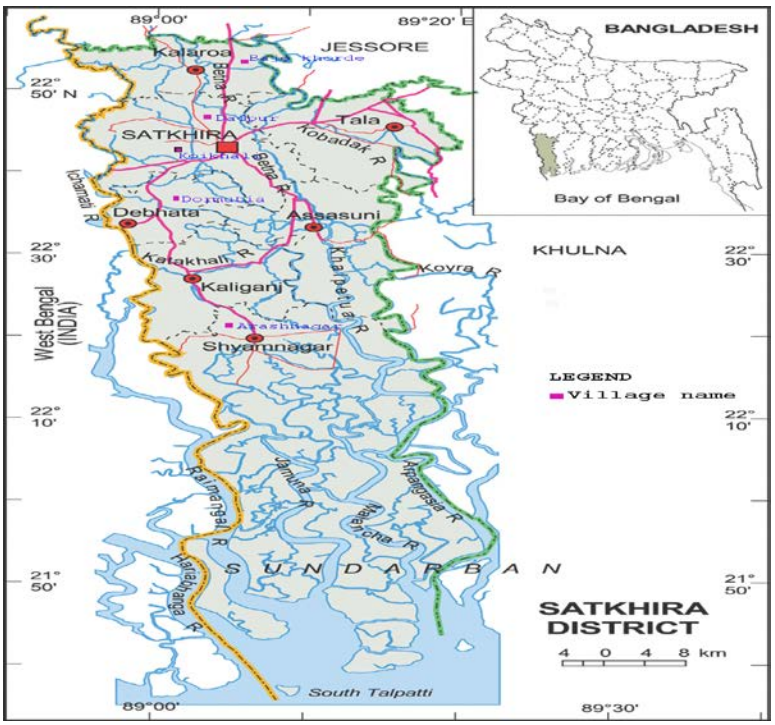

Figure1: Sampling locations in Satkhira, Bangladesh.

2.3 Sample Preparation

All of the locations selected for soil sample collection were open, not prone to flooding or other natural disturbances and were unaffected by human activity during the recent decades, representing undisturbed soils. Ten soil and 10 red amaranth samples (Table 1) were collected from 5 different areas of Satkhira, Bangladesh. For each sample location, 2 1-kg soil samples were collected at a depth of 5 to 10 cm from different locations on the land of a farmer. At the same time, 1 kg red amaranth samples were collected from just above the soil sampling areas. The samples were then placed in polyethylene bags and dried by sunlight.

Table 1: Soil and red amaranth samples collected from different locations of Satkhira, Bangladesh.

Location	Sample no. (Soil)	Dry weight (gm)	Sample no. (Red amaranth)	Dry weight (gm)
Satkhira Sadar	1	355	11	150
	2	332	12	165
Debhata	3	347	13	160
	4	373	14	165
Patkelghata	5	368	15	155
	6	322	16	158
Shyamnagar	7	342	17	156
	8	334	18	170
Kalaroa	9	368	19	157
	10	349	20	150

Each dried vegetable sample was then ground into fine powder using a mortar and pestle, and weighed separately with a balance. The soil and powdered vegetable samples were then sieved using a fine-aperture mesh (2 mm mesh size) to remove extraneous items such as plant materials, roots, pebbles, dust, stones, etc., to obtain a fine-grained sample that would present a uniform matrix to the detector.

At the lab, the samples were transferred from the polyethylene bags to acetone-cleaned stainless steel buckets and dried in an oven at 105°C for 1 h. Finally, each sample was transferred to a cylindrical plastic-container of approximately equal size and shape. The net weight of each sample was calculated using a micrometre. The containers were then sealed tightly, wrapped with thick vinyl tape around the screw necks, and tagged with the location, date of collection, weight and sample number, etc. The samples were stored for at

least 4 weeks to reach secular equilibrium between the ^{238}U and ^{232}Th series and their respective progenies to prepare for measurements.[3]

2.4 Methodology

2.4.1 Determination of Activity

The radioactivity concentrations of ^{226}Ra were determined from the gamma-ray energies of its daughters ^{214}Pb (352.92 and 295.21 keV) and ^{214}Bi (609.31, 1120.30 and 1764.50 keV), while the radioactivity concentrations of ^{232}Th were determined from the gamma-ray energies of its daughters ^{212}Pb (238.63 keV), ^{208}Tl (583.14 and 510.84 keV) and ^{228}Ac (911.07 and 969.11 keV). The radioactivity concentrations of ^{40}K and ^{137}Cs were determined from their gamma-ray energies of 1460.80 keV and 662 keV, respectively. The activity concentrations were calculated using the formula below:[4]

$$A = \frac{N}{P_\gamma \times \varepsilon \times W} \text{ (Bq kg}^{-1}\text{)} \quad (1)$$

where

N = Net counts per second (C.P.S) = (Sample C.P.S – background C.P.S)
P_γ = Intensity of the radionuclide
E = Efficiency in %
W = Weight of sample in kilograms

The activities of the parent ^{232}Th nucleus of the thorium decay series and the head of the uranium decay series, ^{226}Ra, were determined by assuming that they were in radioactive equilibrium with their daughter products ^{212}Pb and ^{214}Pb, respectively.[5] Standard characteristic values of Pγ were used in the present study.[6]

2.4.1.1 Lower Limits of Detection (LLD)

The detection capability of a measurement system under certain conditions is determined by the term Lower Limits of Detection (LLD). The LLD of the detector can be obtained using a high efficiency detector, preferably a large sample and a long counting time. The lower limit of detection is obtained using the expression below:

$$\text{LLD}=\frac{4.66S_b}{\varepsilon(E)\times I_\gamma} \tag{2}$$

where

S_b = the estimated standard error of the net count rate = $\sqrt{b/t^2}$ (with b = background count and t = counting time) $\varepsilon(E)$ = the counting efficiency of the desired energy of the nuclides, and I_γ = the absolute transition probability by γ-decay through the selected energy as for ε.

The LLDs of the examined radionuclides are listed in Table 2.

Table 2: Lower limits of detection for the radionuclides.

Radionuclide	Energy (keV)	LLD (Bq)
Pb-212	238.63	0.4133
Ac-228	911.07	3.1833
Ac-228	969.11	3.1779
Pb-214	295.21	1.3488
Pb-214	352.92	0.9488
Bi-214	609.31	1.2804
Bi-214	1120.30	3.3233
K-40	1460.80	39.969

2.4.2 Dose rate estimation

The gamma dose rates D (in $nGyh^{-1}$) of the radionuclides were estimated using the formula:[7]

$$D = 0.429A_{Ra} + 0.666A_{Th} + 0.042A_K \tag{3}$$

where A_{Ra}, A_{Th} and A_K are the mean specific activity concentrations of Ra, Th and K, respectively, in Bq kg^{-1}.

2.4.3 Radium equivalent activity

The Radium Equivalent Activity (Ra_{eq}) index provides guideline for regulating the safety standards of radiation protection for the general public residing in the investigated area. The Ra_{eq} index represents a weighted sum of the activities of the aforementioned natural radionuclides. The index is given by the following formula:[7]

$$Ra_{eq} = A_{Ra} + 1.43A_{Th} + 0.077A_K \quad (4)$$

where A_{Ra}, A_{Th} and A_k have the same meaning as in Equation 3.

2.4.4 External and internal hazard indices

Soil is used to produce earthen huts, bricks and pottery materials. Consequently, the external radiation hazard index (H_{ex}) due to natural gamma radiation is calculated using the following formula:[3]

$$H_{ex} = A_{Ra}/370 + A_{Th}/259 + A_K/4910 \leq 1 \quad (5)$$

There is also a radiation hazard to respiratory organs due to the ^{226}Ra decay product ^{222}Rn and its short-lived decay products. To account for this hazard, the maximum permissible radium concentration must be reduced to half of the normal limit.[3] The internal hazard index (H_{in}) is calculated using the following formula:[3]

$$H_{in} = A_{Ra}/185 + A_{Th}/259 + A_K/491 \quad (6)$$

where A_{Ra}, A_{Th} and A_k have the same meaning as in Equation 3 and Equation 4.

2.4.5 Transfer factors

Radionuclides in soils are usually transferred to different plant tissues by direct transfer via the root system, as well as radionuclide fallout and resuspension of contaminated soil followed by deposition on plant leaves.[8] The transfer factor (TF) values are calculated according to the equation below.[9]

$$TF = \frac{\text{Activity of radionuclides in plant weight (Bq / Kg dry weight)}}{\text{Activity of radionuclides in soil weight (Bq / Kg dry weight)}} \quad (7)$$

Soil properties that affect uptake may include mineralogical and granulometric composition, organic matter content, pH and fertility.[10]

3. RESULTS AND DISCUSSION

3.1 The Activity Concentration

The present study examined the distribution of naturally occurring radionuclides ^{226}Ra, ^{232}Th and ^{40}K, as well as a fission product resulting from

fallout, in soil and vegetable samples. The activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K in soil samples ranged from 26.05 ± 3.49 to 48.45 ± 4.45 Bq kg^{-1} (average = 35.71 ± 3.92 Bq kg^{-1}), 46.78 ± 3.24 to 62.63 ± 3.68 Bq kg^{-1} (average = 54.06 ± 3.45 Bq kg^{-1}) and 411 ± 67 to 775 ± 77 Bq kg^{-1} (average = 580.6 ± 71.63 Bq kg^{-1}), respectively. In vegetable samples from the same locations, the radioactivity concentrations of ^{226}Ra, ^{232}Th and ^{40}K were 4.04 ± 2 to 12.07 ± 2.36 Bq kg^{-1} (average = 6.86 ± 2.12 Bq kg^{-1}), 9.01 ± 1.54 to 23.62 ± 1.94 Bq kg^{-1} (average = 14.89 ± 1.70 Bq kg^{-1}) and 1124 ± 73 to 1453 ± 76 Bq kg^{-1} (average = 1288.7 ± 74.41 Bq kg^{-1}), respectively. No ^{137}Cs radioactivity was detected in any of the samples.

Table 3 shows that the thorium activity concentration was higher than that of radium in most of the samples, which is evident from the fact that thorium is 1.5-fold more abundant in the Earth's crust than uranium.[11] Table 4 compares the results of the present study to world results. In the present study, activity concentrations of ^{226}Ra and ^{232}Th in soil and vegetable samples are comparable to the values reported in the table, especially those from different regions of Bangladesh.[12–15] It was also observed that the measured ^{40}K activity concentration markedly exceeds the values of both radium and thorium, as it is the most abundant radioactive element considered. High vegetable absorption of potassium used by farmers for commercial production might have enhanced its concentration in the red amaranth samples.

Table 3: Radioactivity concentrations in soil and red amaranth samples from the district of Satkhira, Bangladesh.

Sample no.	Sample	Radioactivity concentrations (Bq kg^{-1})						^{137}Cs
		^{226}Ra		^{232}Th		^{40}K		
		I	R	I	R	I	R	
1	Soil	31.42 ± 3.74	26.05–48.45	58.98 ± 3.58	46.78–62.63	775 ± 77	411–775	–
2		38.83 ± 4.07		55.34 ± 3.49		675 ± 74		–
3		47.17 ± 4.4		51.3 ± 3.37		532 ± 70		–
4		31.5 ± 3.75		51.98 ± 3.39		614 ± 73		–
5		32.01 ± 3.77		46.78 ± 3.24		411 ± 67		–
6		48.45 ± 4.45		49.11 ± 3.31		509 ± 70		–

(*continued on next page*)

Table 3: (*continued*)

Sample no.	Sample	Radioactivity concentrations (Bq kg^{-1})						^{137}Cs
		^{226}Ra		^{232}Th		^{40}K		
		I	R	I	R	I	R	
7	Red amaranth	26.05 ± 3.49	4.04–12.07	60.98 ± 3.64	9.01–23.62	608 ± 72	1124–1453	–
8		42.83 ± 4.23		62.63 ± 3.68		578 ± 72		–
9		30.48 ± 3.7		50.18 ± 3.34		582 ± 72		–
10		28.35 ± 3.6		53.34 ± 3.43		522 ± 70		–
11		8.17 ± 2.17		17.8 ± 1.77		1453 ± 76		–
12		6.66 ± 2.13		12.29 ± 2.65		1124 ± 73		–
13		7.02 ± 2		14.04 ± 1.69		1284 ± 75		–
14		12.07 ± 2.36		17.15 ± 1.8		1160 ± 73		–
15		5.6 ± 2.07		23.62 ± 1.94		1363 ± 75		–
16		4.04 ± 2		11.54 ± 1.6		1206 ± 73		–
17		4.58 ± 2.03		20.48 ± 1.86		1415 ± 76		–
18		9.17 ± 2.25		9.01 ± 1.54		1325 ± 76		–
19		6.18 ± 2.1		12.73 ± 1.64		1281 ± 74		–
20		5.11 ± 2.04		10.21 ± 1.54		1276 ± 73		–

Note: I = individual, R = range.

The absence of ^{137}Cs detection might be due to a very low radionuclide concentration and long half-life, as well as the comparatively lower efficiency of the detector. Consequently, it may be concluded that in this region there is no radioactive fallout from the Chernobyl accident.

Table 4: Comparison of ^{226}Ra, ^{232}Th, ^{40}K and ^{137}Cs (Bq kg^{-1}) activity concentrations in soil samples from different parts of the world with the results of the present study.

Sampling zone	Activity concentration (Bq kg^{-1})			
	^{226}Ra	^{232}Th	^{40}K	^{137}Cs
Volcanic area (Cameroon)[12]	11–17	22–36	201	7.2
Pabna (Bangladesh)[13]	33	47	449	4
Dhaka (Bangladesh)[14]	33	55	574	7
Southern District, Turkey[15]	19–276	8–244	86–1162	1.8–82
Peshawar (Pakistan)[17]	65	84	646	–
All India average[18]	31	63	394	–
Jessore (Bangladesh)[19]	48.32	53.34	481.35	–
Louisiana (U.S.)[20]	43–95	50–190	43–729	5–58
Worldwide average[16]	40 (15–50)	40 (7–50)	580 (100–700)	–
Satkhira (Bangladesh) (Present study)	**35.71 (28–48)**	**54.07 (49–62)**	**580.6 (411–775)**	–

3.2 Radium Equivalent Activity

The calculated Ra_{eq} data is presented in Table 5. The Ra_{eq} for soil and vegetable samples varied in the range of 130.55–176.9 Bq kg^{-1} (average = 157.72 Bq kg^{-1}), and 110.78–145.51 Bq kg^{-1} (average = 127.38 Bq kg^{-1}), respectively. These values are far below the allowable limit (370 Bq kg^{-1}) recommended by the International Atomic Energy Agency (IAEA).[3,7,16]

Table 5: Radium equivalent activities, absorbed dose rates, and external and internal hazard indices in soil and vegetable samples from across the district of Satkhira, Bangladesh.

Sample no.	Sample	Radiological index			
		Ra_{eq} ($Bq\,kg^{-1}$)	D ($nGyh^{-1}$)	H_{ex}	H_{in}
1	Soil	175.44	85.94	0.47	0.56
2		169.94	82.37	0.46	0.56
3		161.49	77.08	0.44	0.56
4		153.11	74.39	0.41	0.5
5		130.55	62.39	0.35	0.44
6		157.87	75.19	0.43	0.56
7		160.07	77.76	0.43	0.5
8		176.9	84.72	0.48	0.59
9		147.05	71.38	0.4	0.48
10		144.82	69.97	0.39	0.47
11	Red amaranth	145.51	78.04	0.39	0.41
12		110.78	59.54	0.3	0.32
13		125.97	67.76	0.34	0.36
14		125.91	66.62	0.34	0.37
15		144.33	76.91	0.39	0.4
16		113.40	61.46	0.31	0.32
17		142.82	76.64	0.39	0.4
18		124.08	67.12	0.34	0.36
19		123.02	66.41	0.33	0.35
20		117.96	64.06	0.32	0.33

3.3 Absorbed Dose Rate

The absorbed dose rates are presented in Table 6. In the present study, the dose rate due to ^{226}Ra, ^{232}Th, and ^{40}K in soil samples varied from 62.39 $nGyh^{-1}$ to 85.94 $nGyh^{-1}$ (average = 76.12 $nGyh^{-1}$); for vegetable samples, these values varied from 59.54 $nGyh^{-1}$ to 78.04 $nGyh^{-1}$ (average = 68.45 $nGyh^{-1}$). The values are comparable to the world average of 55 $nGyh^{-1}$.[7] Table 6 also compares the absorbed doses in different world locations.

Table 6: Comparison of absorbed dose based on different approaches and results in the world.

Countries	Year	No of samples	Absorbed dose rate ($nGyh^{-1}$)	Absorbed dose range ($nGyh^{-1}$)	Methodology
Romania[7]	1979	2372	81	32–210	Soil analysis using gamma spectrometry
Nigeria[21]	2000	20	128	5–186	Analysis of rocks using gamma spectrometry
China[7]	1972	26	69	–	Soil analysis using gamma spectrometry
Chittagong (Bangladesh) [12]	1999	24	75	42–120	Soil analysis using gamma spectrometry
India[22]	2002	39	95.2	–	Soil analysis using gamma spectrometry
World average[7]	–	–	55	30–70	–
Jessore (Bangladesh) [19]	2007	23	77	62–100	Soil analysis using gamma spectrometry
Satkhira, Bangladesh (Present study)	**2008**	**20**	**72.29**	**59–86**	**Soil analysis using gamma spectrometry**

3.4 External and Internal Hazard Indices

The H_{ex} values for soil samples (Table 5) were 0.35–0.48 (average = 0.43) and for vegetable samples they were 0.3–0.39 (average = 0.34). The calculated H_{ex} values for all samples should be below unity, which does not cause harm to the populations of the investigated regions. In the present work, all H_{ex} values were below unity. There is also a radiation hazard for respiratory organs due to the ^{226}Ra decay product ^{222}Rn and its short-lived decay products. To account for this threat, the maximum permissible concentration for radium must be reduced to half of the normal limit.[3] In our study, the soil sample H_{in} values (shown in Table 5) were 0.44–0.59 (average = 0.52) and for vegetable samples, H_{in} values were 0.32–0.42 (average = 0.36).

3.5 Transfer Factor

The TFs from soil to vegetable (shown in Table 7) for ^{226}Ra, ^{232}Th and ^{40}K were 0.08–0.38, 0.14–0.34 and 1.67–3.32, respectively, with average TFs of 0.199, 0.279 and 2.28, respectively. TFs were not determined for ^{137}Cs as ^{137}Cs radioactivity concentration was not detected in either soil or vegetable samples.

The ^{40}K TF was greater than unity in all samples and could be due to excessive use of potassium-containing fertilisers at the sampling sites.[19] Table 8 compares the transfer factors determined by the present work with world transfer factors. Slight differences between the transfer factors of this zone and the values of different world locations could be primarily due to soil type, formation, transport process, media and absorbing capacity of the plants studied.

Table 7: Determination of the radionuclide transfer factors from soil to red amaranth.

Sample number		TF			
Soil	Red amaranth	^{226}Ra	^{232}Th	^{40}K	^{137}Cs
1	11	0.26	0.3	1.87	–
2	12	0.17	0.22	1.67	–
3	13	0.14	0.27	2.41	–
4	14	0.38	0.33	1.89	–
5	15	0.18	0.5	3.32	–
6	16	0.08	0.23	2.37	–
7	17	0.18	0.34	2.33	–
8	18	0.21	0.14	2.29	–
9	19	0.2	0.25	2.2	–
10	20	0.18	0.19	2.44	–

Table 8: Comparison of soil-to-plant TFs obtained in different parts of the world with the results of the present study.

Sample location (transfer media)	^{226}Ra	^{232}Th	^{40}K	^{137}Cs
Thessaloniki, Greece[23] (soil to plant)	–	–	0.73 (0.25–2.42)	0.20 (0.002–7.42)
Kaiga-region, India[24] (soil to leaf)	0.03–0.65	–	0.32–8.04	0.05–3.03
Ramsar, Iran[25] (vegetable to soil)	–	–	–	0.016
Syria[26] (soil to plant)	–	0.11	–	0.04
Mediterranean area[27]	0.17 (0.09–0.05)	1.65 (0.052–4.31)	0.393 (0.12–0.92)	–
Satkhira, Bangladesh (soil to vegetable) (Present study)	**0.199**	**0.279**	**2.28**	–

4. CONCLUSION

In conclusion, the activity concentrations of ^{40}K in both the soil and vegetable samples were found to be higher than those for ^{232}Th, which in turn were greater than those for ^{226}Ra. The activity concentrations, gamma absorbed dose rates (D), radium equivalent activity (Ra_{eq}), hazard indices (H_{ex} and H_{in}) and transfer factors (TFs) of the naturally occurring radionuclides ^{226}Ra, ^{232}Th and ^{40}K as well as the artificial radionuclide ^{137}Cs in soil and vegetable samples matched well with world values. No ^{137}Cs activity concentration was found in any of the samples from this district, indicating the absence of artificial radionuclide fallout from any nuclear accident.

5. ACKNOWLEDGEMENT

The authors would like to thank the Health Physics Division of the Bangladesh Atomic Energy Centre (BAEC) in Dhaka, for allowing us to conduct experiment in their laboratories, and the Department of Physics at the University of Dhaka.

6. REFERENCES

1. Radhakrishna, A. P. et al. (1993). A new natural background radiation area on the southwest coast of India. *Health Phys.*, 65, 390–395.
2. Islam, M. (1994). Evaluation of gamma spectrometric characteristic of an HPGe detector for the radioactivity measurement of food and environment. Technical Report, AERE, Bangladesh Atomic Energy Commission, 02, 50–55.
3. Beretka, J. & Mathew, P. J. (1985). Natural radioactivity of Australian building materials, industrial waste and by-products. *Health Phys.*, 48(1), 87–95.
4. Noordin, I. (1999). Natural activities of ^{238}U, ^{232}Th and ^{40}K in building materials. *J. Environ. Radioact.*, 43, 255–258.
5. Harb, S. (2007). Measurement of the radioactivity of ^{238}U, ^{226}Ra, ^{210}Pb, ^{228}Th, ^{232}Th, ^{228}Ra, ^{137}Cs and ^{40}K in tea using gamma spectrometry. *J. Radioanal. Chem.*, 274(1), 63–66.
6. International Atomic Energy Agency, IAEA. (1989). Measurement of radionuclides in food and the environment. Technical reports series no. 295, Vienna.
7. United Nations Scientific Committee on the Effect of Atomic Radiation, UNSCEAR. (1988). *Exposure from natural sources of radiation.* New York: UNSCEAR.

8. Noordijk, K. E. H. et al. (1992). Impact of ageing and weather conditions on soil-to-plant transfer of radiocesium and radiostrontium. *J. Environ. Radioact.*, 15, 277–286.
9. International Union of Radio Ecologists. (1994). Handbook of parameter values for the prediction of radionuclide transfer in temperate environments. Technical reports series no. 364, IAEA, Vienna.
10. Kuhn, W., Handl, J. & Schuller, P. (1984). The influence of soil parameters on ^{137}Cs uptake by plants from long-term fallout on forest clearings and grassland. *Health Phys.*, 46(5), 1083–1093.
11. Ngachin, M. et al. (2008). Radioactivity level and soil radon measurement of a volcanic area in Cameroon. *J. Environ. Radioact.*, 99(7), 1056–1060.
12. Chowdhury, M. I. et al. (1999). Distribution of radionuclides in the river sediments and coastal soils of Chittagong, Bangladesh and evaluation of the radiation hazard. *Appl. Radiat. Isot.*, 51, 747–755.
13. Roy, S., Hoque, A. & Begum, M. (2001). Distribution of ^{137}Cs and naturally occurring radionuclides in soil at sites of the Roopur nuclear power plant. *Nucl. Sci. Appl.*, 10(1–2), 33–38.
14. Miah, F. K. et al. (1998). Distribution of radionuclides in soil samples in and arround Dhaka city. *App. Radiat. Isot.*, 49(1–2), 133–137.
15. Varinlioglu, A. et al. (2005). Determination of natural and artificial radionuclide levels in soils of western and southern coastal area of Turkey. *J. Water Air Soil Pollut.*, 164, 401–407.
16. UNSCEAR. (1993). *Exposure from natural sources of radiation.* New York: United Nations.
17. Ali, S. et al. (1996). Gamma-ray activity and dose rate of brick samples from some areas of North West Frontier Province (NWFP), Pakistan. *Sci. Total Environ.*, 187, 247–252.
18. Prasad, N. G. et al. (2008). Concentrations of ^{226}Ra, ^{232}Th and ^{40}K in the soils of Bangalore region, India. *Health Phys.*, 94(3), 264–271.
19. Kabir, K. A. et al. (2009). Distribution of radionuclides insurance soil and bottom sediment in the district of Jessore, Bangladesh, and evaluation of radiation hazard. *J. Bangla. Acad. Sci.*, 33(1), 117–130.
20. Delaune, R. D., Jones, G. L. & Smith, C. J. (1986). Radionuclide concentration in Louisiana soils and sediments. *Health Phys.*, 51, 239–244.
21. Ajayi, O. S. (2000). Distribution of natural radioactivity in rocks from Ikogoshi-Ekiti, southwestern Nigeria and its radiological implications. *Health Phys.*, 79(2), 192–195.
22. Selvasekarapandian, S. et al. (2002). Natural radiation distribution of soil at Katagiri Taluk of the Nilgiris biosphere in India. *J. Radioanal. Nucl. Chem.*, 252, 74–78.

23. Papastefanou, C. et al. (1999). Soil-to-plant transfer of ^{137}Cs, ^{40}K and ^{7}Be. *J. Environ. Radioact*., 45(1), 59–65.
24. James, J. P. et al. (2011). Soil to leaf transfer factor for the radionuclides ^{226}Ra, ^{40}K, ^{137}Cs and ^{90}Sr at Kaiga region, India. *J. Environ. Radioact.*, 102(12), 1070–1077.
25. Ghiassi-Nejad, M. et al. (2003). Exposure to ^{226}Ra from consumption of vegetables in the high level natural radiation area of Ramsar-Iran. *J. Environ. Radioact.*, 66(3), 215–225.
26. Al-Masri, M. S. et al. (2008). Transfer of ^{40}K, ^{238}U, ^{210}Pb, and ^{210}Po from soil to plant in various locations in south of Syria. *J. Environ. Radioact.*, 99(2), 322–331.
27. Vera Tome, F. et al. (2003), Soil-to-plant transfer factors for natural radionuclides and stable elements in a Mediterranean area. *J. Environ. Radioact.*, 65(2), 161–75.