Journal of Radiation Research and Applied Sciences 7 (2014) 249–255

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ScienceDirect

Journal of Radiation Research and Applied Sciences

journal homepage: http://www.elsevier.com/locate/jrras

Determination of natural radioactivity and hazard in soil samples in and around gold mining area in Itagunmodi, south-western, Nigeria

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ARTICLE INFO

Article history:
Received 9 February 2014
Received in revised form
30 May 2014
Accepted 1 June 2014
Available online 17 June 2014

Keywords:
Radionuclides
Gold-mining
Itagunmodi
Radiological hazard

ABSTRACT

The activity concentrations of natural radionuclides ^{238}U, ^{232}Th and ^{40}K in soil samples from the gold mining area in Itagunmodi were measured by gamma spectrometry using Sodium Iodide detector. Radiological hazard assessments due to these natural radionuclides were carried out. The average activity concentrations of ^{238}U, ^{232}Th and ^{40}K determined in the mining sites were 55.3 ± 1.2, 26.4 ± 2.7 and 505.1 ± 7.1 Bqkg^{-1}, respectively while in the normal living areas it was respectively 8.8 ± 1.9, 17.5 ± 2.7 and 102.8 ± 12.1 Bqkg^{-1}. The mean annual effective dose in the mining site was 81.3 μSv which is about 16% higher than the world average. The mean radium equivalent activity concentration, the mean external and internal hazard indices in the study areawere less than the world averages. Therefore, mining activities in Itagunmodi poses no radiological hazard to the general public.

1. Introduction

Human environment is radioactive and human beings are exposed to radiation arising from sources including cosmic rays, natural radionuclides in water, air, soil and plants; and artificial radioactivity from fallout in nuclear testing and medical applications. The gamma radiation from natural radionuclides and cosmic rays constitute the external exposure while those derived from inhalation and ingestion through

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Peer review under responsibility of The Egyptian Society of Radiation Sciences and Applications

Production and hosting by Elsevier

http://dx.doi.org/10.1016/j.jrras.2014.06.001

foods and drinking water constitutes internal exposure to humans. IAEA in 1996 estimated that 80% of doses contribution in the environment are derived from the natural radionuclides while the remaining 20% is from cosmic ray and nuclear processes. The natural radionuclides of concern in terrestrial environment are mainly potassium (^{40}K), uranium (^{238}U), thorium (^{226}Th) and the radioactive gas radon which is produced as these naturally-occurring radioisotopes decay. Radon emanates from the ground as a result of the direct decay of naturally-occurring radium and is a major source of radiation exposure (EPA, 2007). Natural radioactivity is widely spread in the earth's environment and depends primarily on the geological and geographical conditions, and appears at different levels in the soils of each region in the world (UNSCEAR, 2000).

Gold is most famous for its use in jewelry which is mostly due to its brilliant yellow luster and its never fading beauty. As a result, it has been used to adorn buildings, artwork, and furniture. Gold is formed deep within the earth's surface by a series of geological processes which involves extraction or mining. Radiation exposures arising in the mining and mineral processing industries are in three ways. These are external gamma radiation from ores, inhalation of dusts containing long-lived alpha-emitting radionuclides and inhalation of the short-lived decay products of radon (UNSCEAR, 2000). Inhalation of radon decay products in poorly ventilated underground mines can lead to exposures in excess of current radiation exposure limits, and this could cause high incidence of lung cancer in mine workers (UNSCEAR, 2000). Environmental problems associated with Naturally Occurring Radioactive Materials (NORM) in solid minerals mines and processing occur in the process of drilling, leaching, handling, storage, transportation of mineral ores and the use of contaminated equipment or waste media without controls. These usually lead to the spread of NORM contaminating the environment, resulting in potential radiation exposure of members of the public (Innocent, Onimisi, & Jonah, 2013). Due to the health risks associated with the exposure to NORM and inhalation of the short-lived decay products of radon, international bodies and governmental organization such as International Commission on Radiological Protection (ICRP, 1991) and Environmental Protection Agency (EPA, 2007) have adopted strong measures at minimizing such exposure.

Although literature showed that studies on radionuclide concentrations in mines have been extensively studied in Nigeria (Abdulkarim & Umar, 2013; Ademola & Ademonehin, 2010; Ademola, Hammed, & Adejumobi, 2008; Ademola & Obed 2012; Innocent et al., 2013; Jibiri & Esen, 2011; Nasiru, Zakari, & Abdullahi, 2013). Mining activities have not been subjected to radiological regulatory control and so there is generally little or no awareness and knowledge of the radiological hazards and exposure levels to NORMs in mining areas.

Our present study aimed at studying the natural radionuclides levels in the soil samples from the gold mining areas in Itagunmodi, south-western, Nigeria. This would be achieved by measuring the activity concentrations of ^{238}U, ^{232}Th and ^{4} K by gamma spectroscopy using NaI (Tl) detector in the representative soil samples, evaluating radiation hazard indices and the effective dose to the public from the soil samples.

2. Geology of the study area

Itagunmodi is located on latitude 7° 31° N and longitude 4° 39° E. It is about 347 m above sea level and has an approximate population of 12655. It is a town very close to Ile-Ife, Ilesa and Modakeke in Osun State, Nigeria (Fig. 1). The gold deposits in Itagunmodi, Osun State, Nigeria is located in the clayey soil types derived from variably migmatised gneiss, biotite-and-biotite-hornblende-gneiss and weathered amphibolites respectively (Adetoyinbo, Bello, & Hammed, 2011). The area is underlain by the Precambrian basement complex of the south-western, Nigeria and it is composed of gneisses, migmatites and Schist associated with amphibolites.

3. Materials and methods.

For this research, Itagunmodi was divided into five divisions. Four divisions (A – D) in the gold mining site and one division (E) in an undisturbed land where people live about 10 km from the mining site. This is done so that the activity concentrations in the mining site can be compared to the living area. Ten samples were collected from each division (A–D) and 20 samples from division (E) at a depth of 20–30 cm using an auger at the various locations. The collected samples were dried in an oven at a temperature of 105 °C overnight in order to remove any available moisture. After drying, the samples were crushed and sieved with a mesh having holes each of diameter of 0.2 mm in order to remove organic materials, piece of stones, gravel and lumps. Afterward, the homogenized samples were weighed and a mass of 200 g of each sample was packed in a cylindrical plastic container of height 7 cm and 6 cm diameter. The plastic containers were hermetically sealed with adhesive tape (AERB, 2003) for 30 days so as to allow for ^{238}U and its short-lived progenies to reach secular radioactive equilibrium (Veiga et al., 2006) before gamma counting.

3.1. *Activity determination*

The gamma-ray spectrometry set-up used in this analysis consists of a highly shielded and well calibrated 7.6 cm by 7.6 cm NaI (Tl) detector enclosed in a 5 cm thick lead shield to

Fig. 1 – Geological map of the study area with the sampling location A–E.

assist in reducing background radiation. In addition, the set-up was coupled with a computer based Multichannel Analyzer (MCA) which was used for the data acquisition and analysis of gamma spectra. The spectrometer was tested for its linearity and then calibrated for energy using gamma sources supplied by the International Atomic Energy Agency, Vienna. This was achieved by collection of spectra data from standard sources with energies in the range 0.511 – 2.62 MeV. The channel numbers of the photopeaks corresponding to the different gamma energies were recorded after 900 s and the energy-channel linear relationship obtained is shown in Fig. 2. The energy and efficiency calibration of the gamma spectrometer was carried out using reference standard source material traceable to a mixed standard gamma source (ENV94084-200g) by Analytic Inc., Atlanta, GA, USA. The efficiency calibration curve for the NaI (Tl) detector was obtained from the efficiency calculated for ^{40}K, ^{238}U and ^{232}Th and is shown in Fig. 3. The detector assembly has a resolution of ~8% at 0.662 MeV of ^{137}Cr. The background count was determined by counting an empty container of the same dimension as those containing the samples and subtracting from the gross count. Each sample was counted for 36,000 s to reduce the statistical uncertainty. The activity concentrations of the samples were determined using the net area under the photopeaks using Equation (1)

$$A_C = \frac{C_n}{P_\gamma M \varepsilon} \quad (1)$$

where A_c is the activity concentration of the radionuclide in the sample given in Bqkg^{-1}, C_n is the net count rate under the corresponding peak, P_γ is the absolute transition probability of the specific γ-ray, M is the mass of the sample (kg) and ε is the detector efficiency at the specific γ-ray energy.

The 1764 KeV γ-line of ^{214}Bi for ^{238}U was used in the assessment of the activity concentration of ^{238}U while 2614.5 KeV γ-line of ^{208}Tl was used for investigating the activity concentration of ^{232}Th. The single 1460 KeV γ-line of ^{40}K was used to determine the concentration of ^{40}K in the samples.

3.2. *Absorbed dose rate and annual effective dose determination*

The average absorbed dose rate in air 1 m from terrestrial sources of gamma radiation in the soil samples were estimated from the results of the activity concentration of the radionuclides determined. The corresponding values relative

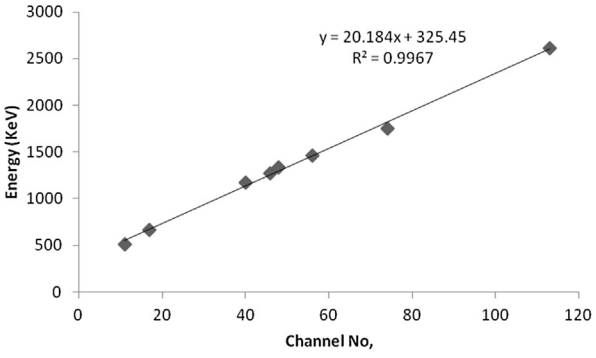

Fig. 2 – Graph of the energy-channel dependence.

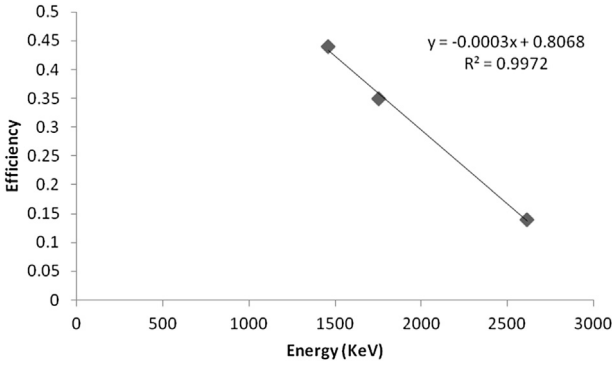

Fig. 3 – The efficiency calibration curve for the Nai(Tl) detector.

to the different types of soil are calculated using the relation given in (UNSCEAR, 1993).

$$D = 0.427A_U + 0.662A_{Th} + 0.043A_K \quad (2)$$

where D is the dose rate (nGy h^{-1}) at 1 m above the ground due to ^{238}U, ^{232}Th and ^{40}K in the soil samples. A_U, A_{Th} and A_K are the activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K in Bqkg^{-1}, respectively.

The annual effective dose due to the natural radionuclides in the soil samples was estimated using the dose conversion coefficient that convert the absorbed dose rate in air to the effective dose (0.7 Sv Gy^{-1}) and the outdoor occupancy of 0.2 (average of 4.8 h spent in the mining site everyday for a year) as proposed by (UNSCEAR, 2000). The annual effective dose rate was calculated using the formula given by (Jibiri & Adewuyi, 2008).

$$E = TfQD\xi \quad (3)$$

where E is the effective dose rate (μSv y^{-1}), T is the time in seconds in a year (8760), f is the occupancy factor which corrects the average time spent outdoors in the sites (0.2), Q is the quotient of the effective dose rate and absorbed dose rate in air (0.7 Sv Gy^{-1}), ξ is the factor converting nano (10^{-9}) into micro (10^{-6}); and D is the absorbed dose rate in air (nGy h^{-1}).

3.3. *Determination of radiation hazards.*

3.3.1. *Radium equivalent dose*

To assess the gamma radiation hazards to human associated with the use of the soil from the mining sites in construction of houses (filling and local brick making), radium equivalent activity was calculated. This gives a single index which describes the gamma output from different mixture of ^{238}U, ^{232}Th and ^{40}K in the samples. Radium equivalent activity (Raeq) is expressed mathematically by (Berekta & Mathew, 1985; UNSCEAR, 2000).

$$Ra_{eq}\,(Bq\ kg^{-1}) = A_{Ra} + 1.43A_{Th} + 0.077A_K \quad (4)$$

where A_{Ra}, A_{Th} and A_K are the activity concentration of ^{226}Ra, ^{232}Th and ^{40}K, respectively.

It may be noted that ^{238}U is replaced by the decay product ^{226}Ra, although there may be disequilibrium between ^{238}U and ^{226}Ra. It is given as 1.03 by UNSCEAR (2000).

Table1 – Average activity concentrations for uranium, thorium and potassium in the soil samples from the mining and living areas in Itagunmodi.

Locations	^{238}U (Bqkg^{-1})	Range	^{232}Th (Bqkg^{-1})	Range	^{40}K (Bqkg^{-1})	Range
A	50.8 ± 1.6	18.5 ± 2.3 – 80.5 ± 1.3	29.0 ± 2.1	21.2 ± 3.5 – 52.4 ± 7.1	389.3 ± 16.1	200.5 ± 23.1 – 560.2 ± 54.6
B	60.4 ± 4.6	23.4 ± 2.5 – 90.3 ± 3.4	18.2 ± 2.5	12.5 ± 1.5 – 32.1 ± 5.5	600.1 ± 3.4	454.2 ± 90.5 – 901.2 ± 28.2
C	51.0 ± 1.4	38.2 ± 9.2 – 65.1 ± 10.1	30.8 ± 3.1	18.2 ± 3.4 – 38.8 ± 3.5	490.6 ± 6.2	372.8 ± 41.2 –560.2 ± 15.3
D	58.9 ± 6.8	28.6 ± 3.5 – 8.5 ± 17.5	27.6 ± 2.8	15.5 ± 2.3 – 40.5 ± 5.8	540.2 ± 5.1	398.6 ± 50.2 – 610.3 ± 20.1
Average (mining site)	55.3 ± 1.2		26.4 ± 2.7		505.1 ± 7.1	
E Average (living area)	8.8 ± 1.9	6.0 ± 0.4 – 22.5 ± 3.8	17.5 ± 2.7	9.1 ± 2.4 – 38.5 ± 6.1	102.8 ± 12.1	60.5 ± 2.2 – 180.5 ± 16.0

3.3.2. *External hazard index*

To limit the external gamma-radiation dose from building materials, an extensively used hazard index, the external hazard index (H_{ex}) was calculated from the equation according to reference (Berekta & Mathew, 1985)

$$H_{ex} = \frac{A_{Ra}}{370} + \frac{A_{Th}}{259} + \frac{A_K}{4810} \leq 1 \quad (5)$$

where H_{ex} is the external hazard index and A_{Ra}, A_{Th} and A_K are the activity concentration of ^{226}Ra, ^{232}Th and ^{40}K, respectively.

3.3.3. *Internal hazard index*

Radon and its short-lived products are also hazardous to the respiratory organs. So internal exposure to radon and its short-lived products is quantified by internal hazard index and is expressed mathematically by (Berekta & Mathew, 1985) in Equation (6)

$$H_{in} = \frac{A_{Ra}}{185} + \frac{A_{Th}}{259} + \frac{A_k}{4810} \leq 1 \quad (6)$$

where H_{in} is the internal hazard index and A_{Ra}, A_{Th} and A_K are the activity concentration of ^{226}Ra, ^{232}Th and ^{40}K, respectively.

3.3.4. *Representative gamma index*

According to the EC, gamma activity concentration index ($I_{\gamma r}$) is derived for identifying whether a dose standard is met (EC, 1999) and it is estimated from Equation (7)

$$I_{\gamma r} = \frac{A_{Ra}}{150} + \frac{A_{Th}}{100} + \frac{A_K}{1500} \quad (7)$$

3.3.5. *Annual gonadal equivalent dose*

According to reference (UNSCEAR, 1982) the gonads, the active bone marrow and the bone surface cells are considered as the organs of interest. Therefore the Annual Gonadal Equivalent Dose (AGED μSv y^{-1}) for the residents of the study area due to the specific activities of ^{226}Ra, ^{232}Th and ^{40}K was calculated using Equation (8) given by (Arafa, 2004) as:

$$\text{AGED } (\mu\text{Sv y}^{-1}) = 3.09A_{Ra} + 4.18A_{Th} + 0.314A_K \quad (8)$$

where A_{Ra}, A_{Th} and A_K are the activity concentration of ^{226}Ra, ^{232}Th and ^{40}K, respectively.

4. Results

Activity concentrations of ^{238}U, ^{232}Th and ^{40}K were determined in sixty soil samples from five divisions in Itagunmodi. Table 1 presents the results of the mean activity concentrations of the investigated soil samples in the division A–E. Using Equation (2), the absorbed dose rates was estimated and is presented in column 1 in Table 2. We assumed here that the contribution from other naturally occurring radionuclides and cosmic radiation at the locations were insignificant. The average absorbed dose rate in the mining site and the living were respectively 66.3 ± 4.1 and 20.4 ± 2.1 nGyh^{-1}. By using Equation (3), the annual effective dose was calculated and the average in the mining and living areas is presented in column 2 in Table 2. Table 3 presents the calculated radiation hazards in the representative soil samples. Using Equations (4)–(7), radium equivalent activity (Ra_{eq}), external hazard index (H_{ex}), internal hazard index (H_{in}) and representative gamma index ($I_{\gamma r}$) were respectively, calculated and their means were presented in column 1–4 in Table 3. Column 5 in Table 3 presented the mean of the calculated annual gonadal equivalent dose to the residents of the study area due to the specific activities of ^{226}Ra, ^{232}Th and ^{40}K and was calculated using Equation (8). The average AGED in the study area was respectively, 439.73 and 132.62 in the mining and living areas in the study area.

5. Discussion

Radionuclide activity concentrations in the soil samples varied within the study area due to the differences in geological structure (Fig. 5). From Fig. 5, the highest activity concentration of ^{238}U was detected in location B (90.3 ± 3.4), the highest activity concentration of ^{232}Th was detected in location A (52.4 ± 7.1 Bqkg^{-1}) and the highest activity concentration of ^{40}K (901.2) was detected in location B. From Table 1, the average activity concentration of ^{238}U in the mining site was

Table 2 – Mean absorbed dose rate (*D*) and effective dose (*E*) estimated for the mining and living areas in Itagunmodi.

Locations	*D* (nGyh^{-1})	*E* (μSv y^{-1})
A	60.5 ± 6.1	74.2 ± 3.8
B	67.8 ± 4.2	83.1 ± 5.1
C	66.7 ± 6.3	81.8 ± 6.7
D	70.1 ± 3.5	86.0 ± 4.5
Mean (mining areas)	66.3 ± 4.1	81.3 ± 5.0
E (mean living areas)	20.4 ± 2.1	25.1 ± 1.5

Table 3 – Radium equivalent (Ra_{eq}), the external hazard (H_{ex}), the internal hazard index (H_{in}) and gamma index ($I_{\gamma r}$) estimated for the mining areas and the normal living areas in Itagunmodi.

Locations	Ra_{eq} (Bqkg^{-1})	H_{ex}	H_{in}	$I_{\gamma r}$	AGED (μSvy^{-1})
A	122.94	0.33	0.47	0.89	400.41
B	132.62	0.36	0.52	0.98	451.12
C	132.81	0.36	0.50	0.98	440.41
D	139.94	0.38	0.54	1.03	466.99
Mean (mining areas)	132.14	0.36	0.51	0.97	439.73
E (mean living areas)	31.75	0.11	0.14	0.33	132.62

55.3 ± 1.2 while in the living area; it was 8.8 ± 1.9 Bqkg^{-1}. For ^{232}Th, the average activity concentration was 26.4 ± 2.7 and 17.5 ± 2.7 Bqkg^{-1} in the mining site and living area, respectively. The average activity concentration of ^{40}K in the mining site was calculated to be 505.1 ± 7.1 Bqkg^{-1} while in the living area; it was 102.8 ± 12.1 Bqkg^{-1}. Activity concentrations of radionuclides are generally higher in the mining site (Location A–D) than in the normal living area (Location E) in Fig. 5. This may be due to the high density dust generated from the mining processes and other related practices which may raise the possibility of exposure to NORMs and radon gas. The activity concentrations of ^{238}U and ^{40}K in the mining site location (A–D) were higher than the world average of 35 and 420 Bqkg^{-1}, respectively while the value of ^{232}Th was lower than the world average of 45 Bqkg^{-1} (UNSCEAR, 2000). In location E, all the values obtained were lower than the world average (UNSCEAR, 2000). Fig. 4 shows the portion of the X-ray spectrum of the NaI (Tl) detector used.

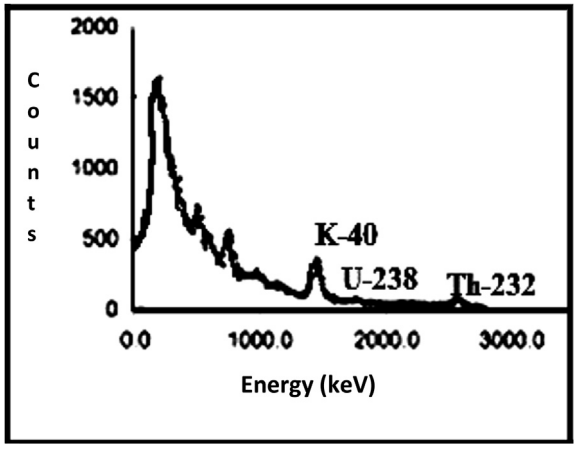

Fig. 4 – The portion of the gamma-ray spectrum of sample 5 from location E.

The average absorbed dose rate in the mining site location A–D were 60.5 ± 6.1; 67.8 ± 4.2; 66.7 ± 6.3 and 70.1 ± 3.5 nGyh^{-1}, respectively (Table 2). The total average absorbed dose rate in the mining site was 66.3 ± 4.1 nGy h^{-1} while in the living area; it was 20.4 ± 2.1 nGy h^{-1}. The absorbed dose rate calculated for the mining site is higher than the world average of 60 nGy h^{-1} (UNSCEAR, 2000). The annual effective dose for the study area due to the soil activity was estimated using the dose conversion coefficient that convert the absorbed dose rate in air to the effective dose (0.7 Sv Gy^{-1}) and the outdoor occupancy of 0.2 (average of 4.8 h spent in the mining site everyday for a year) as proposed (UNSCEAR, 2000). The average annual effective doses in location A–D in the mining site were respectively, 74.2 ± 3.8, 83.1 ± 5.1, 81.8 ± 6.7 and 86.0 ± 5.0 μSvy^{-1} with the total average in the mining site as 84.3 ± 5.0 μSvy^{-1}. The average annual effective dose in the living area was 25.1 ± 1.5 μSvy^{-1}.

The (Ra_{eq}) in the study area as presented in Table 3 shows that the average values obtained in the mining areas location A, B, C and D and in the normal living place location E were lower than the suggested maximal permissible value of 370 Bqkg^{-1} (UNSCEAR, 1982) for material that will be used in building of dwellings. The estimated average values of H_{ex} and H_{in} (0.11 and 0.14) in the study area were lower than unity as desirable. The average values of $I_{\gamma r}$ obtained were below the

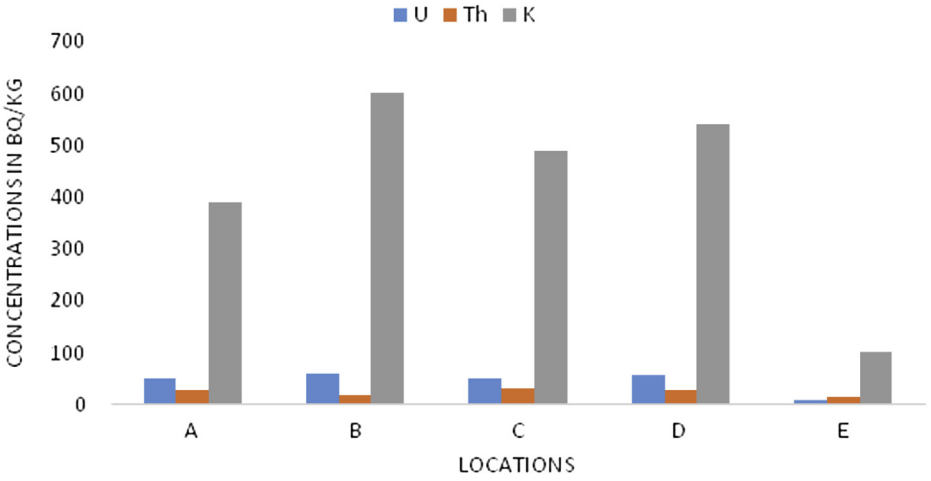

Fig. 5 – Comparison of activity concentrations of ^{238}U, ^{232}Th and ^{40}K in the study locations.

criterion of unity in locations A, B, C and E but the total average in the mining site and in the living area were below the criterion of unity corresponding to an annual effective dose of 0.3 mSv (EC, 1999). The average value of AGED obtained in the living area was 132.62 μSv y^{-1}. This value is lower than the world average of 300 μSv y^{-1} (UNSCEAR, 2000). The average value of AGED in the mining site for locations A–D are 400.41, 451.12, 440.41 and 466.99 μSv y^{-1}, respectively. The total average value of AGED in mining site was 439.73 μSv y^{-1} which is higher than the world average.

Comparison of the results obtained in the mining site with published data from similar investigations in other parts of the country and the world average were presented in Table 4. Lower activity concentration was determined by Ademola and Ademonehin (2010), Ademola and Obed (2012) and Innocent et al. (2013); in Nigeria' Berekta and Mathew (1985) in Australia, Amrani and Tahbat (2001) in Algeria and Faanu, Darko, and Ephraim (2011) in Ghana for ^{238}U compared to this study but lower than the concentration obtained in Brazil (Manlanca, Pessina, & Dallara, 1993) and in Egypt (El Afifi, Hilal, Khalifa, & Aly, 2006). The activity concentration of ^{238}U estimated in this study is higher than the world average (UNSCEAR, 2000). The average activity concentration of ^{232}Th obtained in this study is lower than that obtained in Nigeria by Ademola and Ademonehin (2010) and Innocent et al. (2013). ^{232}Th concentration in the study site is also lower than what is obtained by Amrani and Tahbat (2001), Berekta and Mathew (1985), El Afifi et al. (2006), and Manlanca et al. (1993) but higher than that obtained by Ademola and Obed (2012) and Faanu et al. (2011). The average activity concentration of ^{232}Th is lower than the world average (UNSCEAR, 2000). Similar to what was obtained in this study, other studies as can be seen in Table 4 also indicated that concentrations of ^{40}K in soil samples are significantly higher than the concentration of ^{238}U and ^{232}Th. The average activity concentration of ^{40}K in this study is higher than the world average. The mean average annual effective dose obtained in the mining areas is 81.3 μSv y^{-1}. The annual effective dose obtained is about 16% higher than the world average of 70 μSv y^{-1} (UNSCEAR, 2000) but lower than the Nigeria average of 98 μSv (Farai & Jibiri, 2000)

Table 4 – Comparison of the results of the present study with similar published data.

References	Country	^{238}U (Bqkg^{-1})	^{232}Th (Bqkg^{-1})	^{40}K (Bqkg^{-1})
Present study	Nigeria	55.3	26.4	505.1
Ademola and Obed, 2012	Nigeria	39.8	17.7	384.2
Ademola & Ademonehin, 2010	Nigeria	13.3	40.0	240.2
Innocent et al., 2013	Nigeria	12.1	60.1	426.5
El Afifi et al., 2006	Egypt	78	33	337
Berekta & Mathew, 1985	Australia	51.5	48.1	114.7
Amrani and Tahtat, 2001	Algeria	41	27	422
Manlanca et al., 1993	Brazil	61.7	58.5	564
Faanu et al., 2011	Ghana	13.6	24.2	162.1
UNSCEAR, 2000	World Average	35	30	420

6. Conclusion

The activity concentrations of ^{238}U, ^{232}Th and ^{40}K in soil samples from Itagunmodi area have been studied using NaI (Tl) gamma ray spectrometry. The results obtained showed that the distribution of natural radionuclides in the soil samples was not uniform and artificial radionuclide was not detected in any sample measured.

The mean activity concentrations of ^{238}U, ^{232}Th and ^{40}K in the soil samples from the mining sites were estimated to be 55.3, 26.4 and 505.1 Bqkg^{-1}, respectively while in the living areas it was 8.8, 17.5 and 102.8 Bqkg^{-1}, respectively. The results of the mean activity concentrations of ^{238}U and ^{40}K in the mining sites are higher than the world average. The mean effective dose in this study is about 16% higher when compared the world average (UNSCEAR, 2000). The activity concentrations of ^{238}U, ^{232}Th and ^{40}K and the effective dose in the normal living areas is lower than the world average.

The results in this study area show that there are high levels of the natural radionuclides in the mining areas than the worldwide average but lower than the Nigeria average of 98μSv (Farai and Jibiri, 2000). Therefore, mining activities in Itagunmodi poses no significant radiological hazard to the host communities.

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