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Environmental and Health Impacts: Presence of Radionuclides and Toxic Metals in Mining Areas in Niger State, Nigeria

ABSTRACT

The distribution of radionuclides and heavy metals (carcinogenic and non-carcinogenic) in soil and plants grown in lead contaminated areas were determined using gamma spectrometry and AAS (Atomic Absorption Spectrometry). The values of D_{out} (outdoor dose rate) and D_{in} (indoor dose rate) were found to vary from 49.10 to 96.59 nGy h^{-1} with a mean value of 73.75 nGy h^{-1} and 97.30 to 191.67 nGy h^{-1} with a mean value of 143.39 nGy h^{-1} for Kawo; 51.64 to 105.26 nGy h^{-1} with a mean value of 75.36 nGy h^{-1} and 104.43 to 205. 87 nGy h^{-1} with a mean value of 157.13 nGy h^{-1} for Magiro. The weighted mean concentrations of Pb, Cu, Fe, Cr, Cd and Ni in mg kg^{-1} in soil and crop were (2.20,2.29), (3.14, 0.16), (15.10, 19.33), (0.16, 0.25), (0.24, 0.05), (0.43, 0.06) and (3.00, 3.18), (6.09, 0.34), (16.83, 10.10), (0.22,0.36), (0.25, 0.05), (0.43, 0.06) for Kawo and Magiro respectively. The total incremental lifetime cancer risk ∑ILCR for both adults (0.73 x 10^{-2}) and children (1.03 x 10^{-2}) were above tolerable acceptable risk (given as 1 x 10^{-6} to 1 x 10^{-4}). Therefore, the areas need remedial action and proper monitoring.

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INTRODUCTION

Lead, Chromium, Nickel, Copper, Iron and Cadmium are naturally occurring poisonous metal discovered in the Earth's crust. According to *WHO* (*2015*), its extensive use has resulted in widespread environmental pollution, human exposure and important public health complications in many parts of the world. Human exposure to metals, metalloids and nitrogen is one of the chief consequential issues for public health established on the fact that they are related with polluted groundwater and soil (*Aelion and Davis 2007*; *WHO 2011*) as seen in Kawo and Magiro, in Rafi Local Government of Niger State, Nigeria. Lead was ascertained to be in higher concentration in the blood of affected children in Kawo and Magiro. Soils' composition and water content can be greatly altered in agitated soils (*Mummey 2002*) through leaching, eluviation and illuviation leading to change in the plants in the mining areas. Plants grown in contaminated soil in mining areas tend to build up radionuclides and heavy metals in their various organs; perhaps leading to human exposure when fruits, seeds, leave, stems and roots are being ingested. Young children are more susceptible to the harmful effects of lead and can experience severe, long-lasting health problems, especially those that impact the growth of the brain and neurological system. Adults who consume lead run the risk of developing chronic renal disease and high blood pressure. According to the World Health Organization, pregnant women who are exposed to high amounts of lead can experience miscarriage, stillbirth, early birth, low birth weight, as well as minor abnormalities (*WHO 2015*). Regular use could result in health issues brought on by prolonged metal exposure (*Jung and Thornton 1996*) as seen in the aforementioned communities where gold mining practice has been on for more than 50 years and where 28 children perished between April and May 2015 due to exceptionally high lead levels of 17 to 22 times (171.5-224 g Pb/dl) in their bloodstream following consumption of lead-tainted water (*WHO 2015*).

Disadvantageous effects of mining on environs and surroundings includes erosion, extermination and disruption of ecosystems and habitats, emergence of sinkholes, mislaying of diversity (*Niyogi et al. 2002; Steinhauser et al. 2009; Tarras-Wahlberga et al. 2001*), contamination of soil, contamination of groundwater, surface water, rivers by elements and chemicals, such as arsenic, mercury, Sulphuric acid, gold cyanide and gold chlorine which are poisonous to kidney, liver and other vital organs of the body including fetal (*WHO 2011; Jadhav et al. 2015; IARC 1993*). These are chemical and elements that are available during the processing and extraction of gold. The squashing of rocks into sand in a flour mill gives rise to a great quantity of dust which is extremely polluted with lead (*MSF 2012*) and it turn leads to dust contamination, visual pollution and other forms of pollution. The usual signs revealed by affected persons in the communities are fever, pallor, abdominal pain, vomiting, convulsion, vision impairments and varied level of consciousness (*WHO 2015*).

The inhabitants of the two communities had engaged in illegal gold mining for the past 50 yrs (*The Guardian 2015*). Besides harming the environment, pollution brought by chemical spills from mining sites also affect the health of the local population. Due to the fact that mining activities have significant consequences on the environment and public health, some nations compel the mining industry to adhere to environmental impact assessment standards, such as environmental and rehabilitation rules and restoring mined areas to their pre-mining conditions. However, these laws and by-laws are violated in the two communities as the inhabitants of the area engaged in illegal gold mining. Consequently, this research was done to measure the amounts of radionuclides in soil and heavy metals in selected crops in the areas.

The study examined radioactivity contents in soil in the mining areas; estimated the radiological impact due to long-term exposure (50 yrs); ascertained whether mining activities have caused a rise in soil radioactivity; and determined the concentration of heavy metals in crops and then estimated the concentration accumulation index.

MATERIALS AND METHODS

Study Area

The two settlements, namely Kawo and Magiro are situated in Kagara Districts in Rafi Local Government Area in Niger State with coordinates 10° 11′ 04″N 6° 15′12″E (**Figure 1**) in Nigeria. There are dry and rainy seasons in Niger State, and the yearly rainfall ranges from 1,100 mm (North) to 1,600 mm (South). The highest temperature in the state is 34.4°C. The rainy season varies between 120 and 150 days. The rich soil and hydrology of the areas allows the cultivation of most Nigeria's principal crops (*NSGN 2016*). Agriculture and gold mining are the main livlihood of the residents. Because of the previously mentioned geographic and climatic factors, agriculture is the foundation of the populace's economy. Rafi Local Government Area is flanked in the south by Kaduna River. The inhabitants, time zone, area and density of Rafi are 181,929 (*NPC 2006*), Western African Time (UTC+1), 3,680 km^2 and 71.06 km^2. According to the Nigeria Population Commission, Rafi's population and age distribution are 0-9 yrs (67,862); 10-19 yrs (34,467); 20-29 yrs (30,731); 30-39 yrs (20,367); 40-49 yrs (12,449); 50-59 yrs (6,712); 60-69 yrs (3,657); 70-79 yrs (2,454) and 80+yrs (2,419) (*NPC 2006*). This suggests that 58% of the population is made up of infants and kids between the ages of 0 and 19. For almost 50 years, the local miners have been involved in illegal mining operations.

Sampling

Soil samples were gathered in the affected communities after the lead contamination in 2015. To create a composite sample, roughly four subsamples at a depth of 15-20 cm were combined. Four conglomerate samples were formed in ten plots from each settlement. Each plot is roughly 200 m apart from the others. In total, there are 80 samples collected at dimension 1.5 m x 1.5 m. Using a coring equipment, samples were taken at sampling locations sampling locations following health and safety protocols. The composite samples inside polythene bags were taken to the laboratory for further treatment. The mean concentration from sampling sites was then estimated after counting and was recorded to represent the concentration for that particular plot. Ten soil samples were collected from undisturbed areas (distance 5 km). Edible parts (grains) of *Sorghum bicolor* (L.) Muench grown in the areas were collected in triplicate in a distance (20 m) not far from each plot.

Analysis of radionuclides in the soil

Each soil sample was fully dried at room temperature to achieve uniform weight at 25 ± 2°C to remove the water. In addition, the samples were dried in an oven at 105°C (*Alan et al. 1997*) for about 24 hrs and sieved through a 2 mm mesh to remove stones, pebbles, roots and other impurities. The self-absorption in each of the samples was taken care of by the elimination of moisture. To increase the total emission area, the dry materials were

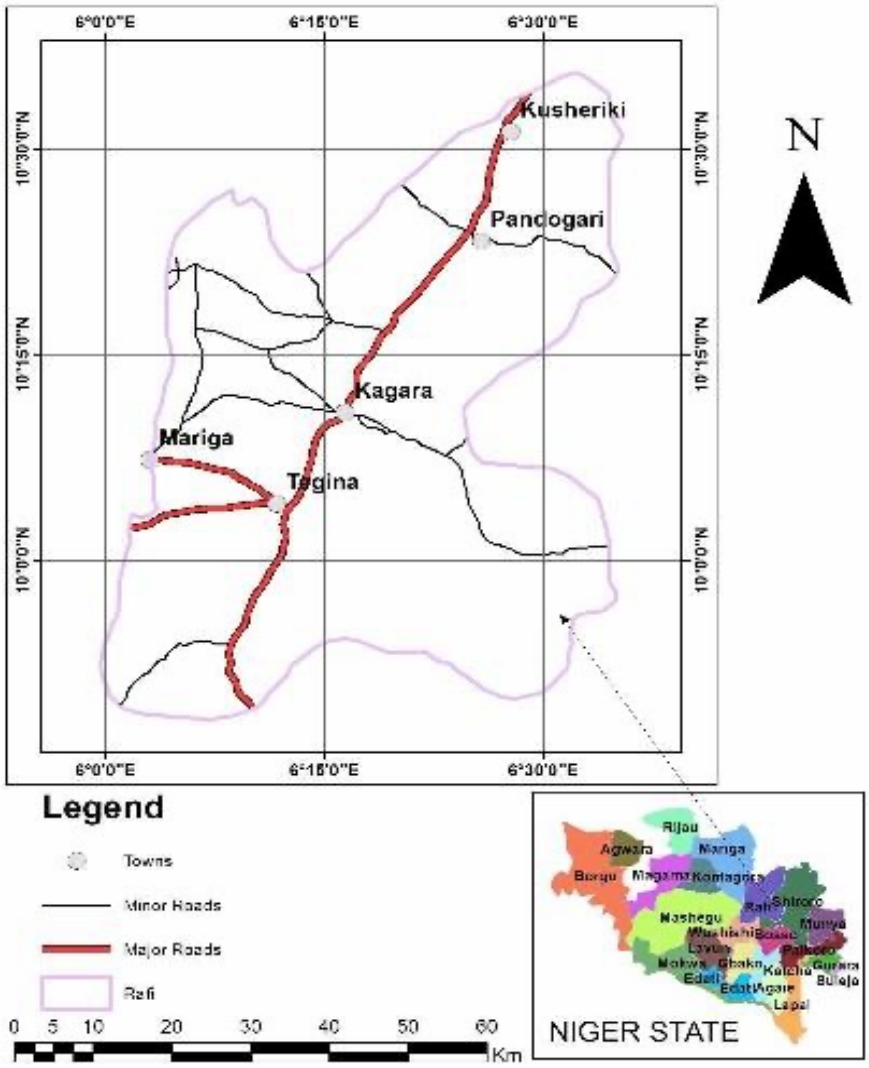

Figure 1: Sampling sites in Rafi Local Government, Nigeria

ground into fine grains (*Papp et al. 2002*). They were then placed in a PVC container at a mass of 250 g. The inner lid was placed in and closed tightly with the outer cap. The container was sealed hermetically and externally using a cellophane tape and kept aside four weeks prior to gamma measurement. This is undertaken in order to permit the ingrowth of ^{238}U and ^{232}Th decay products and accomplishment of secular stability of ^{226}Ra and ^{222}Rn with their corresponding offspring. Activity concentration measurement involves either α or β or γ radiations from samples (*Murugesan et al. 2011*). Due to intrinsic properties of γ ray such as high penetrating power when it interacts with matter, γ radiation measurement provides more valuable fact than α and β radiations. Consequently, the specific activities counting in both contaminated and controlled soil samples were done by gamma-ray spectrometry, utilizing a NaI (Tl) (by Canberra, Inc. USA) detector directly linked to a pre-amplifier, a computer- powered multichannel analyser (MCA). Radium content of samples was estimated from intensity of 1.76 MeV energy that belonged to the absorption of ^{214}Bi corresponding to ^{238}U series that is used to find and measure natural uranium. Thorium concentration was found by the ^{208}Tl peak which belonged to the 2.61 MeV absorption energy. Potassium content was found and measured by means of the absorption of the 1.46 MeV energy that belonged to decay of 40K. Every sample's spectrum was gathered for 15 hours. Lead was used as a barrier on the detector to lessen the background influence (*Faweya et al. 2013*).

Analysis of Heavy Metals in Soil and Crop

The permeability of radionuclides in the soil and their readiness to plants rely on mineralogy constitution, soil makeup, pH, organic matter content and cation constitution of soil solution, Ca and K concentrations (*Faweya et al. 2017*). Therefore, soil and crop samples were analysed for physico-chemical properties. standard procedures by International Soil References

and Information Centre and Food and Agricultural Organization were adopted (*ISRIC/FAO 2002*). The solution was then subjected to Atomic Absorption Spectrophotometer analysis BUCK 210 VGP (*Faweya and Adewumi 2021*).

Calculation of Radiation Hazard Parameters

Calculation of Ra_{eq} and percentage contribution of radionuclides to Ra_{eq}. Since the soil in the areas are used in building construction and the spreading of radionuclides ^{226}Ra, ^{232}Th and 40K in soils and other building materials is not the same throughout the world. Therefore, the radium equivalent was generally introduced based on the presumption that 10 Bq kg^{-1} of ^{226}Ra, 7 Bq kg^{-1} of ^{232}Th and 130 Bq kg^{-1} of 40K will bring out uniform dose rates of gamma rays. With the use of the radium idea, the gamma output from several locations can be described by a single index or number. Values of Raeq in the area were estimated using equation 1 (*Beretka and Matthew 1985; Abd El Raham et al. 2022*):

$$Ra_{eq}=A_{Ra}+1.43A_{Th}+0.077A_{K} \quad (1)$$

Where A_{Ra}, A_{Th}, A_{K} are the specific activity concentrations of ^{226}Ra, ^{232}Th and 40K in Bq kg^{-1}, respectively.

The contribution in % of radionuclides to radium equivalent was estimated using the equation 2:

$$\frac{\eta}{100}=(C_i)f_i.\frac{370}{ERa_{eq}} \quad (2)$$

Where C_i (i=*K,Ra and Th*) is the concentration of radionuclides ^{40}K, ^{226}Ra and ^{232}Th, respectively, f_i (i=*K,Ra and Th*) was the multiplication factors 2.08 x 10^{-4}, 2.7 x 10^{-3} and 3.86 x 10^{-3} for ^{40}K, ^{226}Ra and ^{232}Th respectively, ERa_{eq} is the calculated value of radium equivalent.

Absorbed gamma dose rate (D). Both radiation sources and radiation shields are provided by the building materials, such as soil (*UNSCEAR 1988*). The impact of primordial radionuclides from the soil in the areas to the absorbed dose in air D relies on the primordial specific activity of ^{40}K, ^{226}Ra and ^{232}Th. The largest portion of the gamma radiation comes from terrestrial radionuclides. Therefore, the mean absorbed dose rate in air was estimated. The actual values relative to soil are computed using equation 3 (*UNSCEAR 2000*):

$$D_{out}(n\ Gy\ hr^{-1})=A_i x C_{Fi} \quad (3)$$

Where D is the dose in nGy hr^{-1}, A_i is the activity concentration in Bq kg^{-1}, i.e. A_{Ra}, A_{Th} and A_K respectively, and C_{Fi}(1 = *Ra,Th and K*) is the dose conversion factors in units of nGy hr^{-1} per Bq kg^{-1}. The dose conversion factors as taken from UNSCEAR 2000 report based on the Monte Carlo technique are C_{fRa}(0:462); C_{fTh}(0:604) and C_{fK}(0:0417) for ^{226}Ra, ^{232}Th and ^{40}K, respectively.

The contribution of primordial radionuclides to the absorbed dose rate in indoor air (Din nGy h^{-1}) relies on the activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K (*Kurnaz 2007*). This was determined 1 m beyond the ground's surface for a room of 4 m x 5 m x 2.8 m in a typical masonry construction built using soil samples from the mines in the impacted areas (*UNSCEAR 1993; Markkanen 1995; European Union 1999; Papastefanou et al. 2005*) as follows (equation 4):

$$D_{in}=0.908A_{Ra}+1.06A_{Th}+0.0767A_{K} \quad (4)$$

Where A_{Ra}, A_{Th}, A_{K} are in Bq kg^{-1}. This relation takes into account a wall thickness of 20 cm and a structure's density of 2350 kg m^{-3} (*Turhan and Gunduz 2008*). The absorbed dose rate conversion coefficients of the three radionuclides are used to express the respective contributions to the absorbed dose rate in air from their activity concentrations, are 0.908 nGy h^{-1} per Bq kg^{-1} for ^{226}Ra, 1.06 nGy h^{-1} per Bq kg^{-1} for ^{232}Th and 0.0767 nGy h^{-1} per Bq kg^{-1} for ^{40}K.

Annual effective dose rate (Eout and Ein) and annual gonadal dose equivalent. The Eout and Ein are estimated for an adult person such as miner, farmer and residents, the outdoor and indoor dose rates were translated to effective dose rates using equations 5 and 6 (*UNSCEAR 2000*):

$$E_{out}\ (mSv\ yr^{-1})= D_{out} \times 1.21 \times 10^{-3} \quad (5)$$

$$E_{in}\ (mSv\ yr^{-1})= D_{in} \times 4.91 \times 10^{-3} \quad (6)$$

Where E_{out} and $_{\text{Ein}}$ are annual effective dose rates, D_{out} and D_{in} are the dose rates

Annual gonadal dose equivalent was calculated as a measure of the genetic importance of the yearly equivalent received by the population's reproductive organs (*Morsy et al. 2012*). In similar manner, the mean activities of ^{226}Ra, ^{232}Th and ^{40}K were used to estimate the yearly gonadal dose equivalent (AGDE) for the miners, farmers and residents in the areas using bone marrow activity and bone surface cells as the organs of interest by the following equation (7) (*UNSCEAR 1988*):

$$AGDE(\mu Sv\ y^{-1}) = 3.09A_{Ra} + 4.18A_{Th} + 0.314\ A_K \quad (7)$$

External hazard index (Hext) and internal hazard index (Hin). The external hazard index is another frequently used hazard index that accounts for external exposure. Hext is obtained from the radium equivalent equation by assuming that its highest value allowed which is unit belongs to the upper limit of Ra_{eq} (370 Bq kg^{-1}). H_{ext} is defined as follows (*UNSCEAR 2000*) in equation 8:

$$H_{ext} = \left(\frac{A_{Ra}}{370}\right) + \left(\frac{A_{Th}}{259}\right) + \left(\frac{A_K}{4810}\right) \leq 1 \quad (8)$$

In addition to external hazard index H_{ext}, radon and its short-lived progenies are also detrimental to the breathing organs of miners, residents and farmers. The indoor exposure to radon and its offspring is measured by the internal hazard index H_{in} which is calculated using equation 9:

$$H_{in} = \left(\frac{A_{Ra}}{185}\right) + \left(\frac{A_{Th}}{259}\right) + \left(\frac{A_K}{4810}\right) \leq 1 \quad (9)$$

Exhaled radon in soil and estimated ^{226}Ra from the soil (Ra_{FZ}). In addition to both external and internal hazard indices, the following formula was used to determine the risk component associated with exhaled radon in soil samples (*Szabó et al. 2013*) (equation 10):

$$H_R\% = 100x\left(\frac{H_{in}}{H_{ext}} - 1\right) \quad (10)$$

H_R is the exhaled radon in % showing the percentage of risk posed by radon and its alpha decay products (*Szabó et al. 2013*), H_{in} and H_{ex} are internal and external hazard indices. Excess ^{226}Ra which could be found in soils is often calculated using equation 11 (*Mohannad and Khalil 2014; Faweya et al. 2017*):

$$Ra_{FZ}(Bq\ kg^{-1}) = A_{Ra} - A_{Th} \times \left(\frac{U}{Th_N}\right) \times A \quad (11)$$

Where A_{Ra} and A_{Th} are the activity concentrations of ^{226}Ra and ^{232}Th in the contaminated soil, $\left(\frac{U}{Th_N}\right)$ is the mean value of the primordially observed $\left(\frac{U}{Th_N}\right)$ concentration ratio 0.23, and A is a conversion factor from ^{238}U concentration to ^{226}Ra radioactivity since 1 mg kg^{-1} of ^{238}U is the same as 12.3 Bq kg^{-1} of ^{226}Ra when the two isotopes achieved secular equilibrium.

Activity utilization index (AUI). Activity Utilization Index (AUI) is typically calculated in radiological health hazard survey and assessment studies to determine the dose rates in air from various combinations of the radionuclides present in soils. In Kawo and Magiro soil, equation12 provided it (*Ramasamy et al. 2011; Faweya and Adewumi 2021*):

$$AUI = \left(\frac{A_{Ra}}{50Bq\ kg^{-1}}\right)F_{Ra} + \left(\frac{A_{Th}}{50Bq\ kg^{-1}}\right)F_{Th} + \left(\frac{A_K}{500Bq\ kg^{-1}}\right)F_K \quad (12)$$

where A_{Ra}; A_{Th} and A_K are the activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K in contaminated soils, respectively, F_{Ra}(0.462), F_{Th}(0.604) and F_K(0.041) are gamma radiation's proportional contributions to the air's overall dosage rate.

Excess Lifetime Cancer Risk. Long-term radiation exposure is thought to increase some of the risk of developing cancer. The long-term (50 years) exposure from the radioactivity of mining tailings and soil could cause cancer in the areas. Therefore, the excess lifetime cancer risks as a gauge for the likelihood of getting lung cancer from breathing in gaseous radionuclides indoors (*Al-Hamarneh 2017*) and outdoors recieved over the individual lifetime were calculated using equations 13 and 14:

$$Risk(outdoor) = Eout \times LE \times RF \quad (13)$$

$$Risk(indoor) = Ein \times LE \times RF \quad (14)$$

Where E_{out} and E_{in} are the annual effective doses, LE is the mean lifespan duration of 70 yrs and RF is the lethal hazard factor of 0.05 Sv^{-1} (*ICRP 2008*).

Elemental Concentrations. The activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K were estimated due to their radioactive toxicity using the following relation (IAEA 1989), 1 ppm eU(Ra)=12.35 Bq kg^{-1} of ^{238}U(^{226}Ra), 1ppm eTh=4.06 Bq kg^{-1} of ^{232}Th, and 1% K=313 Bq kg^{-1}.

Concentration accumulation index (CAI). Concentration accumulation index of the heavy metals in crops is the increase of a pollutant concentration in a direct chain from contaminated soil to the crop's edible component. It is calculated by using the concentration factor (Fc). The ratio of concentrations in contaminated soil to concentrations in plants is expressed by the concentration factor. It is calculated using equation 15 (*Faweya et al. 2017*):

$$CAI = \frac{C_c}{S_{cs}} \quad (15)$$

Where C_c and S_{cs} are concentrations of heavy metals in crop and soil, respectively.

Human Health Risk Assessment. The health hazard

constituted to inhabitants of the two villages was computed based on the average constituents of carcinogenic and non- carcinogenic metals obtained in the soils.

The computed mean daily intake ADI (mg kg^{-1} d^{-1}) of heavy metals via soil was estimated by equation 16 (*USEPA 2003*).

$$ADI_{ing} = \frac{C_s \times IR \times EF \times ED}{BW \times AT} \quad (16)$$

Where Cs is the measured heavy metals concentrations in soil, IR (20 and 50 mg d^{-1} for adults and children) is the consumption rate of soil (*USEPA 2011*); EF (350 d y^{-1}) is the exposure recurrence, ED (24 and 6 y for adults and children respectively) is the exposure duration (*USEPA 2011*), BW (65 and 29 kg for adults and children) is the body weight; AT (ED X365 day) is the average time (*Jiang et al. 2017*). Ingestion and dermal absorption play the most notable roles among the prospective exposure tracks (*Fryer et al. 2006; Qu et al. 2012*). These two exposure tracks are well known among the three exposures (*USEPA 2011*). Dermal exposure was then computed using equation 17:

$$ADI_{DER} = \frac{Cs \times SA \times AF \times ABS \times EF \times ED}{BW \times AT} \quad (17)$$

Where *SA* (5700 for adults and 2800 for children) is skin area available for interaction (*USEPA 2001*), AF is the constancy factor (2 x 10^{-7} for adults and 1 x 10^{-6} for children) (*USEPA 2011*) and ABS (0.006, 0.1, 0.04, 0.14 and 0.35) are the retention factors for Pb, Cu, Cr, Cd and Ni respectively (*USEPA 2001*). The inhalation exposure pathway was estimated using equation 18 (*Jiang et al. 2018*):

$$ADI_{inh} = \frac{Cs \times IRinh \times EF \times ED}{PEF \times BW \times AT} \quad (18)$$

Where *IRinh* (16 for adults and 7.6 m^3 d^{-1} for children) is the inhalation rate of soil (*USEPA 2011*) and PEF (1.36 x 109 m^3 kg^{-1}) is the particle emission factor (*USEPA 2001*).

Non-Carcinogenic Risk Assessment. Non-carcinogenic hazards for adults and children were estimated by computing hazard quotient HQ for ADI_{ing} and ADI_{der} respectively. It is defined as the quotient of a mean daily intake divided by the reference dose RFD. It was computed using equations 19 and 20.

$$HQ = \frac{ADI_{ing}}{RfD_0} \quad (19)$$

$$HQ = \frac{ADI_{der}}{RfD_{ABS}} \quad (20)$$

Where RfD_0 (1.4 x 10^{-4}, 4 x 10^{-2}, 1.5, 1 x 10^{-3} and 2 x 10^{-2} mg kg^{-1} d^{-1}) and RfD_{ABS} (1.4 x 10^{-4}, 4 x 10^{-2}, 1.95 x 10^{-2}, 2.5 x 10^{-5} and 8 x 10^{-4} mg kg^{-1} d^{-1}) are the reference doses and reference doses for dermal absorption exposure for Pb, Cu, Cr, Cd and Ni respectively. In order to assess the comprehensive potential for non-carcinogenic effects caused by more than one element, a Hazard Index (HI) was applied (*USEPA 1986*). It is defined as the sum of HQ and was calculated using the following equation:

$$HI = \sum HQ_i = \sum \frac{ADI_{ing\ or\ der}}{RfD_{0\ or\ ABS}} \quad (21)$$

Carcinogenic Risk Assessment. The probability of developing cancer by the resident's lifetime due to exposure to carcinogenic heavy metals was estimated using the incremental lifetime cancer risk. It was calculated using equation 22:

$$ILCR = LADI \times SF \quad (22)$$

Where *LADI* is the lifetime average daily dose and *SF* is the slope factor (0.0085, 6.3 and 0.5 per mg kg^{-1} d^{-1}) for Pb, Cd and Cr respectively. *LADI* was calculated using LT (70 x 365 d) in place of AT in equation 16 through 18.

Environmental (Ecological) Risk. The ecological risk defines the extent an eco-environment is being contaminated. It was estimated using equation 23 (*Jia et al. 2018*).

$$E_s = T_s \times \frac{Cs}{Cb} \quad (23)$$

Where T_s (30,5,5,5 and 1) are the toxic response factors for Cd, Cu, Ni, Pb and Cr respectively, *Cb* is the average shale concentration of each studied metal (*Faweya et al. 2018*).

RESULTS AND DISCUSSION

Radionuclides activities

Cesium (^{137}Cs) was not observed in any of the studied samples. Multivariate analysis was performed on the distribution of radionuclides in the areas using Edraw max 9.1 software, the concentrations of the radionuclides as seen in Spider Web Charts (**Figure 2**) are in the following order AK > ARa > ATh. The highest concentration of ^{40}K among the three radionuclides

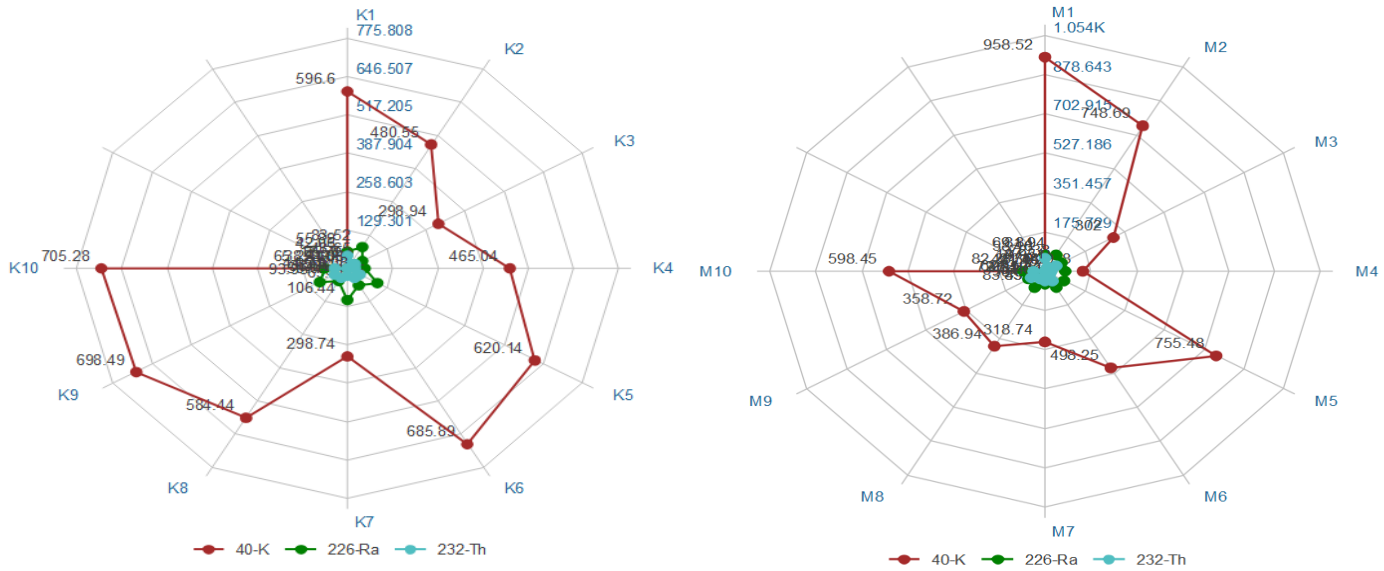

Figure 2. Distribution of radionuclides in Kawo (left) and Magiro (right)

revealed the usual occurrence in most geological medium (*Hassan et al. 2013*) and could be as a result of presence of gold minerals embellishment in the two communities. The average concentration of the primordial radionuclides showed that ^{226}Ra has the concentration greater than 50 and 60 Bq kg^{-1} recommended by United Nation Scientific Committee on Effects of Atomic Radiation and EU (*UNSCEAR 1993; European Union 1999*), while ^{40}K and ^{232}Th are within recommended values 500/640 and 50/60 Bq kg^{-1} (*UNSCEAR 1993; EU 1999*). All the soil samples showed ^{226}Ra concentration significantly higher than these recommended values except at K3, K4 and M7. This could be responsible for higher concentration of lead in the water in the areas (**Table 1** and **2**).

Correlation between two pairs of radionuclides was examined to ascertain the relationship between and the radiological parameters. This was achieved using Microsoft Office Excel 2010 and the SPSS 20.0 statistical package programme. The correlation coefficients are 0.99,

Table 1. Activity concentration of ^{226}Ra, ^{232}Th, ^{40}K, R_{aeq}, R_{aFZ}, AUI, D_{out}, D_{in}, E_{out}, E_{in}. (Kawo is represented by K).

	Activity Concentration (Bq kg^{-1})			Ra_{eq}	Ra_{FZ}	AUI	D (nGy h^{-1})		E (mSv y^{-1})	
	^{40}K±σ	^{226}Ra±σ	^{232}Th±σ	(Bq kg^{-1})	(Bq kg^{-1})		Out	In	Out	In
K1	596.60±82.51	55.89±16.41	42.06±4.12	166.61	-63.10	1.07	74.44	146.01	0.09	0.72
K2	480.55±67.11	83.52±10.12	17.58±2.51	145.66	33.77	1.02	66.98	135.01	0.08	0.70
K3	298.94±98.24	48.65±16.15	25.75±3.84	108.49	-24.22	0.79	49.10	97.30	0.06	0.50
K4	465.04±70.82	47.82±11.18	21.85±5.88	114.88	-14.02	0.74	53.33	105.70	0.06	0.52
K5	620.14±98.11	98.15±14.50	41.08±4.11	204.64	-18.15	1.45	93.26	185.72	0.11	0.91
K6	685.89±76.14	65.51±15.18	38.23±3.29	172.99	-42.68	1.12	80.06	157.88	0.10	0.77
K7	298.74±74.24	106.44±31.24	25.04±4.04	165.25	35.58	0.79	73.68	149.66	0.09	0.73
K8	584.44±75.25	50.09±16.15	38.24±6.12	149.77	-58.12	0.97	69.12	135.47	0.08	0.66
K9	698.49±98.25	93.95±15.17	44.24±4.28	210.99	-31.25	1.46	96.59	191.67	0.12	0.94
K10	705.28±84.19	65.25±10.18	38.14±3.28	174.10	-42.69	1.12	80.71	159.11	0.09	0.78
Control Mean	444.19	62.15	37.56	151.01	-44.11	1.06	70.08	84.18	0.08	0.41
Minimum	298.94	47.82	17.58	108.49	-63.10	0.74	49.10	97.30	0.06	0.50
Maximum	705.28	106.44	44.24	210.99	35.58	1.46	96.59	191.67	0.12	0.94
Mean	543.41	71.53	33.22	160.87	-22.48	1.11	73.75	146.39	0.09	0.72
Kurtosis	-0.72	0.44	-0.53							
Skewness	-0.78	-1.54	-1.52							

Table 2. Activity concentration of ^{226}Ra, ^{232}Th, ^{40}K, R_{aeq}, R_{aFZ}, AUID, D_{out}, D_{in}, E_{out}, E_{in}. (Magiro is represented by M).

	Activity Concentration (Bq kg^{-1})			Ra_{eq}	Ra_{FZ}	AUI	D (nGy h^{-1})		E (mSv y^{-1})	
	$^{40}K\pm\sigma$	$^{226}Ra\pm\sigma$	$^{232}Th\pm\sigma$	(Bq kg^{-1})	(Bq kg^{-1})		Out	In	Out	In
M1	958.52±86.68	69.84±13.15	58.16±4.84	226.82	-94.75	1.43	105.26	205.87	0.13	1.01
M2	748.69±74.18	81.95±18.16	18.95±5.25	166.70	28.32	1.05	78.30	156.88	0.09	0.77
M3	302.00±94.55	72.38±15.42	47.31±3.05	163.28	-61.51	1.27	72.49	143.09	0.09	0.70
M4	142.38±78.18	75.29±16.55	21.48±6.14	115.59	14.50	0.97	51.64	104.43	0.06	0.51
M5	755.48±68.19	84.15±11.45	23.45±2.54	175.85	17.79	1.12	82.24	164.41	0.10	0.81
M6	498.25±78.44	85.21±11.25	51.08±3.45	196.62	-59.35	1.45	88.53	174.94	0.11	0.86
M7	318.74±50.45	58.67±18.25	43.44±8.45	145.33	-64.27	1.09	64.89	127.58	0.08	0.63
M8	386.94±98.26	83.49±15.18	40.59±7.28	153.87	-31.38	1.29	78.85	152.80	0.10	0.75
M9	358.72±51.67	72.84±10.52	62.11±5.92	189.28	-102.93	1.45	83.92	164.39	0.10	0.81
M10	598.45±67.14	87.23±14.83	44.30±3.44	196.66	-38.14	1.40	89.53	177.45	0.11	0.87
Control Mean	444.19	62.15	37.56	151.01	-44.11	1.06	70.08	84.18	0.08	0.41
Minimum	142.38	58.67	18.95	115.59	-102.93	0.97	51.64	104.43	0.06	0.51
Maximum	958.52	87.23	62.11	226.82	28.32	1.45	105.26	205.87	0.13	1.01
Mean	506.82	77.11	41.09	174.90	-39.17	1.25	79.36	157.13	0.09	0.77
Kurtosis	0.46	-0.89	-0.33							
Skewness	-0.64	0.41	-1.15							

0.77, 0.06 and 0.26, 0.12, -0.13 for ^{40}K and ^{226}Ra, ^{40}K and ^{232}Th, ^{226}Ra and ^{232}Th for Kawo and Magiro respectively. The results revealed that ^{40}K and ^{226}Ra, ^{40}K and ^{232}Th were significantly correlated in soil samples from Kawo, while poor correlation was seen between ^{226}Ra and ^{232}Th at Kawo and all the radionuclides at Magiro. The results indicate that mobility of the radionuclides was affected by different sediment processes as reported in literature (*Al-Harmarneh and Awadallah 2009*).

The radium equivalent being one of various radiation hazards indices by which gamma dose could be given to the habitants was assessed. It was observed that the average Ra_{eq} activity in Kawo 160.87 Bq kg^{-1} was below that of Magiro 174.90 Bq kg^{-1} (**Table 1** and **2**). None of the samples has Ra_{eq} above the recommended 370 Bq kg^{-1} the safety limit for constructing materials for residential purposes (*UNSCEAR 2000*). The percentage contribution of ^{40}K, ^{226}Ra and ^{232}Th to radium equivalent content of the soil samples used as building materials were calculated. The percentage contribution 26, 44, 30 % and 22, 44 and 34 % are obtained for Kawo and Magiro, respectively. The Ra_{Fz} values ranged from 63.10 to 35.58 Bq kg^{-1} with mean value of -22.48, -102.93 to 28.32 Bq kg^{-1} with a mean value of -39.17 Bq kg^{-1} for Kawo and Magiro respectively, these values translated to almost 98.8% of the studied points (**Table 1** and **2**). The negative values in the studied soil indicated that a certain percentage of ^{226}Ra activity concentration must have been absorbed by plants or leached away. This might be the cause of excess concentration of lead observed in the water from the areas. The estimated values of AUI varied from 0.74 to 1.46 with a mean value of 1.11 and 0.97 to 1.45 with a mean value of 1.25 for Kawo and Magiro, respectively (**Table 1** and **2**). All estimated values of AUI were less than 2, which belongs to an annual effective dose value < 0.3 mSv y^{-1} (*El-Gamal and Nasr 2007*). The values of Dout and Din were found to vary from 49.10 to 96.59 nGy h^{-1} with a mean value of 73.75 nGy h^{-1} and 97.30 to 191.67 nGy h^{-1} with a mean value of 143.39 nGy h^{-1} for Kawo; 51.64 to 105.26 nGy h-^{1} with a mean value of 75.36 nGy h^{-1} and 104.43 to 205. 87 nGy h^{-1} with a mean value of 157.13 nGy h^{-1} for Magiro. The obtained average values D_{out} 73.75 and 79.36 nGy h^{-1} were 1.25 and 1.35 times higher than the world outdoor average value 59 nGy h^{-1} (*UNSCEAR 2008*); while Din 146.39 and 157.13 nGy h^{-1} were 1.74 and 1.87 times the world indoor average value 84 nGy h^{-1} (*UNSCEAR 2000*). The E_{out} values for the soil samples under considerations were found to vary from 0.06 to 0.12 mSv y^{-1} with mean value of 0.09 mSv y^{-1} and 0.06 to 0.13 mSv y^{-1} with mean value 0.09 mSv y^{-1} for Kawo and Magiro respectively. E_{in} values for the samples ranged from 0.50 to 0.94 mSv y^{-1} with mean value of 0.72 mSv y^{-1} and 0.51 to 1.01 mSv y^{-1} with a mean value of 0.77 mSv y^{-1} for the two communities. The calculated average value E_{out} 0.09 mSv y^{-1} was the same for the two communities and was 1.29 times higher than the world average value of 0.07 mSv y^{-1} (*UNSCEAR 2000*). The

average values E_{in} 0.72 and 0.77 mSv y^{-1} were 1.76 and 1.88 times above the world average value of 0.41 mSv y^{-1} (*UNSCEAR 2000*). The calculated values of H_{ex} for the areas investigated were found to range from 0.29 to 0.57 with a mean value of 0.43 for Kawo and 0.31 to 0.61 with a mean value of 0.48 for Magiro. The calculated values of H_{in} were found to vary from 0.42 to 0.82 with a mean value 0.63 and 0.52 to 0.80 with a mean value of 0.69. H_{ex} and H_{in} values were less than unity, the safety limit (**Table 1**). It is obvious that the average mean values were 45.99 and 42.79 respectively. The results indicated that Kawo with lower levels of H_{ex} and H_{in} compared to Magiro has higher average radon proportionate. High gonadal dose equivalent radiation exposure can cause bone marrow leukemia. The estimated Annual Gonadal Effective Dose (AGDE) in this study varied from 351.84 to 694.56 µSv y^{-1} with a mean value of 530.52 µSv y^{-1} and 367.15 to 759.90 µSv y^{-1} with a mean value of 569.17 µSv y^{-1} for Kawo and Magiro respectively (**Table 3**). All AGDE values were above the world average value of 300 µSv y^{-1} (*UNSCEAR 1988*). The higher values of AGDE in the two communities could be associated to the high levels of activity concentrations of ^{40}K, ^{226}Ra and ^{232}Th which were connected to the mining activities of the areas. The values of risk ranged from 0.21 x 10^{-3} to 0.42 x 10^{-3} with a mean value of 0.32 x 10^{-3} and 1.79 x 10^{-3} to 3.54 x 10^{-3} with a mean value of 0.32 x 10^{-3} for outdoor exposure for the two communities respectively (**Table 3**). The indoor exposure ranged from 1.75 x 10^{-3} to 3.29 x 10^{-3} with a mean value of 2.52 x 10^{-3} for Kawo and 1.79 x 10^{-3} to 3.54 x 10^{-3} with a mean value of 2.70 x 10^{-3} for Magiro (**Table 3**). The total risk exposure values were higher than the world average 1.45 x 10^{-3}; which revealed that cancer risk increases with increasing time of exposure to soil in the areas (**Table 3**).

Elemental concentration of U (Ra) and Th are lower than permissible levels of 10 and 20 ppm (*Khrbish et al. 2007*). The maximum elemental concentrations of potassium were revealed by samples at K10 and M1 with values that each reached 2.25 and 3.06% respectively (**Table 3**). The mean values of ratio of eTh/eU (Ra) are 1.41 and 1.62 which are lower than the lithosphere stated ratio of 3 (*Gautheron and Moreira 2002*).

Heavy Metal in Soil and Crop

The type of plants grown and soil physico-chemical characteristics like pH, CEC, OM, and the distribution of metals in various soil fractions affect how quickly plants absorb elements (*Kos et al. 2003*). The soil pH in the areas is 5.61 and 5.83 which revealed the acidic nature of the soil (**Table 4**). Since the pHs of the studied samples are less than 7, heavy metals are generally more mobile at pH<7. Therefore, the pH conditions revealed hazardous nature of the soil in the areas for agricultural purposes, which might be due to mining activities in the areas. Organic matter content was at medium level and

Table 3. Hazard indices, exhaled radon, AGDE, risk elemental concentrations in soil and concentration accumulation index in crop *Sorghum bicolor* (L.) Muench.

Location	Soil									Plant (ppm)							
	Hazard Indices		H_R%	AGDE µSv y^{-1}	Risk x 10^{-3}		R_T	e(U)	e(Th) ppm	eTh/ eU	K%	Pb	Cu x 10^{-2}	Fe x 10^{-1}	Cr	CD x 10^{-1}	Ni x 10^{-1}
	H_{ex}	H_{in}			R_{out}	R_{in}											
K1	0.44	0.59	33.21	535.81	0.32	2.52	2.84	4.53	10.36	2.29	1.91	1.05	1.05	4.50	1.53	1.67	1.36
K2	0.39	0.62	58.97	482.44	0.28	2.45	2.73	6.76	4.33	0.64	1.54	1.07	1.07	6.90	1.67	0.20	1.40
K3	0.29	0.42	46.20	351.84	0.21	1.75	1.96	3.94	6.34	1.61	0.96	1.05	1.05	8.90	1.60	2.27	1.48
K4	0.31	0.44	41.64	385.13	0.21	1.75	1.96	3.87	5.38	1.39	1.49	1.06	1.06	8.20	1.61	1.90	1.54
K5	0.55	0.82	48.73	669.71	0.39	3.19	3.58	7.95	10.12	1.27	1.98	1.03	1.03	5.80	1.61	1.43	1.43
K6	0.47	0.64	37.06	577.60	0.35	2.70	3.05	5.30	9.42	1.78	2.19	1.03	1.03	5.69	1.50	1.61	0.16
K7	0.45	0.73	63.11	527.36	0.32	2.56	2.88	8.62	6.17	0.72	0.95	1.04	1.04	6.90	1.57	1.67	1.32
K8	0.40	0.53	34.40	498.13	0.28	2.31	2.59	4.06	9.42	2.32	1.87	1.03	1.03	4.80	1.54	2.17	1.67
K9	0.57	0.82	43.86	694.56	0.42	3.29	3.71	7.61	10.90	1.43	2.23	1.04	1.04	7.80	1.67	1.85	1.43
K10	0.47	0.65	37.57	582.51	0.32	2.52	2.84	5.28	9.39	1.78	2.25	1.06	1.06	5.59	1.50	0.20	1.82
Mean	0.43	0.63	45.99	530.52	0.32	2.52	2.84	5.79	8.18	1.41	1.74	1.04	1.04	1.28	1.56	2.08	1.40
M1	0.61	0.80	31.15	759.90	0.45	3.54	3.99	5.66	14.33	2.53	3.06	1.06	1.06	5.90	1.50	0.15	1.61
M2	0.45	0.67	48.58	567.53	0.32	2.70	3.02	6.64	4.67	0.70	2.39	1.03	1.03	9.10	1.50	1.81	1.45
M3	0.44	0.63	43.18	516.24	0.32	2.45	2.77	5.86	11.65	1.99	0.96	1.03	1.03	8.50	1.60	2.08	1.57
M4	0.31	0.52	67.74	367.15	0.21	1.79	1.99	6.10	5.29	0.87	0.45	1.11	1.11	7.96	1.60	1.74	1.43
M5	0.47	0.70	48.30	595.26	0.35	2.84	3.19	6.81	5.78	0.85	2.41	1.03	1.03	4.21	1.67	1.82	1.58
M6	0.53	0.76	42.83	633.26	0.39	3.01	3.40	6.90	12.58	1.82	1.59	1.05	1.05	9.10	1.50	1.88	1.64
M7	0.39	0.55	41.03	462.95	0.28	2.21	2.49	4.75	10.70	2.25	1.02	1.06	1.06	6.00	1.60	1.67	1.63
M8	0.46	0.69	49.28	549.15	0.35	2.63	2.98	6.76	9.99	1.48	1.24	1.05	1.05	8.00	1.63	1.90	1.46
M9	0.51	0.70	38.04	597.34	0.35	2.63	2.98	5.90	15.30	2.59	1.15	1.06	1.06	9.10	1.56	2.05	1.62
M10	0.53	0.76	43.40	642.62	0.39	3.05	3.44	7.06	10.90	1.54	1.91	1.06	1.06	6.00	1.64	0.20	1.40
Mean	0.48	0.69	42.79	569.17	0.32	2.70	3.02	6.24	10.12	1.62	1.62	1.08	1.08	1.13	1.53	1.84	1.54

the values for the two communities are 2.02 and 4.06%. The areas could be classified as moderately calcareous because the calcium contents are 4.93 and 6.02 C_{mol} kg^{-1} for Kawo and Magiro respectively. The sand, silt and clay contents for the areas are (73.00, 79.90), (6.00, 8.50), and (21.00, 13.50). The results revealed varying soil compositions among the sampling points. The mean concentrations of Pb, Cu, Fe, Cr, Cd and Ni in mg kg^{-1} in both soil and crop were (2.20, 2.29), (3.14, 0.16), (15.10, 19.33), (0.16, 0.25), (0.24, 0.05), (0.43, 0.06) and (3.00, 3.18), (6.09, 0.34), (16.83, 10.10), (0.22, 0.36), (0.25, 0.05), (0.43, 0.06) for Kawo and Magiro respectively. The sequence of heavy metals in the soil are Fe>Cu>Pb>Ni>Cd>Cr; while the sequence in the plant are Fe>Pb>Cr>Cu>Ni>Cd for the two communities. The mean concentration of Pb and Cr are higher in crop compared to soil, an indication that surface soil in the areas is contaminated. Since the crop has higher concentration of lead and chromium compared to soil, a certain percentage of their concentrations must have been leached away as revealed by R_{aFz} calculated, which showed that 98% of ^{226}Ra has leached into the deep soil at depth beyond 15-20 cm considered in the study.

Heavy metals of the soil profile in the areas should be analyzed to determine the level of lead concentration. The concentrations of these metals in in the control site are 2.54, 7.01, 15.28, 0.20, 0.18 and 0.38 for Pb, Cu, Fe, Cr, Cd and Ni respectively. Eighty-three percent of the samples showed heavy metals are slightly higher in contaminated soils to uncontaminated soils. This indicates that heavy metals in polluted samples are mobile and bonded to other phase (*Rauret 1998*), while they are largely stationary in their native mineral and bound silicate states (*Sungur et al. 2014*). The heavy metals concentrations are higher in crops than the FAO/WHO guideline that recommended 0.1 mg kg^{-1} Pb, 0.1 mg kg^{-1} Cu, 0.3 mg kg^{-1} Fe, 0.1-0.2 mg kg^{-1} Cr, 0.02 mg kg^{-1} Cd and 0.1 mg kg^{-1} Ni. Plants grown in the areas had absorbed heavy metals. Translocation effects from one medium to another are largely osmotic in nature. Therefore, the effect of heavy metals from the soil to the plant in the areas was estimated using Concentration Accumulation Index (CAI) (*Faweya et al. 2017*). *Sorghum bicolor* (L.) Muench was chosen because it was readily available during the sampling exercise aside been one of staple dietary crops in the area. The metal CAI in *S. bicolor* was used to indicate the extent of build-up of an element in an associated biological system. The mean CAI in S. bicolor in the areas showed a trend in order of Cu<Fe<Ni<Cd<Pb<Cr.The CAI of the metals in the ingested parts of *S. bicolor* was less than 1, except for lead and chromium which are greater than 1 (**Table 3**). Lead and chromium with CAI values in the range of (1.03 to 1.11), (1.50 to 1.67) were the most accumulated, which indicated that lead and chromium transport to the edible part (grains) of the plant above those with a low CAI (*Luo et al. 2011*) and crops grown in the areas should be properly monitored (**Figure 3**).

Non-Carcinogenic Risk

Six heavy metals (Pb, Cu, Fe, Cr, Cd and Ni) were considered for both non-carcinogenic and carcinogenic health risks. Among these, Fe had the highest ADI_{ing} values of 3.1 x 10^{-1}, 4.3 x 10^{-1} and 3.4 x 10^{-1}, 4.8 x 10^{-1} mg kg^{-1} d^{-1} at Kawo and Magiro for adults and children respectively. Chromium had the least values. The estimated ADI_{ing} for the studied metals followed order-Fe (0.31, 0.43/0.34,

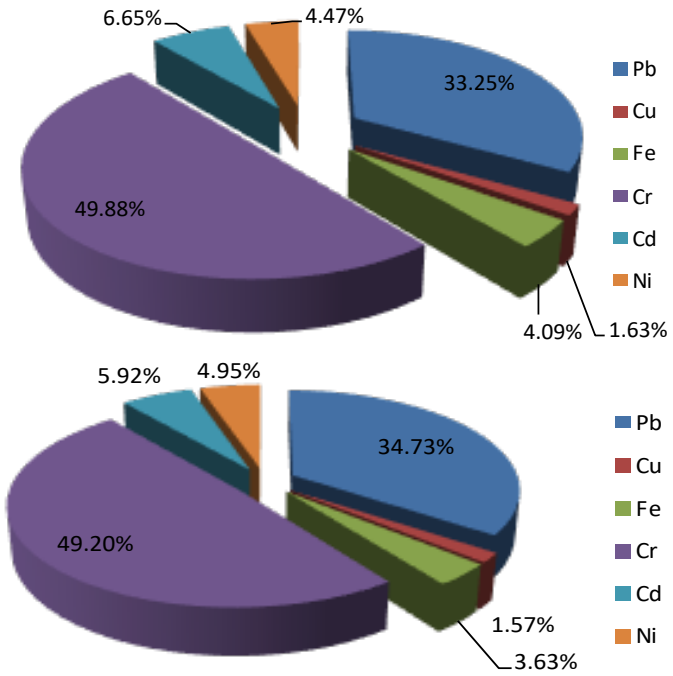

Figure 3. The percentage of concentration accumulation index values of each metal in Kawo (top) and Magiro, Nigeria (bottom)

Table 4. Physico-chemical properties of the soil samples, Niger State, Nigeria.

Location	pH (H_2O)%	OM%	N%	P mg kg^{-1}	K	Ca	Na	Mg	CEC	Exch acidity	Sand %	Silt %	Clay %	Textural
					Cmol kg^{-1}									
Kawo	5.61	2.02	0.48	10	0.22	4.93	0.35	4.84	11.44	0.02	73.00	6.00	21.00	Sandyclayloamy
Magiro	5.83	4.06	0.72	18	0.38	6.02	0.41	4.60	13.67	0.02	79.90	8.50	13.50	Sandyclayloamy

0.48) > Cu (0.06, 0.09/0.06, 0.08) > (0.04, 0.06/0.06, 0.08) >Ni (0.009, 0.008/0.008, 0.008) >Cd (0.005, 0.007/0.005, 0.007) >Cr (0.003, 0.005/0.004, 0.005) for adults and children at both Kawo and Magiro. The ADI_{der} peaked for Cu at both communities, while the ADI_{inh} peaked for Fe at both communities for adults and children. The HQ_{ing} among the investigated heavy metals revealed that Cr and Ni are below 1 (hazard quotient threshold), while Pb, Cu and Cd are above 1. The obtained values of HQ_{ing} increased in the order – Cr < Ni < Cu <Pb <Cd. Also, the HQ_{ing} for Pb, Cu and Cd are above 1, the threshold H_Q value. This implies that there are considerable non-carcinogenic adverse health effects from these metals. The HQ_{der} values for all the studied metals are below 1. Thus, it suffices to mention that the contamination suffered by the residents of the areas, most especially by the children, was not through dermal exposure to these heavy metals. The estimated H_I values for ingestion at both Kawo and Magiro were 9.85, 12.75 and 13.95, 17. 35 for adults and children respectively. These values were higher than 1, indicating that there are significant non-carcinogenic risks. Compared with H_I for ingestion, H_I value for dermal was much lower than 1, indicating that there is no significant non-carcinogenic risk through dermal exposure.

Carcinogenic Risk

The *ILCR* decreased in the order- Cd (0.75×10^{-4}, 1.1×10^{-4}) < Cr (3.94×10^{-4}, 5.48×10^{-4}) < Pb (6.94×10^{-3}, 9.67×10^{-3}) for adults and children in the two communities. The *ILCR* values for adults were lower than those for children, resulting in 1.41 times higher in $\sum ILCR$ for the children with respect to adults. The $\sum ILCR$ carcinogenic risks for both adults (0.73×10^{-2}) and children (1.03×10^{-2}) were above tolerable acceptable risk (given as 1×10^{-6} to 1×10^{-4}). This made the children more susceptible to harmful effects than the adults in the areas.

Ecological Risk

Ecological risks assessment were defined based on the following four grades- $Er \leq 150$, low risk; $150 < Er \leq 300$, considerable risk; $300 < Er \leq 600$ and $Er > 600$,high risk. Lead (550, 800), Cu (349, 677), Cr (3.2, 4.4), Cd (2400, 2500) and Ni (43, 43) for Kawo and Magiro respectively. The results revealed considerable risks of Cu and Pb and high risk of Cd at Kawo, while there is high risk of Cd, Cu and Pb at Magiro.

CONCLUSIONS AND RECOMMENDATIONS

The Radiometric and heavy metals investigations showed the presence of the three primordial radionuclides and six heavy metals. Average outdoor and indoor radiation doses received were higher than the world's average. The concentration of heavy metal was higher in crop than in soil, which was due to mining activities in the areas. The concentration of heavy metals in the crop has provided baseline data as basis for intensive sampling of all food grown in the areas to avoid entry of heavy metals into the food because the rates of absorption of heavy metals by plants differ. The results of lead concentration in contaminated soil revealed that lead entered the body of residents of the affected communities via ingestion. The rate of ingestion of lead by children is higher than the adults. This was in agreement with that of *Tarrago* (*2012*). This confirms the report from a previous study in 2015 indicating higher concentration of lead in the blood serum of the infants and children in the areas. Therefore, the areas need proper monitoring and measures should be taken to reduce carcinogenic risks. The results also provided important information for policy making in order to reduce the potential effects of soil contamination through mining on human and other ecosystems.

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