

Primordial Radionuclide Concentrations and Gamma Ray Dose Rates of Black Construction Sand from Longonot Sand Mines in Naivasha, Kenya

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Abstract

The radiological characteristics of black construction sand from the Longonot volcanic region of Kenya were investigated using a calibrated NaI(Tl) gamma spectrometer. Activity concentrations of ²³²Th, ²³⁸U, and ⁴⁰K, together with the associated radiological screening parameters, were evaluated in 56 sand samples. The reported mean activity concentrations exceeded the corresponding global UNSCEAR averages. The absorbed dose rate had a mean value of 222.08 ± 6.51 nGy h⁻¹, corresponding to annual effective doses of 0.82 ± 0.02 mSv y⁻¹ indoors and 0.55 ± 0.02 mSv y⁻¹ outdoors. The mean radium equivalent activity was 481.38 ± 14.32 Bq kg⁻¹, exceeding the conventional reference value of 370 Bq kg⁻¹. The mean external and internal hazard indices exceeded unity, indicating that the material exceeded the conventional screening criterion for further radiological assessment. Further assessment, including radon exhalation and indoor radon measurements, is recommended to characterize the potential exposure pathways associated with the use of the sand in construction.

Keywords: Gamma-ray spectroscopy; activity concentration; black sand; Longonot sand mine.

I. INTRODUCTION

Natural radioactivity is an inherent component of the Earth's crust and arises primarily from primordial radionuclides, including uranium-238 (²³⁸U), thorium-232 (²³²Th), and potassium-40 (⁴⁰K). These radionuclides occur naturally in rocks, soils, and sediments, with their activity concentrations varying according to geological composition, mineralogical characteristics, and geochemical processes [1–2].

The decay of naturally occurring radionuclides contributes

to external gamma-ray exposure, while radon and its short-lived progeny may contribute to internal exposure through inhalation [3]. Consequently, the radiological characterization of construction materials is important for evaluating their potential contribution to radiation exposure and identifying materials that may require further investigation [4]. Materials such as sand, rocks, cement, and concrete can contain varying concentrations of naturally occurring radionuclides depending on their geological sources and constituent minerals. The use of materials with elevated radionuclide concentrations may contribute to indoor gamma radiation and, where radon is

released from the material, potential internal exposure [5–6]. The resulting exposure depends on factors such as radionuclide concentrations, the quantity and distribution of the materials, building geometry, occupancy patterns, and ventilation conditions [7–8].

Increasing construction activity has raised demand for sand and encouraged the exploitation of alternative sources, including volcanic deposits. Assessing the radiological characteristics of these materials is important, particularly where they are used in buildings. Volcanic terrains may exhibit variations in radionuclide concentrations due to differences in parent-rock composition, magmatic differentiation, mineral crystallization, weathering, and sediment redistribution. In some geological settings, accessory minerals such as zircon and monazite may contribute to uranium and thorium enrichment; however, mineralogical characterization is needed to confirm their specific contributions to individual sand deposits [9–10].

The Kenyan Rift Valley contains extensive volcanic formations and areas of elevated natural background radiation [11]. Previous studies have reported enhanced radionuclide concentrations at locations such as Olkaria, Homa Hills, and Mrima Hill, where geological and mineralogical characteristics influence radionuclide distribution [12]. Mount Longonot is a relatively young stratovolcano within the Rift

Valley system, and the surrounding black sand deposits are likely associated with the weathering and erosion of volcanic rocks, including basaltic and trachytic formations [13]. Despite the growing extraction and use of volcanic sand for construction, published information on the activity concentrations of ^{238}U , ^{232}Th , and ^{40}K in black sand from the Longonot region remains limited.

This study therefore determined the activity concentrations of ^{238}U , ^{232}Th , and ^{40}K in black construction sand collected from mining sites in the Longonot region of Naivasha, Kenya. The associated radiological parameters, including absorbed dose rate, annual effective dose, radium equivalent activity, and external and internal hazard indices, were evaluated to characterize the radiological properties of the sampled material and assess its potential implications for construction use.

II. MATERIALS AND METHODS

A. Study Area

Fig. 1 shows the study area with sampling locations. The study was conducted in the Longonot region of Naivasha Sub-County, Nakuru County. Geographically, it lies between latitudes 0.85°S and 0.95°S , and longitudes 36.35°E and 36.50°E . The area is characterized by Quaternary volcanic formations like basalts and tuffs [14].

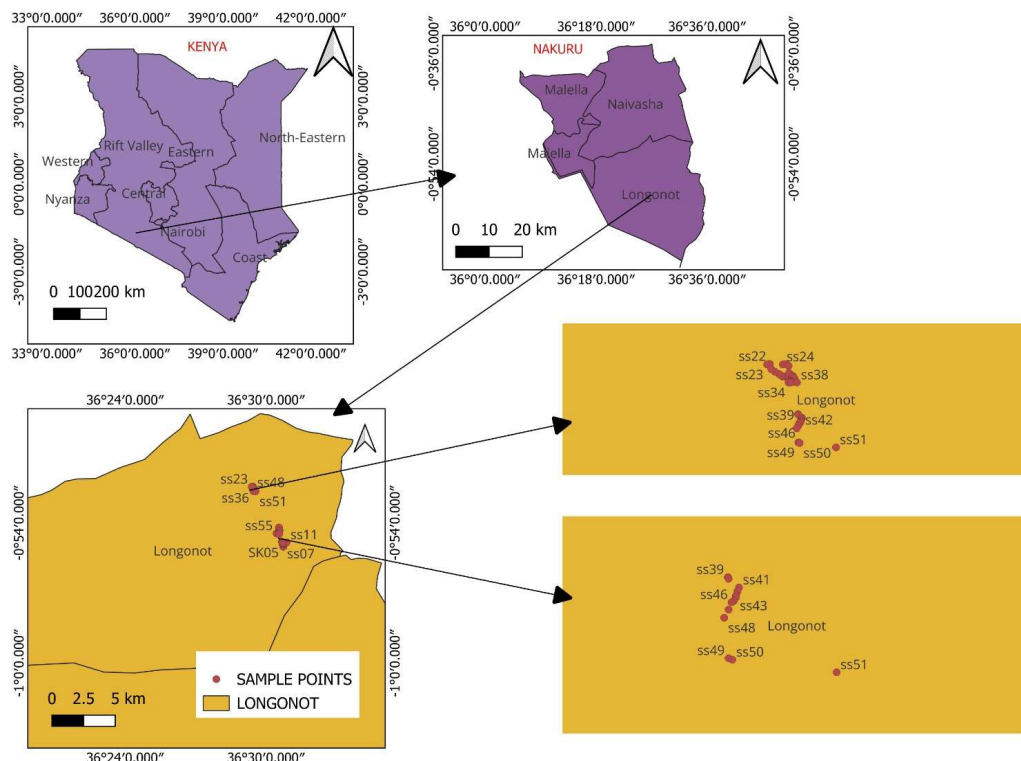


Fig. 1. Map of the study area showing sampling locations in Longonot.

Mount Longonot is a stratovolcano within the Kenya Rift Valley, characterized by volcanic deposits and associated

sedimentary materials. The sampled black sand may reflect the weathering, erosion, and redistribution of volcanic

materials in the region; however, its specific sedimentary provenance was not established through mineralogical analysis in the present study [15]. Sampling was conducted at active sand mines within the Longonot volcanic field. The sand is mined for construction and supplied to urban centres, particularly Nairobi and Naivasha.

B. Sample Collection and Preparation

A total of 56 black sand samples were collected from five active mining sites within the Longonot volcanic region using a systematic random sampling approach. The samples were distributed among the five sites as follows: Site 1 ($n = 12$), Site 2 ($n = 11$), Site 3 ($n = 10$), Site 4 ($n = 13$), and Site 5 ($n = 10$). The number of samples collected from each site varied according to the size and extent of the respective mining area. Sampling points were identified by traversing along the symmetry lines of the mines at approximately 10–15 m intervals, with the initial sampling point selected randomly. Sampling was conducted to a maximum depth of 1 m. At each location, a circular area with a radius of 0.5 m was delineated using a wooden marker, and the sand within the marked area was thoroughly mixed using a hand shovel to improve sample representativeness. Approximately 0.5 kg of bulk material was collected from the centre of each sampling area using a shovel rinsed with sand from the corresponding location. Each sample represented an individual sampling location and was placed in a clean, labelled, tightly sealed plastic container to minimize cross-contamination during transport. The sample identification number, GPS coordinates, and collection date were recorded.

The samples were transported upright in a carton to the drying site and laboratory. Each sample was air-dried separately in sunlight until visibly dry and subsequently oven-dried at 105 °C for 24 h to remove residual moisture. The dried samples were ground using an electronic pulveriser and sieved through a 1 mm mesh to improve homogenization and particle-size consistency before gamma-ray measurement.

From each homogenized sample, 250 g was accurately weighed into a 300 mL measurement bottle matching the geometry of the reference samples. The bottles were sealed and stored for 28 days before gamma-ray spectrometric analysis. This period was primarily intended to allow the ingrowth of short-lived ^{222}Rn progeny, particularly ^{214}Pb and ^{214}Bi , following sealing and the associated re-establishment of equilibrium with ^{226}Ra . Accordingly, ^{226}Ra activity was inferred from the selected gamma-ray emissions of ^{214}Pb and ^{214}Bi , assuming equilibrium between ^{226}Ra and these short-lived progeny at the time of measurement. The 28-day storage period was therefore not considered sufficient to establish secular equilibrium throughout the entire ^{238}U decay series.

For ^{232}Th , the activity concentration was determined from selected gamma-ray emissions of its progeny, particularly ^{212}Pb and ^{228}Ac , under the conventional assumption of radioactive equilibrium between ^{232}Th and the relevant progeny. The 28-day storage period was not interpreted as

establishing secular equilibrium throughout the entire ^{232}Th decay series. Rather, the ^{232}Th activity estimates represent equivalent ^{232}Th -series concentrations under the stated equilibrium assumption [16].

C. Gamma Spectrometry

Gamma-ray measurements were performed using a NaI(Tl) scintillation detector (model VG-BB-98/6D1) coupled to a photomultiplier tube (PMT), manufactured by Luxium Solutions, USA. The detector assembly accommodated a 250-g plastic sample bottle with an internal sample volume of approximately 300 cm³. The NaI(Tl) detector had a nominal energy resolution of approximately 7% full width at half maximum (FWHM) at 662 keV, consistent with the typical performance specified for NaI(Tl) scintillation detectors. The detector was used for gamma-ray spectrometric measurements of the black sand samples. The selected gamma-ray lines and the assumptions used to estimate the parent radionuclide activity concentrations are described in Section II.C.3.

1) Counting Geometry and Data Acquisition

The counting geometry, live time, and spectral processing of the geological samples were kept identical to those used for the standard reference sample. The detection efficiency for ^{238}U , ^{232}Th , and ^{40}K was determined using the certified reference materials RGK-1, RGU-1, and RGTh-1, respectively, traceable to the International Atomic Energy Agency (IAEA). A 250-g portion of each sample was placed in a bottle having the same dimensions as the reference-sample container and positioned above the detector entrance window. Background measurements were performed weekly using a deionised-water blank under the same counting geometry and measurement conditions as the geological samples, with a counting duration of 28,800 s. The measured background spectrum was used to correct the sample spectra for background contributions. Net peak counts were obtained by subtracting the corresponding background counts from the gross sample peak counts, using the same counting time and spectral-processing procedures described in (1).

$$A_n = A_g - A_w \quad (1)$$

where A_n is the net peak count, A_g is the corresponding gross peak count for the geological sample, and A_w is the background contribution obtained from the deionised water.

All samples were measured for a live time of 28,800 s to obtain adequate counting statistics, particularly for radionuclides with relatively low gamma-ray emission intensities. The system operated with a dead time of approximately 0.01%, ensuring high counting efficiency throughout data acquisition. The spectra were acquired and saved for analyses using Maestro Ortec software.

2) Energy Calibration

Energy calibration was performed using four reference gamma-ray lines from ^{241}Am , ^{137}Cs , and ^{60}Co at 59.5, 661.7, 1173.2, and 1332.5 keV, respectively, covering the experimental energy range of 59.5–1332.5 keV. The calibration data were fitted using a second-order polynomial

relationship between photon energy and MCA channel number, as given in (2) [17].

$$y(x) = A + Bx + Cx^2 \quad (2)$$

where $y(x)$ is the photon energy, C is the MCA channel number, and A , B , and C are the fitted calibration coefficients. The fitted parameters were $A = (-0.12569 \pm 0.31138)$, $B = (0.23763 \pm 2.74561 \times 10^{-4})$, and $C = (5.5892 \times 10^{-8} \pm 4.58458 \times 10^{-8})$. The principal ^{40}K photopeak occurs at 1460 keV, which is 127.5 keV above the highest experimental calibration point at 1332.5 keV. The second-order calibration function was extrapolated over this limited 127.5-keV interval to determine the MCA channel corresponding to the 1460-keV ^{40}K photopeak [18]. This extrapolation was applied only to the energy-channel calibration and not to the detector-efficiency calibration. The gamma-ray emission probabilities used for subsequent radionuclide quantification were 0.3534 for ^{214}Pb at 351 keV, 0.4516 for ^{214}Bi at 609 keV, 0.4316 for ^{212}Pb at 238 keV, 0.258 for ^{228}Ac at 911 keV, and 0.1066 for ^{40}K at 1460 keV, as reported by IAEA (2007).

3) Efficiency Calibration

Detector efficiency calibration was performed using the certified reference materials RGK-1, RGU-1, and RGTh-1, traceable to the International Atomic Energy Agency (IAEA). The efficiency calibration covered the gamma-ray energy range of 238–1460 keV using selected reference gamma-ray

emission lines at 238, 351, 609, 911, and 1460 keV. The experimentally determined full-energy peak efficiencies were calculated from the net full-energy peak counts using (3).

$$\eta = \frac{N_p}{A \cdot P_\gamma \cdot t} \quad (3)$$

where N_p is the net full-energy peak count, A is the activity of the reference material, P_γ is the corresponding gamma-ray emission probability, and t is the counting live time. The experimentally determined efficiency values were fitted as a function of gamma-ray energy using a third-order polynomial relationship, as shown in Fig. 2.

The resulting efficiency–energy relationship is given in (4).

$$\eta(E_\gamma) = a_3 E_\gamma^3 - a_2 E_\gamma^2 + a_1 E_\gamma - a_0 \quad (4)$$

where $a_3 = 2.6166 \times 10^{-12}$, $a_2 = -6.5343 \times 10^{-9}$, $a_1 = 4.1074 \times 10^{-6}$ and $a_0 = -8.2039 \times 10^{-5}$ are the efficiency fit coefficients with a goodness-of-fit of $R^2 = 0.99745$.

The efficiency at 1460 keV, corresponding to the principal ^{40}K gamma-ray emission, was determined directly from the ^{40}K reference material and therefore lay within the experimentally measured efficiency range. Consequently, no extrapolation of the efficiency–energy relationship was required or used at 1460 keV, and no extrapolation uncertainty was introduced at this energy. The efficiency determined at 1460 keV was used directly in the subsequent quantification of ^{40}K activity.

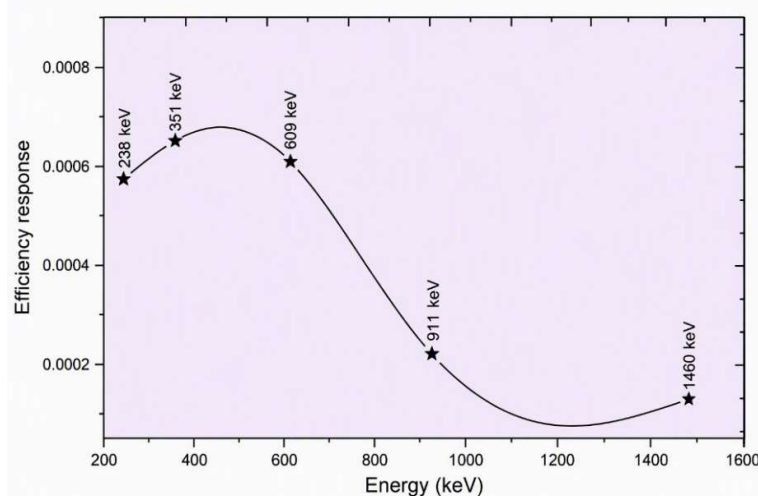


Fig. 2. Detector efficiency as a function of gamma-ray energy determined using certified reference materials.

4) Radionuclide Identification and Quantification

Radionuclides were identified using their characteristic gamma-ray energies: the ^{238}U series (^{214}Pb at 352 keV, ^{214}Bi at 609 keV), the ^{232}Th series (^{212}Pb at 238 keV, ^{228}Ac at 911 keV), and ^{40}K (at 1460 keV). The ^{238}U was quantified from the average intensities of ^{214}Pb and ^{214}Bi at their emission gamma lines at the energies of 352 keV and 609 keV, respectively, while ^{232}Th was quantified from the average emissions of ^{212}Pb at 238 keV and ^{228}Ac at 911 keV. The activity of ^{40}K was quantified from the intensity of its single photopeak at an energy of 1460 keV. The calculation employed an

intercomparison method in which the known mass, activity concentration, and net peak intensity of each reference material were compared with the corresponding quantities measured for the sample under study, as summarized in (5) [19].

$$A_s = \frac{I_s M_r A_r}{I_r M_s} \quad (5)$$

where I_s and I_r are the net peak intensities of the sample and reference standard, M_s and M_r are the masses of the sample and reference, and A_r represents the known activity of the standard source in Bq kg^{-1} , and for this study was $4,940 \text{ Bq kg}^{-1}$.

kg^{-1} , $3,250 \text{ Bq kg}^{-1}$, and $14,000 \text{ Bq kg}^{-1}$ for ^{238}U , ^{232}Th , and ^{40}K , respectively. The uncertainties were calculated at the 95% confidence level.

III. RESULTS AND DISCUSSION

A. Activity Concentration Calculation

The activity concentrations of ^{232}Th , ^{238}U , and ^{40}K were calculated using (5). The results are shown in Fig. 3, while the minimum, maximum, and mean values are presented in Table I.

The activity concentrations varied across the sampled area, with ^{232}Th ranging from 74.53 ± 3.73 to $201.74 \pm 10.09 \text{ Bq kg}^{-1}$, ^{238}U from 89.84 ± 4.49 to $284.60 \pm 14.23 \text{ Bq kg}^{-1}$, and ^{40}K from 181.30 ± 9.07 to $1720.75 \pm 86.04 \text{ Bq kg}^{-1}$. The observed variation in radionuclide activity concentrations demonstrates heterogeneity among the sampled locations. However, without formal spatial statistical analysis, the data do not establish a systematic spatial pattern. Further spatial and mineralogical investigations are required to identify the factors controlling this variability [1].

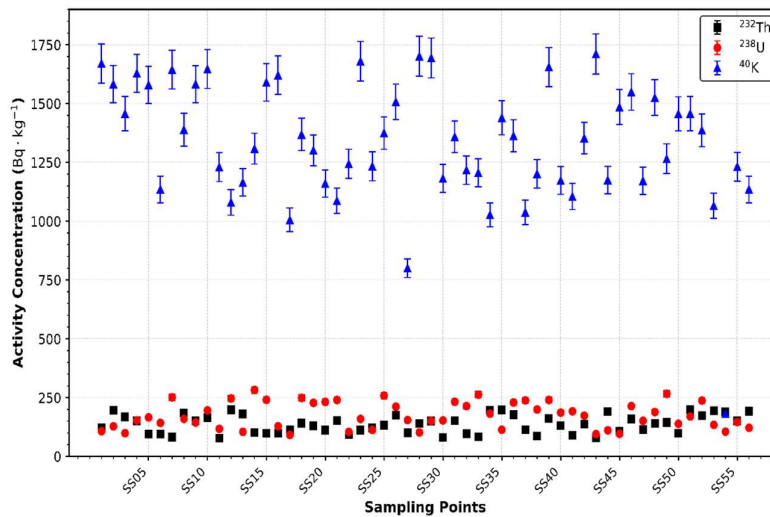


Fig. 3. Activity concentration of ^{232}Th , ^{238}U and ^{40}K .

Table I. Activity concentrations of ^{232}Th , ^{238}U and ^{40}K in black sand samples.

Radionuclide	Mean (Bq kg^{-1})	Minimum (Bq kg^{-1})	Maximum (Bq kg^{-1})
^{232}Th	138.18 ± 6.91	74.53 ± 3.73	201.74 ± 10.09
^{238}U	180.32 ± 9.02	89.84 ± 4.49	284.60 ± 14.23
^{40}K	1326.3 ± 66.32	181.30 ± 9.07	1720.75 ± 86.04

The relatively high activity concentrations observed at SS25 may reflect differences in the geological materials contributing to the sampled sand. The regional occurrence of trachytic volcanic rocks provides a possible geological context for the observed radionuclide concentrations; however, the specific mineralogical controls were not determined in this study [13]. In contrast, SS54 exhibits comparatively lower radionuclide concentrations, likely reflecting weathered, reworked, or leached surface materials that have undergone sediment redistribution, resulting in the dilution of radionuclide-bearing phases.

The observed variation in activity concentrations indicates heterogeneity in the radionuclide distribution within the sampled material, potentially reflecting differences in source-material composition and sediment redistribution. The relatively elevated ^{40}K concentrations may reflect variability in potassium-bearing phases, while variations in ^{238}U and ^{232}Th may arise from differences in the abundance and distribution of radionuclide-bearing phases [20]. However, the specific mineralogical parameters were not determined in the present study. Table II compares activity concentration in Longonot with other similar studies.

Table II. Comparison of activity concentration of Longonot with other studies.

Study Area	^{238}U (Bq kg^{-1})	^{232}Th (Bq kg^{-1})	^{40}K (Bq kg^{-1})	Reference
Longonot	180.3 ± 9.02	138.2 ± 6.91	1326.3 ± 66.32	This study
West Coast (Chilaw)	1243 ± 15	6257 ± 38	647 ± 37	[21]
Rashid (Rosetta)	107.0 ± 5.0	148.0 ± 7.0	162.0 ± 8.0	[22]
Mrima Hill	207.0 ± 11.3	500.7 ± 20.3	805.4 ± 20.7	[12]
Lambwe Valley (Ruri Hills)	179 ± 12	1397 ± 97	509 ± 33	[23]
Homa Hills	195.3 ± 8	409.5 ± 4	915.6 ± 3	[24]

The activity concentration pattern observed in Longonot, characterized by moderately high ^{238}U and ^{232}Th and relatively high ^{40}K , is consistent with the heterogeneous geological composition of volcanic-derived sediments in the region. These differences may reflect variations in the source materials and sedimentary redistribution processes; however, the specific mineralogical mechanisms responsible for the observed radionuclide concentrations were not determined in the present study. Compared with carbonatite-dominated regions such as Mrima Hill, Lambwe Valley, and Homa Hills, Longonot exhibits lower thorium concentrations, possibly due to the absence of thorium-rich carbonatite intrusions, but

higher potassium levels associated with potassium-bearing volcanic minerals [10]. These differences suggest that the radiological characteristics of Longonot may be influenced by volcanic processes rather than carbonatite enrichment.

B. Absorbed Dose Rate

The absorbed dose rate was computed using the conversion factors for primordial radionuclides [3]. The dose rate (D) in nGy h^{-1} was obtained using (6) [1]. The results are presented in Table III and Fig. 4.

$$D(\text{nGy h}^{-1}) = 0.462A_U + 0.604A_{Th} + 0.0417A_K \quad (6)$$

Table III. Absorbed dose rate and annual effective doses.

Parameter	Mean	Maximum	Minimum
D (nGy h^{-1})	222.08 ± 6.51	325.35 ± 9.65	94.08 ± 3.09
$AED_{in}(\text{mSv y}^{-1})$	0.82 ± 0.02	1.20 ± 0.04	0.35 ± 0.01
$AED_{out}(\text{mSv y}^{-1})$	0.55 ± 0.02	0.80 ± 0.02	0.23 ± 0.01

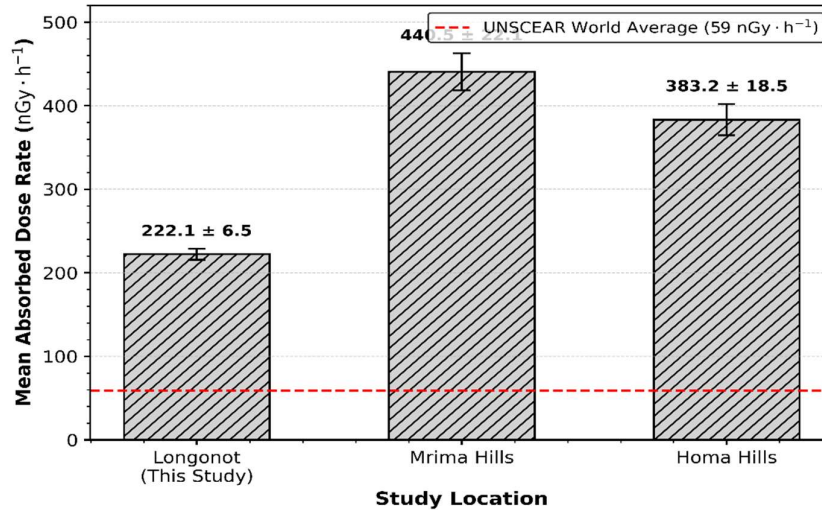


Fig. 4. Comparison of the mean absorbed dose rates for Longonot (this study), Mrima Hills, and Homa Hills. Error bars represent the reported uncertainties.

Table III shows that the absorbed dose rates (D) ranged from 94.08 ± 3.09 to $325.35 \pm 9.65 \text{ nGy h}^{-1}$, with a mean of $222.08 \pm 6.51 \text{ nGy h}^{-1}$. These elevated values are associated with the activity concentrations of ^{238}U , ^{232}Th , and ^{40}K . Samples with higher concentrations of ^{238}U and ^{232}Th , particularly SS25 and SS32, also exhibited higher dose rates. The volcanic origin of the sand and possible mineralogical enrichment through erosion and sedimentary processes may contribute to these values. The mean dose rate exceeds the global average of 59 nGy h^{-1} [1], indicating elevated natural radioactivity in the sampled Longonot sand.

Fig. 4 shows that the mean absorbed dose rate of $222.08 \pm 6.51 \text{ nGy h}^{-1}$ for Longonot is lower than $383.2 \pm 18.5 \text{ nGy h}^{-1}$ and $440.5 \pm 22.1 \text{ nGy h}^{-1}$ reported for Homa Mountain and Mrima Hills, respectively [24], [12]. Both are known high background radiation areas in Kenya. This difference shows

variations in geological setting and mineralogy. Mrima Hills is dominated by carbonatite intrusions rich in uranium- and thorium-bearing minerals [12]. Similarly, Homa Mountain has elevated dose rates due to enhanced concentrations of radiogenic heavy minerals [10]. In contrast, Longonot is an alkaline volcanic system where radiological contributions arise mainly from potassium-bearing minerals and moderate uranium and thorium enrichment, resulting in comparatively lower dose rates. Therefore, although Longonot sand exhibits enhanced radioactivity relative to global averages, its absorbed dose rate remains lower than that of carbonatite-dominated and highly mineralized volcanic regions within Kenya.

C. Annual Effective Dose (AED)

The annual effective dose (AED) was estimated from the absorbed dose rate in air (D) using (7) [25].

$$AED = D \times 8760 \times 0.7 \times OF \times 10^{-6} \quad (7)$$

where (D) is the absorbed dose rate in nGy h⁻¹, 8760 is the number of hours per year, 0.7 Sv Gy⁻¹ is the dose conversion coefficient, and (OF) is the occupancy factor. Occupancy factors of 0.6 for indoor exposure and 0.4 for outdoor exposure were adopted, consistent with values used in previous public-exposure assessments in Kenya [25]. These factors represent assumed fractions of annual occupancy and were not measured at the study locations.

The mean absorbed dose rate was 222.08 ± 6.51 nGy h⁻¹. Indoor AED values ranged from 0.35 ± 0.01 to 1.20 ± 0.04 mSv y⁻¹, with a mean of 0.82 ± 0.02 mSv y⁻¹, while outdoor values ranged from 0.23 ± 0.01 to 0.80 ± 0.02 mSv y⁻¹, with a mean of 0.55 ± 0.02 mSv y⁻¹. Approximately 14% of the samples had estimated indoor AED values exceeding the conventional 1 mSv y⁻¹ reference level used for exposure from building materials, whereas all estimated outdoor values remained below this level. Because the same measured absorbed dose rates underpin both estimates, the approximately 1.5-fold higher indoor AED reflects the different occupancy factors (0.6 versus 0.4), rather than a measured difference in dose rate.

The AED values are modelled estimates conditioned on the measured absorbed dose rates and the specified occupancy and dose-conversion assumptions. Their uncertainty is therefore dominated by the uncertainty in the absorbed dose rate, with the occupancy factor and dose-conversion coefficient constituting additional methodological assumptions [26]. The 14% exceedance should consequently be interpreted as a screening-level result under the stated exposure assumptions, rather than as evidence that 14% of occupants would receive annual effective doses exceeding 1 mSv y⁻¹ [9].

D. Radium Equivalent Activity (Ra_{eq})

The radium equivalent activity (Ra_{eq}) was used to assess the combined radiological contribution of the naturally occurring radionuclides ²³⁸U, ²³²Th, and ⁴⁰K in the black sand samples. This index provides a single value representing the total gamma radiation output, allowing comparison with the conventional reference value of 370 Bq kg⁻¹ for construction material. Radium equivalent activity (Ra_{eq}) was calculated using (8).

$$Ra_{eq} = A_U + 1.43A_{Th} + 0.077A_K \quad (8)$$

where A_U , A_{Th} , and A_K represent the activity concentrations of ²³⁸U, ²³²Th, and ⁴⁰K, respectively, in Bq kg⁻¹.

The component uncertainties used in the Ra_{eq} uncertainty propagation are expressed at the 95% confidence level, and the uncertainty was propagated using the corresponding 95% confidence uncertainties [27].

Table IV and Fig. V present the Ra_{eq} values for the sampled area.

Table IV. Radium equivalent for the studied samples.

Sample ID	RaEq (Bq kg ⁻¹)
SS01	349.62 ± 2.53
SS02	526.96 ± 3.42
SS03	458.26 ± 3.18
SS04	507.62 ± 3.56
SS05	437.51 ± 3.11
SS06	486.21 ± 3.71
SS07	471.21 ± 3.30
SS08	507.11 ± 3.70
SS09	430.02 ± 3.06
SS10	429.18 ± 3.04
SS11	498.80 ± 3.83
SS12	534.50 ± 3.73
SS13	538.57 ± 3.82
SS14	491.12 ± 3.74
SS15	413.62 ± 3.34
SS16	417.44 ± 3.35
SS17	372.12 ± 3.09
SS18	474.04 ± 3.77
SS19	545.53 ± 4.13
SS20	404.14 ± 3.12
SS21	500.48 ± 3.37
SS22	533.24 ± 3.55
SS23	474.77 ± 3.46
SS24	459.52 ± 3.52
SS25	587.59 ± 3.98
SS26	467.80 ± 3.53
SS27	308.20 ± 3.01
SS28	512.37 ± 3.19
SS29	567.80 ± 3.65
SS30	495.65 ± 3.56
SS31	555.41 ± 3.82
SS32	634.98 ± 3.48
SS33	437.43 ± 3.13
SS34	515.84 ± 3.61
SS35	532.71 ± 3.07
SS36	505.72 ± 3.36
SS37	491.26 ± 3.26
SS38	470.20 ± 3.22
SS39	499.00 ± 3.30
SS40	499.77 ± 3.29
SS41	590.78 ± 3.93
SS42	489.64 ± 3.34
SS43	558.54 ± 3.44
SS44	557.97 ± 3.43
SS45	556.67 ± 3.61
SS46	490.98 ± 3.01
SS47	508.63 ± 3.36
SS48	471.78 ± 3.32
SS49	547.31 ± 3.61
SS50	500.45 ± 3.27
SS51	513.19 ± 3.42
SS52	510.43 ± 3.37
SS53	378.55 ± 3.09
SS54	235.40 ± 3.25
SS55	389.77 ± 3.08
SS56	313.61 ± 2.93
Mean	481.38 ± 14.32

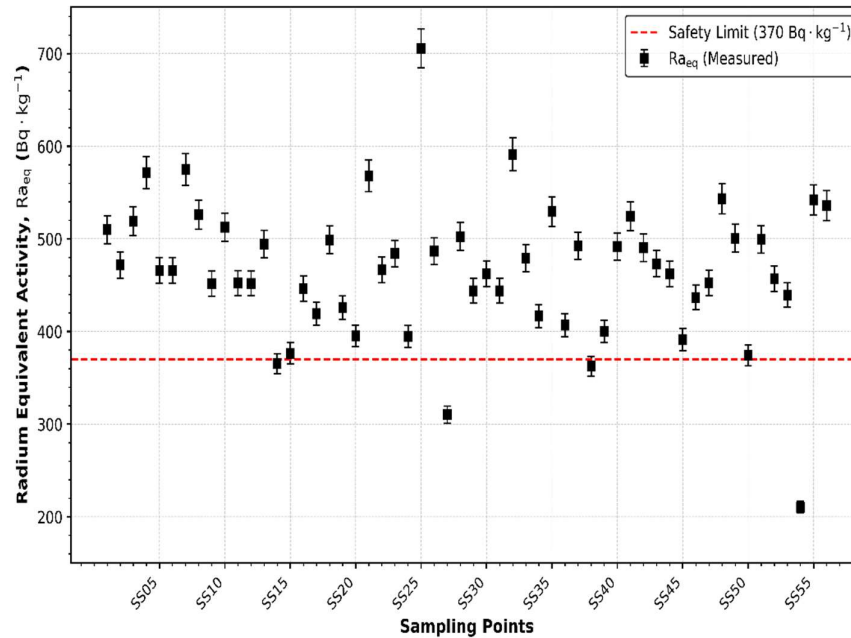


Fig. 5. Radium equivalent of black sand samples.

The Ra_{eq} values for the 56 samples ranged from 235.40 ± 3.25 to 634.98 ± 3.48 Bq kg⁻¹, with a mean of 481.38 ± 14.32 Bq kg⁻¹. Of the 56 samples, 52 (92.9%) exceeded the conventional reference value of 370 Bq kg⁻¹. This indicates that most of the sampled sand had Ra_{eq} values above the conventional screening criterion. The observed variation may reflect differences in the composition and distribution of radionuclide-bearing materials; however, specific mineralogical controls and spatial patterns were not established in the present study [19].

E. Hazard Indices

While radium equivalent activity provides a screening-level measure of total gamma radiation, the external (H_{ex}) and internal (H_{in}) hazard indices give more detailed information about potential exposure pathways [28]. These indices were calculated from the activity concentrations of ²³⁸U, ²³²Th, and ⁴⁰K using standard methodologies. H_{ex} estimates the risk from direct, external gamma radiation exposure from these radionuclides. Conversely, H_{in} is an indirect indicator of potential internal exposure associated with the radionuclides present in the materials [28]. The hazard indices and their corresponding uncertainty propagation, assuming independent uncertainties in the measured activity concentrations, were calculated using (9) and (10).

$$H_{ex} = \frac{A_U}{370} + \frac{A_{Th}}{259} + \frac{A_K}{4810} \quad (9)$$

$$H_{in} = \frac{A_U}{185} + \frac{A_{Th}}{259} + \frac{A_K}{4810} \quad (10)$$

where A_U , A_{Th} and A_K represent the mean concentration (in Bq kg⁻¹) of ²³⁸U, ²³²Th, and ⁴⁰K, respectively. The corresponding uncertainties were propagated using (11) and (12).

$$\sigma(H_{ex}) = \sqrt{\left[\left(\frac{\sigma_u}{370}\right)^2 + \left(\frac{\sigma_{Th}}{259}\right)^2 + \left(\frac{\sigma_K}{4810}\right)^2\right]} \quad (11)$$

$$\sigma(H_{in}) = \sqrt{\left[\left(\frac{\sigma_u}{185}\right)^2 + \left(\frac{\sigma_{Th}}{259}\right)^2 + \left(\frac{\sigma_K}{4810}\right)^2\right]} \quad (12)$$

where σ_u , σ_{th} , and σ_k represent the uncertainties in the measured activities of ²³⁸U, ²³²Th, and ⁴⁰K, respectively, expressed at the 95% confidence level. The potassium term uses 4810 Bq kg⁻¹ as the denominator because the conventional hazard-index formulation is derived from the radium-equivalent relationship, in which the ⁴⁰K contribution is expressed using its corresponding radium-equivalent conversion coefficient. The results of H_{in} and H_{ex} are presented in Table V.

Table V. External and internal hazard indices.

Index	Mean	Maximum	Minimum
H_{ex}	1.30 ± 0.04	1.91 ± 0.10	0.57 ± 0.03
H_{in}	1.78 ± 0.06	2.68 ± 0.13	0.81 ± 0.04

The H_{in} for the analyzed black sand samples ranged from 0.81 ± 0.04 to 2.68 ± 0.13 , while the H_{ex} ranged from 0.57 ± 0.03 to 1.91 ± 0.10 .

The mean values of H_{ex} (1.30 ± 0.04) and H_{in} (1.78 ± 0.06) both exceeded unity, the conventional screening criterion for further radiological assessment. The exceedance of H_{in} indicates the need for further assessment of potential internal exposure pathways; however, H_{in} is not a measurement of actual radon exposure. Radon exhalation rates and indoor radon concentrations were not measured in this study; therefore, direct radon measurements are recommended to characterize potential internal exposure pathways [29–30]. An H_{ex} or H_{in} value greater than unity is conventionally used as a

screening criterion warranting further radiological assessment of materials intended for construction use. For Kenya, these findings emphasize the importance of considering radiological screening when evaluating construction materials containing naturally occurring radionuclides. Further studies, especially measurements of radon exhalation and indoor radon concentrations, are encouraged to characterize potential exposure pathways.

IV. CONCLUSION

This study evaluated natural radioactivity in black construction sand from Longonot, Kenya. The mean activity concentrations of ^{232}Th , ^{238}U , and ^{40}K exceeded the corresponding global averages, and the mean absorbed dose rate was higher than the reported global average. The mean indoor annual effective dose was $0.82 \pm 0.02 \text{ mSv y}^{-1}$, with approximately 14% of the samples yielding screening-level indoor AED estimates above 1 mSv y^{-1} under the stated occupancy and dose-conversion assumptions. The mean radium equivalent activity was $481.38 \pm 14.32 \text{ Bq kg}^{-1}$, exceeding the conventional reference value of 370 Bq kg^{-1} . The mean external and internal hazard indices were 1.30 ± 0.04 and 1.78 ± 0.06 , respectively. Both exceeded unity, indicating the need for further assessment of potential external gamma exposure and radon-related exposure pathways.

Overall, the results indicate elevated radiological characteristics in the sampled Longonot black sand. However, this study did not measure radon exhalation rates or indoor radon concentrations; therefore, it cannot quantify the potential internal exposure pathway from the present data. Further studies should directly measure radon exhalation and indoor radon concentrations in buildings constructed with this sand to better characterize potential exposure.

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