

Vertical Distribution and Geochemical Relationships of ^{137}Cs and Natural Radionuclides in Soils of the Gilan Mountain Region, Uzbekistan

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Abstract

This study assessed the vertical distribution of natural (^{226}Ra , ^{232}Th , ^{40}K) and anthropogenic (^{137}Cs) radionuclides in soil profiles on two opposing hillslopes of the Gilan mountainous region in Uzbekistan, with their relationships with elemental composition (EDXRF), radon flux density, and ambient equivalent dose rate. Activity concentrations ranged from 21.9 to 30.2 Bq kg⁻¹ for ^{226}Ra , 39.7 to 45.6 Bq kg⁻¹ for ^{232}Th , and 569 to 706 Bq kg⁻¹ for ^{40}K , while ^{137}Cs peaked at the surface (up to 32.4 Bq kg⁻¹) and then declined with depth to 8.2–10.7 Bq kg⁻¹, which confirmed its anthropogenic origin. Spearman analysis revealed a strong ^{232}Th – ^{40}K correlation ($\rho = 0.73$; $p < 0.001$), whereas ^{226}Ra showed weaker, complex behavior and ^{137}Cs correlated weakly with elemental composition. Radon flux density and dose rate showed divergent relationships with soil radionuclides. These findings clarify radionuclide migration and geochemical controls in mountainous ecosystems.

Article Highlights

- Radioactive cesium accumulates in the topsoil of mountain slopes and decreases rapidly with depth.
- Natural radioactivity in the soil varies between the two slopes because they have different rock types and weathering processes.
- Radon release from soil and gamma radiation levels are influenced by multiple soil properties rather than by a single factor.

1. Introduction

Natural ionizing radiation remains one of the primary sources of radiation exposure to the global population [1]. According to the most recent assessments by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), a substantial portion of the annual effective dose received by the public from natural sources is attributable to radon, thoron and their short-lived decay products, cosmic radiation, and naturally occurring radionuclides present in the Earth's crust [2].

Soil constitutes one of the most important natural matrices for evaluating the environmental behavior of radionuclides, as it integrates the combined influences of parent rock composition, weathering processes, moisture regime, particle redistribution, organic matter accumulation, and hydrological factors [3, 4]. The International Atomic Energy Agency (IAEA) recommends soil and vegetation sampling programs as a fundamental methodological basis for environmental radiological monitoring. In mountainous regions, where slope exposure, surface runoff, infiltration, erosion, and accumulation processes act simultaneously, the spatial and vertical distribution of radionuclides is frequently heterogeneous [5]. Accordingly, natural mountain soils serve as convenient model environments for jointly evaluating lithogenic controls and secondary migrational redistribution processes.

Among the naturally occurring radionuclides, ^{226}Ra , ^{232}Th , and ^{40}K are of particular radioecological significance, as they represent the principal sources shaping the natural gamma-ray background of soils [6, 7]. Their concentrations and mutual ratios in soil are governed primarily by the mineral composition and geochemical characteristics of the parent rocks, as well as by the degree of weathering. Recent studies have demonstrated that ^{40}K is typically closely associated with the overall mineral matrix of the soil, ^{232}Th exhibits relatively conservative behavior as a lithogenic radionuclide, whereas ^{226}Ra tends to be more mobile and susceptible to redistribution processes. Consequently, analyzing the interrelationships among these radionuclides and their correlations with soil elemental composition is of considerable importance for identifying the geochemical controlling factors [8–10].

In contrast to naturally occurring radionuclides, ^{137}Cs is predominantly of anthropogenic origin and is characteristically concentrated in the upper soil layers. Contemporary research has shown that ^{137}Cs typically exhibits elevated values within the 0–5 or 0–10 cm surface layer, decreasing with increasing depth. Its distribution pattern is intimately linked to fine-grained particles, the clay fraction, organic matter content, and erosion–accumulation processes. For this reason, radiocesium is regarded not only as an artificial radionuclide but also as a sensitive indicator of surface layer redistribution on sloped terrains [11–13].

Radon flux density represents an important parameter in the assessment of environmental radioactivity, linking the radium content in soils to the process of radon gas emanation at the surface. The World Health Organization (WHO) recognizes radon as a significant environmental risk factor for lung cancer [1]. However, radon exhalation from soil is determined not solely by ^{226}Ra content but also by porosity, gas permeability, moisture content, meteorological conditions, and the availability of migration pathways [14–16]. Therefore, evaluating the relationships between radon flux density and radionuclide concentrations, particularly under natural mountain conditions, is essential for understanding the complex mechanisms of radiological processes.

The ambient equivalent dose rate, in turn, is determined not by a single radionuclide but by the cumulative contribution of gamma-emitting radionuclides in the soil [17]. Consequently, its spatial distribution does not always correspond directly with radon flux density. In mountainous terrains, redistribution along hillslopes may alter the ratios of ^{226}Ra , ^{232}Th , and ^{40}K in the surface layer, with the result that radon exhalation and the external gamma-ray background may be governed by different factors [18, 19]. Recent studies indicate that dose rate is often more strongly correlated with the overall natural radionuclide background of the soil, while radon-related indicators are more sensitive to the radium source and the physical properties of the soil [20–22]. For this reason, the combined analysis of radionuclide vertical profiles, elemental composition, radon flux density, and gamma dose rate enables a more complete characterization of radioecological processes.

Although the number of studies on environmental radioactivity continues to grow, investigations that integrate the vertical profiling of radionuclides, their distribution along opposing hillslopes, correlations with EDXRF-determined elemental composition, radon flux density, and ambient equivalent dose rate

within a unified framework remain relatively scarce. This gap is particularly relevant for natural mountainous ecosystems with minimal anthropogenic influence. From this perspective, the present study aimed to determine the vertical distribution of ^{226}Ra , ^{232}Th , ^{40}K , and ^{137}Cs across soil profiles on two opposing hillslopes of the Gilan mountain region, to evaluate their correlational relationships with EDXRF-determined elemental composition, and to analyze the associations of radon flux density and ambient equivalent dose rate with near-surface radionuclide concentrations. Such an integrated approach contributes to a deeper understanding of radionuclide migration, geochemical controlling factors, and the formation mechanisms of radiological indicators in natural mountainous environments.

2. Materials and Methods

2.1. Study Area

The study area is located in the Gilan mountainous region within the Shahrisabz district of Kashkadarya province, in the southern part of Uzbekistan. The village of Gilan is situated on the western slopes of the Pamir–Alay mountain system, in a mountainous zone near the border with Tajikistan, characterized by high-altitude terrain and a strongly dissected relief. The study site represents a natural mountainous ecosystem with a relatively low degree of anthropogenic influence. The Gilan area belongs to the natural mountain landscapes associated with the Hisar mountain system, where the annual precipitation has been recorded at approximately 550–750 mm [23]. Such natural conditions directly influence soil moisture, surface runoff, infiltration processes, and both the vertical and lateral migration of radionuclides.

The climate of the region is montane-continental in character, with relatively cold winters, moist springs, and hot, dry summers. This area serves as a convenient model site for evaluating the interrelationships among radionuclide distribution, radon flux density, and gamma dose rate under natural conditions.

The principal rationale for selecting this area as the study site is that it affords the opportunity to evaluate the distribution of radionuclides across soil layers in a natural mountain setting, their migration under hillslope conditions, and the relationships of radon flux density and gamma dose rate with naturally occurring radionuclides. Accordingly, the Gilan mountainous region may be regarded as a representative model site for investigating natural radioecological processes.

2.2. Sample Collection and Analysis

2.2.1. Determination of natural radionuclides in soil

Two hillslopes of the mountain ridge were selected as sampling areas. A total of 10 sampling points were established on the first hillslope (Side A) and 6 on the second hillslope (Side B) (Fig. 1). At each sampling point, a $30 \times 30 \text{ cm}^2$ plot was delineated, and soil samples were collected from four depth intervals: 0–5 cm, 5–10 cm, 10–15 cm, and 15–20 cm, yielding a total of 64 individual samples. Each soil sample weighed approximately 2 kg; all samples were labeled using paper tags and markers and

were placed in polyethylene bags for transport to the laboratory. The geographic coordinates of each sampling point were recorded using a portable GPS device.

Organic matter and small stones were removed from the soil samples, which were then passed through a 5 mm sieve. The samples were first air-dried in an open environment and subsequently oven-dried at 110°C for 24 hours until a constant dry weight was achieved. To establish secular equilibrium between the decay products of the ^{238}U and ^{232}Th decay chains, the dried samples were sealed in standard Marinelli plastic containers and stored for 30 days in accordance with the IAEA guidelines for measuring radionuclides in food and the environment [24].

The prepared soil samples were analyzed using a NaI(Tl) scintillation gamma-ray spectrometer with a crystal size of $\varnothing 80 \times 80$ mm and an energy resolution of 8.5% at the 661 keV gamma line of ^{137}Cs . The spectrometer was calibrated using a standard ^{232}Th source. For measurement, the samples were placed in Marinelli containers with a layer thickness of 8 cm, positioned within lead shielding, and measured for 2 hours under controlled laboratory conditions of temperature and humidity. The gamma-spectrometric analysis of the soil samples was performed following a previously described methodological approach [25], employing a NaI(Tl) detector-based gamma spectrometer, Marinelli geometry, and standard density correction coefficients.

The activity concentrations of ^{226}Ra , ^{232}Th , and ^{40}K were calculated by normalizing the measured spectra against standard sources provided by OMACH, with density-dependent correction coefficients applied to convert the spectra to unit density. The quantitative determination utilized the normalized form of spectral area ratios, i.e., $\alpha = S_{\text{ref}}/S_0$, based on reference source spectra [26].

To ensure the reliability of the analytical results, a 250 mL mixed certified volumetric standard containing radium, thorium, potassium, and cesium was employed. The determination of radionuclide activity concentrations in soil samples was carried out following the certified methodology of STC RADEK (St. Petersburg, Russia) [27].

The minimum detectable activity concentrations at a 2-hour measurement time were 8 Bq kg⁻¹ for ^{226}Ra , 6 Bq kg⁻¹ for ^{232}Th , and 30 Bq kg⁻¹ for ^{40}K (RADEK, 2005). The relative measurement uncertainty for Naturally Occurring Radioactive Material (NORM) radionuclides was accepted as 10%.

The overall measurement uncertainty incorporated contributions from sample preparation, sample mass measurement (5%), and gamma-spectrometric measurement uncertainty (10%). Uncertainties were evaluated using standard calculation methods.

2.2.2. Elemental analysis of soil samples by EDXRF

Well-dried soil samples were ground in a ball mill to a particle size of 0.5–2 mm. To homogenize the granulometric composition of the ground samples and to improve analytical accuracy, the samples were passed through a uniform-sized sieve. The macro- and micro-elemental compositions of the samples

were determined using a Rigaku NEX DE VS (Rigaku, Japan) energy-dispersive X-ray fluorescence (EDXRF) spectrometer [28, 29]. For analysis, the lower surfaces of dedicated measurement cups were covered with a 4 µm thick X-ray transmitting film, and the prepared soil samples were loaded into these cups. Elemental concentrations were calculated using the fundamental parameters (FP) method [30]. Measurements were performed with a 10 mm diaphragm, without sample spinning, and with an optimized measurement sequence. Two excitation conditions were applied: a Low-Z regime with an open filter, a tube voltage of 6.5 kV, a current of 502 µA, and a measurement time of 100 s; and a Mid-Z regime with a C filter, a tube voltage of 35.0 kV, a current of 342 µA, and a measurement time of 100 s. All calculations and spectral processing were carried out using the standard FP algorithm of the Rigaku system [31].

2.2.3. Radon flux density determination

To achieve a more complete radiological characterization of the study area, radon flux density (RFD) from the soil surface was also measured. A further objective was to determine the correlation between RFD and ^{226}Ra in the soil in natural settings. The measurement method is based on the passive adsorption of gaseous radon onto an activated carbon sorbent, followed by gamma-spectrometric determination of the activity of its short-lived decay products. A passive radon sorption detector (PRSD) based on activated carbon was employed for sampling. Prior to measurement, the detector was degassed in a drying cabinet for 1 hour [27]. Its mass was then weighed to a precision of 0.01 g, and the detector was placed in a hermetically sealed transport cassette. Ambient air sampling from the soil surface was conducted in accordance with a previously published methodology [32]. During sampling operations, meteorological parameters—including temperature, atmospheric pressure, relative humidity, and wind speed—were monitored in the field and were found to remain essentially stable throughout the measurement period.

The PRSD exposure duration was set at 3 hours. Following exposure, the detector was retrieved from the collection chamber and transported to the laboratory for gamma-spectrometric analysis (Fig. 2). The interval between the end of exposure and gamma-spectrometric measurement of the PRSD sorbent was 3 hours, corresponding to the time required for the re-establishment of radioactive equilibrium between ^{222}Rn and its short-lived decay products (^{214}Pb , ^{214}Bi , ^{214}Po , and ^{218}Po). The gamma spectrum of the exposed PRSD was analyzed using the ASW (Analysis Software for Windows) software developed by STC RADEK, in which the ^{222}Rn component was isolated (Fig. 3) [33]. Measurement uncertainties were calculated at a 0.95 confidence level in accordance with GOST-R-8.736–2011 [34] and ISO 5725 [35].

The spectrometer geometry was calibrated using a standard ^{226}Ra source matching the sorbent geometry (Ø30 × 10 mm). Using the ASW software, the gamma spectrum of the sample sorbent was acquired over 1800 s, and the volumetric activity of ^{222}Rn gas was calculated. The activities at the 609, 1120, and 1764 keV gamma lines of ^{214}Bi , a decay product of ^{222}Rn , were determined by the “intervals” method. In this approach, the energy scale of the gamma spectrometer is divided into three ranges 125–

500 keV, 500–1000 keV, and 1000–1900 keV, and the summed activities within each interval are used as an aggregate indicator [36].

For quantitative determination of ^{222}Rn , the PRSD sorbent was measured for 1800 s using a NaI(Tl) crystal gamma spectrometer positioned within 5 cm thick lead shielding. The sorbent exposure duration was entered into the ASW software package. The quantitative estimate of ^{222}Rn was based on the relative intensities of the γ -lines corresponding to its short-lived decay products, specifically the 352 keV and 1764 keV γ -lines, which are characterized by high quantum yields. Based on the acquired spectral data, the ASW software computed the radon flux density in units of $\text{mBq m}^{-2} \text{ s}^{-1}$.

2.2.4. Ambient equivalent dose rate

At each sampling point, the ambient equivalent dose rate (EDR) of environmental gamma radiation was measured directly using a portable RadEye G20 Gamma Survey Meter. Measurements were conducted in situ under field conditions at a height of approximately 1 m above the ground surface, and results were recorded in $\mu\text{Sv h}^{-1}$. The instrument was used for rapid determination of the gamma-radiation dose rate, and the measurement results were documented together with the corresponding sampling data for each observation point [37, 38].

2.2.5. Statistical analysis

Statistical analysis and graphical visualization of the data were performed using the Google Colab Python environment. Given the relatively small sample size and the likelihood that the distributions of radionuclides and radiological parameters in mountain slope soils may deviate from normality and strict linearity, Spearman's rank correlation coefficient was employed to assess relationships among variables.

3. Results and Discussion

3.1. Vertical distribution of radionuclides across soil profiles on two hillslopes of the mountain ridge

To determine the vertical distribution of radionuclides on the two hillslopes (Side A and Side B) of the mountain ridge, the specific activities of ^{226}Ra , ^{232}Th , ^{40}K , and ^{137}Cs were evaluated across four depth intervals. The mean values and their variations with depth for each hillslope are presented in Tables 1 and 2. These data facilitate a comparative analysis of the migration behavior of natural radionuclides and the surface accumulation characteristics of ^{137}Cs .

Table 1
Summary of radionuclide activity concentrations by depth in soils on Side A of the ridge

Depth (cm)	^{226}Ra (Bq kg ⁻¹)	^{232}Th (Bq kg ⁻¹)	^{40}K (Bq kg ⁻¹)	^{137}Cs (Bq kg ⁻¹)
0–5	21.93 ± 4.43	42.80 ± 5.05	666.59 ± 126.95	32.36 ± 19.18
5–10	24.04 ± 3.32	45.20 ± 4.15	688.37 ± 120.78	16.63 ± 9.61
10–15	23.84 ± 4.66	44.73 ± 1.92	706.33 ± 100.14	10.10 ± 6.88
15–20	23.73 ± 3.87	45.56 ± 3.62	702.33 ± 99.28	8.24 ± 2.04

On Side A of the mountain ridge, a slight enrichment of natural radionuclides was observed in the subsurface layers relative to the topsoil. The mean activity concentration of ^{226}Ra was 21.93 Bq kg⁻¹ in the 0–5 cm layer, increasing to 24.04 Bq kg⁻¹ in the 5–10 cm layer, and remaining relatively stable at 23.84 and 23.73 Bq kg⁻¹ in the 10–15 and 15–20 cm layers, respectively. ^{232}Th followed a similar pattern, rising from 42.80 Bq kg⁻¹ at the surface to 45.20–45.56 Bq kg⁻¹ in the deeper intervals. ^{40}K likewise showed higher values in the subsurface layers (688.37, 706.33, and 702.33 Bq kg⁻¹) compared with the surface layer (666.59 Bq kg⁻¹). In marked contrast, ^{137}Cs exhibited its highest value at the surface (32.36 Bq kg⁻¹), declining consistently with depth to 16.63, 10.10, and 8.24 Bq kg⁻¹. Overall, Side A was characterized by a modest vertical redistribution of natural radionuclides and a pronounced surface accumulation of ^{137}Cs .

Table 2
Summary of radionuclide activity concentrations by depth in soils on Side B of the mountain range

Depth (cm)	^{226}Ra (Bq kg ⁻¹)	^{232}Th (Bq kg ⁻¹)	^{40}K (Bq kg ⁻¹)	^{137}Cs (Bq kg ⁻¹)
0–5	28.34 ± 5.14	40.64 ± 5.62	569.08 ± 80.68	25.88 ± 12.54
5–10	30.18 ± 6.87	40.40 ± 6.05	592.16 ± 85.94	23.24 ± 19.88
10–15	27.52 ± 9.87	39.70 ± 4.51	579.42 ± 84.05	13.50 ± 9.78
15–20	27.10 ± 6.66	40.68 ± 1.29	589.56 ± 45.27	10.72 ± 5.65

On the second hillslope (Side B), the vertical profiles of natural radionuclides were relatively more uniform compared with Side A. The mean ^{226}Ra values were 28.34, 30.18, 27.52, and 27.10 Bq kg⁻¹ across the four depth intervals, with only a slight increase at the 5–10 cm depth followed by a decline. ^{232}Th remained nearly invariant, ranging from 39.70 to 40.68 Bq kg⁻¹. ^{40}K similarly showed limited variation, with values of 569.08, 592.16, 579.42, and 589.56 Bq kg⁻¹, respectively. ^{137}Cs on Side B also exhibited a surface maximum (25.88 Bq kg⁻¹), decreasing with depth to 23.24, 13.50, and 10.72 Bq kg⁻¹. Thus, on Side B, the migrational trend for natural radionuclides was less pronounced, although the depth-dependent distribution of ^{137}Cs remained clearly evident.

A comparison between the two hillslopes revealed that radionuclide concentrations differ not only with depth but also in relation to opposing aspects of the relief. ^{226}Ra was consistently higher on Side B across all depth intervals. For instance, in the 0–5 cm layer, the mean values were 21.93 Bq kg^{-1} on Side A and 28.34 Bq kg^{-1} on Side B; in the 5–10 cm layer, these were 24.04 and 30.18 Bq kg^{-1} , respectively. Conversely, ^{232}Th and, particularly, ^{40}K were higher on Side A. Notably, the mean ^{40}K in the 10–15 cm layer was $706.33 \text{ Bq kg}^{-1}$ on Side A compared with $579.42 \text{ Bq kg}^{-1}$ on Side B.

Regarding vertical trends, a notable difference was observed between the hillslopes: on Side A, ^{226}Ra , ^{232}Th , and ^{40}K exhibited a slight enrichment in the subsurface layers, whereas on Side B these radionuclides displayed a relatively flat profile. On both hillslopes, ^{137}Cs attained its maximum in the surface layer and decreased consistently with depth; however, this decline was substantially more pronounced on Side A (from 32.36 to 8.24 Bq kg^{-1}) than on Side B (from 25.88 to 10.72 Bq kg^{-1}). Overall, Side A showed a comparatively more active vertical redistribution of natural radionuclides, while Side B exhibited a more stable profile; the surface enrichment of ^{137}Cs on both hillslopes confirms its anthropogenic origin and strong association with the upper soil horizon [39].

As illustrated in Fig. 4, on the first hillslope, a slight enrichment trend for ^{226}Ra , ^{232}Th , and ^{40}K is observed in the lower layers, whereas on the second hillslope these radionuclides form a relatively stable profile [37, 39, 40]. ^{137}Cs on both hillslopes reaches its maximum in the surface layer and decreases consistently with increasing depth. This pattern clearly demonstrates the contrasting behavior of natural radionuclides and anthropogenic ^{137}Cs within the soil profile [41].

3.2. Correlational relationships between radionuclides and soil elemental composition

To provide a geochemical interpretation of the observed differences in vertical radionuclide distribution, this section evaluates the inter-radionuclide correlations and their associations with the EDXRF-determined elemental composition of the soil. The relationships among ^{226}Ra , ^{232}Th , ^{40}K , and ^{137}Cs , as well as their correlations with major elements, were analyzed to identify the probable factors controlling their distribution.

Figure 5 presents the correlational relationships among the radionuclides. In the scatter plots, data points represent individual samples and the lines indicate the overall trend. According to the Spearman correlation analysis, the most pronounced and statistically significant association was observed between ^{232}Th and ^{40}K ($\rho = 0.73$; $p < 0.001$). This strong positive correlation indicates that these two natural radionuclides co-vary in relation to the overall mineral matrix and lithogenic components of the soil [42].

The associations between other radionuclide pairs were substantially weaker. In particular, the correlation between ^{232}Th and ^{226}Ra was virtually absent ($\rho = -0.10$; $p = 0.452$), indicating that radium

and thorium do not exhibit concordant migrational behavior within the soil profile [43]. Thorium, being a relatively immobile element, largely retains its original distribution in the soil [44], whereas radium, being prone to aqueous transport, undergoes redistribution across soil layers [45]. Similarly, the correlation between ^{226}Ra and ^{40}K was weak and statistically insignificant ($\rho = -0.19$; $p = 0.150$). A weak negative association was noted between ^{232}Th and ^{137}Cs ($\rho = -0.23$; $p = 0.075$), though this too did not reach statistical significance [39, 43]. Collectively, these results confirm that, among the natural radionuclides, the coherent co-variation of ^{232}Th and ^{40}K is particularly robust, while ^{137}Cs , as an anthropogenic radionuclide distributed by a fundamentally different mechanism, exhibits distinctive behavior.

The Spearman correlational relationships of ^{226}Ra , ^{232}Th , ^{40}K , and technogenic ^{137}Cs with the major elements determined by EDXRF were also evaluated. The correlation analysis revealed that, among the natural radionuclides, ^{40}K and ^{232}Th formed relatively stable and geochemically interpretable associations with the soil mineral composition, while ^{226}Ra exhibited substantially weaker and more complex geochemical behavior. The correlations of ^{137}Cs , in the majority of cases, were markedly different from those of the natural radionuclides, being consistently weak, which further corroborated its anthropogenic origin and association with the surface layer (Figs. 6 and 7) [46, 47].

According to the heatmap results for the first hillslope, ^{40}K displayed a very strong positive correlation with EDXRF-determined K, demonstrating excellent mutual consistency between the gamma-spectrometric and EDXRF results. Furthermore, ^{40}K showed notable positive associations with Al, Fe, and Rb, indicating a close linkage with potassium-bearing aluminosilicate and lithogenic mineral components. This finding is geochemically logical and suggests that ^{40}K may be considered an indicator reflecting the overall silicate and potassium-bearing mineral matrix of the soil.

Several important associations were also observed for ^{232}Th , which showed positive correlations with Al, Fe, Rb, Zr, K, and SiO_2 , indicating that thorium distribution is governed predominantly by lithogenic and mineralogical factors. The statistically significant positive association observed in the Ti– ^{232}Th scatter plot further supports the linkage of thorium with heavy mineral fractions and relatively resistant mineral components. Overall, the coherent co-variation of ^{232}Th with multiple elements signifies that it is a comparatively conservative radionuclide, more stably associated with the lithogenic background within the soil profile.

In contrast, the associations of ^{226}Ra with elements were considerably weaker. The first hillslope heatmap showed weak positive associations with Al and K and a weak negative association with Ca; however, the selected scatter plots for the Fe– ^{226}Ra and Ca– ^{226}Ra pairs did not reveal statistically significant correlations. This finding indicates that the distribution of ^{226}Ra is shaped not by any single element but rather by the combined influence of multiple geochemical factors. Although radium is often considered a geochemical analog of Ca, Sr, or Ba, its distribution in the present soil samples cannot be explained by a simple linear Ca-controlled model. This suggests that, in the studied area, the migration

and retention of ^{226}Ra may be related not only to carbonate phases but also to sorption centers, mineral surfaces, and other soil properties.

The correlational behavior of ^{137}Cs differed markedly from that of the other radionuclides. In the heatmaps, its associations with the majority of elements were weak or statistically insignificant. This indicates that ^{137}Cs is not subject to the lithogenic geochemical controls characteristic of natural radionuclides but rather possesses a distribution governed predominantly by surface accumulation, association with fine-grained fractions, and anthropogenic fallout. Thus, the correlation analysis confirmed that ^{137}Cs requires separate evaluation as a radionuclide with a fundamentally distinct distribution mechanism compared with ^{226}Ra , ^{232}Th , and ^{40}K .

The selected scatter plots visually confirmed the principal tendencies observed in the heatmaps. In particular, the strong positive association between K and ^{40}K is clearly expressed by the relatively consistent trend formed by the data points. In contrast, the Fe– ^{226}Ra and Ca– ^{226}Ra pairs exhibited widely scattered and inconsistent data points, confirming that radium does not form a simple monotonic relationship with these elements. The Ti– ^{232}Th pair showed a positive trend, reinforcing the conclusion of lithogenic association for thorium with mineral components (Fig. 8).

In summary, the correlation analysis demonstrated that the distribution of ^{40}K and ^{232}Th is predominantly controlled by the overall mineral composition and lithogenic components of the soil, while ^{226}Ra is characterized by more complex and weaker geochemical associations. The essentially independent behavior of ^{137}Cs once again confirms its anthropogenic origin and its migration mechanism linked to the surface horizon. The correlational differences within the radionuclide–element system indicate that radionuclide distribution in the studied area is shaped not by a single mechanism but by a combination of diverse geochemical and migrational factors.

3.3. Radon flux density (RFD) and ambient equivalent dose rate (EDR): distribution patterns and relationships with radionuclides

Radon flux density (RFD) and the ambient equivalent dose rate (EDR) exhibited pronounced spatial variability on both hillslopes of the study area (Fig. 9). On Side A, RFD values ranged from 47.16 to 217.7 $\text{mBq m}^{-2} \text{s}^{-1}$, with particularly notable local maxima at points 4–7. On Side B, RFD values increased progressively toward the higher-numbered points, reaching 156.5 $\text{mBq m}^{-2} \text{s}^{-1}$ at point 12 and 143.98 $\text{mBq m}^{-2} \text{s}^{-1}$ at point 15. Despite local differences, the overall mean levels of RFD were of comparable magnitude on both hillslopes, indicating that the potential for radon emanation is established in both hillslope systems, although its spatial manifestation differs.

EDR values varied within a narrower range relative to RFD: from 0.129 to 0.233 $\mu\text{Sv h}^{-1}$ on Side A, and from 0.110 to 0.200 $\mu\text{Sv h}^{-1}$ on Side B. In general, EDR values were higher on Side A, with prominent maxima at points 8 and 9, while on Side B the distribution was comparatively smoother, with a notable

increase only at point 15. This indicates that external gamma radiation is distributed somewhat more uniformly than radon exhalation, yet remains sensitive to local enrichments in gamma-emitting radionuclides.

An important observation from Fig. 9 is that the spatial trends of RFD and EDR do not fully coincide. At certain points where RFD was high, EDR was not at its maximum; conversely, at some points with elevated EDR, the RFD was not particularly high. Although these two radiological parameters are interrelated, they are partially governed by different factors [48]. RFD is primarily associated with radon generation in near-surface radium-rich layers and its exhalation from the soil, whereas EDR reflects the cumulative external gamma contribution of ^{226}Ra , ^{232}Th , and ^{40}K . Consequently, the lack of complete spatial correspondence between RFD and EDR maxima indicates that radon exhalation is also appreciably dependent on physical soil properties such as porosity, moisture content, and gas permeability [49–51].

The associations of the specific activities of natural radionuclides (^{226}Ra , ^{232}Th , and ^{40}K) with RFD and EDR were evaluated using Spearman's correlation coefficient (ρ). The results of the statistical analysis are presented graphically in Fig. 10(a–d).

As shown in Fig. 10a, a strong positive correlation was observed between the ^{226}Ra content of the soil and the surface radon exhalation rate (RFD) ($\rho = 0.62$, $p = 0.011$). This statistically significant association ($p < 0.05$) confirms that the radium content of the soil is a primary factor governing radon generation and its diffusive release to the ground surface [52–54].

The association between ^{226}Ra and EDR (Fig. 10b) was found to be weak and statistically insignificant ($\rho = -0.24$, $p = 0.365$). This indicates that, in the study area, the overall gamma-radiation background is not determined solely by the radium content. In contrast, the EDR exhibited notably higher positive correlations with ^{232}Th ($\rho = 0.55$, $p = 0.027$) and ^{40}K ($\rho = 0.59$, $p = 0.017$) (Figs. 10c and 10d). These results demonstrate that the contributions of thorium and potassium to the external gamma dose rate at the measurement points are more substantial than that of radium [9, 48, 55].

The correlational results obtained across the study area indicate that RFD and EDR are not governed by the same mechanisms in their relationships with radionuclides. Radon exhalation is primarily dependent on the radium content and the physical properties of the soil, whereas the ambient equivalent dose rate represents the aggregate effect of all naturally occurring gamma-emitting radionuclides in the upper soil layer.

4. Conclusions

The results showed that two distinct patterns govern the vertical distribution of radionuclides in the soils of the Gilan Mountains. The distribution of natural radionuclides was generally lithogenic, particularly for ^{232}Th and ^{40}K . This minor subsurface enrichment was noted on Side A, whereas Side B had a more

uniform profile, which may be due to the varying weathering and soil redistribution on the two sides of the slope. In contrast, the mobility of ^{226}Ra was not readily controlled by Ca; it was not lithogenic, indicating that mobility is not only regulated by sorption, but also by mineral surface processes. The distribution of ^{137}Cs was also different from that of the natural radionuclides, with the consistent high concentration in the topsoil on both slopes, as expected due to its origin in the atmosphere. Both radon flux density and ambient dose rate had different controlling factors. Exhalation of radon was mainly related to ^{226}Ra , and ambient dose rate to ^{232}Th and ^{40}K . This is also because their spatial maxima did not match, which indicates that both these radiological indicators offer complementary information.

Therefore, no single radiological parameter is sufficient to fully characterize soil radioactivity, and a combination of radiometric measurements and geochemical interpretation is necessary to better understand radionuclide behavior in mountainous environments.

The limitations of this study include the relatively small number of sampling points and the investigation of only two hillslopes within a single mountain range. The moisture content and porosity of the soil were not directly measured, but could affect radon exhalation. Further studies involving more mountain systems, more extensive sampling networks, seasonal sampling, and direct measurements of soil physical properties would help confirm these relationships and improve the applicability of these findings to radiological monitoring in other natural environments.

Declarations

Competing Interests

The authors declare no competing interests.

Ethics Approval

Not applicable.

Consent to Participate

Not applicable.

Consent to Publish

Not applicable.

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Author Contribution

Material preparation and data collection were performed by U.H., U.T., and D.H. Data analysis and review were performed by U.H., A.M., and S.K. The manuscript was written by U.H. and S.K. The study was supervised by A.M. and U.T. All authors read and approved the final manuscript.

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Data Availability

The data supporting the findings of this study are available from the corresponding author upon reasonable request.

References

1. World Health Organization (2023) Radon
2. United Nations Scientific Committee on the Effects of Atomic Radiation (2025) Sources, Effects and Risks of Ionizing Radiation: UNSCEAR 2024 Report, vol II. United Nations, New York
3. Osman R, Dawood YH, Melegy A et al (2022) Distributions and risk assessment of the natural radionuclides in the soil of Shoubra El Kheima, South Nile Delta. Egypt Atmos 13:98
4. Abbasi A, Kurnaz A, Turhan S, Mirekhtiary F (2020) Radiation hazards and natural radioactivity levels in surface soil samples from dwelling areas of North Cyprus. J Radioanal Nucl Chem 324:203–210
5. IAEA (2018) Regulations for the Safe Transport of Radioactive Material. International Atomic Energy Agency, Vienna
6. Lopez-Perez M, Martin-Luis C, Hernandez F et al (2021) Natural and artificial gamma-emitting radionuclides in volcanic soils of the Western Canary Islands. J Geochem Explor 229:106840
7. Pehlivanioğlu SA, Mancini S, Özden S et al (2024) Characterization of a typical urban soil in terms of natural radionuclide content: the case study of a university campus. Heliyon 10
8. Rihs S, Lascar E, Pourcelot L et al (2021) U- and Th-series disequilibria in separated soil mineral fractions: insight into the mechanism and timescale of U, Th and Ra redistribution. Chem Geol 583:120455
9. Chen T-C, Zeng F-C, Lin C-C et al (2024) Assessment of soil radioactivity associated with risk and correlation with soil properties near Maanshan Nuclear Power Plant, Taiwan. Appl Sci 14:9239
10. La Manna L, Gaspar L, González Comunian D, Navas A (2025) Assessment of environmental radionuclides and controlling factors in volcanic soils of Andean Patagonia. Geosciences 15:348

11. Guan Y, Cui L, Huang C et al (2024) Migration of fallout radionuclides and soil erosion of Hongsongwa mountainous in China. *ACS Earth Space Chem* 8:1000–1010
12. Sekudewicz I et al (2022) Factors controlling ¹³⁷Cs distribution in bottom sediments of Koronowo Reservoir (Poland). *J Soils Sediments*. <https://doi.org/10.1007/s11368-022-03326-5>
13. Koarashi J, Atarashi-Andoh M, Nishimura S (2023) Effect of soil organic matter on the fate of ¹³⁷Cs vertical distribution in forest soils. *Ecotoxicol Environ Saf* 262:115177
14. Guarino A, Cicchella D, Lima A, Albanese S (2022) Radon flux estimates, from both gamma radiation and geochemical data, to determine sources, migration pathways, and related health risk: the Campania region (Italy) case study. *Chemosphere* 287:132233
15. Čeliković I, Pantelić G, Vukanac I et al (2022) Overview of radon flux characteristics, measurements, models and its potential use for the estimation of radon priority areas. *Atmosphere* 13:2005
16. Hidayath M et al (2023) Studies on radon exhalation rate and activity of radioactive elements in soil samples. *J Geol Soc India*. <https://doi.org/10.1007/s12594-023-2340-y>
17. Günay O, Özden S, Aközcan Pehlivanioğlu S (2024) The current status of natural and artificial radiation in İstanbul 36 years after Chernobyl, preceding a potential nuclear threat. *J Radioanal Nucl Chem* 333:3819–3831
18. Lee J, Kim H, Kye YU et al (2023) Activity concentrations and radiological hazard assessments of ²²⁶Ra, ²³²Th, ⁴⁰K, and ¹³⁷Cs in soil samples obtained from the Dongnam Institute of Radiological & Medical Science, Korea. *Nucl Eng Technol* 55:2388–2394
19. Mekongtso Nguele EJ, Moyo Ndontchueng M, Motapon O (2016) Determination of ²²⁶Ra, ²³²Th, ⁴⁰K, ²³⁵U and ²³⁸U activity concentration and public dose assessment in soil samples from bauxite core deposits in Western Cameroon. *SpringerPlus* 5:1253
20. Hamed Y et al (2024) Environmental radioactivity, ecotoxicology (²³⁸U, ²³²Th and ⁴⁰K) and potentially toxic elements in water and sediments from North Africa dams. *Sustainability* 16:490
21. Ramzaev V, Bernhardsson C, Vodovatov A et al (2024) Ambient dose equivalent rates of gamma radiation from natural radionuclides and ¹³⁷Cs at grasslands and forests in the area of the Belarusian NPP in the pre-commissioning period (2019). *Radiat Prot Dosimetry* 200:496–503
22. Popoola OJ, Olubi OE, Adewalure OS et al (2025) Elevated natural radionuclides in soils and stream sediments: pollution, spatial distribution, radiological hazards, and cancer risks in peri-urban Emure-Ekiti, southwest Nigeria. *Discover Soil* 2:70
23. World Bank Group (2020) Climatology Data (1991–2020). In: Climate Change Knowledge Portal. <https://climateknowledgeportal.worldbank.org>. Accessed 31 May 2026
24. IAEA (1989) Measurement of Radionuclides in Food and the Environment: A Guidebook. International Atomic Energy Agency, Vienna, Austria
25. Azimov AN, Hushmuradov SK, Muminov IT et al (2008) Gamma-spectrometric determination of natural radionuclides and ¹³⁷Cs concentrations in environmental samples: the improved scintillation technique. *Radiat Meas* 43:66–71

26. Tukhtaev U, Khasanov S, Safarov A, Proshad R (2023) Assessment of radioactivity of the Sobirsoy reservoir and adjacent areas of the Nurabad district, Uzbekistan. *Radiochim Acta* 111:781–791. <https://doi.org/10.1515/ract-2023-0153>
27. RADEK (2005) Methodology for measuring the specific activity of radionuclides of Ra-226, Th-232, K-40, Cs-137, Sr-90 in samples of products from industrial enterprises, agricultural enterprises and environmental objects. Analytical complex. No. 805/05. D.I. Mendeleev All-Russian Institute for Metrology
28. Rigaku Corporation NEX DE Series: direct excitation energy dispersive X-ray fluorescence spectrometers
29. Takahara H, Kobayashi H (2021) Standardless FP XRF analysis for lithium ion battery electrode materials. *Rigaku J* 37:6–11
30. Croffie MET, Williams PN, Fenton O et al (2022) Rubidium measured by XRF as a predictor of soil particle size in limestone and siliceous parent materials. *J Soils Sediments* 22:818–830
31. Croffie MET, Williams PN, Fenton O et al (2020) Optimising sample preparation and calibrations in EDXRF for quantitative soil analysis. *Agronomy* 10:1309
32. Khasanov S, Akhmedova G, Tukhtaev U et al (2023) Evaluation of soil radioactivity in the areas of underground nuclear explosions at the Pomuk gas field site in Uzbekistan. *Environ Monit Assess* 195:1412
33. RADEK (2013) ASW software
34. GOST (2013) State system for ensuring the uniformity of measurements. State Standards of Russian Federation: Multiple direct measurements. Methods of measurement results processing. Main principles. State Standards of Russian Federation
35. International Organization for Standardization (1994) Accuracy (trueness and precision) of measurement methods and results – Part 2: Basic method for the determination of repeatability and reproducibility of a standard measurement method. International Organization for Standardization
36. Safarov A, Safarov A, Khasanov S et al (2023) Evaluation of radon hazards at the rural settlements of Uzbekistan. *Environ Monit Assess* 195:915. <https://doi.org/10.1007/s10661-023-11493-2>
37. Khasanov S, Tukhtaev U, Mamatkulov O et al (2024) Natural radionuclide profiles and radiological risks in soils and rocks of the Koytash–Ugam Range, Uzbekistan. *Environ Monit Assess* 196:847. <https://doi.org/10.1007/s10661-024-13015-0>
38. Thermo Fisher Scientific (2007) RadEye G20-10 / G20-ER10: X-ray and gamma survey meters for ambient equivalent dose and dose rate measurement
39. Ahmad AY, Al-Ghouti MA, AlSadig I et al (2019) Vertical distribution and radiological risk assessment of ¹³⁷Cs and natural radionuclides in soil samples. *Sci Rep* 9:12196
40. Yıldız Yorgun N, Oto B, Gür F et al (2022) Measurement of the concentration of ²²⁶Ra, ²³²Th, ⁴⁰K in the soil in settlements on the coastline of Lake Van, Turkey. *Radiat Prot Dosimetry* 198:1575–1584

41. Ivanov MM, Ivanova NN (2023) Express analysis of vertical distribution of ^{137}Cs to assess the rates of erosion and accumulation processes in the zone of intense radioactive contamination. *Eurasian Soil Sci* 56:524–533
42. Becegato VA, Becegato VR, Baum CA et al (2019) Multivariate statistical analysis correlating ^{238}U , ^{232}Th , and ^{40}K equivalent activities in soil to geochemical data from an agricultural area. *J Soils Sediments* 19:1901–1910
43. Ozaydin Ozkara R, Eke C, Boztosun I (2021) A study on the activity concentrations of ^{226}Ra , ^{232}Th , ^{40}K , ^{137}Cs and radiological risk assessments in soil samples from Seydisehir and Beysehir districts of Konya in Turkey. *J Radioanal Nucl Chem* 330:1017–1025
44. Ivanov MM, Kuzmenkova NV, Garankina EV, Tulyakov ED (2019) The ^{232}Th distribution in modern sediments near radioactive lovchorrite mine, the Khibiny Mountains, Kola Peninsula. *J Geochem Explor* 200:77–83
45. Chevychelov A, Sobakin P, Gorokhov A et al (2021) Migration of ^{238}U and ^{226}Ra radionuclides in technogenic permafrost taiga landscapes of Southern Yakutia. *Russia Water* 13:966
46. de Castilhos ND, Melquiades FL, Thomaz EL, Bastos RO (2015) X-ray fluorescence and gamma-ray spectrometry combined with multivariate analysis for topographic studies in agricultural soil. *Appl Radiat Isot* 95:63–71
47. Belyaeva O, Movsisyan N, Pyuskyulyan K et al (2024) Assessing radioactivity in agricultural soils: spatial distribution patterns, area-specific influences, and radiological hazard. *ACS Agricultural Sci Technol* 4:916–928
48. Devi V, Kumar A, Chauhan RP (2019) A study on radionuclides content and radon exhalation from soil of Northern India. *Environ Earth Sci* 78:506
49. Pyngrope A, Saxena A, Khardewsaw A et al (2022) Effect of soil's porosity and moisture content on radon and thoron exhalation rates. *J Radioanal Nucl Chem* 331:1975–1984
50. Pérez B, Stieff LC, Ponce-Amanca RE et al (2025) Edaphic and meteorological parameters as determinants of radon exhalation and its environmental implication in Peruvian agroecosystems. *Sci Rep* 15:22140
51. Zhao L, He Y, Cai Z et al (2024) Preliminary investigation of radon-222 flux density in the soil surface: a case study of campus, China. *J Radioanal Nucl Chem* 333:1107–1118
52. Mehta V, Kaur R, Shikha D et al (2024) A study of radon exhalation rates and activity concentration of ^{226}Ra , ^{232}Th and ^{40}K in soil samples of Roopnagar district, Punjab, India. *J Radioanal Nucl Chem* 333:2967–2974
53. Alene Assefa N (2025) Analysis of ^{226}Ra content and ^{222}Rn exhalation rates in soil samples from Wukro, Tigray, using SSNTDs. *F1000Research* 14:289
54. Djeufack LB, Bineng GS, Modibo OB et al (2022) Correlation between ground ^{222}Rn and ^{226}Ra and long-term risk assessment at the bauxite bearing area of Fongo-Tongo, Western Cameroon. *Radiation* 2:387–404

55. Onjefu SA, Ebibuloami BP, Kamunda C et al (2025) Natural radioactivity assessments and radiological hazards in agricultural soils around mining activities in Karibib. Namibia Nuclear Anal 4:100155

Figures

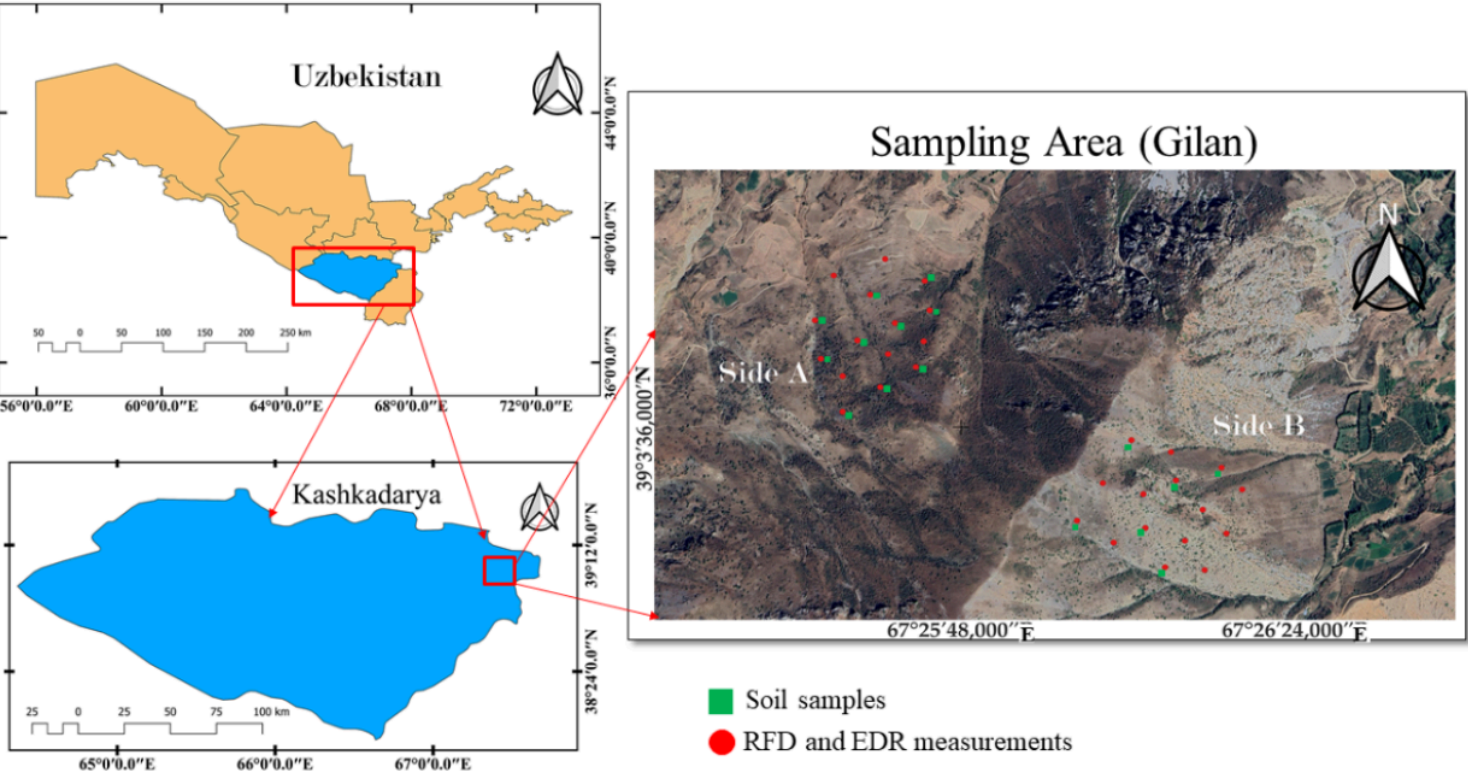


Figure 1

Location of the surveyed sites



Figure 2

Enclosing the sorbent in a transparent plastic bucket for radon sampling

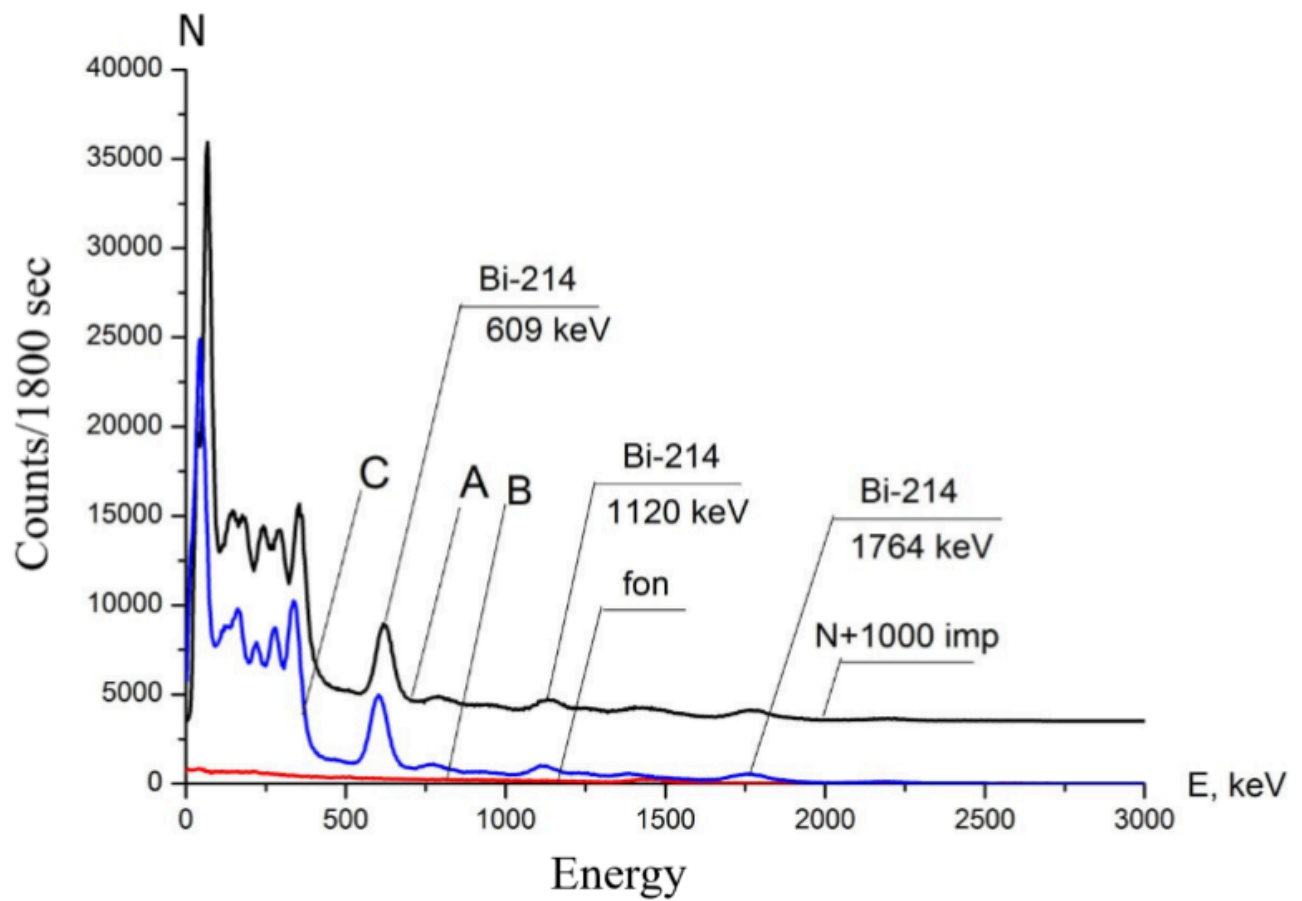


Figure 3

Spectrum of the exposed PRSD (A – Sample, B – Background, C – Component of ^{222}Rn)

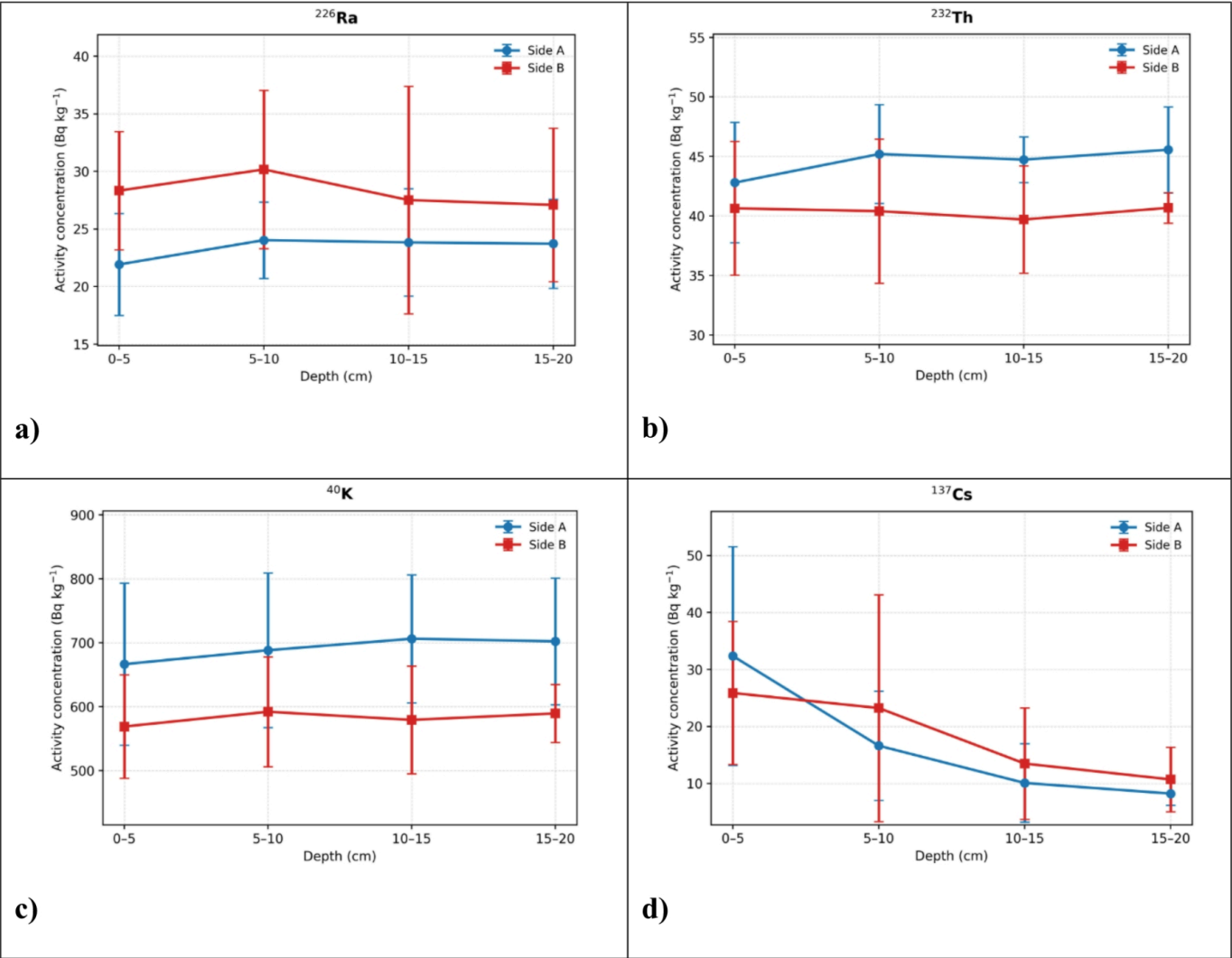


Figure 4

Depth-dependent changes in mean specific activities of (a) ^{226}Ra , (b) ^{232}Th , (c) ^{40}K , and (d) ^{137}Cs on the two hillslopes of the mountain ridge. Data points represent mean specific activities for each layer; vertical bars indicate $\pm\text{SD}$

Side A Side B

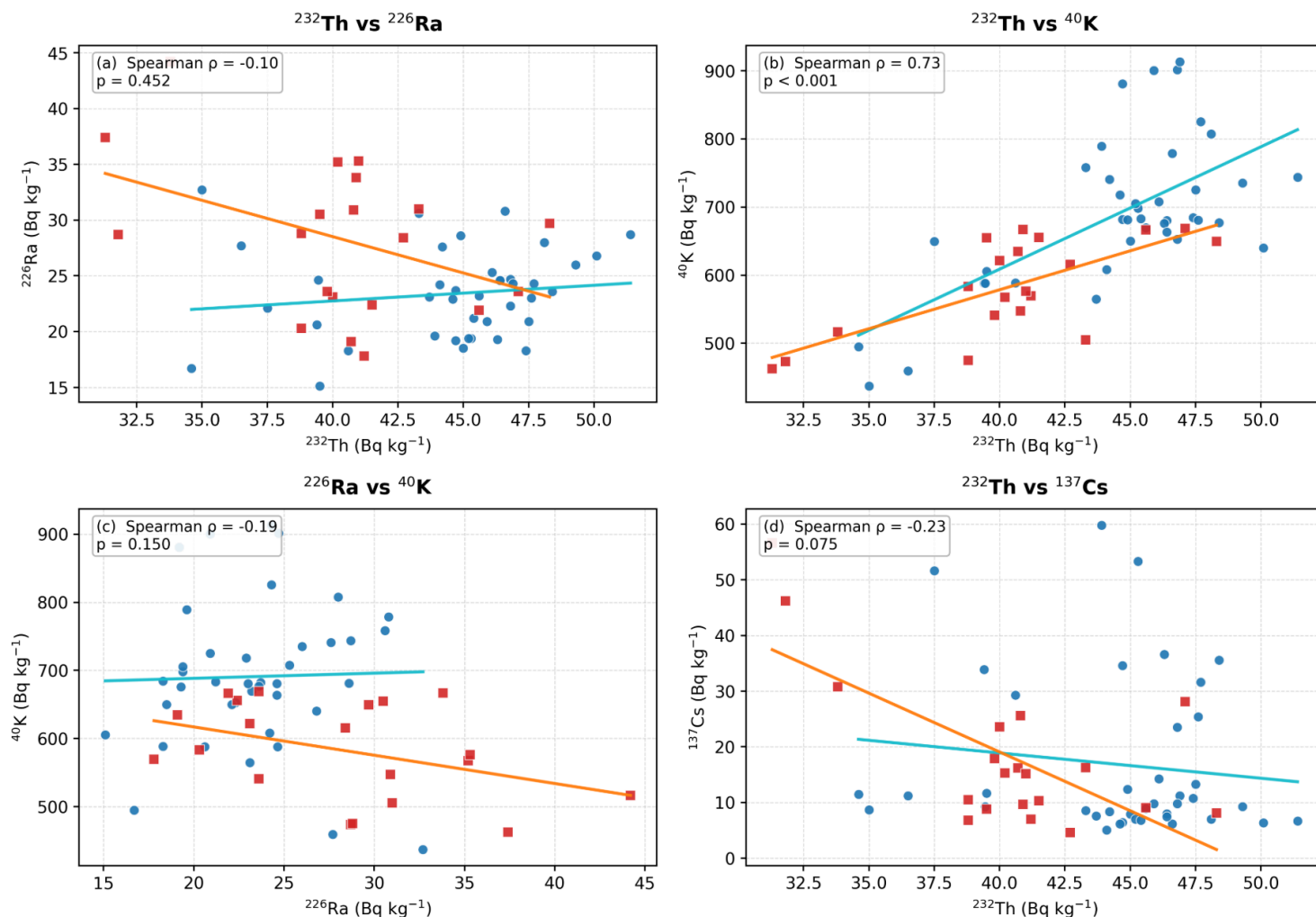


Figure 5

Correlational relationships among radionuclides

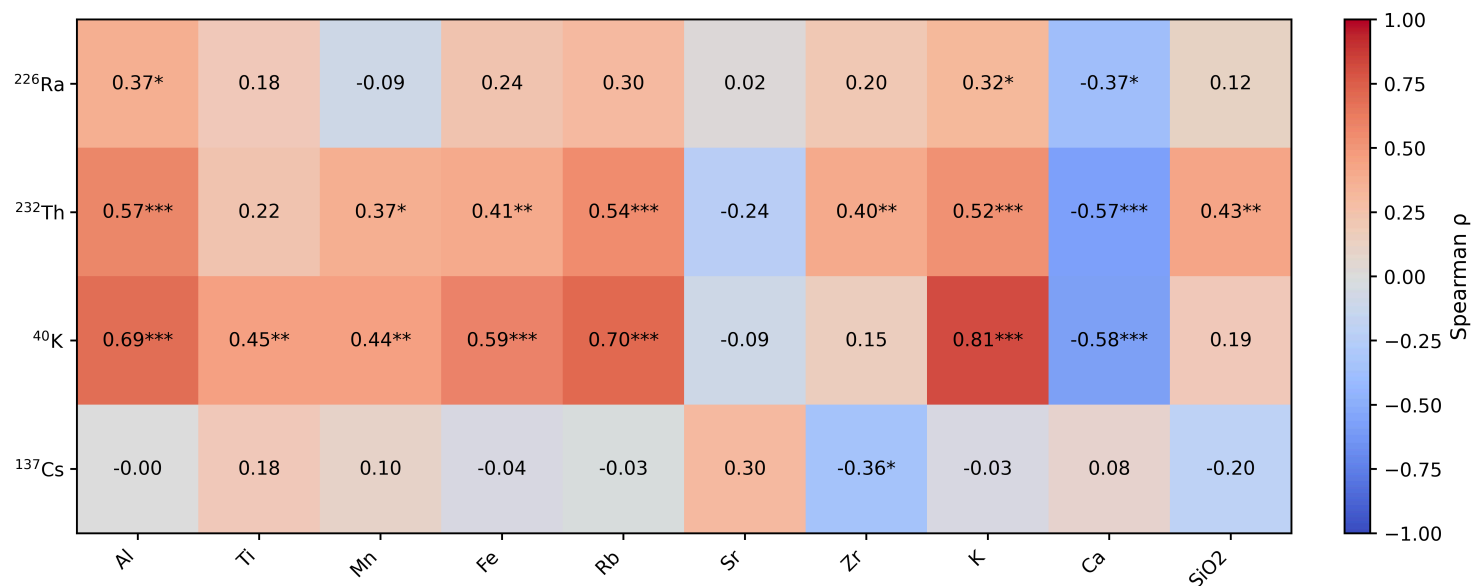


Figure 6

Correlation heatmap between radionuclides and EDXRF elements on Side A

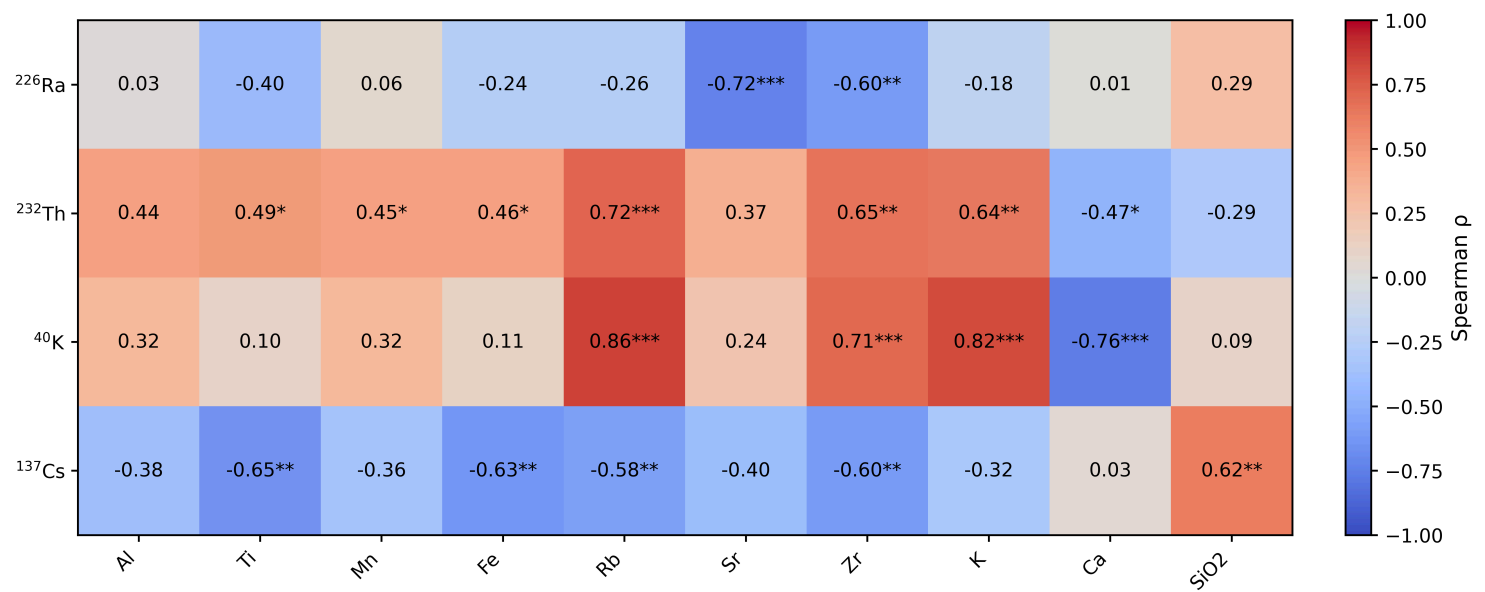


Figure 7

Correlation heatmap between radionuclides and EDXRF elements on Side B

Side A Side B

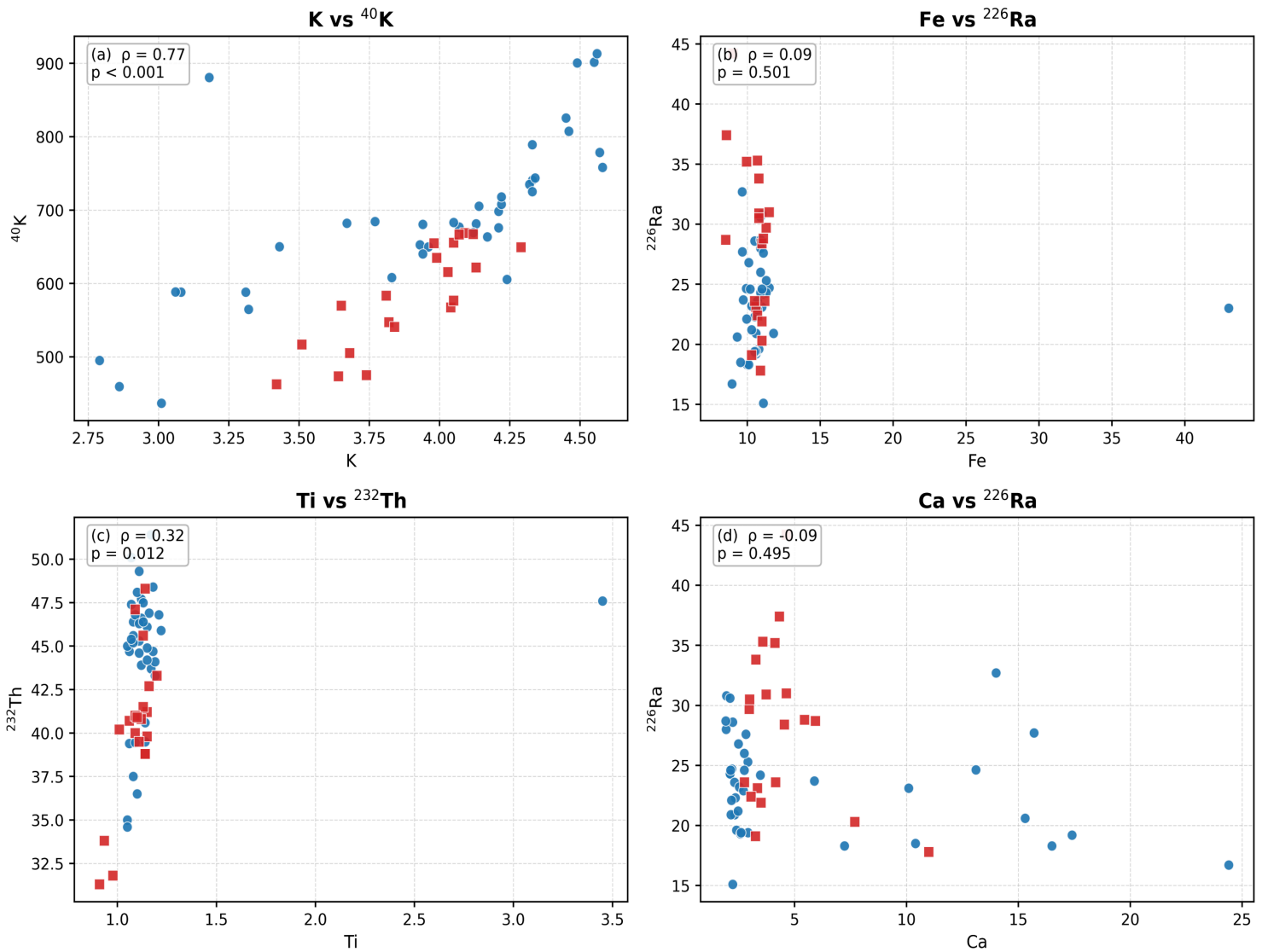


Figure 8

Selected correlational scatter plots between radionuclides and EDXRF-determined major elements for Side A and Side B soil samples

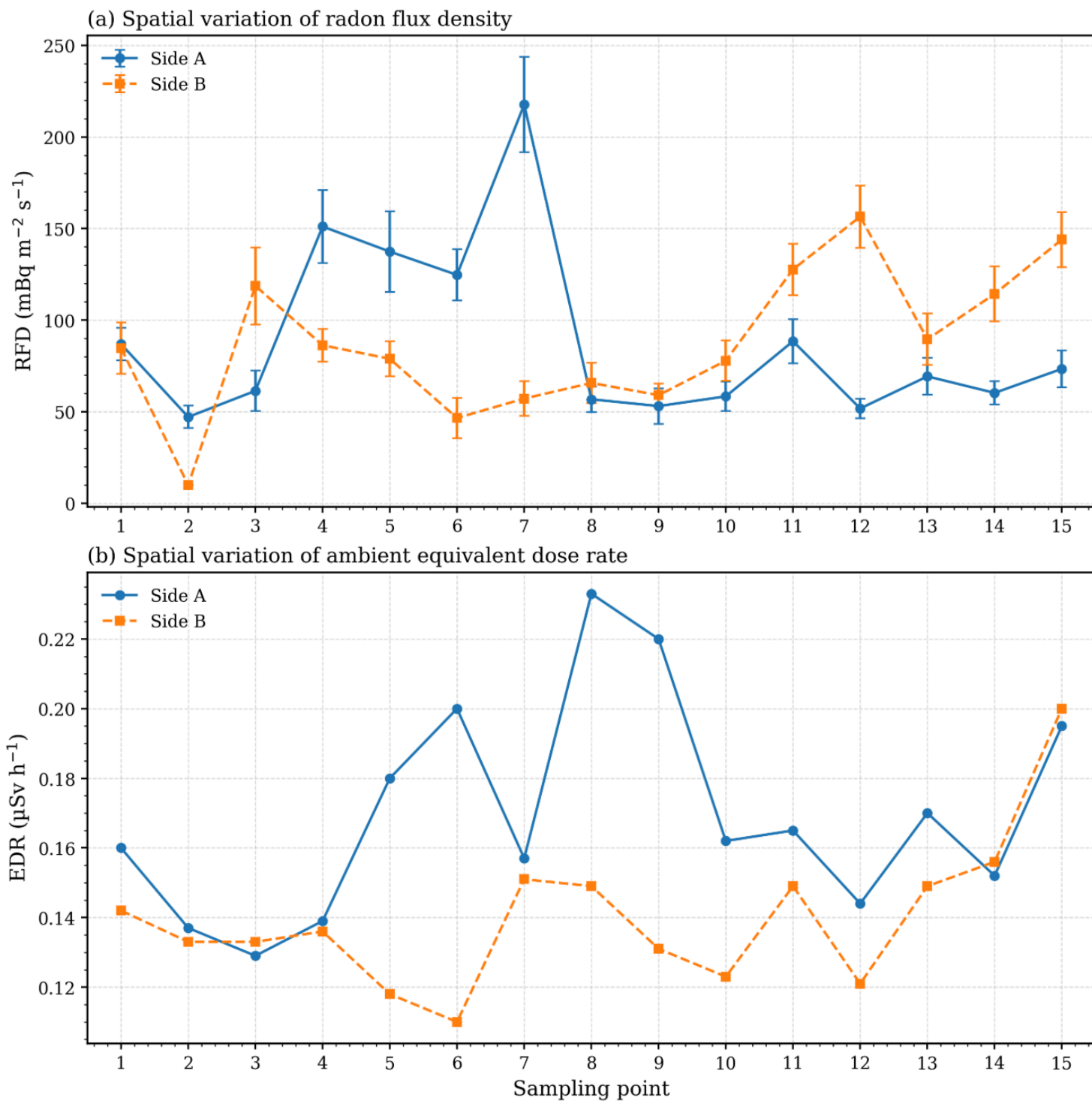


Figure 9

Spatial variation of radon flux density and ambient equivalent dose rate (EDR) measured at 15 points on Side A and Side B of the study area. Error bars for radon flux density indicate measurement uncertainty

• All sampling points

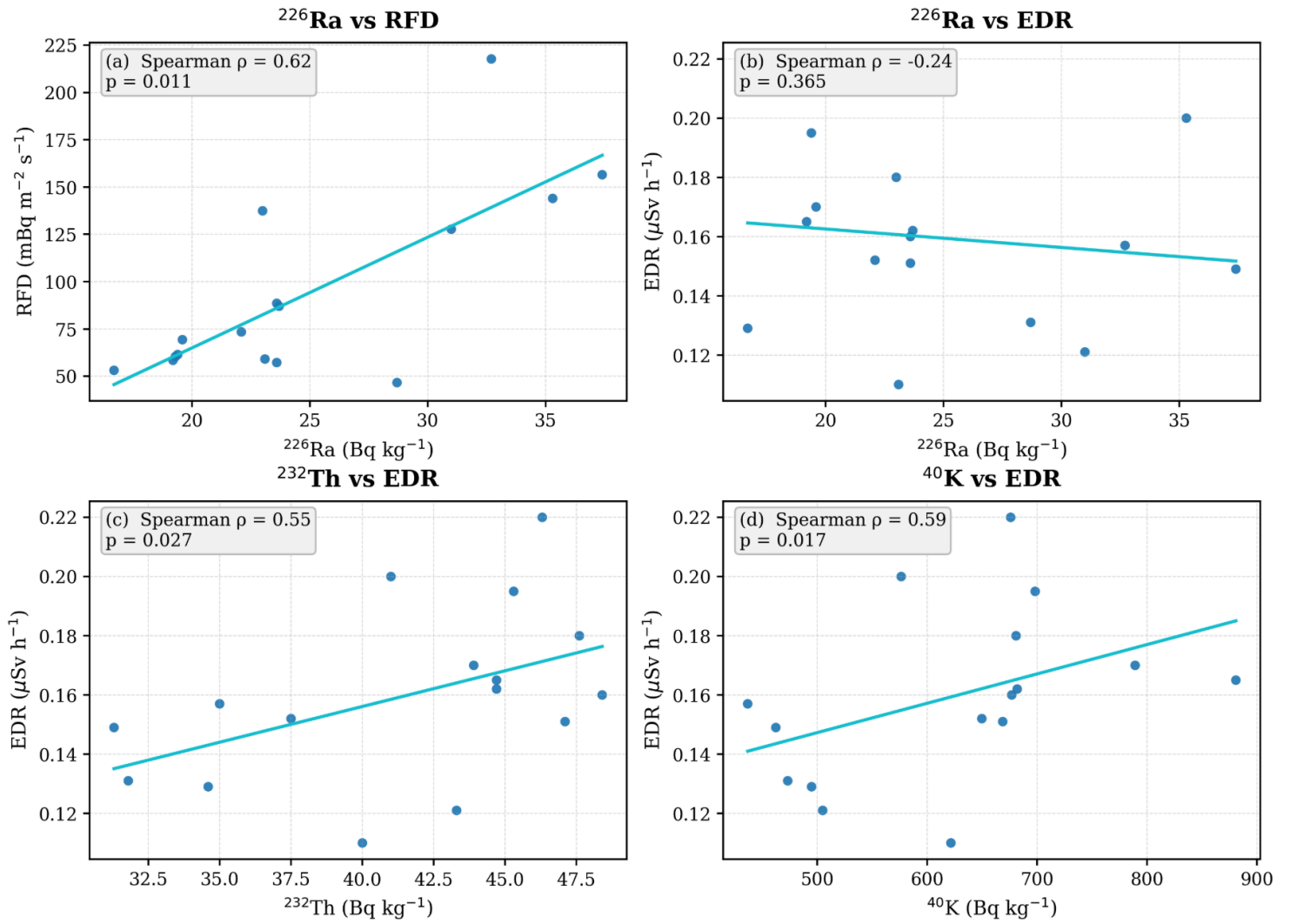


Figure 10

Spearman correlation plots showing the relationships of radon flux density (RFD) and ambient equivalent dose rate (EDR) with near-surface radionuclide activity concentrations (0–5 cm soil layer) for all available sampling points in the study area