Transfer Factors of ^{238}U, ^{232}Th, ^{40}K, ^{210}Pb to selected some Crops and radiation impact assessment in Semi-Arid Environment

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Abstract

Research on the safety of staple agricultural food products has always been one of humanity's priorities and provides input for dose assessment models. Within this important priority, activity concentrations, transfer factors (TF), and radiological effects of ^{238}U, ^{232}Th, ^{210}Pb, and ^{40}K were studied for selected crops in a village close to the NORM area, located in Central Anatolia region of Turkey. The RESRAD-onsite code has been used to assess the total dose rate. The simulation of the risk analysis covered 80 years. The maximum total dose of 0.5 mSv/yr was obtained at t = 30 years.

Keywords

Soil-to-plant transfer factor, ^{238}U, ^{232}Th, ^{210}Pb, ^{40}K, RESRAD-onsite.

Introduction

The consumption of plants containing radionuclides is one of the sources of internal dose for humans and thus contributes to internal radiation. Primordial radionuclides accumulate in the plant either as a result of direct deposition from the atmosphere onto the external surfaces of plants or through direct absorption from the plant's roots. The radionuclides in the soil are then transferred to plants together with crucial minerals and are distributed to different parts of the plant, including edible parts [1]. When consumed by humans, these plants would act to expose as a source of radiation [26]. Soil-to-plant pathway has been identified as one of the most significant ways of radionuclide transfer to plants and subsequently to humans. The concentration of radionuclides in soil is the main source of radionuclide concentration in plants. Therefore, it is important to investigate the distribution of radionuclides in plants and related soils, as well as the long-dated act of radionuclides, including motility, transfer and relocation [2]. Transfer factor (TF) is a proper method used to describe the relationship between plant and soil radionuclide concentrations. This approach also reveals how much the plant absorbs radionuclides from the soil [3]. It has been used to identify the transport of radionuclides via food and assessment of internal radiation dose through the ingestion pathway. TF is considered one of the important input parameters to radiological risk assessment models for nuclear facilities [4]. It represents the ratio between the concentration of the radionuclide in the plant and the concentration of a similar radionuclide in the soil where the plant grow. [4]. Since TF varies with climate, soil and vegetation, several studies have been carried out in different parts of the world [5 – 8]. Terrestrial radionuclides (^{238}U, ^{232}Th, and their decay products) and ^{40}K are abundant in nature as components of natural

radiation and have the potential to cause significant effects on humans [9, 25]. A significant amount of new data has been obtained on the absorption of radionuclides by the roots of tropical crops [4, 9]. Furthermore, the absence of global data for arid and semi-arid environments emphasizes the importance of such data in these environments. Moreover, the impact of global warming on the world has led to an increase in semi-arid and arid regions [10]. One of the noteworthy issues is that some semi-arid and arid countries have begun or are in the process of building nuclear power plants [11]. Transport and mobility characteristics of radionuclides in soil vary depending on soil properties, climatic conditions, plant species, cultivation and management practices [12, 24]. Therefore, the International Atomic Energy Agency (IAEA) initiated to assess extensively the radionuclide transfer factors present in various foods [3, 4, 10].

This study is carrying out in the framework of a Coordinated Research Project (CRP) of IAEA launched in 2021. In this paper, the analysis of the first harvesting season's data and TF of some primordial radionuclides were evaluated and the annual effective dose was calculated by using data from the first year of CRP project. Therefore, the present study was aimed to establish baseline data on the activity concentrations of ^{238}U, ^{232}Th, ^{210}Pb and ^{40}K in soils and cereal (barley), legume (white beans), tomato and apple as well as soil-to-plant TF collected in a semi-arid region of Türkiye. To determine the potential risk of radiation exposure through ingestion and external pathways, the total annual effective dose was also estimated.

Study area

The study was conducted in the southeast of Eskişehir province in the Central Anatolian Region of Türkiye. There is a Beylikova-Sivrihisar complex ore site located between Kızılcaören, Karkın and Okçu villages. The total area of this complex is about 17.2 km² and this area is the biggest rare earth element oxide deposit of Türkiye and also one of the largest thorium deposits in the world with an apparent reserve of 380,000 tons of thorium oxide [13]. This area can also be called as a naturally occurring radioactive materials (NORM) area. A semi-private company in the region has received a license from national regulatory body of Türkiye and is establishing a pilot facility for the processing of rare earth elements. There is no settlement in the NORM area. According to the köppen climate classification system, the study area a has semi-arid climate (Bsh) characterized by hot, dry summers and cold winters with relatively low precipitation throughout the year [14]. In the summer months, temperatures range from 25°C to 35°C, with occasional heatwaves that can push temperatures higher. In the winter months, temperatures typically range from 0°C to 10°C. Annual precipitation levels are relatively low, with an average of around 350 - 400 mm per year, most of which falls in the spring and autumn months.

The samples were collected from the agricultural areas and gardens around a village, which is nearest to and located in the direction of the prevailing wind from the NORM area. We collected barley, beans, tomato and apple samples from 4 different locations (Fig. 1). The cultivated fields were divided into several plots so that three replicates (2m × 2m) were dedicated for each crop; adjacent plots were selected randomly.

Fig. 1 Satellite map of the study area where soil, meat, milk and plant samples are collected. Sample collection locations; where wheat (1), tomato (2), white bean (3) and apple (4).

Materials and methods

Soil characteristics

Physical parameters of soil samples collected from the plant sample harvested locations are given in Table 1. The pH value of the soil sample varies between 7.1 and 8.2, with an average value of 7.7, signifying that it is alkaline. The average value of organic matter is 0.6 %. The soil salinity ranges from 0.0 to 0.15, with an average of 0.12 $mS.cm^{-1}$.

Table 1 Physicochemical characteristics of the soils [15].

	Study area
Sand(%)	48
Clay(%)	23
Silt(%)	29
Texture Class	Clay
pH	7.1 – 8.2
EC ($mS.cm^{-1}$)	0.12
$CaCO_3$ (%)	5 – 15
Salt content (%)	0.0– 0.15
Water content (%)	1-5
Organic matter, OM (%)	0.76
Density ($g.mL^{-1}$)	1.5 – 1.6
Stone fraction (%)	5.0 – 10.0

The plant species

Four plant species were selected in this investigation. The species were comprised of white beans (Leguminous), barley (Cereal), apple (Woody trees) and tomato (Fruit). Four crops that belong to four plant groups and eight crop compartments according to the International Atomic Energy Agency, IAEA, classification of plants were studied taking into account the edible and vegetative parts as the main plant components (Table 2).

Table 2 Plant groups, crops, crop compartments according to the IAEA classification [4].

Plant group	**Crop**	**Crop compartment**	**Sowing period**	**Harvesting period**
Cereal	Barley	Grain Stem and shoot	July – October	May – June
Leguminous	White beans		April – May	August – September
Fruit	Tomato		April – May	July – August
Woody trees	Apple		-	July – October

Sample preparation

Each soil sample was dried under laboratory conditions until constant weight was achieved. The dried samples were then homogenized using a motorized grinder and allowed to pass through a sieve of 250 μm mesh size. The homogenized soil sample was dried in a temperature-controlled oven at 105 ^{o}C for overnight to eliminate organic matter content and achieve constant weight. Three grams of soil sample were transferred to a moisture analyzer (OHAUS MB45) for the determination of moisture content at each step of the drying process. The samples were then packed and sealed in a 1L Marinelli beaker to prevent the escape of radiogenic gases. The weights of the sealed samples were

recorded using an electronic weighing balance. To achieve radioactive secular equilibrium between parent radionuclides and their respective daughters, the sealed containers were stored for 4 weeks.

Vegetable and fruit samples were washed with distilled water to remove dust and soil. All plant samples were pre-weighed and dried in an oven at 105 ^{o}C until a constant dry weight was obtained. Dried samples were grinded and homogenized. After homogenization, the samples were packed in 1L Marinelli beaker for gamma spectrometric analysis. Fifty grams of samples were placed into plastic bags for the analysis to be carried out in alpha spectrometry and beta laboratories.

Instrumentation and measurement methods

Measurement of ^{40}K, ^{226}Ra and ^{210}Pb activity concentration by gamma spectrometry

^{40}K, ^{226}Ra, and ^{210}Pb activities in the soil samples, and ^{40}K and ^{226}Ra activities in the vegetable and the fruit samples were measured by using two p-types high purity germanium crystal (HPGe) detectors of 20 % (Baltic Scientific Instruments, GCD-20180) and 50 % (Princeton Gamma Tech, IGC-50195) relative efficiencies respectively. The efficiency calibration of the detectors for the marinelli beaker was performed by using a multi-nuclide gamma reference source in three different matrices (water, vegetable, and sand). The beakers were filled fully for calculating efficiencies. The efficiency transfer method by Monte Carlo was used to calculate the efficiencies of samples that have different densities and filling heights. The validation of the method was performed by using the measured efficiencies of these three reference sources with different filling

heights. The Monte Carlo code GESPECOR was used for the calculation of the geometry transfer factors and the density correction for differences between the calibration source and the samples [16]. Minimum detectable activity activities (MDA) were calculated according to the well-known Curie formula as about 1.7 and 11.9 $Bq.kg^{-1}$ for soil and plant samples, respectively. Analyses were carried out using Genie2K software (V3.4) and uncertainties were reported at a 95% confidence level (k = 2).

Radiochemical separation of soil and plant samples for the measurement of ^{238}U and ^{232}Th activity concentration by alpha spectrometer

The massic activities of uranium and thorium radioisotopes were determined by using an alpha particle spectrometer from the Canberra model: Alpha Analyst-Integrated Alpha Spectrometer containing Passivated Implanted Planar Silicon (PIPS) detector with an active surface area of $450mm^2$. The energy calibration of the spectrometer was performed by using an electroplated mixed alpha standard source containing an active surface diameter of 17.2 mm supplied from Eckert&Ziegler Isotope Products, which is the thin sources of uranium and thorium for alpha-particle spectrometry. Samples to be measured were prepared by electrodeposition on a stainless steel disc. An electrodeposition cell assembly, consisting of the disposable plastic vial, cap assembly containing stainless steel for plating planchet (cathode) and platinum electrode (anode) combined with power supply was used to accumulate uranium and thorium on stainless steel disc of 17.2 mm in diameter.

Grinded and homogenized forms of soil (2 g) and plant (10 g) samples were transferred into porcelain crucibles of appropriate volume. The soil and plant samples were ashed in a furnace at 600 °C and 350° C for about 9 h respectively. Ashed samples were

transferred into a Teflon beaker for dissolution. ^{232}U tracer and ^{229}Th were added to the samples. Then 50 mL of concentrated HNO_3 and 150 mL of concentrated HCl were added in a 1:3 proportion to the samples. Teflon beakers were covered with Teflon watch glass and then the sample was evaporated gently for 12 h. The final residue was separated by centrifugation. This residue was again subjected twice more to the leaching process, with the same initial amount of acid, but reducing the stirring time to about 5 h. This ensures maximum recovery of the elements of interest. Finally, all the resulting supernatants combined. After complete dissolution, the Teflon watch glass was removed and the solution evaporated near to dryness. 50 mL of concentrated HF was added to soil samples to dissolve silicates. The ashed sample was evaporated gently to dryness on a hot plate. The evaporation of solutions was performed at 220 °C. Finally, the residue was ready for chemical separation of uranium and thorium [17]. The residue dissolved in 15 mL HNO_3 and evaporated to dryness. The dried residue was dissolved with 10 mL of 3M HNO_3-1M $Al(NO_3)_3$. Uranium and Thorium were purified and isolated radiochemically using UTEVA and TEVA resins prior to source preparation and measurement by alpha particle spectrometry [18]. After separation, uranium and thorium were deposited on 17.2 mm diameter stainless-steel discs by electro-deposition technique. Electrodeposited discs (source) were measured by an alpha spectrometer.

Radiochemical separation of plant samples for the measurement of ^{210}Pb activity concentration by proportional counter

The activity concentration of ^{210}Pb radioisotope was determined using the MPC 9604 model a gas flow proportional counter. The standard of ^{210}Pb source was used by Isotope Products (2.609 (80) $kBq.gr^{-1}$, k=2) for detector yield. Different amounts of Pb^{2+}

carrier solution were added to the ^{210}Pb / ^{210}Bi standard solution to obtain different surface densities of $PbSO_4$. For detector efficiency, HNO_3 was added and evaporated. The sample was dissolved in H_2O. Then the sample was precipitated with sulfuric acid from $PbSO_4$. The precipitate was transferred to a plastic tube with 2 ml of ethanol and then centrifuged. The supernatant was discarded. The precipitates were deposited on the planchet, was dried under an infrared lamp. The precipitates generated in this manner were weighed on an analytical scale to evaluate the chemical yield of separation, which was then measured in the proportional counter over a total in-growth duration of ^{210} Bi. For the analysis of the plant samples, an amount of 30-35 g was first taken from the sample with predetermined moisture at 105 ^{o}C and placed in a crucible. Then an exactly known amount of 30 mg of Pb^{2+} carrier was added as lead nitrate. The sample was ashed at 600 ^{o}C for 9 hours. The sample was dissolved with 20 ml of concentrated HF and 20 ml of concentrated HNO_3 and evaporated to dryness on a hot plate. The dried sample was dissolved and ^{210}Pb was separated by using a Sr-resin column. Separation of lead can be achieved from a hydrochloric acid solution [19].

^{210}Po was stripped with 60 ml 6 M HNO_3 and ^{210}Pb was stripped with 60 ml 6 M HCl. From the 6 M HCl eluate, lead was heated to near dryness at 160 ^{o}C on a hot plate. 1 mL conc. (% 65) HNO_3 was added and evaporated to dryness. The precipitate was dissolved in 40 mL pure water and 2-3 mL concentrated H_2SO_4 was added. Lead sulfate was centrifuged on a measuring planchette to form a thin layer. Chemical recovery was determined gravimetrically. After at least 21 days to allow ^{210}Bi to grow, the activity of ^{210}Pb / ^{210}Bi beta emission was measured on a gas flow proportional counter.

Results and discussion

Activity concentrations in plants and soil

The activity concentrations of plants and soil samples are presented in Table 3. Based on the activity concentrations of ^{238}U, ^{232}Th, ^{40}K and ^{210}Pb in the soil and plant crops, TFs were calculated and presented in Table 3. In addition, expected slight variation is observed between the activity concentrations of radionuclides in the soil around different plant crops. It can be said that these differences arise from differences in local conditions. Although the highest activity concentrations of ^{40}K, ^{232}Th and ^{238}U were found in tomato plant, the highest ^{210}Pb concentration was measured in the apple sample. By contrast, the lowest activity concentration of the four radionuclides was found in apple plant. The variations in activity concentrations of radionuclides in plants may be due to the metabolic characteristics and availability of radionuclides in each plant. The activity concentrations of ^{40}K in all of the plants were considerably greater than those of ^{238}U, ^{232}Th and ^{210}Pb. This is probably because potassium is a primary nutrient required for the growth of vegetables [20].

222 **Table 3** Activity concentrations of ^{238}U, ^{232}Th, ^{40}K, ^{210}Pb and Soil to plant Transfer Factors in study area.

Sample location	Plant type	Plant species	No. of samples	Sample type	Activity concentrations (Bq / kg)				AM of Transfer factor (TF) x10^{-2}			
					^{238}U	^{232}Th	^{40}K	^{210}Pb	^{238}U	^{232}Th	^{40}K	^{210}Pb
1	Barley	Cerals	3	G	0.61 (5)	12.5 (1.4)	131 (15)	4.3 (4)	1.3 (3)	1.7 (3)	7.9 (1)	3.8 (0.7)
			3	S	45.7 (9.1)	729 (90)	1666 (147)	113 (19)				
2	White beans	Leguminous	3	G	0.07 (1)	0.55 (7)	462 (55)	3.0 (4)	0.31 (1)	0.14 (3)	51 (7)	3.5 (5)
			3	S	23.6 (6.9)	394 (51)	904 (51)	86.2 (7.8)				
3	Apple	Woody tree	3	G	0.023 (2)	0.09 (1)	127 (23)	11.0 (1.2)	0.08 (2)	0.030 (5)	32 (4)	12 (2)
			3	S	28.8 (5.8)	288 (38)	1033 (99)	91.6 (8.6)				
4	Tomato	Fruit	3	G	2.6 (2)	55.6 (6.1)	1503 (146)	7.4 (8)	4.9 (5)	6.7 (8)	77 (10)	6.6 (1)
			3	S	52.7 (1.1)	834 (32)	1956 (160)	112.9 (8.9)				

223 G: Grain, S: Soil, AM: Arithmetic mean.

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Annual effective dose

In this study ingestion of plant foods, meat, milk and soil pathways and external gamma exposure pathways were only considered for dose calculations. In the study area, villagers consume fruits and vegetables grown in their gardens only in the spring and summer seasons. During the winter months, they consume fresh fruits and vegetables from outside the region. Only one or two farmer families in the region are engaged in animal husbandry, and at least half of the products used in meat and milk consumption are brought from the nearest city. The amount of food consumption used in dose calculations was taken as half of the food consumption amounts determined by Uzel al in 1972 [21]. It is also assumed that villagers spend 70 % of their time at home and 30 % outside. Although the villagers spend most of their time outside only in the spring and summer months, working in the fields and gardens (about 5 hours), they spend more time in and around the residential area of the village. Therefore, the concentration of a soil sample taken from a residential area was used for the soil concentration values entered into the RESRAD-onsite computer code. Water consumption not considered for dose assessment because residents use the city water supply in the village. ^{238}U, ^{232}Th, ^{210}Pb and ^{40}K activity concentrations of soil sample are measured as 44.5 (9) Bq.kg^{-1}, 110.6 (51) Bq.kg^{-1}, 75.0 (58) Bq.kg^{-1} and 177.0 (82) Bq.kg^{-1} respectively. These activity concentrations were used as soil concentration input in the RESRAD computer code.

The potential human radiation exposure due to environmental contamination was investigated using the RESRAD code family. Pathway analysis is used to determine the radiation exposure and risk factors. The monitoring of radionuclides in the environment (soil, water, and air) through a pathway analysis method is used by RESRAD to model

the way these molecules reach receptors through direct exposure, ingestion, or respiration. The RESRAD-onsite computer code is designed to calculate the radiation exposure and excess cancer risk associated with living or working in an area contaminated with radionuclides. To simulate radiological dose, we used RESRAD-onsite and assumed an event where a person settled near the NORM site in the simulated population.

A typical exposure scenario as described in the RESRAD user manual was used with some modifications in this study [22]. In this scenario, farmers are assumed to be permanent residents. To identify exposure scenario, soil activity concentrations, geological features, resident lifestyles and others were specified. The dose constraint was set to 3 $mSv.y^{-1}$, due to secular balance, the progeny's concentrations were identical to the primordial radionuclides.

Default values and site-specific parameters taken from the literature were used for dose estimation as shown in Table 2. Parent and daughter radionuclides were considered by selecting the cutoff half-life as 180 days. In this study area soil was classified to be clay hence the distribution coefficients (Kd) of unsaturated zone were set to, $2.8x10^{1}$, $4.5x10^{3}$, $6.6x10^{4}$ and $4.3x10^{1}$ for ^{238}U, ^{232}Th, ^{210}Pb and ^{40}K respectively. Default values are used for "the storage times before use".

274 **Table 4** Basic default and site-specific input data used in RESRAD-onsite computer code.

S/N	Parameters	Site specific data	Default data	Reference
	Soil concentration ^{238}U ^{232}Th ^{210}Pb ^{40}K	 0.044 $Bq.g^{-1}$ 0.11 $Bq.g^{-1}$ 0.075 $Bq.g^{-1}$ 0.17 $Bq.g^{-1}$	-	-
1	Study area	1000000 m^2	-	-
2	Thickness of contaminated zone		1 m	RESRAD default
3	Density of contaminated zone	1.5 $g.m^{-3}$	-	[15]
4	Cover depth	0	-	-
5	Length parallel to aquifer flow	300 m	-	[15]
6	Contaminated erosion rate	-	0.001 $m.y^{-1}$	RESRAD default
7	Contaminated zone total porosity	0.498	-	[15]
8	Contaminated zone b-parameter	7.582	-	[15]
9	Evapotranspiration coefficient	0.72	-	[15]
10	Wind speed	2.6 $m.s^{-1}$	-	[15]
11	Precipitation rate	0.415 $m.y^{-1}$	-	[15]
12	Irrigation rate	0.038 $m.y^{-1}$	-	[15]
13	Hydraulic conductivity	3680 $m.y^{-1}$	-	[15]
14	Runoff coefficient	0.5		[15]
15	Density of saturated zone	-	1.5 $g.cm^{-3}$	RESRAD default
16	Saturated zone total porosity	0.498	-	[15]
17	Saturated zone effective porosity	0.498	-	[15]
18	Saturated hydraulic gradient	0.50	-	[15]
19	Saturated zone b- parameter	7.582	-	[15]
20	Water table drop rate	-	0.001 $m.y^{-1}$	RESRAD default
21	Well pump intake depth	10 m	-	[15]
22	Exposure duration	-	30 y	RESRAD default
23	Fruits,vegetable and grains consumption rate	77.9 $kg.y^{-1}$	-	[15]
24	Milk consumption rate	99.3 $kg.y^{-1}$	-	[15]
25	Meat consumption rate	3.5 $kg.y^{-1}$	-	[15]
26	Leafy vegetable consumption rate	-	14 $kg.y^{-1}$	[15]
27	Soil ingestion rate	-	36.5 $gr.y^{-1}$	[15]
28	Contaminated fraction of plant food Contaminated fraction of meat Contaminated fraction of milk	0.4 0.04 0.5	-	-
29	Transfer factor for plants ^{238}U ^{232}Th ^{210}Pb ^{40}K	 0.016 0.021 0.065 0.58	-	Table 3 (mean value of all plants for each radionuclide)

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Fig. 2 Results of RESRAD onsite scenario.

Fig. 2 shows the result for the RESRAD-onsite scenario for a resident. In this scenario, we calculated the total dose ($mSv.y^{-1}$) obtained from RESRAD-onsite with activity concentrations for an onsite resident who settles on a village near to NORM area. The radiological dose was calculated as 0.13 $mSv.y^{-1}$. Fig. 2 shows that the values were all below the ICRP-recommended public dose limit of 1 $mSv.y^{-1}$, with a maximum value of 0.51 $mSv.y^{-1}$ in the simulation. Although mine processing facility operation has not yet started the process, the distribution of the naturally occurring thorium and uranium within the shallow subsurface contaminates soils, farmlands and streams within the study area, constituting the source of radioactivity due to the presence of NORMs in the rare earth minerals.

Conclusions

The radioactivity concentrations of ^{238}U, ^{232}Th, ^{210}Pb and ^{40}K were measured in 12 soil samples and 4 types of plant samples collected from selected farms in Sivrihisar, Türkiye. The Arithmetic mean of ^{238}U, ^{232}Th, ^{210}Pb and ^{40}K concentrations in soils were 37.7 (7.2), 470.6 (60.8), 101.0 (10.8) and 1390 ± (110) $Bq.kg^{-1}$, respectively, while the concentrations in plant samples were an order of magnitude lower than in soils at most. The plants were treated as two separate compartments: crops and stems-leaves. ^{238}U, ^{232}Th, and ^{40}K concentrations were found to be higher in tomatoes than in other collected samples. In fruits ^{210}Pb concentration was the highest in apple. Moreover, except ^{40}K concentration differences in ^{238}U, ^{232}Th and ^{210}Pb concentrations of soil were proportionally reflected in the plant concentrations.

The ratio of radionuclide concentrations of each crop to its substrate soil gives the compartmental TF. In conclusion, the interpretation of the U, Th, Pb, K and TF data presented here for the studied crops along with the data gathered from the literature strongly suggests that regional TF parameters are essential to obtain an accurate radiological assessment and dose calculations. Finally, the results of this study will form valuable baseline data on the behavior and concentrations of natural radioactivity in soils and vegetation grown in the semi-arid environment of Türkiye, which are not yet available. This work contributes to establishing a database on the concentration of radioactive elements (^{234}U, ^{232}Th, ^{210}Pb, and ^{40}K) in soils and plants and their TF in the semi-arid environment of Türkiye. These transfer factors are entered into RESRAD-onsite code and used to determine the dose received by dwellers through the ingestion pathway.

In this study evaluation of radiological dose from existing NORM area containing naturally occurring radioactive materials in Türkiye was conducted using soil activity concentration below 1 $Bq.g^{-1}$. Results for RESRAD-onsite show that the doses were below the ICRP recommended public dose limit of 1 $mSv.y^{-1}$ for both pathways, which makes the site safe for onsite dwellers.

The ICRP dose limits are intended to serve as a margin state by avoiding deterministic consequences and regulating the possibility of stochastic effects. Doses above 1 $mSv.y^{-1}$ requires the implementation of public protection measures. In this study, the radiological dose from the existing activity concentration of NORM was performed using RESRAD-onsite code version 7.2. The aim of this dose assessment is to determine the background of the village residential area and to determine the dose currently received by dwellers

through ingestion and external exposure pathways before the established open pit rare earth elements mining facility starts to operate.

Since the expected doses are less than the permissible limit of 1 $mSv.y^{-1}$, the results obtained indicate that the sites may not have a negative impact on the health of the resident farmers due to radioactivity from the agricultural soil. However there could be radiological health consequences after the mining facility starts to operate so it would be useful to carry out radiological monitoring activities in the settlements around the NORM area as long as the mining processing plant is operated.

This study, carrying out within the scope of CRP "Transfer of Radionuclides in Arid and Semi-Arid Environments for Radiological Environmental Impact Assessment (K41022)" launched by IAEA in 2021. The first samples were collected during the harvest period of 2022 and 11 radionuclides were analyzed. The same samples were collected from the same locations in 2023, and sample preparation efforts are continuing. In this article, data for only four radionuclides are presented. It is planned to publish all data in the year when the CRP project is completed.

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