Carpathian Journal of Earth and Environmental Sciences, August 2014, Vol. 9, No. 3, p. 75 - 84

NATURAL RADIONUCLIDES CONTENT IN THE RIVER SEDIMENT AND RELATED HEALTH RISK ASSESSMENT FOR THE WEST MORAVA RIVER BASIN, SERBIA

Ljubica ZUPUNSKI[1], Vesna SPASIC JOKIC[1] & Vojin GORDANIC[2]

[1]Faculty of Technical Sciences, University of Novi Sad, 21000 Novi Sad, Trg Dositeja Obradovica 6, Serbia, svesna@uns.ac.rs, ljubicaz@uns.ac.rs;

[2]Geoscience, Kosovska 18, 11000 Belgrade, Serbia, gordanicv@gmail.com

Abstract: Purpose of this study was to measure activity concentration for radionuclides ^{232}Th, ^{40}K, and total uranium ($^{\Sigma}$U) using HPGe detector in samples collected from the three levels of bank profiles (A-alluvial soil level, OB-overbank level, S-active stream level) in the West Morava River basin, Serbia. External dose equivalent was estimated as well as inhalation and ingestion lifetime cancer morbidity risks. External dose equivalent was obtained using Monte Carlo package FOTELP. Mean activity concentrations are (393 Bq/kg - A, 378 Bq/kg - OB, 346 Bq/kg – S) for ^{40}K, (29 Bq/kg - A, 24 Bq/kg - OB, 23 Bq/kg - S) for $^{\Sigma}$U and (31 Bq/kg - A, 30 Bq/kg - OB, 22.5 Bq/kg - S) for ^{232}Th. Mean value of calculated cancer morbidity risk from ^{232}Th inhalation is 0.94×10^{-6}, range (0.35 – 1.4) $\times10^{-6}$ and for ingestion 1.7×10^{-6}, range (0.6 - 2.5) $\times10^{-6}$. Mean value of calculated cancer morbidity risk from ^{238}U inhalation is 0.2×10^{-6}, range (0.07-0.3) $\times10^{-6}$ and from ingestion 0.43×10^{-6}, range (0.17-0.58) $\times10^{-6}$.

Keywords: natural radionuclides, health risk assessment, Monte Carlo method, river sediment

1. INTRODUCTION

Terrestrial natural radionuclide content is important component of the natural radiation which has influence on the exposure of human population and other ecological receptors (Životić et al., 2008; Tereesh et al., 2013). A great effort is invested in research of the terrestrial radionuclide content, both naturally occurring and man-made, because of the possible radiological hazard (Radosevic et al., (2002); Radenković et al., (2008); Spasić-Jokić et al., (2011a)). Also, this kind of research can reveal various geological processes that influenced on the study area as well as possible contamination and industrial activity. Gamma emitters from ^{238}U and ^{232}Th series and ^{40}K in the Earth's crust contribute to the absorbed dose via external exposure. Alpha emitters represent hazard in the case of dermal or internal exposure. Internal exposure arises from the inhalation or ingestion of the radionuclides. Inhalation of the radionuclides from soils is possible due to their intake through dust particles that are resuspended in the air. Internal exposure via ingestion includes radionuclides transfer from soil to vegetables wherefrom they enter into the food chain (Unscear, 2000; IARC, 2001). Direct ingestion of the dust particles also appears to be one of the possible exposure pathways (EPA, 2000a).

West Morava represents the tributary of the Great Morava River which flows through the most densely populated and the most fertile area in Central Serbia. West Morava River drains the catchment area of 15,849 km^2. River length is 208 km and has a great number of tributaries (85) among the longest are Ibar (272 km), Gruza (77 km), Cemernica (41 km), Dicina (35 km), Kamenica (Fig. 1). Structure of streambed varies, including stones, gravel and sand. Because of the flash-flood nature of tributaries, river rises and overfloods its banks so territories and plains along the river are usually flooded causing damage to properties and crops. Radionuclide content of the particular river sediment depends on the type and origin of the sedimentary rocks or soil constituents as well as on the geological and geographical conditions (Dragović et al., 2006; Krmar et al., 2009; Papaefthymiou et al., 2011). The main part of West Morava with its tributaries belongs to the Vardar geotectonic zone.

Figure 1. Map of central part of Serbia with emphasized West Morava River basin with tributaries and sampling locations for each river.

The Vardar geotectonic zone is one of the most complex domains of the Balkans Peninsula. This zone is composed of several blocks which have different composition and origin such as: Paleozoic strata and Triassic volcanogenic-sedimentary formation, ultramafics, mélange, Cretaceous flysches and paraflysches, Tertiary granitoids, volcanics (Jadar and Kopaonik blocks), granitoide area with ultramafics (Ibar, Stolovi), post-Triassic metamorphites, paraflysch, Ophiolite mélange and metamorphites (Central Vardar block), crystalline schists of unknown age, Flysch and Tertiary volcanics (The Internal Vardar subzone).

Radionuclide content of the river sediments may reflect contaminated areas as result of industrial activity. Non-nuclear technologies, coal burning, fertilizers, chemistry industry, may influence on the elevated natural radionuclides concentrations and deposition of the Technologically-Enhanced Naturally-Occurring Radioactive Material (TENORM) in some areas (Bikit et al., 2005).

Internal and external exposure to terrestrial radionuclides should be taken into account during health risk assessment process (EPA, 1999; Spasić-Jokić et al., 2011b). Methods for health risk assessment should provide prediction of biological effects after exposure to ionizing radiation. Health Risk assessment based on dose estimation is an important part of radiation protection program. Health risk can be expressed in terms of cancer mortality and morbidity risks which are derived from various radiogenic risk models (National research council, 2006). These risk

models are based on the Linear – no – threshold Hypothesis (LNT) that deals with dose-effect relationship. The LNT model uses epidemiological data at higher doses and extrapolates the excess cancer risk linearly down to zero doses so the possibility of cancer developing exists even for very low doses, just above zero doses (Brenner et al., 2003; Brenner & Sachs, 2006). However, risk assessment related to low dose exposures, such as exposure to the natural radionuclides from soil is characterized with high uncertainty because of the high background cancer rates in population. A great effort should be invested in epidemiology researches that should cover areas with high natural radionuclides content.

Purpose of this study was to measure activity concentration of total uranium, ^{232}Th and ^{40}K, using HPGe detector in three levels of bank profiles in the West Morava basin. These measurements should represent basis for environmental quality control and spatial planning regarding main uses of this river and tributaries such as irrigation, fisheries, sand usage as constructing material, or wastewater collectors. This area is populated so external dose equivalent was estimated as well as inhalation and ingestion cancer morbidity risks, as contributors to the total risk of the internal and external exposure.

2. METHODS

2.1. Samples

In stage of geochemical prospecting we collected samples from the area of 15000 km^2. During the geochemical prospection, the samples of stream sediments, overbank sediments and samples located at A horizon in alluviums were processed according to the WEGS (Western European Geological Surveys) methodology. Sampling network was adjusted to hydrographic characteristic of terrain. From 4 to 6 samples were collected from A, OB and S horizon of each river so total of 160 samples were collected. The following rivers are included in this study: 1. Bjelica; 2. Gruza; 3. West Morava 1; 4. Ribnica; 5. Lopatnica; 6. Ibar; 7. Dicina; 8. Cemernica; 9. Kamenica; 10. South Morava; 11. West Morava 2. West Morava 1 and West Morava 2 are two different locations along the same River. West Morava 1 sampling area is in the vicinity of the city Kraljevo and the location of West Morava 2 sampling area is in the vicinity of the confluence of the South Morava and the West Morava. Distance between these two sampling areas is approximately 70 km. The locations of the sampling areas were chosen regarding sites of increased industrial activity and sand exploitation areas. Figure 1 presents sampling areas for each river. There was one sampling area for each river (except West Morava with two sampling areas), but on each location samples were taken at three levels (alluvium, overbank and stream sediments (Fig. 2).

Bank sampling profiles (Fig. 2) were performed at following levels:

- A Horizon in alluviums at the depth of (15-20) cm with sample weight of 5 kg. This layer was considered as free from human influences, suitable to show level of natural geochemical changes;
- OB - overbank sediment at average depth of 1 m, in dependence of lithological changes and development of sediment, with sample weight between 5 kg and 15 kg. Overbank sediments show sedimentation history. Samples from overbank sediment were proposed to be a measure of the state of pollution in environmental studies because they represent anthropogenic and natural environment back through time (Bogen et al., 1992).

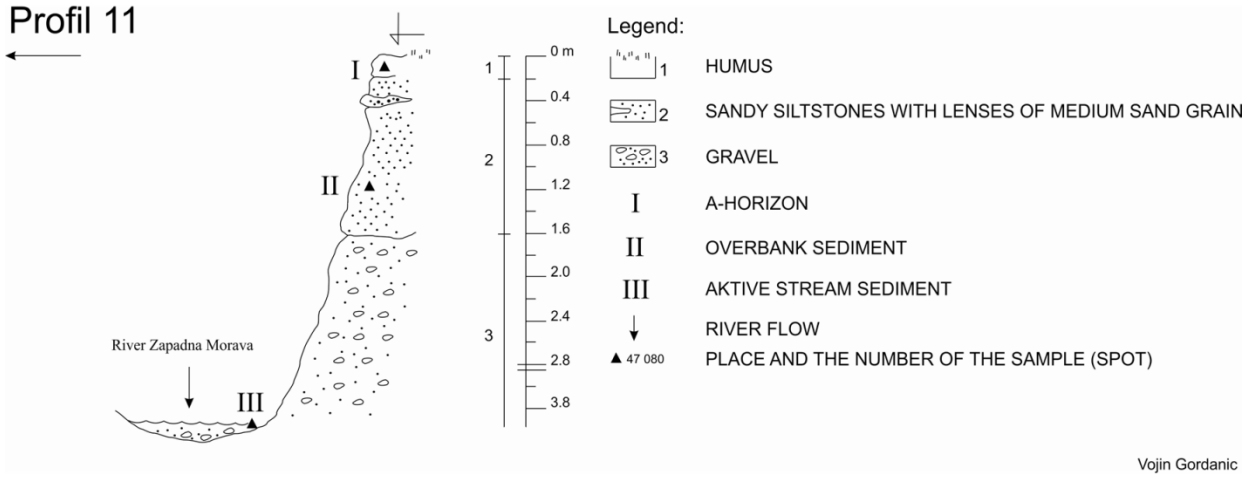

Figure 2. Sampling profile (no. 11) of West Morava River

S - Horizon of active stream sediments, with sample weight of 5 kg, suitable to show correlation with another geochemical data.

All the samples were air dried, homogenized and griddled to grain size of 0.60 mm. Samples of about 0.5 kg were prepared and placed in cylindrical gas-tight containers that have the same geometry as sample container used for efficiency calibration. The samples were kept for at least three weeks before the measurement in order to reach secular equilibrium between radium and thorium, and their decay products.

2.2. Determination of natural radionuclide activity concentrations and estimation of external dose equivalent

For collected samples, determination of $^{\Sigma}$U, ^{232}Th and ^{40}K were performed. According to obtained activity concentrations, external dose equivalent and inhalation/ingestion cancer morbidity risk were calculated, for all three bank profiles in West Morava river basin.

Gamma spectrometry determination was performed with high purity, low energy, germanium semiconductor detector (HPGe), manufactured by ORTEC, with accompanying electronic equipment and ORTEC software for spectra evaluation. The relative efficiency of HPGe detector was 28 % with energy resolution of 2 keV in measurements at the 1.33 MeV reference transition of ^{60}Co (Kandić et al., 1993). Calculated minimal detectable concentrations were 1.3 Bq/kg for ^{232}Th, 1.1 Bq/kg for ^{238}U, and 2.4 Bq/kg for ^{40}K. The counting geometry was the same for all radionuclides. Detectable energy range was up to 2 MeV. The expanded uncertainty of measured radionuclide concentration was 15 % ($k=2$) in each case. Calibration was performed using reference sources: (1) NBL 107, 0.1 % Th; (2) Certified mix source, Amersham, 1988 (^{55}Fe, ^{60}Co, ^{137}Cs, ^{226}Ra and ^{241}Am); (3) NBL 103 (New Brunswick Laboratory), content of U: 0.05 %; (4) K in form of potassium-chloride. It was assumed that soil contains only natural uranium with content: 99.284 % ^{238}U + 0.711 % ^{235}U + 0.0058 % ^{234}U. The activity concentrations were determined using gamma lines from ^{214}Pb (352 keV), ^{214}Bi (609 keV) for the ^{238}U series and ^{212}Pb (photon energy of 238 keV), ^{208}Tl (583 keV), ^{212}Bi (727 keV) and ^{228}Ac (911 keV) for ^{232}Th series. For ^{40}K measurement, only gamma line on 1460 keV was used.

External dose equivalent was calculated from the determined gamma absorbed dose rate in air 1 m above the ground. Gamma absorbed dose rate in air (nGy/h) at 1 m above the ground was calculated using Monte Carlo package FOTELP (Stankovic, (2002)). The absorbed dose rate was calculated from photon flux energy distribution at the given position where photon flux consists of the unscattered flux (photons of energy $E\gamma$ reaching the given position without interaction with the soil and the air above) and the scattered flux (photons of energies less than Eγ reaching the given position after scattering, mainly in soil) (Clouvas et al., 2000).

Input data relevant for the simulation were:

- *Geometry*: 2π geometry was assumed. The soil was simulated as a square slab with length of 40 m and depth of 1 m (sufficient size in order to consider the emission photon geometry as half space geometry). Air was above the soil, simulated by a square slab with the same side as the soil and a height of 1.5 m (Župunski et al., 2010).
- *Material characterization*: The atomic compositions of soil was: 2.1% C , 7.3% Al, 3.9% Fe, 1.4% Ca, 0.9% K, 0.6% Na, 0.5% Mg, 0.1% N, 50.1% O, 32.7% Si, 0.4%Ti, with density of 2.3 g/cm^3 (usually used in the other dedicated codes). Atomic composition of soil was not a critical parameter of the flux energy distribution (Clouvas et al., 2000). Density of dry air was 1.29 kg/m^3 and atomic composition of air used in simulation was 21 % O and 79 % N.
- *The source distribution*: It was assumed that photon radiation is uniformly distributed in the soil with different photon energies in the range from 200 keV to 3000 keV. The number of simulated histories (emitted photons) was 2×10^6.
- *Desired answer*: energy distribution of photon flux in the given position in the air at one meter above the soil (Chen, 1991; Saito & Jacob, 1995; Likar et al., 1998).

For the determination of the photon flux energy distribution at one meter above soil, one virtual detector was used, simulated by a square surface with side of 2 m, located in the center of the square slab which represents the soil. Assumed the low-energy cutoff for the calculations was 50 keV (photons below 50 keV contribute a negligible amount to the dose rate) and the high-energy cutoff was 2614 keV as this energy is the highest important gamma energy of natural radionuclide.

2.3. Risk assessment

Risk assessment was performed from exposure to natural radionuclides ^{238}U and ^{232}Th in river sediment via inhalation and ingestion. Risk was

assessed only for isotope ^{238}U in natural uranium. Contribution to the risk from isotope ^{235}U is small compared to isotope ^{238}U because of its abundance in natural uranium. Lifetime cancer morbidity risk per capita for lifetime exposure was assessed using EPA coefficients (risk/Bq). Coefficients can be used for acute exposure or for population exposed throughout life to a constant concentration of a radionuclide in an environmental medium (EPA, 2000a; EPA, 1999; EPA, 2000b). Coefficients are provided for different radionuclides regarding type, energy, and intensity of emitted particles and exposure scenarios. Risks refer to average member of population because they are averaged over age and gender distribution of hypothetical closed stationary population. It is assumed that stationary population has time-invariant gender specific birth rates and survival functions. Survival functions and mortality rates are based on US decennial life table (EPA, 1999). In the case of inhalation scenario, it was assumed average dust concentration in the air of 50 $\mu g/m^3$ and breathing rate of 20 m^3/day. For the calculation of cancer morbidity risk via direct ingestion, it was assumed daily incidental dust intake of 100 mg/day.

Lifetime cancer morbidity risk per capita from lifetime exposure via inhalation pathway was calculated using equation:

$$Risk = R_c \times A \times C \times IR_i \times T \qquad (1)$$

where R_c represents risk conversion factor (risk/Bq), A is activity concentration of radionuclide in soil (Bq/kg), C is dust concentration in air (50×10^{-9} kg/m^3), IR_i is inhalation rate (20 m^3/day), T is exposure time (75 years).

Lifetime cancer morbidity risk per capita from lifetime exposure via direct ingestion pathway was calculated using equation:

$$Risk = R_c \times A \times IR_s \times T \qquad (2)$$

where R_c represents risk conversion factor (risk/Bq), A is activity concentration of radionuclide in soil (Bq/kg), IR_s is ingestion rate (100×10^{-6} kg/day), T is exposure time (75 years).

Lifetime cancer morbidity risk coefficient per capita for ^{232}Th inhalation exposure is 1.17×10^{-6} risk/Bq and for ^{238}U is 2.53×10^{-7} risk/Bq. Lifetime cancer morbidity risk coefficient per capita for ^{232}Th direct ingestion exposure is 0.62×10^{-8} risk/Bq and for ^{238}U is 0.57×10^{-8} risk/Bq (EPA, 2000b).

3. RESULTS AND DISCUSSION

Results of the total uranium ($^{\Sigma}U$), ^{232}Th and ^{40}K activity concentration (Bq/kg) measurements of river sediments in three horizons (A – alluvial soil, OB – overbank sediment, S – stream sediment) are presented in Table 1, Table 2, and Table 3. Measurement uncertainty of activity concentration is 15 % (k = 2).

Mean activity concentrations with range of activities of natural radionuclides $^{\Sigma}U$, ^{232}Th, and ^{40}K for A, OB and S horizon, for all locations, are shown in Table 4. Also, Table 4 presents results from the literature of activity concentration measurement in river sediments in the region (Danube river - northern part of Serbia, Tisa river - northern part of Serbia, Bega channel - northern part of Serbia) (Todorovic et al., 2008). Mean activity concentration of radionuclides ^{40}K and ^{232}Th could be compared to the world average values of 420 Bq/kg and 45 Bq/kg for ^{40}K and ^{232}Th respectively. It can be seen that mean activity concentration of ^{40}K (393 Bq/kg - A, 378 Bq/kg - OB, 346 Bq/kg – S), and ^{232}Th (31 Bq/kg – A, 30 Bq/kg – OB, 22.5 Bq/kg – S) for all three horizons are lower than world average values. Uranium world average concentration is 2.8 ppm in the Earth's crust (Kidd, (2011)). Average values of U concentration in this study are 2.2 ppm, 1.9 ppm and 1.6 ppm for A, OB and S horizons, thus, they are lower than world average value. Also, it can be seen that mean activity concentration of ^{40}K, $^{\Sigma}U$ and ^{232}Th are the highest in sediments collected in the A horizon (alluvial soil) and the lowest in the S horizon. Figure 3 presents parallel comparison between activity concentrations in all three horizons for sediments of 10 rivers. Rivers are presented with numbers which can be found in the section Methods. ^{232}Th and $^{\Sigma}U$ concentrations vary depending on the type of parent rock, alteration or metamorphic processes. Because of the correlation between Th and U concentration in soil, it is expected that Th/U concentration ratio should be relatively constant. Th/U mass concentration ratio should be in the range of 2.8±1.3 for sedimentary rocks (Papaefthymiou et al., 2011). Any significant alteration of the Th/U ratio may indicate enrichment or depletion of natural uranium. Significant deviation from this expected value may be indication for the contamination due to increased industrial activity (Krmar et al., 2009).

Figure 4 shows individual Th/U mass concentration ratios for A, OB and S horizons for all locations. Point values are represented with standard uncertainty bars (k = 1). Arithmetic mean was calculated (central line) with combined uncertainty (bands above and below central line). Mean value of Th/U mass concentration ratio with combined uncertainty for A, OB and S horizons are 3.50±0.56, 4.09±0.65 and 2.83±0.48.

Table 1. Activity concentration (Bq/kg) of radionuclides $^{\Sigma}U$, ^{232}Th, ^{40}K in A horizon of river sediments and calculated external dose equivalent

Location	A horizon					
	$^{\Sigma}U$		^{232}Th		^{40}K	
	A (Bq/kg)	**H* (nSv/y)	*A* (Bq/kg)	**H* (nSv/y)	*A* (Bq/kg)	**H* (nSv/y)
Bjelica	27.3±4.1	8.87	34.1±5.1	34.7	468±70	58500
Gruza	31.2±4.7	10.15	37.4±5.6	38.2	456±68	57000
West Morava 1	38.5±5.8	12.56	43.8±6.6	44.6	443±67	55500
Ribnica	16.0±2.4	5.21	16.1±2.4	16.5	175±26	21900
Lopatnica	11.0±1.6	3.58	11.6±1.7	11.8	175±26	21900
Ibar	34.7±5.2	11.32	26.9±4.0	27.4	377±57	47200
Dicina	35.7±5.4	11.64	46.7±7.0	47.7	489±73	61200
Cemernica	33.5±5.1	11.00	35.5±5.3	36.0	425±64	53200
Kamenica	21.7±3.3	7.39	26.3±3.9	26.9	302±45	37800
South Morava	35.8±5.4	11.68	27.1±4.1	27.6	480±72	60000
West Morava 2	31.8±4.8	10.38	40.9±6.1	41.7	528±79	66100

**H* - External dose equivalent at 1 m above the soil surface for radionuclides evenly distributed through soil layer at specific horizon

Table 2. Activity concentration (Bq/kg) of radionuclides $^{\Sigma}U$, ^{232}Th, ^{40}K in OB horizon of river sediments and calculated external dose equivalent

Location	OB horizon					
	$^{\Sigma}U$		^{232}Th		^{40}K	
	A (Bq/kg)	**H* (nSv/y)	*A* (Bq/kg)	**H* (nSv/y)	*A* (Bq/kg)	**H* (nSv/y)
Bjelica	27.0±4.0	8.77	32.4±4.9	33.0	432±65	54100
Gruza	25.4±3.8	8.26	40.7±6.1	41.5	492±74	61600
West Morava 1	18.6±2.8	6.07	30.6±4.6	31.2	341±51	42700
Ribnica	12.3±1.8	4.00	15.8±2.4	16.1	154±23	19300
Lopatnica	10.1±1.5	3.29	12.4±1.9	12.6	154±23	19300
Ibar	27.0±4.1	8.81	33.7±5	34.3	429±64	53700
Dicina	40.6±6.1	13.24	41.9±6.3	42.7	516±78	64600
Cemernica	33.0±5.0	10.84	36.3±5.4	37.0	365±55	45700
Kamenica	13.0±2.0	4.43	16.9±2.5	17.2	257±39	32200
South Morava	30.5±4.6	9.95	45.5±6.8	46.4	640±96	80100
West Morava 2	27.0±4.1	8.81	26.7±4.0	27.2	381±57	47700

**H* - External dose equivalent rate at 1 m above the soil surface for radionuclides evenly distributed through soil layer at specific horizon

Table 3. Activity concentration (Bq/kg) of radionuclides $^{\Sigma}U$, ^{232}Th, ^{40}K in S horizon of river sediments and calculated external dose equivalent

Location	S horizon					
	ΣU		^{232}Th		^{40}K	
	A (Bq/kg)	**H* (nSv/y)	*A* (Bq/kg)	**H* (nSv/y)	*A* (Bq/kg)	**H* (nSv/y)
Bjelica	20.0±3.0	6.49	13.8±2.1	14.1	293±44	36700
Gruza	32.9±4.9	10.71	40.0±6.0	40.8	471±71	58900
West Morava 1	0**	/	0**	/	0**	/
Ribnica	11.6±1.7	3.80	7.8±1.2	8.0	172±26	21500
Lopatnica	12.9±1.9	4.22	9.8±1.5	10.0	144±22	18000
Ibar	31.7±4.8	10.33	31.9±4.8	32.5	462±69	57800
Dicina	41.0±6.1	13.37	43.1±6.5	43.0	568±85	71100
Cemernica	24.2±3.6	7.89	20.0±3.0	20.5	326±49	40800
Kamenica	8.7±1.3	2.83	4.21±0.63	4.3	121±18	15100
South Morava	25.7±3.9	8.39	36.5±5.5	37.2	559±84	70000
West Morava 2	23.9±3.6	7.80	17.5±2.6	17.9	338±51	42300

**H* - External dose equivalent rate at 1 m above the soil surface for radionuclides evenly distributed through soil layer at specific horizon ** - value below the detection limit

It can be seen that all values for Th/U ratio are inside the expanded uncertainty interval of $k = 2$. We could expect that values outside the expanded uncertainty interval of $k = 3$ could be considered as beyond expected ratios for thorium and uranium.

Table 4. Mean activity concentration with range of activities of natural radionuclides $^{\Sigma}U$, ^{232}Th, and ^{40}K for A, OB and S horizon for all locations compared to the results from the literature of activity concentration measurement in river sediment in the region.

Radionuclide	Σ_U			^{232}Th			^{40}K		
Horizon	A	OB	S	A	OB	S	A	OB	S
Mean activity concentration (Bq/kg) of River sediments, present study									
Mean	29	24	23	31	30	22.5	393	378	346
Range	11-39	10-41	9-41	12-47	12-46	4-43	175-529	154-640	121-568
Mean activity concentration (Bq/kg) of River sediments reported in the region, data from the literature									
Danube river, Serbia (Todorovic et al., (2008))	(^{238}U) 42±12			36±9			445±88		
Bega channel, Serbia (Todorovic et al., (2008))	71±15			49±4			520±48		
Tisza river, Serbia (Todorovic et al., (2008))	37.3±7.3			36.5±5.0			486.5±20.7		

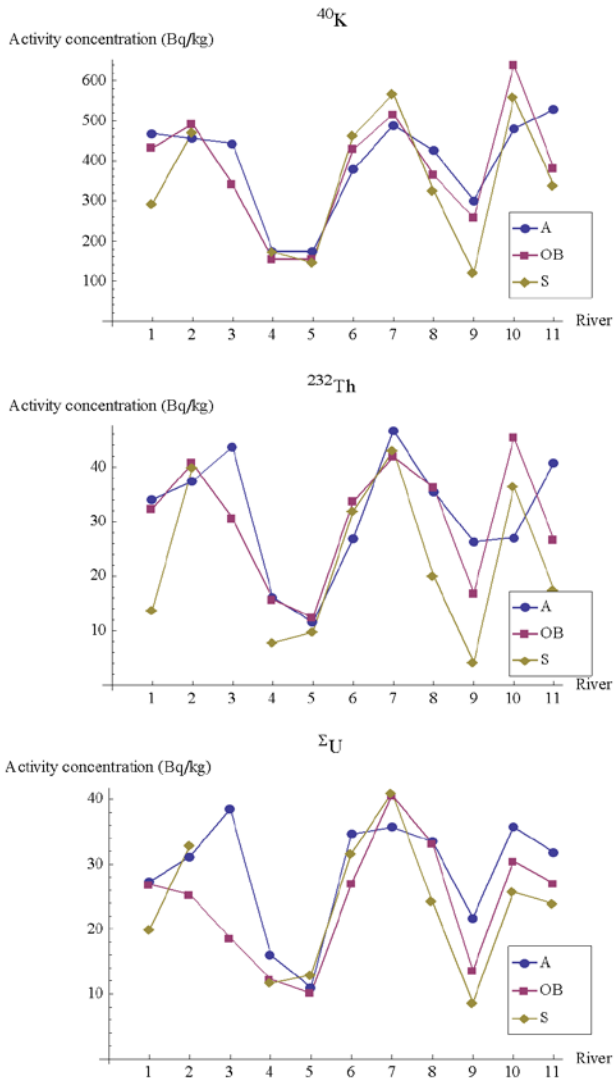

Figure 3. Comparison between activity concentrations in samples from the A, OB and S horizons for 10 rivers.

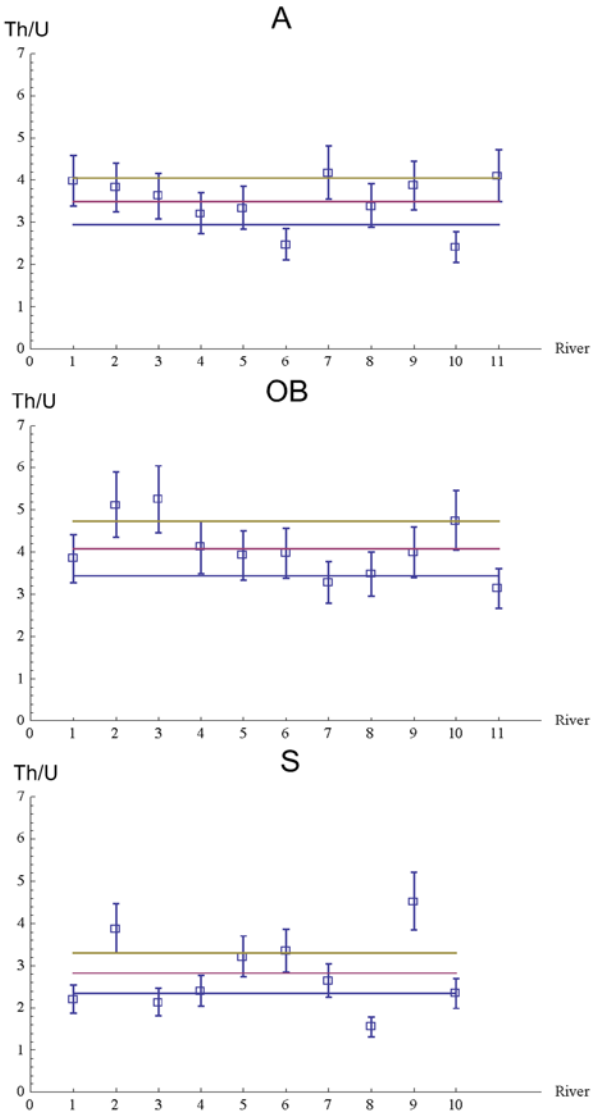

Figure 4. Th/U ratios for A, OB and S horizons for rivers under investigation.

Bearing in mind that all calculated values for Th/U ratio are inside the expanded uncertainty of k = 2, it can be concluded that there is no indication for the enrichment or depletion of $^{\Sigma}$U in the river sediments under investigation. Observed local variation of Th/U ratio could be explained with different solubility and mobility of uranium compared to thorium. That could lead to dissolution and readsorption of uranium, introducing local variations (Powell et al., 2007). Graph representing S horizon on figure 4, contains one location less than for A and OB horizons, because location "West Morava 1" with activities under detection limit is not presented (see Table 3).

Tables 1, 2, and 3, present calculated external dose equivalent for radionuclides from the ^{238}U and ^{232}Th series and ^{40}K in the river sediment for A, OB, S horizons. Mean value of total external dose equivalent per year for A, OB, and S horizons are 49160 nSv, 47402 nSv and 43250 nSv, respectively.

After inhalation and ingestion of radionuclides, that are primarily alpha particle emitters, increment of cancer risk is related to tissues that are deposition sites. In the case of ^{238}U and ^{232}Th intake, deposition sites like lungs, colon, bones, liver contribute to the total risk (IARC, 2001; Župunski et al., 2010). Table 5 presents cancer morbidity risk for lifetime exposure via inhalation and direct ingestion. Calculation was performed for A horizon because area is suitable for extended and long term stay (Fig. 2). Cancer morbidity risk was calculated for ^{238}U and ^{232}Th because radionuclides from these series contribute to the dose which reflects their intake from the air and diet (IARC, 2001). Concentration of potassium in soft tissues is constant and under homeostatic control. Besides from specific routes of exposure, this relatively constant ^{40}K body content contributes to the annual effective dose equivalent with 165 μSv (IARC, 2001). Mean value of calculated cancer morbidity risk of ^{232}Th inhalation is 0.94×10^{-6}, range (0.35 – 1.4) $\times10^{-6}$ and for ingestion 1.7×10^{-6}, range (0.6 – 2.5) $\times10^{-6}$. Mean value of calculated cancer morbidity risk of ^{238}U inhalation is 0.2×10^{-6}, range (0.07 – 0.3) $\times10^{-6}$, and for ingestion 0.43×10^{-6}, range (0.17 – 0.58) $\times10^{-6}$. It can be seen that the highest risk is related to the exposure to radionuclide ^{232}Th via direct ingestion exposure.

4. CONCLUSIONS

This paper deals with activity concentration measurements of natural radionuclides $^{\Sigma}$U, ^{232}Th and ^{40}K in the samples of the three levels of bank profiles in the West Morava basin. River deposits are massively exploited, so results of natural radionuclides influence have special significance in protection of living environment. Regarding content of natural radionuclides, equivalent dose was determined as well as inhalation and ingestion morbidity risks as contributors to the total risk of the internal and external exposure.

It can be concluded that mean activity concentration for ^{40}K (393 Bq/kg – A, 378 Bq/kg – OB, 346 Bq/kg – S), $^{\Sigma}$U (29 Bq/kg – A, 24 Bq/kg - OB, 23 Bq/kg – S) and ^{232}Th (31 Bq/kg – A, 30 Bq/kg – OB, 22.5 Bq/kg – S) for all three horizons are lower than world average values.

Performed statistical analysis showed only local insignificant variation of the Th/U mass concentration ratio in sediments, so it can be concluded that there is no indication for the enrichment or depletion of $^{\Sigma}$U in the sediments under investigation.

Health risk assessment showed very small cancer morbidity risks after inhalation and ingestion of the radionuclides ^{232}Th and ^{238}U, but very large uncertainties are related to the available cancer risk coefficients. A great effort should be invested in the future in the epidemiology studies of the population exposed to elevated concentrations of the radionuclides in their environment. Also, available biokinetic models should be improved, regarding lower associated uncertainties.

Table 5. Lifetime cancer morbidity risk after exposure to ^{238}U and ^{232}Th through inhalation and ingestion

	A horizon			
	^{238}U		^{232}Th	
	risk inhalation $\times10^{6}$	*risk ingestion* $\times10^{6}$	*risk inhalation* $\times10^{6}$	*risk ingestion* $\times10^{6}$
Bjelica	0.18	0.41	1.0	1.8
Gruza	0.21	0.47	1.1	2.0
West Morava 1	0.26	0.58	1.3	2.3
Ribnica	0.11	0.24	0.48	0.85
Lopatnica	0.07	0.17	0.35	0.6
Ibar	0.23	0.52	0.81	1.4
Dicina	0.24	0.54	1.4	2.5
Cemernica	0.23	0.50	1.1	1.9
Kamenica	0.15	0.33	0.79	1.4
South Morava	0.24	0.54	0.81	1.4
West Morava 2	0.22	0.48	1.23	2.1

Results of our investigations may form a basis for investigations in other relevant fields as environment protection, geomedicine, agriculture, forestry, soil quality and mineral resource investigation.

ACKNOWLEDGMENTS

This research is a part of the projects "Physics and Chemistry with Ion Beams" [III45006], "Joint research of measurements and effects of ionizing and UV radiation in medicine and environmental protection" [III43011] and „Geologic and ecotoxicologic research in identification of geopathogenic zones of toxic elements in drinking water reservoirs – research into methods and procedures for reduction of biochemical anomalies" [OI176018]. The authors are sincerely grateful to Ministry of Education and Science, Republic of Serbia, for supporting this study.

REFERENCES

Bikit, I., Varga, E., Čonkić, Lj., Slivka, J., Mrđa, D., Ćurčić, S., Žikić-Todorović, N. & Veskovic, M., 2005. *Radioactivity of the Bega sediment—case study of a contaminated canal*. Appl. Radiat. Isot., 63, 261–266.

Brenner, J. D., Doll, R., Goodhead, T. D. et al., 2003. *Cancer risks attributable to low doses of ionizing radiation: Assessing what we really know*. PNAS, 100, 13761-13766.

Brenner, J. D. & Sachs, K. R., 2006. *Estimating radiation-induced cancer risks at very low doses: rationale for using a linear no-threshold approach*. Radiat. Environ. Biophys., 44, 253-256. DOI: 10.1007/s00411-006-0029-4.

Bogen, J., Bolviken, B. & Ottesen, R. T., 1992. *Environmental studies in Western Europe using overbank sediment. Erosion and Sediment Transport Monitoring Programmes in River Basins*. Proceedings of the Oslo Symposium, IAHS Publ. no. 210, 317-325

Chen, S. Y., 1991. *Calculation of effective dose-equivalent responses for external exposure from residual photon emitters in soil*. Health Phys., 60, 411-426.

Clouvas, A., Xanthos, S., Antonopoulos-Domis, M. & Silva, J., 2000. *Monte Carlo calculation of dose rate conversion factors for external exposure to photon emitters in soil*. Health Phys., 78, 3, 295-302.

Dragović, S., Janković, Lj., Onjia, A. & Bačić, G., 2006. *Distribution of primordial radionuclides in surface soils from Serbia and Montenegro*. Radiat. Meas., 41, 611-616.

International agency for research on cancer (IARC), 2001. *Monographs on the Evaluation of Carcinogenic Risks to Humans, Ionizing Radiation Part 2: Some Internally Deposited Radionuclides, Volume 78*. IARCPress, Lyon, 1-46.

Kandić, A., Spasić Jokić, V. & Todorović, D., 1993. *Intercomparison of semiconductor HPGe detectors*. Proceedings of full papers, XVII Conference of Yugoslav Society of Radiation Protection Yugoslav Society of Radiation Protection. Beograd-Vinca 1993, 195-198 (in Serbian).

Kidd, S., 2011. *Uranium supply for the nuclear future.* Energy and Environment, volume 22, No. 1+2. Multi-science publishing co. LTD., United Kingdom.

Krmar, M., Slivka, J., Varga, E., Bikit, I. & Vesković, M., 2009. *Correlations of natural radionuclides in sediment from Danube*. J. Geochem. Explor., 100, 20–24.

National Research Council, 2006, *Health Risks from Exposure to Low Levels of Ionizing Radiation: BEIR VII Phase 2*. Washington, DC: The National Academies Press, 259-290.

Likar, A., Vidmar, T. & Pucelj, B., 1998. *Monte Carlo determination of gamma-ray dose rate with the geant system*. Health Phys., 75, 165-169.

Papaefthymiou, H. V., Chourdakis, G. & Vakalas, J., 2011. *Natural radionuclides content and associated dose rates in fine-grained sediments from Patras-Rion Sub-Basins*. Radiat. Prot. Dosim., 143, 1, 117–124.

Powell, B. A., Hughes, L. D., Soreefan, A. M., Falta, D., Wall, M. & DeVol, T. A., 2007. *Elevated Concentrations of Primordial Radionuclides in Sediments from the Reedy River and Surrounding Creeks in Simpsonville, South Carolina*. J. Environ. Radioactiv., 94, 3, 121-128.

Radenković, M. B., Cupać, S. A., Joksić, J. D. & Todorović, D. J., 2008. *Depleted Uranium Mobility and Fractionation in Contaminated Soil (Southern Serbia).* ESPR, 15, 1, 61–67.

Radosevic, B., Kovacevic, J. & Jovic, V., 2002. *Natural Occurring Uranium in Serbia and Possible Environmental Effect*. JEPE, 3, 706-718.

Saito, K. & Jacob, P., 1995. *Gamma ray fields in the air due to sources in the ground.* Radiat. Prot. Dosim., 58, 29-45.

Spasić-Jokić, V., Župunski, Lj., Janković, Lj. & Gordanić, V., 2011 a. *Effective dose estimation and lifetime cancer mortality risk assessment from exposure to Chernobyl 137Cs on the territory of Belgrade City and the region of Vojvodina, Serbia*. ESPR, 18, 708–715.

Spasić-Jokić, V., Župunski, Lj., Janković-Mandić, Lj., Gordanić, V., Mitrović, Z., Vujičić, B. & Župunski, I., 2011 b. *Environmental risk of natural radiation sources*. Proceedings of full papers, IEEE International Symposium on Medical Measurements and Applications (MeMeA). Bari, Italy, 638 – 641.

Stankovic, S. J. & Ilic, R. D., 2002. *Applications of Monte Carlo software package FOTELP in doseimetry and radiation protection*. Proceedings of full papers, Applied physics in Serbia-APS. Belgrade, Serbia, 337-340.

Tereesh, M. B., Radenkovic, M. B., Kovacevic, J. & Miljanic, S. S., 2013. *Terrestrial Radioactivity of the Jabal Eghei Area in Southern Libya and Assessment of the Associated Environmental Risks*. Radiat. Prot. Dosim. 153, 4, 475–484.

Todorovic, N., Bikit, I., Mrdja, D., Slivka, J., Forkapic, S. & Veskovic, M., 2008. *Search for TENORM in the Tisza River sediment*. J. Radiat. Res., 49, 3, 241-248.

UNSCEAR, 2000. *Sources and effects of ionizing radiation. Report to General Assembly, with Scientific Annexes, Annex b*, United Nations, New York, 89-94.

U.S. Environmental Protection Agency (EPA), 1999. *Cancer Risk Coefficients for Environmental Exposure to Radionuclides, Federal Guidance Report 13. EPA 402 –R-99 -001*. Oak Ridge National Laboratory, 1-9.

U.S. Environmental Protection Agency (EPA), 2000 a. *Soil Screening Guidance for Radionuclides: Technical Background Document, Publication 9355.4-16*. Washington, DC 20460, Part 2, 1-33.

U.S. Environmental Protection Agency (EPA), 2000 b. *Soil Screening Guidance for Radionuclides: User's Guide, EPA/540-R-00-007*. Washington, DC 20460, 1-7.

Životić, D., Gržetić, I., Lorenz, H. & Simić, V., 2008. *U and Th in Some Brown Coals of Serbia and Montenegro and Their Environmental Impact*. ESPR, 15, 2, 155–161.

Županski, Lj., Spasić-Jokić, V., Trobok, M. & Gordanić, V., 2010. *Cancer risk assessment after exposure from natural radionuclides in soil using Monte Carlo techniques*. ESPR, 17, 9, 1574–1580.

Received at: 16. 12. 2013
Revised at: 05. 04. 2014
Accepted for publication at: 07. 04. 2014
Published online at: 23. 04. 2014