Бюллетень Почвенного института им. В.В. Докучаева. 2019. Вып. 100.
Dokuchaev Soil Bulletin, 2019, 100

УДК 631.4
DOI: 10.19047/0136-1694-2019-100-117-132

Cite this article as:
Sanzharova N.I., Ratnikov A.N., Fesenko S.V., Sviridenko D.G., Review on uranium in soil: levels, migration, Dokuchaev Soil Bulletin, 2019, V. 100, pp. 117-132, DOI: 10.19047/0136-1694-2019-100-117-132

Review on uranium in soil: levels, migration

© 2019 г. N. I. Sanzharova[*], A. N. Ratnikov[**],
S. V. Fesenko[***], D. G. Sviridenko[****]

Russian Institute of Radiology and Agroecology,
109 km Kievskoye shosse, 249032 Obninsk,
Kaluga region, Russian Federation,
[*]*e-mail: natsan2004@mail.ru,*
[**]*e-mail: ratnikov-51@mail.ru,*
[***]*e-mail: corwin_17f@mail.ru,*
[****]*e-mail: sedelnikov167@gmail.com.*

Received 05.09.2019, *Revised* 05.09.2019, *Accepted* 18.12.2019

Abstract: The problems of radioactive contamination of biosphere with the ^{238}U in recent years have attracted a large number of experts in various fields of knowledge. Natural radionuclides are an integral part of the biosphere. They are concentrated everywhere: in rocks, in soil, water, air and food. Various types of phosphorus fertilizers and other natural ameliorants used in agriculture are also an additional source of heavy natural radionuclides. The review describes the behaviour and migration of ^{238}U in biosphere, which depend on various factors acting simultaneously. It is noted that the behaviour of ^{238}U in the soil depends on the forms of its presence in it and the physical and chemical properties of the soil. The processes of sorption and desorption of ^{238}U by different soil types are significant indicators.

Keywords: ^{238}U; radioactive contamination; soils, behaviour of ^{238}U, uranium forms.

INTRODUCTION

Intensification of energy development on nuclear and fossil organic fuels, extraction of rare and non-ferrous metals, production of phosphorus fertilisers with a high content of heavy natural radionuclides (HNR) can lead to further dispersion of uranium in the biosphere. A sharp increase in technogenic flows of HNR in the biosphere, including in the field of agricultural production, necessitates the study of patterns of inclusion and migration of HNR in biological chains. To reliably assess the sanitary and hygienic danger of HNR, it is necessary to study the processes of interaction of these radionuclides with the soil. Migration of HNR in the terrestrial biosphere is a serious theoretical and practical problem, largely unsolved by modern science.

ROCKS AND SOILS

Isotopes of uranium are widely distributed in the Earth's crust. Uranium mass concentrations in parent rocks vary from 0.1 to 1,250 ppm with a typical range of 2.0–6.1 ppm or 25–75 Bq kg^{-1} ^{238}U (Table 1) (Rubtsov, Pravdina, 1971). The estimated average ^{238}U concentration in the continental crust is 32.9 Bq kg^{-1}.

Table 1. Typical U concentrations in various rocks (adapted from Gascoyne, 1992)

Rock types	**U (mg kg^{-1})**	**^{23S}U (Bq kg^{-1})**
Silicic rocks (granites-dacites)	2.2–6.1	27–75
Basalts	0.1–1	1–12
Gneiss	2	25
Schist	2–5	25–62
Limestone	2	25
Black shale	3–1,250	37–15,000
Phosphates	50–300	620–3,700

Relatively high uranium concentrations are observed in black shale and phosphate rocks. The high uranium concentrations in shale are likely due to associations of clay-rich material of organic origin, whereas phosphate rocks of sedimentary origin are composed of minerals rich in uranium. Silicic rocks, granites and basalts tend to have similar uranium concentrations.

Concentrations of uranium in the soil depend, to a large extent, on its concentration in the parent rock. Parent rocks are affected by many environmental factors, resulting in soil formation. In particular, because of uranium weathering from the parent rocks, uranium can be transferred and deposited as loess, silt placers and Tertiary soils (Table 2).

Table 2. Uranium concentrations in soils (Alexakhin et al., 1990; Baranov, Morozova, 1971; Saric et al., 1995)

Landscape	***Type* of soil**	**Texture**	**^{238}U *mg kg^{-1}***
Tundra	Soddy-Podzolic*	Loam	2.53 ± 0.6
Taiga	Soddy-Podzolic*	Loam	20.1 ± 2.5
Taiga	Soddy-Podzolic	Loam	4.7 ± 2.9
Mixed forests	Typical chernozem	Clay	2.7 ± 0.2
Deciduous forest	Grey forest	Loam	1.2 ± 0.1
Meadows, steppe	Chernozem, Brown earth[b]	Clay, clay-loam	3.03 ± 0.7
	Chernozem	Clay	2.6 ± 0.2
	Light brown	Clay; clay-loam	2.9 ± 0.7
Dry Subtropics	Red earth	Clay-loam	5.1 ± 0.3
Desert	Light brown desert soil		5.33 ± 0.5

*Area with elevated natural background, the Komi Republic;
[b] Samples taken from the arable soil.

The data in table 2 show that ^{238}U concentrations in soils in areas with normal natural radiation background vary from 0.4 to 20 mg kg^{-1}. The highest mean values of 20.1 + 2.5 mg kg^{-1} were found in Soddy-Podzolic Loam soils of the taiga.

Available data on uranium measurements in soils of different countries of the world are also reported in an UNSCEAR report (Saric et al., 1995), table 3 shows that ^{238}U concentrations in soil vary over a wider range than that reported in table 2, most likely because of the wider range of soil types investigated, inhomogeneity of the soils studied, the wider range of uranium concentrations in the parent rocks and other properties (in some cases localised) favouring uranium accumulation in the topsoil. Even within some countries, the ranges of reported ^{23S}U concentrations in soils covered a large fraction of ^{238}U concentrations range presented in table 3. For example, this is the case for China (0.2–55 mg kg^{-1}) and Thailand (0.2–29.7 mg kg^{-1}).

Table 3. ^{238}U-concentrations in soil on a country-by-country basis (adapted from Saric et al., 1995)

Region	Country	Bq kg^{-1}		mg kg^{-1}	
		Mean	**Range**	**Mean**	**Range**
Africa	Algeria	30	2–110	2.4	0.2–8.8
	Egypt	37	6–120	3.0	0.5–9.6
North America	Costa Rica	46	11–130	3.7	0.9–10.5
	United States	35	4–140	2.8	0.3–11.3
East Asia	China	33	2–690	2.7	0.2–55.5
	India	29	7–81	2.3	0.6–6.5
	Japan	29	2–59	2.3	0.2–4.7
	Kazakhstan	37	12–120	3.0	1.0–9.6
	Malaysia	66	49–86	5.3	3.9–6.9
	Thailand	114	3–370	9.2	0.2–29.7
West Asia	Armenia	46	20–78	3.7	1.6–6.3
	Syrian Arab Republic	23	10–64	1.8	0.8–5.1
North Europe	Lithuania	16	3–30	1.3	0.2–2.4
	Norway	50		4.0	
West Europe	Ireland	37	8–120	3.0	0.6–9.6
	Switzerland	40	10–150	3.2	0.8–12.1
East Europe	Bulgaria	40	8–190	3.2	0.6–15.3
	Hungary	29	12–66	2.3	1.0–5.3
	Poland	26	5–120	2.1	0.4–9.6
	Romania	32	8–60	2.6	0.6–4.8
	Russian Federation	19	1–67	1.5	0.1–5.4
	Slovakia	32	15–130	2.6	1.2–10.5
South Europe	Albania	23		1.8	
	Croatia	110	21–77	8.8	1.7–6.2
	Greece	25	1–240	2.0	0.1–19.3
	Portugal	49	26–82	3.9	2.1–6.6
Median		35	17–60	2.8	1.4–4.8

The median value for the ^{23S}U concentration in soil calculated based on these data is 35 Bq kg^{-1} or 2.8 mg kg^{-1}. However, there are some areas in the world where the normal range of variation of naturally occurring radionuclides in soils or waters is greatly exceeded. These high radiation background regions are known as “Enhanced Natural Radiation Areas” or “ENRA”.

Maximum values were found to be for some Azerbaijan sites, Ramsar, Iran and the Komi Republic of the Russian Federation, where ^{238}U activity concentrations were two to three orders of magnitude higher than the average worldwide value of 32 Bq kg^{-1} reported in UNSCEAR (2008).

^{238}U AND AGRICULTURAL ECOSYSTEMS

Heavy natural radionuclide uranium participates in the formation of the Earth crust. Atomic mass of the natural uranium is 238. It was discovered by the German chemist M.H. Klaproth in 1789. Uranium is widely spread in nature (Tananaev, 2011). It was discovered in considerable concentrations in earth material, oceans, the Earth mantle and meteorites. The half-life of ^{238}U is 4.49 × 10^{9} years (4.49 billion years) ^{238}U is the α-emitter with maximum energy of 4.7 MeV.

In areas where rocks containing ^{238}U and mine dumps come to the surface, there is an increased level of contamination of the biosphere with uranium. In the industrial production of phosphate fertilizers, there is also additional contamination of soils and vegetation with uranium (Ishchenko, 1988)

The content of mobile forms of ^{238}U in ammophos and phosphogypsum comes up to 5.95 and 23% from gross content in water-soluble form.

Content of uranium in earth crust is 2·10^{-4}% (Tananaev, 2011). Average content of ^{238}U in all earth soils is 2.4 × 10^{-4}% (Alexakhin et al., 1992). The ^{238}U content in grey forest soils is equal to 3.4 × 10^{-4}%, in chernozems – 3.0 × 10^{-4}%, in the upper layers of sierozemic soils of semideserts it comes up to 2.2 × 10^{-4}%. The average content of ^{238}U in soils all over the world is 24.4 Bq kg^{-1}. Global technogenic flows of ^{238}U in biosphere were approximately 1.5 × 10^{14} Bq at the operation of fuel cycle enterprises working on nuclear and fossil fuels and 1 × 10^{14} Bq at the application of phosphoric fertilizers. Volume of ^{238}U in ploughed horizon of agricultural areas is 1.4×10^{17} Bq (Alexakhin et al., 1992).

Uranium belongs to the class of water migrants, and in natural media, it exists in (IV) and (VI) valency forms. In continental surface waters, the distribution of ^{238}U follows the natural and climatic zone sequence, where the concentration of this radionuclide varies from $n \times 10^{-8}$to $n \times 10^{-5}$ L g^{-1}. In the waters of large rivers flowing through different climatic zones, the concentration of ^{238}U for the Northern Hemisphere increases from north to south (Alexakhin et al., 1990). In continental surface waters, the distribution of ^{238}U follows the natural and climatic zone sequence, where the concentration of this radionuclide varies from $n\times10^{-8}$to $n\times10^{-5}$ lg^{-1}. In the waters of large rivers flowing through different climatic zones, the concentration of ^{238}U for the Northern Hemisphere increases from north to south (Alexakhin et al., 1990).

^{238}U in natural waters exists as a uranyl ion UO^{2+}, which forms complex compounds with inorganic and organic acids. The ability of the uranyl ion to form complexes with ions such as SO_4^{2-}, Cl^-, F^-, NO_3^-, CO_3^-, CH_3COO^- etc., to become a part of the negatively charged colloidal sols of the hydroxide and to form complexes of uranium-organic compounds plays a major role in the migration of ^{238}U to natural waters (Drichko, 1983). The forms of ^{238}U in surface waters depend on pH and the content of CO_2 and C_{org}. In neutral and slightly acidic waters of the cold and temperate zones of the humid zone, the dissolved ^{238}U migrates in the form of fulvates, hydroxycomplexes or pseudocolloids. For weakly alkaline waters of the arid zone containing carbonate-ion, the main forms of finding ^{238}U are stable carbonate complexes $[UO_2(CO_3)_2(H_2O_2)_2]^{2-}$ and $[UO_2(CO_3)_3]^{4-}$. The researchers also assume the presence of bicarbonate complex compounds ^{238}U (Alexakhin et al., 1990).

Anthropogenic sources of environmental pollution by ^{238}U, including agricultural ecosystems, are: extraction of uranium in mines and a formed uranium trace, the atomic and nuclear power industry, agriculture (mineral fertilizers, primarily phosphoric, may contain ^{238}U in high concentrations), uranium content in irrigation water (Alexakhin et al., 1992; Drichko, 1983), the use of radioactive projectiles in military conflicts.

With the systematic use of waste (phosphogypsum) and complex phosphoric fertilizers, accumulation of gross and mobile forms of heavy natural radionuclides can occur in soils. It can reach the levels exceeding background concentrations by 1.5–2.0 times (Drichko, 1983).

The nature and direction of the soil formation processes determine the distribution of uranium in the soils of certain landscape zone. These factors result in the great variety of uranium content in soils. Uranium is accumulated mainly in the sod and the upper arable soil horizon.

^{238}U IN SOILS OF AGRICULTURAL LANDS

The major factor determining the behaviour of the ^{238}U, which enters the land cover, is its high mass concentration in the soils, whereby the chemical properties of the element itself play a more significant role in the behaviour of the radionuclide than the stable analogues (Alexakhin et al., 1990). At the same time, for ^{238}U present in soils in ultramicroconcentrations, in some cases, the carrier may not be ^{238}U, but some chemical elements being typomorphic for individual landscapes, which is confirmed by the deviation of the ratio ^{234}U/^{238}U from 1 (Titaeva, 1992).

Sorption of ^{238}U in soils depends on many factors (Alexakhin et al., 1990). The degree of oxidation of ^{238}U in soils is +4 and +6. In a highly reducing medium, ^{238}U is predominantly in 4-valence, and in oxidative – in 6-valence forms.

Sorption of ^{238}U in soils depends on the chemical properties, the physical and chemical state and concentration of elements, the granulometric and mineralogical composition of soils, the content of organic matter, the presence of certain ions in the solution, migratory-capable colloids, complexing agents, etc. (Alexakhin et al., 1990; Alexakhin et al., 1992; Drichko, 1983).

When pH varies from 5 to 6 ^{238}U appears in soils mainly in the form of well-sorbed hydrolysed forms of uranyl, and for this reason, the transition of the radionuclide into water extract from these soils is negligible. At a small (about $1M \times 10^{-3}$) content of carbonates, hydrolysis forms of uranyl prevail in the pH range of 2–12 (Iskra, Bakhurov, 1981). Therefore, even at a high pH of the solutions, the recovery of

^{238}U from the soils does not increase. Thus, at pH 11, the fraction of radionuclide that has passed from leached chernozem to water extract is only 1.5%. In neutral and alkaline media, carbonate and phosphate compounds predominate (Sultanbaev, Grigoryev, 1979), and this can reduce and weaken the absorption of ^{238}U in the soil (Iskra, Bakhurov, 1981). At pH above 10, the equilibrium shifts again towards the formation of hydrolysis forms, the recovery of ^{238}U decreases. In media, rich in $CaCO_3$ and $MgCO_3$, the absorption strength can increase due to the precipitation of $Ca_2(UO_2(CO_3)_3$ and $Ca,Mg(UO_2(CO_3)_3)$. The minimum absorption of ^{238}U by sod horizons of sod-podzolic and sod-meadow soils was found at pH –3, maximum (97–98%) at pH 5–8.

A significant concentration of humus, as a rule, increases the sorption of radionuclides in the soil (Rusanova, 1971; Sultanbaev, Grigoryev, 1979). Organic substances of the soil solution bind ^{238}U to complex compounds, differing in their solubility and stability (Evseeva et al., 1974). For example, the strength of the element complexes with humic acids is 1.5–2 times higher on average than with fulvic acids (Kichigin, Nosova, 2004). Under reducing conditions at pH 5, humic acids and, at pH 6.0–6.6, fulvic acids precipitate ^{238}U, in oxidative conditions, soluble compounds are formed. The carbonates concentration of $1M \times 10^{-3}$ almost eliminates the formation of soluble hydrolysis products and prevents the humate binding at the content of the humic acids lower than $0.1M \times 10^{-6}$. At their concentration of $50M \times 10^{-9}$ and low ($0.5M \times 10^{-3}$) carbonate content, the formation of uranium humates dominates (Rachkova et al., 2010). The effect of organic substances on the processes of absorption in the soil is due to the fact that they are themselves collectors of radioactive elements, or modify the absorption properties of other sorbents, and also form mobile compounds with radionuclides. Mechanisms of sorption and fixation of ^{238}U by soils have been poorly studied. Absorption process includes adsorption, ion exchange, precipitation, particularly due to oxidation-reduction reactions. Some authors (Evseeva et al., 1974) believe that the last of the above mechanisms is the most important in the process of sorption of uranium in the soil. At the same time, it was experimentally established that hexavalent uranium retains its valence in complexes with natural humic acid (Teterin et al., 2001). The mechanism of sorption consists of the exchange of hydrogen cations and uranyl UO^{2+}, that is why the capacity and rate of absorption depend on the acidity of the medium. Fe^{3+} ions inhibit the formation of uranyl humates due to sorption on the surface of particles of colloidal humic acid (Teterin et al., 2001) that prevents the migration of ^{238}U in the form of absorbed or molecular solutions of uranyl humates. At the same time, the addition of humic acid to the hematite suspension reduces the absorption of the radionuclide, the degree of reduction depends on the pH of the medium and the amount of humic acid. In addition, the dominant factor is the fraction of ^{238}U bound to humates. In the presence of fulvates, the sorption of the radionuclide by the clayey soil components is completely inhibited (Kornilovich et al., 2001).

The sorption of ^{238}U in soil is related to the size and specific surface area of soil particles (Arkhipov et al., 1986; Ishchenko, 1988; Ishchenko, Butnik, 1990; Rubtsov, Pravdina, 1971). For soil with a high content of physical clay and silt, the high values of the distribution coefficients were set (Drichko, 1983; Rubtsov, Pravdina, 1971; Rudnick, Gao, 2003; Saric et al., 1995; Titaeva, 1992). The content of ^{238}U in the solid phase of clay soils is 200–7.9 × 10^5 times higher than in the equilibrium solution. For desert-sandy, meadow, meadow-swamp soils, light and typical sierozems, direct dependence of the absorption and mobility of ^{238}U on the content of humus, potassium, calcium and physical clay has been established (Ishchenko, 1988; Ishchenko, Butnik, 1990). According to some estimates, the content of ^{238}U in the clay fraction is up to 3 times higher than its concentration in unfractionated soil. At the same time, little information indicates a lack of a close relationship between the sorption of radionuclides and the granulometric composition of soils. For example, for soils in the taiga zone, up to 50% of radionuclides of the uranium series are sorbed on the surface of particles with a size of 1–100 μm, and consolidation of aggregates in this range is accompanied by a decrease in the content of radioelements.

Fixation of ^{238}U in soils occurs due to its sorption by organic substances and clay minerals (illite, montmorillonite and kaolinite) (Alexakhin et al., 1990; Ratnikov et al., 2007; Rusanova, 1998; Taskaev, 1979). Limestone weakly absorbs ^{238}U, and crystallised earth silicon does not absorb the uranium at all (Alexakhin et al., 1990; Savenko, 2001). The sorption centres of two types are significant for sorption of radionuclides by clay minerals. These centres are located on the basal surfaces of mineral particles, where ion exchange takes place, and on the lateral faces with the formation of strong sorption complexes. According to Kornilovich et al. (2001), uranium was primarily absorbed on the lateral faces of clay minerals The X-ray diffraction analysis (Iskra, Bakhurov, 1981) showed that binding takes place according to the ion-exchange mechanism. ^{238}U competes with cations Ca^{2+}, Ba^{2+}, Mg^{2+} for sorption centres, but not with Na^+ and K^+. According to the structural data, uranyl forms dissolved and absorbed by clay minerals are identical. Under sub-acidic conditions, they correspond to the hexahydrated ion (Kornilovich et al., 2001). In the pH range 6–12, the sorbed forms of uranyl are represented by UO_2OH^+ and $(UO_2)_2(OH)$. Absorption of UO_2OH^+ predominates at pH 6, with a decrease in acidity, the ratio of sorption of $(UO_2)_2(OH)$ increases. The isotherms of ^{238}U (VI) sorption by clay minerals have a maximum at pH 3–6 with a sharp decrease in the absorption parameters to the sides of this interval (Kornilovich et al., 2001).

Influence of the capacity of cation exchange on the sorption of ^{238}U by soils is practically absent. For example, the removal of organic matter and free ferrous oxides from the clayey fraction of silty loam reduces its cation-exchange capacity, without changing the absorption rate of ^{238}U. Along with this, there is an opinion (Kornilovich et al., 2001) that in surface media the Fe content controls the pH-dependent adsorption of ^{238}U. Oxide forms of metals (goethite, hematite, boehmite, etc.), on the one hand, themselves serve as sorbents of a radionuclide, and on the other – when they are adsorbed, these oxides form films on the surface of clay minerals changing their ion-exchange properties. The distribution coefficient of the radionuclide between solutions of its salts and Fe minerals reaches 2×10^6 ml g^{-1}, which is 100 times greater than for secondary minerals.

Behaviour of ^{238}U and its forms in soils are determined both by the state of the radionuclide itself and by the physical and chemical properties of the soils. The original form of the radionuclide fallout plays an essential role in the processes of sorption and desorption of ^{238}U. For example, the sorption of ^{238}U by 5 types of soil (17 soil differences) with contrasting physical and chemical properties and indicators of texture when introducing a radionuclide in the form of a solution of uranyl nitrate is 82.2–99.2%, desorption by water is 1.1–7.5%, by 1N ammonium acetate solution – 2.4–40.8%, and by 2N HNO_3 solution – 51.3–79.2% (Alexakhin et al., 1985) (Table 4).

Table 4. Rate and strength of sorption of ^{238}U in different types of soils (in relation to the original quantity), %

Soil type	**Sorption**	**Desorption**		
		H_2O	**1N CH_3COONH_4**	**2N HNO_3**
Sod-podzolic	97.0	1.7	2.4	54.6
Dark-grey forest	98.4	1.2	11.7	59.5
Grey forest	96.9	1.4	3.2	51.3
Brown forest	88.4	1.2	36.2	79.2
Chernozem				
leached	98.5	1.2	13.4	68.4
podzolic	82.2	5.9	24.1	51.3
usual	85.9	7.5	9.6	57.0
southern	90.9	3.6	25.6	68.4
firm	98.1	1.1	22.2	76.4
Meadow chernozem	89.1	5.6	22.3	69.5
Sod	82.2	5.9	24.1	51.3
Meadow	99.2	1.5	23.6	60.8
Humic-gleyed	98.1	1.1	14.8	72.8
Meadow-swampy	87.5	3.6	23.0	69.2
Dark kastanozem	93.6	3.8	32.8	77.2
Kastanozem	87.0	4.8	40.8	61.2
Light kastanozem	87.4	4.4	26.4	72.0

In works of Arkhipov et al. (1986), there were established the limits of the absolute and relative content of gross, acid-soluble and exchangeable forms of natural radionuclides (^{238}U, ^{232}Th, ^{226}Ra, ^{210}Po and ^{210}Pb) in the soils of the agricultural territory of the country. The gross content of ^{238}U is 0.2–0.5 pCi g^{-1} in sod-podzolic soils, 0.6–1.0 pCi g^{-1} in chernozems, 0.7 pCi g^{-1}in kastanozems, 0.8 pCi g^{-1}in brown semi-desert soils, 0.7 pCi g^{-1}in sierozems. The content of the acid-soluble forms of ^{238}U is 6–26% of the gross content, depending on the agrochemical properties of the soil. The amount of exchangeable and water-soluble forms of ^{238}U in sod-podzolic soils varies between 18–40%, in chernozem – 9–12% of gross soil content (Arkhipov et al., 1986).

By this means, a generalisation of the accumulated data indicates that intensive and strong absorption of ^{238}U in soils occurs mainly due to organic matter and clay minerals. The processes of ion exchange, complexation, hydrolysis and oxidation-reduction designate the fixation of ^{238}U in soils.

To date, there is a lack of the investigation of the temporal aspects of ^{238}U absorption processes in various soils. The existing research methodologies do not allow evaluating fully the transformation of forms and mechanisms of radionuclide binding in soils. One of the ways to overcome these difficulties can be carrying out complex ^{238}U full-scale and model studies.

Study of Ishchenko and Butnik (1990) conducted on different soil types in Central Asia gives a comprehensive assessment of the effect of ^{238}U on the bioavailability, depending on the time of interaction of the radionuclide with the soil.

Transformation of the ^{238}U forms in time was studied on a typical sierozem. Soil samples were taken from the plots, where cotton was grown in the first and second years, in the third year – wheat. The concentration of ^{238}U was 53 mg kg^{-1} of soil.

^{238}U from soil samples was successively extracted with distilled water, acetate-ammonium buffer solution (pH 4.8), solution of citric acid (0.033M), mixture of citric (0.033M) and oxalic (0.5M) acids, hydrochloric acid. The content of the ^{238}U forms was calculated in %, assuming the initial value to be 100%.

It was found that uranium, introduced into the soil as a solution of $^{238}U(NO_3)_2$, comprises mainly exchangeable forms with soil components in desert-sandy soil, light and typical sierozems, and in meadow soils, the uranium interaction with soil components results in acid-soluble form. From such soils as desert-sandy, light and typical sierozems, most of the ^{238}U passed into aqueous and acetate-ammonium extracts (from 48.4 to 51.5%), into which readily soluble, exchangeable compounds and organomineral complexes are displaced. There is an inverse relationship between the physical and chemical soil properties and the amount of water-soluble and exchangeable forms of ^{238}U, and with the number of fixed ^{238}U compounds, the relationship is direct. Correlation coefficients between the amount of radionuclide in water-soluble and exchangeable forms and all indicated soil parameters turned out to be rather high (R = 0.73–0.88).

The ratio of water-soluble, exchangeable, acid-soluble and fixed forms had been determined for three years in order to study the influence of the time factor on the transformation of the occurrence of ^{238}U in a typical sierozem, the ratio of the forms of radionuclide was determined two weeks after its introduction into the soil. A year later, the number of exchangeable compounds of ^{238}U increased, and the number of acid-soluble compounds decreased. After two years of radionuclide stay in the soil, no further changes in the content of the various forms of ^{238}U were detected. With a 3-year incubation of ^{238}U in the soil, the number of fixed compounds increased significantly in comparison with their content in the first year, while the fraction of the exchangeable forms of the radionuclide remained unchanged (Ratnikov et al., 2019; Kornilovich et al., 2001) (Table 5).

Table 5. The content of various forms of ^{238}U, depending on the time of their stay in the soil, % of application

Timing of determination	**Water soluble**	**Exchangeable**	**Acid-soluble**	**Fixed**
In 2 weeks	7.5	41.5	27.6	23.8
In 1 year	5.9	47.0	21.6	25.5
In 2 years	4.1	49.1	20.8	25.6
In 3 years	2.7	47.5	10.7	39.2

The amount of water-soluble and essentially acid-soluble forms has decreased. This occurs because acid-soluble compounds of ^{238}U have been transformed into fixed ones with time. As for the water-soluble compounds of the radionuclide, it is possible that some of them migrated down the soil profile, and a part entered the plants, so their content in the plough horizon of the soil decreased.

Describing, in general, the behaviour and migration of ^{238}U in biosphere, it should be noted that all this occurs under the influence of various factors acting simultaneously. For the chemical element as a whole, it will depend mainly on three main parameters: 1) the chemical properties of the element; 2) the physical and chemical conditions of the environment; 3) the forms of finding the element in the initial rocks. The second factor is a function of climatic and partly geological conditions; the third one is completely determined by geological conditions.

REFERENCES

1. Aleksakhin R.M., Bufatin O.I., Malikov V.G., Perepelyatnikov G.P., Parashchukov N.P., Sanzharova N.I., Prister B.S., Zhukov B.I., Perepelyatnikova L.V., Shukhovtsev B.I., Fomkina N.D., Sakharov I.F., Kuznetsov V.K., *Radioekologiya oroshaemogo zemledeliya* (Radioecology of Irrigated Agriculture), Moscow: Energoatomizdat, 1985, 224 p.
2. Aleksakhin R.M., Arkhipov N.P., Burkhudarov R.M., Basilenko I.Ya., Drichko V.F., Ivanov Yu.A. ,Maslov V.I., Maslova K.I., Nikiforov V.S., Polikarpov G.G., Popova O.N., Sirotkin A.N., Taskaev A.I., Testov B.V., Titaeva N.A., Fevraleva L.T., *Tyazhelye yestestvennye radionuklidy v biosphere. Migratsiya I biologicheskoye deistviye na populyatsii i biogeotsenozy* (Heavy natural radionuclides in the biosphere. Migration and biological effect on populations and biogeocenoses), Moscow: Nauka, 1990, 367 p.
3. Aleksakhin R.M., Vasil'ev A.V., Dikarev V.G., Egorova V.A., Zhigareva T.L., Ivanov Yu.A., Koz'min G.V., Kruglikov B.P., Kruglov S.V., Morgunova E.A., Panteleev L.I., Povalyaev A.P., Popova G.I., Rasin I.M., Ratnikov A.N., Sanzharova N.I., Sarapul'tsev I.A., Sirotkin A.N., Sokolov V.A., Spirin E.V., Fesenko S.V., Filipas A.S., Khvostunov I.K., Shevchenko A.S., Shukhovtsev B.I., *Selskokhozyajstvennaya radioekologiya* (Agricultural radioecology), Moscow: Ekologiya, 1992, 400 p.
4. Arkhipov N.P., Fedorova T.A., Fevraleva L.T., Ratio of forms of compounds of heavy natural radionuclides in soils, *Pochvovedeniye*, 1986, Vol. 1, pp. 69–72.
5. Baranov V.I., Morozova N.G., Behavior of natural radionuclides in soils, In: Alexakhin R.M., Klechkovsky V.M. (Eds), *Sovremennye problemy radiobiologii* (Current problems of radiobiology), Moscow: Atomizdat, 1971, pp. 13–40.
6. Drichko V.F., Behaviour of heavy natural radionuclides in the natural environment, In: *Itogi nauki i tekhniki: Radiatsionnaya Biologiya,* 1983, Vol. 4, pp. 66–98.
7. Evseeva L.S, Perelman A.I., Ivanov K.E., *Geokhimiya urana v zone gipergeneza* (Geochemistry of uranium in the hypergenesis zone), Moscow: Atomizdat, 1975, 280 p.

8. Gascoyne M., Geochemistry of the actinides and their daughters, In: Ivanovich, M., Harmon, R.S. (Eds), *Uranium-series disequilibrium: applications to earth, marine, and environmental sciences*, Oxford, Clarendon Press, 1992, pp. 34–61.

9. Ishchenko G.S., *The patterns of migration of uranium-238 and thorium-232 in the soil-plant system in Central Asia, Cand. biol. sci. thesis*, RIARAE, Obninsk, 1988.

10. Ishchenko G.S., Butnik A.S., The forms of ^{238}U and ^{232}Th in the soils of Central Asia, *Agrokhimiya*, 1990, No. 1, pp. 92–96.

11. Iskra A.A., Bakhurov V.G., *Estestvennye radionuklidy v biosfere* (Natural radionuclides in the biosphere), Moscow: Energoizdat, 1981, 124 p.

12. Kichigin O.V., Nosova O.A. (2004). Potentiometric study of the stability of U (VI), Ni (II), Sr (II) complexes with soil humic acids, *Izvestiya vysshikh uchebnykh zavedeniy: Khimiya. Khimicheskaya Tekhnologiya*, 2004, Vol. 47, No. 1, pp. 94–96.

13. Kornilovich B.Y., Pshinko G.N., Kovalchuk I.A., Influence of fulvic acids on the interaction of U (VI) with clayey components of soils, *Radiokhimiya*, 2001, Vol. 43, No. 5, pp. 404–407.

14. Malikov V.G., Zhukov B.I., Perepelyatnikova L.V., Behavior of radionuclides in the soils of the North Caucasus, *Agrokhimiya*, 1988, No. 3, pp. 84–89.

15. Megumi K., Lagmay N.R., Concentration of natural radionuclides in soil particles in relation to their surface area, *J. Radiat. Res*., 1980, Vol. 21, No. 1, pp. 48–53.

16. Rachkova N.G., Shuktomova I.I., Taskaev A.I., The state of natural radionuclides of uranium, radium, and thorium in soils, *Eurasian Soil Sc*., 2010, 43, pp. 651–658, DOI: 10.1134/S1064229310060050.

17. Ratnikov A.N., Zhigareva T.L., Sviridenko D.G., Popova G.I., Petrov K.V., Kasyanenko A.A., Kuliyeva G.A., Behaviour of the ^{238}U in the soil-plant system, *Proc. Intern. Symposium "Agroecological Safety under the Conditions of Technogenesis"*, Part I, 2007, Kazan, pp. 307–311.

18. Ratnikov A.N., Sviridenko D.G., Popova G.I., Sanzharova N.I., Mikailova R.A., The Behaviour of Uranium in Soils and the Mechanisms of Its Accumulation by Agricultural Plants. In: Gupta D., Walther C. (eds) *Uranium in Plants and the Environment. Radionuclides and Heavy Metals in the Environment*, Springer, Cham, 2019, pp. 113–135, DOI: 10.1007/978-3-030-14961-1_5.

19. Rubtsov D.M., Pravdina E.I., The content and distribution of natural radioactive elements (uranium, radium, thorium) in the soils of some landscapes of the Northern Urals, *Inf. byul. nauch. Soveta po problemam radiobiologii AN SSSR*, 1971, Iss. 13, pp. 130–134.

20. Rudnick R.L., Gao S., Composition of the continental crust, *Treatise on Geochemistry*, 2003, Vol. 3, pp. 1–64.
21. Rusanova G.V., Content and regularities of the distribution of radium-226 in the soil cover of the region with the increased natural radiation, In: Popova O.N., Kodaneva R.P., Maslov V.I. (Eds), *Materials of Radioecological Research in Natural Biogeocenoses*, Syktyvkar, 1971, pp. 32–65.
22. Rusanova G.V., Forecast of long-term behaviour of radionuclides in soils of the taiga zone, *Proc. of VI International Symposium "Nuclear Urals. Industrial Urals"*, Ekaterinburg, 1998, pp. 52–54.
23. Savenko A.V., Sorption of UO_2^{2+} on calcium carbonate, *Radiokhimiya*, 2001, Vol. 43, No. 2, pp. 174–177.
24. Saric M.R., Stojanovic M., Babic M., Uranium in plant species grown on natural barren soil, *Journal of Plant Nutrition*, 1995, Vol. 18, Iss. 7, pp. 1509–1518.
25. Sheppard I.C., Campbell M.J., Retention of radionuclides by mobile humic components and soil particles, *Environ. Sci. Technol.*, 1980, Vol. 14, No. 11, pp. 1349–1353.
26. Sultanbaev A.S., Grigoryev A.F., The content of uranium in the soils and plants of the Tien Shan, In: *Perfection and cultivation of agricultural crops as the scientific basis for the intensification of plant growing in Kyrgyzstan*, Frunze, 1979, Iss. 16, pp. 240–250.
27. Taylor S.R., Trace element abundances and the chondritic Earth model, Geochimica et Cosmochimica Acta, Vol. 28, Iss. 12, 1964, pp. 1989–1998.
28. Tananaev I.G., *Uran: Uchebnoe posobie* (Uranium: Educational guidance), Moscow: NRNU MEPhI, 2011, 92 p.
29. Taskaev A.I., *Regularities in the distribution and migration of U, Th and Ra isotopes in the soil-vegetation cover of the region of increased natural radiation, Cand. biol. sci. thesis*, Syktyvkar, 1979.
30. Teterin Y.A., Nefedov V.I., Nikitin A.S., Teterin A.Y., Ivanov K.E., Maslakov K.I., Utkin I.O., Bubner M., Reich T., Pompe S., Heise K.H., Nitsche H., Interaction of UO_2^{2+} and Fe^{3+} ions with natural humic acid, *Russ. J. Inorg. Chem.*, 2001, Vol. 46, No. 6, pp. 886–891.
31. Titaeva N.A., *Yadernaya Geokhimiya* (Nuclear Geochemistry), Moscow: MSU, 2000, 336 p.
32. United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), Report to the General Assembly, with scientific annexes, 2008.