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RADIONUCLIDES

Current Status of Studies of ^{90}Sr Behavior in the Soil–Agricultural Plant System (Overview)

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Abstract—The necessity to study the mechanisms of ^{90}Sr behavior in ecosystems is conditioned by the radioactive contamination of the environment due to global fallout as a result of nuclear weapon tests, as well as incidents and accidents at radiation-hazardous facilities. The basic regularities of radionuclide behavior in soils and in the soil–agricultural plants system are described. The role of various factors affecting radionuclide migration in ecosystems is assessed. The key characteristics required for the parameterization of migration mathematical models used for predicting the radionuclide uptake by plants are presented.

Keywords: agricultural ecosystems, crops, migration, radionuclide, sorption, fixation, accumulation

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INTRODUCTION

Tests of nuclear weapons and radiation accidents and incidents have resulted in global and local contamination of agricultural land, which has determined the need to develop systems of protective and rehabilitation measures. The decisions to use polluted areas in agriculture are based on understanding the mechanisms of behavior of radionuclides in agroecosystems and on assessing the role of various factors that affect their migration and accumulation in agricultural products and then in the food of the population. The problem of the behavior of the radioactive isotope ^{90}Sr in agricultural and natural ecosystems is related to global fallout as a result of nuclear weapon tests and accidents at radiation-hazardous facilities. During the period of intensive global fallout in the temperate zone of the Northern Hemisphere, the pollution density of ^{90}Sr was about 2.1×10^4 Bq/m^2 [1]. As a result of the accident in the southern Urals (the Mayak Scientific-Production Association, 1957), the area of the East Ural radioactive trace excluded from economic use by 1959 due to its radioactive contamination was about 106000 km^2. Agricultural land occupied about 54% of its total area, including more than 29% of plowed lands. ^{90}Sr was the main dose-forming radionuclide of the fallout. The contamination density of the alienated land was 230–300 kBq/m^2 [2, 3]. After the accident at the Chernobyl nuclear power plant, about 150000 km^2 of the area of the former Soviet Union was contaminated by ^{137}Cs with a pollution density over 37 kBq/m^2 [4]. The emergency release of ^{90}Sr was equal to 2.3×10^{17} Bq, but due to specific features of precipitation and the formation of pollution zones, the role of the radionuclide as a source of radiation hazard was significantly smaller. The fallout of ^{90}Sr mainly occurred in the zone near the Chernobyl nuclear power plant, as well as in some adjacent regions of Belarus and Ukraine [5]. In six polluted regions of Belarus, the ^{90}Sr density was more than 5.55 kBq/m^2 over an area of 374900 ha of agricultural land and more than 111 kBq/m^2 over an area of 70 ha [6]. In seven oblasts of Ukraine, the pollution density of ^{90}Sr was more than 5.55 kBq/m^2 over an area of 307800 ha and more than 111 kBq/m^2 over an area of 1400 ha [7].

The biological hazard of ^{90}Sr is determined by several factors: the high yield during uranium and plutonium fission, the long half-life period (about 30 years), and the high mobility in natural media. The migration rate of the radionuclide depends on the form of radioactive fallout, soil properties, biological mobility of the radionuclide, weather conditions, crop cultivation technologies, etc. [8, 9].

The radionuclide migration in agricultural and food chains is determined by the effect of both natural factors and production technologies. The migration rate of the radionuclide is mainly determined by its mobility in soils and the soil–plant system [10]. A combination of rehabilitation measures has been elaborated to limit the input of radionuclides in food for humans. An important place among them is occupied by technologies that reduce the migration rate of radionuclides on the one hand and rise soil fertility on the other hand [11–13]. The effect of agrotechnical and agrochemical technologies on ^{90}Sr mobility is

related to changes in the agrochemical parameters, soil acidity, cation exchange capacity, the content of competing cations, the conditions of mineral nutrition of plants, and other factors, which, in turn, result in modification of the mobility of the radionuclide at its initial migration stage—in soil—and limit its transition to agricultural products and human food.

THE MAIN MECHANISMS OF ^{90}Sr BEHAVIOR IN SOILS

^{90}Sr is a β-emitter with a maximal energy of 0.544 MeV and with $T_{1/2}$ = 28.79 years. It is assigned to biologically mobile radioisotopes [14]. ^{90}Sr is characterized by high mobility, enters ecological migration chains, and is able to create sources of long-term radiation in living organisms. In global fallout, ^{90}Sr is present in soluble and insoluble fractions, and most of it falls out with atmospheric precipitation as a soluble fraction [15]. The insoluble fraction is, on average, 18% and consists of a mixture of various mineral compounds, soil particles, industrial dust, soot, etc. In the case of emergency emissions, ^{90}Sr may fall out as a component of fuel particles [16–18].

After entering the soil, ^{90}Sr is quickly involved in physicochemical processes [19]. The radionuclide is absorbed by several mechanisms, which is related to heterogeneity of the soil–absorbing complex and physical, chemical, and biochemical processes in it.

The absorption capacity of the soil is mainly determined by the content of highly dispersed particles with a large specific surface area [20, 21]. Experimental studies have shown that the exchange capacity of particles increases with their dispersion rate. It is also necessary to take into account the mineralogical composition of fractions: coarse fractions are dominated by primary minerals (quartz, feldspar, and mica), and fine fractions are mainly represented by secondary clay minerals of the montmorillonite, kaolinite, and hydromica groups. Soils containing a large amount of highly dispersed mineral particles (<0.001 mm) are characterized by great absorption capacity. Such particles are surrounded by films-gels mainly composed of sesquioxides (SiO_2, Fe_2O_3, Al_2O_3, etc.), silicic acid, organic substances, and various salts. Since radionuclides are absorbed at the interface between the solid phase and the soil solution, the nature of films–gels covering the mineral particle and the crystal lattice of the mineral on which these films are adsorbed are of great importance [15, 22].

The fixation and distribution of ^{90}Sr in soil is determined by the behavior of the isotopic carrier represented by stable strontium and its chemical analog: stable Ca, the content of which in the Earth's crust is 2.96% [23, 24].

N.V. Timofeev-Resovskii et al. (1966) proposed to assign strontium to the second group of radionuclides with the exchange type of behavior in the soil–plant system. Ion exchange is its main fixation mechanism in the soil [25]. The presence of other cations in the solution is the most important migration factor. Three groups of ^{90}Sr ions may be distinguished: (1) those located in the soil solution; (2) exchangeable ions on the surface of mineral and organic particles; and (3) ions in practically insoluble compounds [26]. Most of ^{90}Sr (60–90%) occurs in soil in exchangeable form (the second group according to the Timofeev–Resovskii classification). There is a dynamic equilibrium between radionuclide forms in soils [27]. The radionuclide may be present in the liquid phase in the form of cations and as a part of complex compounds and colloidal particles. In the solid phase, the radionuclide occurs in the exchange-adsorbed status, i.e. in an adsorption–desorption equilibrium with the liquid phase; it can be the components of insoluble compounds or fallen insoluble particles (for example, fuel); or it may be irreversibly adsorbed [28].

Isomorphic replacement in minerals containing calcium and magnesium—calcite and limestone ($CaCO_3$), gypsum ($CaSO_4 \cdot 2H_2O$), and dolomite ($CaMg(CO_3)_2$)—is important for ^{90}Sr [22]. Since ion exchange is the main process of ^{90}Sr absorption by the solid soil phase, similarly to the adsorption of stable Sr and Ca, ^{90}Sr sorption by the solid soil phase depends on the macroconcentration of cations in the soil solution. With respect to the effect on sorption of ^{90}Sr by the solid soil phase, the competing cations may be arranged in the following sequence: $Al^{3+} > Fe^{3+} > Ba^{2+} > Ca^{2+} > Mg^{2+} > K^+ > NH_4^+ > Na^+$ [9, 15]. Ion exchange substitution of Ca or another cation or the formation of insoluble strontium phosphates occurs on the surface of the crystal lattice [23, 29, 30].

The status and fixation of ^{90}Sr in the soil absorbing complex depends on the composition of the mineral part of the soil: the radionuclide is more firmly fixed in soils with a high content of clay particles [8]. Clay soil minerals may absorb up to 99% of the radionuclide. Minerals of the montmorillonite group (ascanite, bentonite, and gumbrine) are characterized by a high sorption capacity with regard to ^{90}Sr. They absorb from 92 to 99.9% of the radionuclide. The absorption of ^{90}Sr by other minerals comprises from 40 to 68% for the kaolinite group (kaolite, rodalite, and terratolite), from 71 to 87% for micas (hydromuscovite and phlogopite), from 80 to 88% for hydromica (vermiculite) [15], and from 10 to 50% for minerals of the calcite, feldspar, quartz, and gypsum group [22].

The behavior of ^{90}Sr in soil is affected by soil organic matter, first of all, the content and qualitative composition of humus. ^{90}Sr is mainly bonded to low-molecular fulvic acids, which explains its high migration ability [31]. The radionuclide mainly occurs in soils as a component of complexes, which also include Ca, Fe, and Al, and does not form individual compounds with nonspecific organic substances and

Table 1. Mobility factors of ^{90}Sr in the soil–soil solution system for different soil types [67]

Soil type	Exchangeable ^{90}Sr, %	$K_s(^{90}Sr/Ca)$
Soddy–podzolic	93 ± 0.07	2.5 ± 0.5
Light gray forest	76 ± 0.09	1.5 ± 0.2
Leached chernozem	73 ± 0.10	2.1 ± 0.3
Podzolized chernozem	82 ± 0.11	1.3 ± 0.1
Ordinary chernozem	68 ± 0.01	1.8 ± 0.1
Typical chernozem	69 ± 0.02	1.9 ± 0.2
Meadow–chernozemic	64 ± 0.02	1.4 ± 0.1
Chestnut	76 ± 0.09	1.7 ± 0.5
Lowland peat bog	79 ± 0.01	1.1 ± 0.3

humic acids as such. A significant amount of ^{90}Sr in chernozem and gray forest soils is bonded to organic matter and is present in acid solutions after the precipitation of humic acids. In addition to fulvic acids, there are organic substances of particular nature (polysaccharides, amino acids, low-molecular acids, carbohydrates, etc.) there. Contrary to soils rich in organic matter, the main portion of ^{90}Sr in soddy–podzolic soils is absorbed by nonhydrolyzable soil residue due to the great content of sesquioxides and the high absorption capacity of the underlying rocks, and such soils are characterized by high saturation with bases. The main part of ^{90}Sr in the nonhydrolyzable residue is concentrated in the lighter fraction, which contains a larger amount of Fe, Al, and organic carbon as well. This testifies that, similarly to other elements, ^{90}Sr is a component of complex compounds of humic acids with highly dispersed minerals [32]. The destruction of organic matter favors an increase in the mobility of ^{90}Sr as a result of its transformation into the exchangeable form [15, 33].

The processes of sorption and desorption of exchangeable cations, including strontium and its chemical analog calcium, occur in soil on a continuous basis. As a result, an equilibrium between the solid soil phase and the soil solution is established. The content of exchangeable cations in soil, on the one hand, and in the soil solution, on the other hand, is one of the important factors affecting the amount of adsorbed radionuclides [15]. With the increase in the concentration of associated cations in the soil solution, the amount of radionuclides adsorbed by the solid soil phase drops. Calcium in soil solution causes a decrease in the sorption of ^{90}Sr in soil [34]. When the concentration of ^{90}Sr in the soil solution becomes smaller as a result of assimilation by plants or leaching to the deeper soil horizons, the radionuclide migrates from the soil absorbing complex to the liquid phase [8].

As per the modern theory of selective sorption, ion-exchange sorption sites for the radionuclide are specified into three types according to the selectivity: (i) nonselective sorption centers (Regular Exchange Sites, RES) located on the surface of the solid soil phase; (ii) sorption centers (Frayed Edge Sites, FES) located between the layers of the crystal lattice in their expanded edges; and (iii) centers of particularly high selectivity (High Affinity Sites, HAS) [35, 36]. Radiostrontium is characterized by a simple and almost complete exchange at RES of clay particles [28].

The selectivity of sorption sites in relation to the radionuclide (R) is quantitatively characterized by the constant of the ion exchange equilibrium of the radionuclide with one of the competing ions M (the selectivity coefficient K_s):

$$K_s = \frac{[R]_{exch} \times [M]_W}{[M]_{exch} \times [R]_W}, \quad (1)$$

where $[R]_{exch}$ and $[M]_{exch}$ are the content of exchangeable forms of the radionuclide and the competing cation in the solid phase, and $[R]_W$ and $[M]_W$ are their concentrations in the water extract [37].

The migration of ^{90}Sr into the soil solution increases parallel to Ca migration, because there is a particular ratio between Sr and Ca in the solution. However, the distribution of ^{90}Sr between the soil solution and the soil absorbing complex differs from that of Ca. The ^{90}Sr to Ca ratio in the soil solution ranges from 0.49 to 0.78 of the ratio between these ions in the soil, which is related to the stronger sorption of ^{90}Sr as compared to Ca [38]. The selectivity coefficient of exchange between strontium and calcium cations (K_s(Sr/Ca)) (Eq. 1) varies in most soils from 1 to 3 [37]. It has been revealed that the content of the exchangeable form of the radionuclide ^{90}Sr varies from 64 to 93%, depending on soil properties; and the selectivity coefficient K_s(Sr/Ca) varies by a factor of 1.8 (Table 1).

The distribution coefficient K_d is another indicator characterizing the partition of radionuclides between the solid and liquid phases of the soil. It is determined as the ratio between the equilibrium concentration of the radionuclide in the solid phase and its concentration in the solution [39]. A simple K_d-based model is

substantiated by the hypothesis that the radionuclide in the solid phase is characterized by the equilibrium with the radionuclide in solution, which makes possible the exchange process. There is also the concept of an exchangeable distribution coefficient K_D^{exch}, which is equal to the ratio between the concentration of the exchangeable form of the radionuclide and its concentration in the solution [37]. This parameter takes into account the fact that some part of the radionuclide in the solid phase is present in nonexchangeable form and does not participate in desorption into the liquid phase.

The distribution of ^{90}Sr (RN) between the solid and liquid phases may be better understood, taking into account the distribution of the analog ion (AN): a competitive ion, which is characterized by similar behavior during sorption [40]. In this approach:

$$K_d(\mathrm{RN}) = K_d(\mathrm{AN}) \times K_s\left(\frac{\mathrm{RN}}{\mathrm{AN}}\right), \quad (2)$$

where K_d(RN) is calculated by the linear increase in K_d(AN) by a coefficient equal to the selectivity coefficient of RN–AN at the sorption centers K_s(RN/AN).

K_d(Sr) may be predicted by the ratio of Ca + Mg in the exchangeable complex of the solid soil phase (mol/kg) to the sum of the concentrations of Ca and Mg in the soil solution (mmol/kg) [27, 41] multiplied by the trace selectivity coefficient (K_s(Sr/Ca–Mg)), which reflects the exchange of trace quantities of strontium ions with calcium and magnesium ions [42]:

$$K_d(\mathrm{Sr}) = \frac{K_s\left(\frac{\mathrm{Sr}}{\mathrm{Ca}} - \mathrm{Mg}\right)(\mathrm{Ca_{exch}} + \mathrm{Mg_{exch}})}{\mathrm{Ca_{ss}} + \mathrm{Mg_{ss}}}. \quad (3)$$

Since K_s(Sr/Ca–Mg) is close to 1 [28], similar trace selectivity coefficients of Sr to Ca and of Sr to Mg can be usually assumed to obtain a simplified model. Thus, Eq. (3) can be simplified as follows:

$$K_d(\mathrm{Sr}) = \frac{\mathrm{Ca_{exch}} + \mathrm{Mg_{exch}}}{\mathrm{Ca_{ss}} + \mathrm{Mg_{ss}}}. \quad (4)$$

If there are no data on exchangeable cations, the ratio of the cation exchange capacity (CEC, mol/kg) to the sum of the Ca and Mg concentrations in the soil solution may be used. This approach provides satisfactory results when evaluating K_d(Sr), especially in the case of soils with the saturated exchangeable complex.

MIGRATION OF ^{90}Sr IN THE SOIL–PLANT SYSTEM

Numerous studies have shown that the leading factors determining the radionuclide behavior in the soil–plant system include the physicochemical status of the radionuclide, physicochemical soil properties, the content of exchangeable Ca in the soil, and the biological features of plants [8, 9, 34, 43–47].

Weak fixation of ^{90}Sr by soils results in its increased availability for plants over a long period of time. It is known that radionuclides in the soil solution and in the exchangeable part of the soil absorbing complex (SAC) are the most available for plants. Hence, the content of the exchangeable form is the most important parameter that determines the uptake of radionuclides by plants [8, 15]. This process is the result of not only soil chemical processes, but also a biological (physiological) mechanism related to the absorption of the radionuclide by the root system of plants. Similarly to any other mineral nutrients, ^{90}Sr enters plants from the soil solution. The plant initially absorbs the radionuclide as a result of ion transportation by free diffusion inside the cell wall of the root cortex [48]. A dynamic equilibrium is established between the soil solution (free water space) and the root exchange complex (donnan free space) due to the adsorption and exchange of ions. The donnan electropotential at the interface between the water medium and the cation-exchange complex determines the absorption selectivity of charged elements. Cations (two- and trivalent in particular) are more actively fixed by the root exchange complex as compared to anions [49].

The assimilation of ultramicroconcentrations of radionuclides by plants depends on the content of isotopic and nonisotopic carriers. The uptake of ^{90}Sr by plants under conditions of its similar concentration in soil may vary due to different concentrations of the carriers. Numerous studies have shown that the migration of strontium radioisotopes in soils and their transfer to plants are closely related to the behavior of exchangeable Ca, which is the main ^{90}Sr carrier in soil [10, 15, 50–52].

The accumulation of radionuclides by plants also depends on the soil type. There is an understanding in agricultural radiobiology that the uptake of ^{90}Sr by plants is more intensive from low fertility, acid, and calcium-depleted soils of light texture as compared to highly fertile and calcium-enriched soils [9]. It is shown that the bond of ^{90}Sr becomes weaker in the following sequence: chernozem > solonets > soddy–podzolic medium-loamy soil > soddy–podzolic sandy–loamy soil [8].

The accumulation of ^{90}Sr in crops strongly depends on their biological characteristics related to the plant family, genus, species, and variety. For example, the accumulation of ^{90}Sr in tested crops grown on the same soil differs 85 times for varieties of cereals and legumes, 350 times for varieties of root crops and vegetables [10], and 10–15 times for varieties of corn [53]. The accumulation of the radionuclide in seeds and fruits of plants (in the economically valuable part of crops) is much smaller than in other aboveground organs. The content of ^{90}Sr in stems and shoots of cereals is 10–15 times higher than in grain. The same regularity is seen for corn. The distribution of strontium in plant organs is similar to that of calcium. In the

Table 2. The ratio between differences in ^{90}Sr accumulation in products determined by varieties of plant species [68]

Crop	Number of varieties tested	Ratio between differences in the minimal and maximal accumulation
Spring wheat	20	3.1
Winter rye, barley, oats	12	3.1
Corn (grain)	5	5.1
Peas (grain)	28	3.1
Potato	58	10.0
Red beet	25	3.5
Carrot	7	8.3
White cabbage	24	3.1
Onion	18	2.3
Cucumbers	10	1.6
Tomatoes	19	2.5
Seeded herbs:		
Cereals	4	1.8
Legumes	3	1.9

experiments by A.V. Marakushin and E.A. Fedorov, the following sequence of field crops with respect to ^{90}Sr accumulation was determined: clover > corn > timothy > vetch > potatoes > oats > barley > rye [11]. The studies after the Chernobyl accident revealed pronounced differences between crop species in the accumulation of ^{90}Sr from soddy–podzolic soils [17]. Legumes accumulated ^{90}Sr much more intensively than cereals [54].

Among the factors that favor smaller radionuclide accumulation in agricultural crops, specific features of not only species, but also of varieties are important. Differences in the radionuclide accumulation by crops within one species may reach ten times, depending on the variety (potatoes) (Table 2). This regularity is used to choose plants with low radionuclide accumulation for cultivation in polluted areas. However, other requirements must also be met: productivity, resistance to insects, and the susceptibility of plants to diseases. This method of reducing contamination of agricultural plants is the most efficient and simple.

The experiments performed by A.G. Podolyak et al. after the accident at the Chernobyl nuclear power plant show that the differences in the accumulation of ^{90}Sr by leguminous crops (soy, lupine, and pea) may reach 2.5–2.7 times, depending on the variety (Table 3) [55].

The analysis of the experimental data on ^{90}Sr accumulation by various crops shows that the removal of ^{90}Sr with the vegetation mass is insignificant and does not exceed 4% of the total radionuclide reserve in the soil. This value is similar to the self-purification of soils as a result of the annual radionuclide decay (2.5%). The accumulation of ^{90}Sr is maximal in clover and rape and is minimal in cereals.

Radionuclide removal with the disposed portion of the harvest in the crop rotation is insignificant and is equal to 0.03–0.04% of its total content in the soil per year for cereals, 0.07–0.010% for rape, 1.53–2.73% for clover, and 0.03–0.07% for lupine.

A large portion of the radionuclide (to 95%) is accumulated in the non-marketed part of the harvest (straw), which may be disposed of by plowing into the soil on which the crop was cultivated.

Table 3. The effect of peculiarities of varieties of peas, lupine, and soy on ^{90}Sr accumulation in grain (T_{ag}, Bq kg^{-1}) [55]

Peas		Lupine		Soy	
variety	T_{ag}, Bq/kg	variety	T_{ag}, Bq/kg	variety	T_{ag}, Bq/kg
Aist	25 ± 10	Rannii	56 ± 22	Mageva	30 ± 6
Kudesnik	63 ± 25	Mikhas'	66 ± 26	Yasel'da	46 ± 11
Bogatyr'	33 ± 13	Adradzhenne	65 ± 26	Pina	50 ± 13
Agat	31 ± 12	Mif	73 ± 29	Severnaya zvezda	22 ± 3
Gomel'skii	33 ± 13	BSKhA-382	128 ± 51		
Svetanik	34 ± 14	Krok	145 ± 58	No. 37–15	61 ± 5
Eva	33 ± 12	Pershatsvet	108 ± 43		
Pegas	42 ± 17	Mirtan	105 ± 42		
D-15	36 ± 14	Vladlena	85 ± 34		
Truzhenik	48 ± 13	Svetannik	84 ± 32		
Polesskii	36 ± 12	Mitan	75 ± 30		
Belus	44 ± 13	Belokruz	85 ± 34		
Ovoshchnoi	40 ± 12	Khval'ko	61 ± 24		

The intensity of radionuclide release from organic residues plowed into the soil depends on their type and size. The release of ^{90}Sr from straw plowed into the soil is very slow and does not exceed 3.5% of its total content in the plowed biomass. This makes the radionuclide practically unavailable for the next crops for two years [13].

Experience shows that agricultural crops are not useful for the phytoremediation of soils which requires a long time period and the disposal of a significant amount of biomass.

It is difficult to estimate migration parameters, because a large number of physicochemical, biological, geochemical, and other factors affect the behavior of radionuclides in ecosystems. Various parameters are used to quantify the amount of radionuclides accumulated in plants. The Transfer Factor (TF) and the Aggregated Transfer Factor (T_{ag}) are most often used in scientific works [9, 47, 56–60].

The Transfer Factor (TF) (or the Concentration Ratio (CR)) is determined as the ratio of the radionuclide content in a unit of plant mass (or a part of a plant) to the unit soil mass (Table 4):

$$TF = \frac{\text{Specific activity in the plant (Bq/kg)}}{\text{Specific activity in the soil (Bq/kg)}}. \quad (4)$$

The Aggregated Transfer Factor (T_{ag}) is calculated as the ratio of the radionuclide concentration in a plant (or in a part of a plant) to soil contamination density per unit area (Table 4):

$$T_{ag} = \frac{\text{Specific activity in the plant (Bq/kg)}}{\text{Soil contamination density (Bq/kg}^2\text{)}}. \quad (5)$$

Taking into account the dependence of the ^{90}Sr behavior on the presence of the nonisotopic calcium carrier, the following special parameters are proposed: strontium units, the Fredrickson parameter, and the Klechkovskii complex parameter.

Strontium units (s.u.) represent the ratio between the strontium and calcium concentrations: 1 pCi ^{90}Sr/1 g Ca (3.7×10^{-2} Bq ^{90}Sr/1 g Ca, respectively) [8].

The Fredrickson parameter is determined as the ratio of the number of s.u. in plants to the density of soil contamination by ^{90}Sr [61].

The Klechkovskii complex parameter was proposed to account for the factors that affect the ^{90}Sr input into plants: uneven contamination of the soil surface by ^{90}Sr, plant calciphilous properties, and the content of exchangeable Ca in the soil [56]:

$$K = \frac{\text{s.u. in harvest}/(\mu\text{Ci } ^{90}\text{Sr/km}^2)}{\text{mg eq Ca}/100\,\text{g soil}}. \quad (6)$$

The use of the complex parameter enables the comparison of the plant cover contamination in different soil-climatic zones and the effect of various ecological conditions on the ^{90}Sr uptake by plants [19, 62].

The migration parameters are improved with the accumulation of experimental data. After receiving information about the migration of radionuclides as a result of the Chernobyl accident in 2003, the International Atomic Energy Agency initiated the international project IMRAS (Environmental Modeling for RAdiation Safety). The experts analyzed quantitative data of numerous scientific studies of radionuclide transfer from soil to plants. For the classification of data, 13 crop groups were specified: cereals, maize, leaf vegetables, nonleaf vegetables, legumes, root crops, tubers, seeded grasses, forage legumes, pasture plants, spices, and other species. The accumulation coefficients of radionuclides were estimated for four groups of soils: clay, loam, sand, and organic [63].

MATHEMATICAL MODELS DESCRIBING THE BEHAVIOR OF ^{90}Sr IN THE PLANT–SOIL SYSTEM

The development of methods for predicting the input of artificial radionuclides into plants is one of the important tasks of agricultural radioecology. This is related to the fact that, under certain conditions after radioactive contamination of the environment, the migration of radionuclides in agricultural chains may become the main source of radiation hazard for man.

V.M. Prokhorov contributed much to the elaboration of models of vertical migration of radionuclides in soils. Among them, there is a dynamic model of vertical migration of ^{90}Sr in soils [64, 65]. The analysis of the initial data for the model has shown that the vertical migration of ^{90}Sr is affected by a complicated combination of factors, which include filtration of atmospheric precipitation into the soil, capillary rise of moisture to the surface as a result of evaporation, thermal transfer of moisture under the effect of the temperature gradient, and others.

Table 4. Aggregated transfer factor of ^{90}Sr [69]

Plant group	Plant part	Soil type	*N*	Geometric mean	T_{ag}	Arithmetic mean	Standard deviation	Min	Max
Cereals	Grain	All soils	282	1.1×10^{-1}	1.0	1.8×10^{-1}	1.9×10^{-1}	3.6×10^{-3}	1.0
		Clayey	72	7.8×10^{-2}	8.9×10^{-1}	1.2×10^{-1}	1.3×10^{-1}	5.3×10^{-3}	7.1×10^{-1}
		Loamy	71	1.1×10^{-1}	8.7×10^{-1}	1.6×10^{-1}	1.5×10^{-1}	1.6×10^{-2}	7.2×10^{-1}
		Sandy	123	1.4×10^{-1}	1.1	2.3×10^{-1}	2.3×10^{-1}	3.6×10^{-3}	1.0
		Organic	10	9.7×10^{-2}	1.4	1.8×10^{-1}	1.4×10^{-1}	1.2×10^{-2}	3.6×10^{-1}
Corn	Grain	All soils	39	3.2×10^{-1}	1.4	5.9×10^{-1}	6.1×10^{-1}	2.0×10^{-3}	2.6
		Clayey	7	6.9×10^{-2}	1.9	1.6×10^{-1}	1.5×10^{-1}	2.0×10^{-3}	3.9×10^{-1}
		Loamy	13	3.6×10^{-1}	4.6×10^{-1}	4.0×10^{-1}	1.9×10^{-1}	1.5×10^{-1}	8.6×10^{-1}
		Sandy	19	5.2×10^{-1}	1.2	8.8×10^{-1}	7.5×10^{-1}	4.0×10^{-2}	2.6
Leaf vegetables	Leaves	All soils	217	7.6×10^{-1}	1.8	1.9	1.8	3.9×10^{-3}	7.8
		Clayey	54	1.5×10^{-1}	1.8	5.9×10^{-1}	8.1×10^{-1}	3.9×10^{-3}	2.2
		Loamy	84	1.2	1.4	2.0	1.2	4.1×10^{-2}	5.0
		Sandy	72	1.7	1.4	2.9	2.1	6.4×10^{-2}	7.8
		Organic	6	2.1×10^{-1}	3.1×10^{-1}	2.2×10^{-1}	6.8×10^{-2}	1.5×10^{-1}	3.0×10^{-1}
Nonleaf vegetables	Heads, berries, buds	All soils	19	3.6×10^{-1}	1.7	9.8×10^{-1}	1.8	7.1×10^{-3}	7.9
		Clayey	8	1.3×10^{-1}	1.8	3.2×10^{-1}	3.1×10^{-1}	7.1×10^{-3}	8.6×10^{-1}
		Loamy	3	1.4	4.7×10^{-1}	1.5	7.1×10^{-1}	9.0×10^{-1}	2.3
		Sandy	5	8.7×10^{-1}	1.4	2.1	3.3	2.0×10^{-1}	7.9
		Organic	2	–	–	2.2×10^{-1}	4.2×10^{-2}	1.9×10^{-1}	2.5×10^{-1}
Legumes	Seeds, pods	All soils	148	1.4	8.2×10^{-1}	1.8	1.3	1.3×10^{-1}	6.0
		Clayey	25	6.2×10^{-1}	7.9×10^{-1}	8.1×10^{-1}	6.2×10^{-1}	1.3×10^{-1}	2.6
		Loamy	68	1.3	6.4×10^{-1}	1.5	8.5×10^{-1}	1.7×10^{-1}	4.6
		Sandy	55	2.2	7.2×10^{-1}	2.7	1.5	3.0×10^{-1}	6.0
Root crops	Roots	All soils	56	7.2×10^{-1}	1.4	1.5	1.4	3.0×10^{-2}	4.8
		Clayey	13	4.1×10^{-1}	1.5	1.0	1.3	5.2×10^{-2}	3.9
		Loamy	16	6.1×10^{-1}	1.5	1.3	1.5	4.4×10^{-2}	4.5
		Sandy	26	1.1	1.3	1.8	1.4	3.0×10^{-2}	4.8

Table 4. (Contd.)

Plant group	Plant part	Soil type	*N*	Geometric mean	T_{ag}	Arithmetic mean	Standard deviation	Min	Max
Tuber crops	Tubers	All soils	106	1.6×10^{-1}	1.1	2.4×10^{-1}	2.2×10^{-1}	7.4×10^{-3}	1.6
		Clayey	21	1.3×10^{-1}	8.5×10^{-1}	1.7×10^{-1}	1.5×10^{-1}	2.6×10^{-2}	6.7×10^{-1}
		Loamy	41	1.3×10^{-1}	1.1	2.0×10^{-1}	1.4×10^{-1}	7.4×10^{-3}	4.5×10^{-1}
		Sandy	39	2.2×10^{-1}	9.6×10^{-1}	3.3×10^{-1}	3.0×10^{-1}	2.6×10^{-2}	1.6
		Organic	4	5.8×10^{-2}	1.5	1.0×10^{-1}	9.8×10^{-2}	8.0×10^{-3}	2.3×10^{-1}
Seeded herbs	Stems, shoots	All soils	50	9.1×10^{-1}	6.5×10^{-1}	1.1	6.2×10^{-1}	2.5×10^{-1}	2.8
		Clayey	7	7.9×10^{-1}	2.4×10^{-1}	8.1×10^{-1}	1.7×10^{-1}	4.8×10^{-1}	9.7×10^{-1}
		Loamy	6	6.0×10^{-1}	9.2×10^{-1}	8.9×10^{-1}	8.6×10^{-1}	2.9×10^{-1}	2.0
		Sandy	34	1.1	5.1×10^{-1}	1.3	5.9×10^{-1}	2.6×10^{-1}	2.8
		Organic	3	2.6×10^{-1}	5.8×10^{-2}	2.6×10^{-1}	1.5×10^{-2}	2.5×10^{-1}	2.8×10^{-1}
Forage legumes	Stems, shoots	All soils	35	3.7	6.5×10^{-1}	4.6	3.6	1.3	1.8×10^{-1}
		Clayey	10	2.8	5.5×10^{-1}	3.2	1.6	1.3	5.8
		Loamy	11	3.3	5.7×10^{-1}	3.8	2.4	1.4	9.8
		Sandy	14	4.9	7.1×10^{-1}	6.2	4.9	1.3	1.8×10^{-1}
		Organic	1	3.9×10^{-1}	–	3.9×10^{-1}	–	–	–
Pasture plants	Stems, shoots	All soils	172	1.3	7.9×10^{-1}	1.7	1.2	5.6×10^{-2}	7.3
		Clayey	22	8.0×10^{-1}	8.1×10^{-1}	1.0	7.6×10^{-1}	9.0×10^{-2}	2.8
		Loamy	58	1.1	4.9×10^{-1}	1.2	5.6×10^{-1}	3.7×10^{-1}	2.6
		Sandy	87	1.7	1.7	2.2	1.4	9.8×10^{-2}	7.3
		Organic	4	3.5×10^{-1}	1.3	5.5×10^{-1}	4.7×10^{-1}	5.6×10^{-2}	1.2
Spices	Stems, shoots	All soils	1	4.5	–	4.5	–	–	–
Other species		All soils	9	8.8×10^{-1}	1.8	2.4	3.1	2.0×10^{-2}	8.2
Cereals	Stems, shoots	All soils	37	1.1	9.3×10^{-1}	1.7	1.9	1.5×10^{-1}	9.8
		Clayey	20	7.5×10^{-1}	8.6×10^{-1}	1.0	7.8×10^{-1}	1.5×10^{-1}	2.8
		Loamy	3	1.8	8.4×10^{-1}	2.2	1.5	7.2×10^{-1}	3.6
		Sandy	11	2.1	8.4×10^{-1}	3.0	2.9	9.3×10^{-1}	9.8
Corn	Stems, shoots	All soils	36	7.3×10^{-1}	1.8	9.9×10^{-1}	7.6×10^{-1}	1.2×10^{-1}	3.0
		Clayey	6	5.0×10^{-1}	6.5×10^{-1}	5.9×10^{-1}	3.4×10^{-1}	1.8×10^{-1}	1.1
		Loamy	7	7.0×10^{-1}	5.2×10^{-1}	7.8×10^{-1}	3.7×10^{-1}	2.8×10^{-1}	1.4
		Sandy	23	8.2×10^{-1}	9.5×10^{-1}	1.2	8.7×10^{-1}	1.2×10^{-1}	3.0

After the Chernobyl accident, the elaboration and parameterization of mathematical models has become one of the most important areas of scientific research. The calculation of the parameters of radionuclide migration in food chains is an important task, because its accuracy determines the correctness of forecasts of the radiation situation.

Since ^{137}Cs is the main radionuclide of the fallout after the Chernobyl accident, the great attention was paid to elaboration of its migration models. However, ^{90}Sr migration modeling was then developed.

M.C. Roca et al. elaborated a static model for predicting the concentration ratio of ^{90}Sr (CR) [66] under experimental as part of the study for the determination of the main soil parameters responsible for radionuclide migration in the soil–plant system.

The input of ^{90}Sr into the plant is proportional to its concentration in the soil solution and depends on the biological features of plants (Plant Factor):

$$R\mathrm{Sr}_{(\mathrm{Plant})} = R\mathrm{Sr}_{(\mathrm{SoilSoiution})} \times \mathrm{Plant\ Factor}. \quad (7)$$

The Plant Factor includes the physiological features of the plant and depends on the plant species and the element analyzed. It is assumed that the Plant Factor is a linear function of the inverse total concentration of Ca and Mg in the soil solution:

$$\mathrm{Plant\ Factor} = \frac{1}{(\mathrm{Ca} + \mathrm{Mg})_{\mathrm{ss}}}. \quad (8)$$

The concentration of radiostrontium in the soil solution—$R\mathrm{Sr}_{(\mathrm{SoilSolution})}$—depends, on the one hand, on the total concentration of the radionuclide in the solid phase ($R\mathrm{Sr}_{(\mathrm{Soil})}$) and, on the other hand, on the coefficient of the radionuclide distribution between the solid and liquid phases (K_d):

$$R\mathrm{Sr}_{(\mathrm{SoilSolution})} = R\mathrm{Sr}_{(\mathrm{Soil})} \frac{f_{\mathrm{av}}}{K_\mathrm{d}}. \quad (9)$$

The distribution coefficient (K_d) may be calculated using the cation exchange capacity (CEC) and the concentration of competing elements $(\mathrm{Ca} + \mathrm{Mg})_{\mathrm{ss}}$:

$$K_\mathrm{d} = \frac{\mathrm{CEC}}{(\mathrm{Ca} + \mathrm{Mg})_{\mathrm{ss}}}. \quad (10)$$

In this work, we assumed the values of the Sr/Ca and Sr/Mg selectivity coefficients to be equal. The parameters of the available fraction of ^{90}Sr (f_{av}), the distribution coefficient (K_d), and the Plant Factor were used based on the equation for determining the concentration ratio of ^{90}Sr (CR) in the soil–plant system to predict the relative content of radiostrontium for each soil:

$$\mathrm{CR}(^{90}\mathrm{Sr}) = \frac{f_{\mathrm{av}} \times \mathrm{Plant\ Factor}}{K_\mathrm{d}} = \frac{f_{\mathrm{av}}}{\mathrm{CEC}}. \quad (11)$$

This simple model includes a number of soil parameters (cation exchange capacity (CEC), the composition of the soil solution, the distribution coefficient, and the available form of the radionuclide). In the opinion of the authors, these are the main data for a relatively accurate forecast of the coefficient of radiostrontium accumulation by plants.

CONCLUSIONS

The study of the behavior and bioavailability of radioactive strontium in agricultural ecosystems is of prime importance for assessing the real danger of radioactive contamination for humans. Radioactive strontium is one of the most comprehensively studied elements. The analysis of the results of the research has shown an evolution in understanding the radionuclide behavior in agricultural ecosystems: from general regularities to the modern theory of selective sorption. Soil-geochemical processes, which determine the biological availability of the radionuclide, are quantitatively described; and the main parameters characterizing vertical migration, as well as the rate and strength of radionuclide sorption, depending on the properties of the soil absorbing complex, are determined.

Estimation of the migration parameters is a complicated task, because the radionuclide behavior is affected by a large number of physical, chemical, biological, geochemical, and other factors. Various quantitative characteristics are proposed to solve this problem. They permit the assessment of the dependence of radionuclide accumulation in plants on the modifying factors: the uneven contamination of the soil surface by ^{90}Sr, the calciphilous properties of plants, the content of exchangeable Ca in the soil, etc. Despite the considerable accumulated information, the forecast of the radionuclide behavior in agroecosystems is one of the most urgent tasks of agricultural radioecology. It is necessary to continue long-term comprehensive research to refine the quantitative migration parameters for higher accuracy of forecasting the consequences of contamination of agroecosystems by radioactive strontium on the basis of migration mathematical models.

COMPLIANCE WITH ETHICAL STANDARDS

The authors declare that they have no conflict of interest. This article does not contain any studies involving animals or human participants performed by any of the authors.

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