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INVESTIGATION OF ^{137}Cs AND ^{90}Sr TRANSFER FROM SANDY SOIL TO SCOTS PINE (*PINUS SYLVESTRIS* L.) RINGS

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Abstract. Artificial radionuclides entered the environment mostly as a result of nuclear explosions, accidents at nuclear power plants and are entering due to the operation of the nuclear industry. After entering the environment, radionuclides spread globally at the world level, affect all environmental components and accumulate here. One of such environmental components is the tree. It, as if a historical chronicle, fixes the previous contamination and the former climatic conditions. One of the ways to read that chronicle is to estimate the radionuclide soil-to-tree transfer factors and coefficients.

This work presents and analyses the experimental data of a study on the transfer of radionuclides from the soil to Scots pine (*Pinus sylvestris* L.). The report investigates the transfer of ^{90}Sr and ^{137}Cs from a soil layer of 0–20 cm to Scots pine wood. For investigation, Scots pine (*Pinus sylvestris* L.) was selected. Its growing site is in Alytus district, in a woody territory, where it falls into an increased radioactive contamination patch. On this growing site, sandy soils are prevailing. It is identified that ^{90}Sr transfer factor to the pinewood under study ranges from 0.005 ± 0.002 to 0.315 ± 0.002, and transfer coefficients from $(0.2 \pm 0.1) \cdot 10^{-4}$ to $(4.0 \pm 0.6) \cdot 10^{-4}$ m^2/kg. Meanwhile, the coefficients of ^{137}Cs transfer from the soil to Scots pine wood ranges from – $(4.0 \pm 1.0) \cdot 10^{-4}$ m^2/kg to $(8.0 \pm 2.0) \cdot 10^{-4}$ m^2/kg. The identified mean transfer coefficients of ^{137}Cs are approximately five times higher than mean coefficients of ^{90}Sr transfer from soil to wood.

Keywords: ^{90}Sr, ^{137}Cs, radionuclide specific activity, radionuclide transfer, Scots pine (*Pinus sylvestris* L.), sandy soil.

1. Introduction

The biosphere and its components are under constant impact of ionizing radiation, its main sources being: radioactive substances resulting from the operation of nuclear power plants, nuclear weapon tests and accidents at nuclear power plants, cosmic radiation, etc. (Lamarque *et al.* 2005; Butkus *et al.* 2002, 2003; Von Fircks *et al.* 2002; Vaca *et al.* 2001; Momoshima *et al.* 1994).

Radionuclides migrate with nutrient materials from the soil to woody plants through their system of roots. Accumulation of the major part of radionuclides in the annual tree ring is proportionate to the soil contamination existing that year. In addition, a sudden increase of the specific activity of a radionuclide in the annual ring of a tree may mean the plant's contamination through its surface part (Malek *et al.* 2002; Butkus *et al.* 2002; Butkus 2004; Чернобыльская … 2004; Nedveckaitė 2004; Цыбулька и др. 2004; Щеглов и др. 2004).

Radioactive strontium (strontium isotopes, the atomic mass of which is 77÷83.85, 89÷99) is the β radiator and biologically hazardous radionuclide. It is especially hazardous after entering the organism, since it causes internal radiation (Василенко, И., Василенко, О. 2002; Strontium 2002; Botezatu, Iacob 1999). ^{90}Sr gets into plants through their contaminated surface parts, when atmospheric fallout is deposited on them, from the soil through roots. A type of soil, its pH value, humidity, Ca and the amount of organic substances have a strong impact on ^{90}Sr migration (Василенко, И., Василенко, О. 2002; Kanapickas, Raupelienė 2003). ^{90}Sr, deposited on the plant surface, due to active biological processes, is absorbed inside the plant. Uptake of ^{90}Sr by the plant depends on the solubility of the radionuclide in water and kind of a plant. Assimilation of ^{90}Sr from the soil depends on the biological accessibility of ^{90}Sr to plants, its solubility in water, agrochemical properties of the soil, metabolic amount of Ca in the soil (Василенко, И., Василенко, О. 2002). The author of the work (Dušauskienė-Duž 1997, 2001) mentions that ^{90}Sr joins the mineral (biological) metabolism cycle. Solubility in water predetermines the migration of radioactive strontium in the environment.

^{137}Cs releases β and γ radiation, it entered the environment as a result of nuclear explosions and accidents that occurred in nuclear reactors. Small amounts of this radionuclide in the environment of Lithuania could appear as a result of nuclear tests in 1950–1960. The consequence of the accident at the Chernobyl Nuclear Power Plant in 1986 is the radioactive contamination of the environment in many states in Europe (Radioisotopes 2004). As any other radionuclide, ^{137}Cs may enter plants in two ways: through the root system, and a radionuclide entering the plant through the leaf absorption mechanism directly or through the secondary contamination (with resuspension going on) (Malek *et al.* 2002; Butkus *et al.*

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2002; Butkus 2004; Nedveckaitė 2004). The author of the work (Kiponas 2001) mentions that ^{137}Cs accumulation in plants depends on the amount of organic substances in the soil as well as on the morphological properties of a plant. The author of this work also maintains that from 60 to 87% of the total amount of ^{137}Cs, accumulated in the soil, is on the upper 5 cm layer of the soil.

For estimation of radionuclide transfer from the soil to a plant, the radionuclide transfer factor from the soil to a plant is used (Nedveckaitė 2004).

The main goal of this report is to determine the transfer coefficients of artificial radionuclides ^{137}Cs and ^{90}Sr from the soil to Scots pine (*Pinus sylvestris* L.) rings at different stages of the tree growth.

2. Methods for determination of ^{137}Cs and ^{90}Sr transfer from the soil to Scots pine (*Pinus sylvestris* L.) rings

A Scots pine (*Pinus sylvestris* L.), the growing site of which is in Alytus district, in a woody territory, was selected for investigation. This growing site falls into an increased radioactive contamination patch – prior to the accident at the Chernobyl Nuclear Power Plant (ChNPP), the specific activity of ^{137} Cs in the soil was 12–26 Bq/kg. After the accident, it was identified that ^{137}Cs radioactive contamination is 3700–7400 Bq/m^2. ^{90}Sr contamination after the ChNPP accident was estimated from 550 to 1300 Bq/m^2; and in the natural landscape from 300 Bq/m^2 (Butkus *et al*. 1999; Буткус *et al.* 2001). On this growing site, sandy soils prevail.

Radionuclide vertical migration in the soil-plant system occurs due to diffusion, convection and migration via root systems (Butkus *et al*. 2002; Щеглов и др. 2004; Malek *et al.* 2002). To estimate radionuclide intake via the root system, the radionuclide transfer coefficient is used. Often in the soil-plant system, radionuclide transfer is estimated by the transfer factor, i.e. the relation of the radionuclide specific activity in the plant with the radionuclide specific activity in the soil (Nedveckaitė 2004; Modelling 2002).

In order to establish the radionuclide transfer coefficient from the soil to a plant, radiological plant-soil investigation should be conducted. The sequence of plant-soil investigation is shown in Fig. 1.

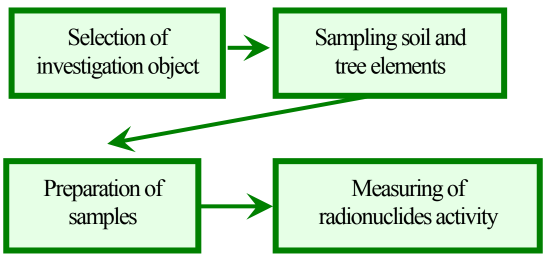


Fig. 1. Scheme of determination of radionuclide transfer from soil to tree

For investigation of ^{137}Cs and ^{90}Sr specific activity in the wood, trunk samples, wood disks, the diameter of which equals the diameter of that trunk area, and height is 3 and 5 cm, are taken. A pine was selected at some distance from strategic objects in a woody territory. Trunk samples were taken at a height of 1 m from the litter. A trunk disk in the direction of the tangent around the ring is split into splinters using chisels of a different breadth. Truck samples are formed. They are weighed, dried at a temperature of 105 °C, burned for 3 hr at 480 °C, and when concentrating ^{90}Sr – for 15 hr more at a temperature of 610 °C. A soil sample is formed of several soil samples, taken according to depths. Radiochemical sample concentration is performed according to methods "LAND 64–2005, Determination of Radioactive Strontium-90 in the Specimens of Environmental Elements, Radiochemical Method".

The activity of ^{90}Sr, which is in chemical balance with its decay product ^{90}Y, is determined by measuring 2.27 MeV energy ^{90}Y beta radiation intensity with a low-background beta radiometer. In the case of radioactive balance, the measured ^{90}Y activity in the sample equals ^{90}Sr activity. Formulae, used for calculation of ^{90}Sr activity, are given in (Butkus *et al*. 2008; Pliopaitė Bataitienė *et al*. 2007) works and LAND 64–2005.

^{137}Cs specific activity in Scots pine (*Pinus sylvestris* L.) rings was determined according to the methods, described in LAND 36–2000 "Measuring of Contamination of Environmental Elements – Gamma Spectral Analysis of Specimens with a Spectrometer, Possessing a Semiconductor Detector". For the determination of radionuclide activity in a sample, a semiconductor Ge(Li) spectrometer is used, the efficiency of which for 662 keV energy, using a 52 ml cuvette, is 0.26%. The prepared for investigation ones were measured in standard "Denta" cuvettes. Measuring duration of specimens is 1–0.5 day. Energy calibration of the spectrometer is performed with dot pattern sources, and geometric one – RP 395, RP 392, RP 396 sources. According to the radionuclide activity in a sample and the sample weight, the radionuclide specific activity in a sample is calculated (Bq/kg). While calculating ^{137}Cs specific activity in samples, formulae provided in the works (Butkus *et al*. 2007, 2006; Pliopaitė *et al.* 2005; Rimeika *et al*. 2008) and LAND 36–2000 were used.

To ensure the data reliability, control tests were carried out by specialists in the Radiation Protection Centre.

Applying the concept of the radionuclide transfer coefficient, certain presumptions are made: the root zone depth is 20 cm; radionuclide accumulation in plants is the same as in their stable analogues (Nedveckaitė 2004; Щеглов и др. 2004).

Computation procedure of radionuclide transfer coefficients. According to the data of the fully formed marginal ring of the growing pine, radionuclide transfer coefficients from soil to pinewood are calculated using the formula from (Nedveckaitė 2004; Pliopaitė Bataitienė *et al.* 2006; Rimeika *et al.* 2008):

$$PF_{kr} = \frac{A_m}{A_d}, \tag{1}$$

where PF_{kr} – factor of transfer from soil to wood; A_m – specific activity in wood, Bq/kg; A_d – specific activity in soil, Bq/kg.

If in formula (1) we substitute the soil specific activity by the soil surface contamination density, then soil-to-plant transfer coefficient, $m^2 \cdot kg^{-1}$, is evaluated according to it.

Radionuclide specific activities in each annual ring are determined according to formula (2), evaluating a specific activity decrease due to radioactive decay:

$$A_0 = A \cdot e^{\lambda \cdot t}, \qquad (2)$$

where A_0 – specific radionuclide activity at the time of flute formation, Bq/kg; A – specific radionuclide activity after some time t, s, Bq/kg; λ – radioactive fissure constant, 1/s; t – time from flute formation to investigation, s.

To determine the transfer factor to an annual ring of a tree in any Scots pine growth year, normative factor K_1 is determined. We take that it is equal to the relation of the specific activity in the annual ring (with account taken of radionuclide decay) to the average radionuclide specific activity in the wood. Factor K_1 is computed by formula (3):

$$K_1 = \frac{A_{mr}}{\bar{A}_{mr}}, \qquad (3)$$

where K_1 – factor of rating; A_{mr} – specific activity in tree ring, according by radioactive fissure, Bq/kg; $\bar{A}_{mr}$ – mean of specific activity in wood, investigated by pine growing time, Bq/kg.

If the determined radionuclide transfer factor from soil to trees of the fully formed marginal ring of the growing pine and the normative coefficient K_1 are known, the coefficient of radionuclide soil-to-ring transfer at a certain moment of time is calculated:

$$PC_r = K_1 \cdot PC_{kr}, \qquad (4)$$

where PC_r – coefficient of transfer from soil to wood at the time of flute formation; K_1 – factor of rating; PC_{kr} – coefficient of transfer from soil to wood by the latest tree ring year data.

For determining the radionuclide soil-to-pinewood transfer coefficient, the soil surface contamination with radionuclides under study is necessary. It is determined using formula (5):

$$Q = A_d \cdot h \cdot \rho, \qquad (5)$$

where Q – superficial density of soil contamination, Bq/m^2; A_d – specific activity in soil layer of 0–20 cm, Bq/kg^3; ρ – density of sandy soil, kg/m^3, h – thickness of soil layer, m.

For determination of the soil activity, a specimen was made by mixing samples taken by depth: 0–5 cm, 5–10 cm, 10–15 cm, 15–20 cm. The specimen is investigated according to the methods provided, and the specific ^{90}Sr activity in the soil is determined $A_m = 7.61 \pm 0.09$ Bq/kg. The defined density of the sandy soil under study is 1628 kg/m^3. The determined volumetric ^{90}Sr activity in the soil is 12389 Bq/m^3. The surface density of ^{90}Sr in the soil, estimated according to formula (5), in the growing site of Scots pine (*Pinus sylvestris* L.) in the Alytus district is 2478 Bq/m^2. The ^{137}Cs soil surface contamination density is established as 3221 Bq/m^2. While computing the transfer factor, ^{90}Sr soil contamination was evaluated as specific activity.

3. Investigation results of ^{137}Cs and ^{90}Sr transfer from the soil to Scots pine (*Pinus sylvestris* L.) rings

Fig. 2 provides investigation results of variation of ^{90}Sr transfer coefficients from the soil to pinewood (at a height of 1 m from the litter).

From Fig. 2 it is seen that ^{90}Sr soil-to-pinewood transfer factor at a height of 1 m from the stump in 1955–2000 varied from $0.2 \cdot 10^{-4} \pm 0.1 \cdot 10^{-4}$ m^2/kg to $4.0 \cdot 10^{-4} \pm 0.6 \cdot 10^{-4}$ m^2/kg, and the mean value of soil-to-pinewood transfer coefficient at the bottom of the trunk within the entire growing period is $1.4 \cdot 10^{-4} \pm 0.3 \cdot 10^{-4}$ m^2/kg. It is possible to note that ^{90}Sr soil-to-pinewood transfer at the beginning of Scots pine growth is higher than average – $(0.2 \pm 0.1) \cdot 10^{-4}$ m^2/kg. It could be predetermined by radioactive ^{90}Sr contamination due to nuclear tests conducted. A statement cannot be ignored that in a growing pine ^{137}Cs and ^{90}Sr migrate towards the core of the tree. Meanwhile, maximum transfer, according to the investigation results obtained, was determined for the wood of 1970–1975. Such an increment of radioactive strontium soil-to-wood transfer could be evaluated as a consequence of secondary radioactive contamination through the root system from the soil. From 1975, an exponential decrease of ^{90}Sr soil-to-wood transfer has

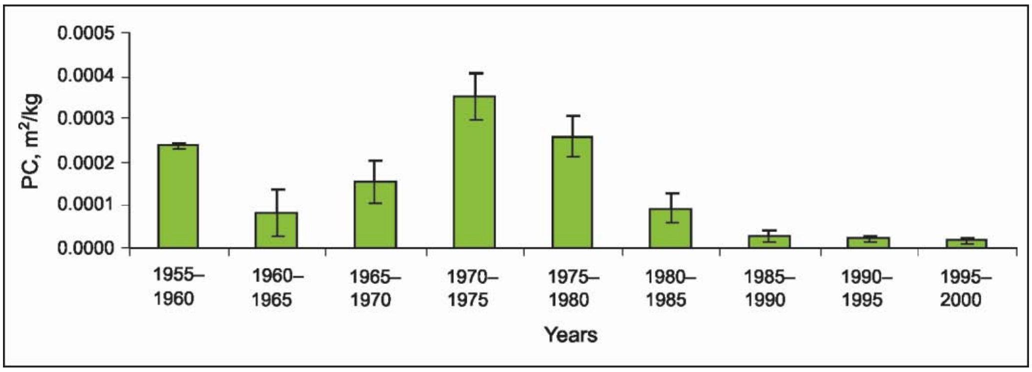


Fig. 2. Fluctuation of ^{90}Sr transfer coefficient from soil to pinewood (1 m from stump)

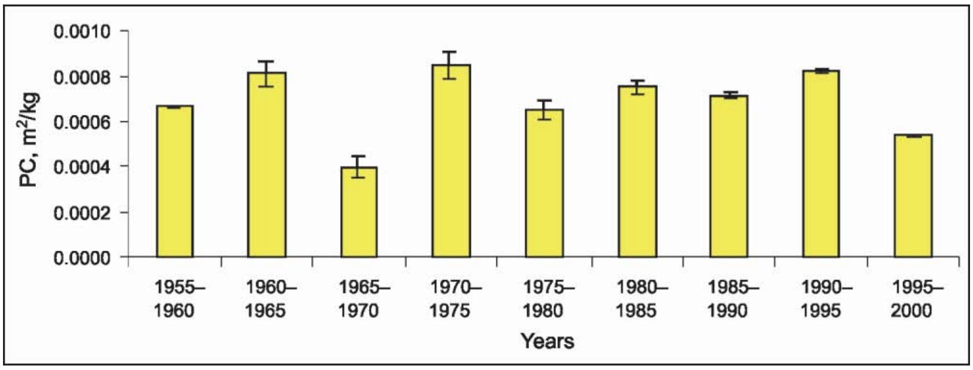


Fig. 3. Fluctuation of ^{137}Cs transfer coefficient from soil to pinewood (1 m from stump)

been observed. For explanation of that phenomenon, we could distinguish several reasons: migration of radioactive strontium along the soil horizon deeper and the absence of airborne ^{90}Sr radioactive contamination in the environment. The authors of the work (Cigna 2000) mention that ^{90}Sr diffusion coefficient depends on the soil and it is $(0.4–3.4)\cdot 10^{-4}$ m^2/year. The authors of this work maintain that the highest contamination is in the soil layer of 20 cm, and as it is known the main biomass of the active roots of Scots pine (*Pinus sylvestris* L.) is at a depth of 0–30 cm in the soil (Данусевичюс 1984).

Meanwhile, ^{90}Sr transfer-evaluating factor is, on the average, even 325 times higher than the established coefficients, but it does not mean that transfer is higher. Radionuclide soil-to-wood transfer factor and coefficient characterize the same transfer phenomenon, though due to dimension differences their values vary.

Fig. 3 presents the coefficients of ^{137}Cs transfer from the soil to Scots pinewood.

^{137}Cs transfer coefficients from the soil to Scots pinewood vary from $(4.0\pm1.0)\cdot 10^{-4}$ m^2/kg to $(8.0\pm2.0)\cdot 10^{-4}$ m^2/kg, and the mean value is $(7.0\pm1.0)\cdot 10^{-4}$ m^2/kg. The determined ^{137}Cs transfer coefficients are approximately 5 times higher than ^{90}Sr transfer soil-to-wood coefficients. This variation probably occurred due to the peculiarities of radionuclide migration in the soil and specific features of dispersion in wood.

The author of the work (Marčiulionienė 2001) mentions that the major part of the total amount of ^{137}Cs environmental contamination is accumulated at the 5 cm upper layer of the soil, i.e. even 60–87%. It is also stated that soils, which contain more organic matter or consist of small fraction particles, tend to accumulate ^{137}Cs more, and at the same time to contaminate plants. The author of the work (Чилимов, Богачев 2000) mentions that of special importance for radioactive soil-to-plant contamination is opportunity for assimilation of nutrient materials from the upper soil layer, the acidic-restorative potential of the soil in the growing site, transpiration conditions, and temperature regime. Nevertheless, even though the author mentions that the most important contamination of this radionuclide comes from the upper layer of the soil, it is also necessary to evaluate the fact that radiocesium migrates along the soil horizon deeper. The mechanism of this migration is predetermined by the processes of diffusion, occurring in the soil. ^{137}Cs diffusion coefficient in sandy soils is $0.17\cdot 10^{-4}$ m^2/year. Also, it is very important to evaluate the fact that the main part of the active root biomass of the plant under study, Scots pine, is in the 0–30 cm soil layer (Данусевичюс 1984).

Looking at the types of variation of ^{137}Cs and ^{90}Sr transfer coefficients, it is possible to notice that they are different: for ^{90}Sr transfer coefficients prior to 1970, an exponential increment is characteristic, and from 1970 – an exponential decrease. Meanwhile, in the variation of ^{137}Cs transfer coefficients, there are several specific periods – approximately every ten years an increment of the specific activity of this radionuclide in wood is observed. Such a tendency may be explained by the migration of that radionuclide along the soil horizon deeper, towards the main biomass of pine roots. Types of variations could be predetermined by the properties of radionuclides under study and the radioactive pollution of the environment. Such a difference in the variation of transfer coefficients could be related with the properties of radionuclides under study and the radioactive pollution of the environment.

^{90}Sr, after entering the wood of gymnosperms, is relatively immobile, whereas ^{137}Cs is noted for a higher radical mobility among rings than ^{90}Sr – it moves towards the core. On the other hand, in the territory of Lithuania pollution with radioactive cesium was higher than that with radioactive strontium.

A standard scale of annual increment of ring variation is given in Fig. 4, which was made according to (Stravinskienė 2002).

According to the results presented in Fig. 4, in the area under study within the period under analysis (1955–2000), the most favourable pine growth years, taking into account the coefficient of annual radial increment indices (IARI), are distinguished: 1958–1976, 1982–1994. The

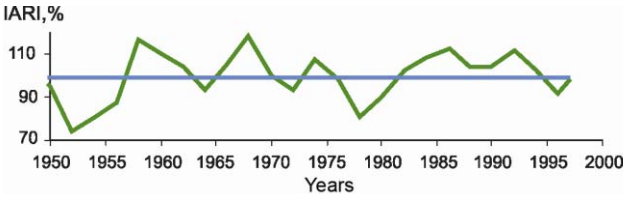
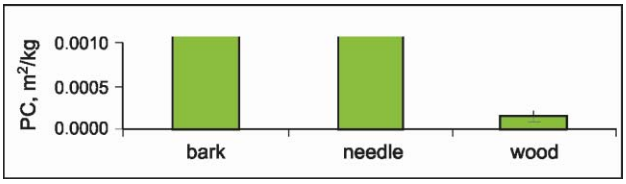
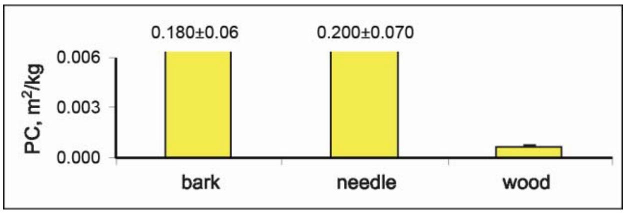


Fig. 4. Variation in the indexes of annual radial increment (IARI) of *P. sylvestris* in *Pinetum myrtillo-sphagnosum* during the period 1930–1997 by (Stravinskienė 2002)

values of ^{137}Cs transfer coefficients from 1960 to 1970 decrease, whereas within the period 1982–1994 an increase of these transfer coefficients is observed. While making an analysis of the variation of ^{90}Sr transfer from the soil to pinewood during the favourable pine growth years an opposite view is noticed than in the variation of ^{137}Cs transfer coefficients. Within the period 1965–1975, ^{90}Sr transfer coefficients increase and reach the maximum value close to 1970–1975. Even though ^{90}Sr transfer coefficients calculated from 1975 go on decreasing, but not so distinctly as from 1985.



a)



b)

Fig. 5. ^{90}Sr (a) and ^{137}Cs (b) transfer coefficients from soil to elements of Scots pine

Fig. 5 provides ^{90}Sr (a) and ^{137}Cs (b) transfer coefficients to certain rings of Scots pine.

According to the determined ^{137}Cs and ^{90}Sr transfer coefficients from the soil to Scots pine needles and bark, it is possible to notice that ^{137}Cs transfer to the bark is established 18 times higher than that of ^{90}Sr, and to the needles – 12 times higher. According to the investigation results, it is also possible to state that ^{137}Cs transfer both to the needles and bark is very similar. Meanwhile, ^{90}Sr transfer to the needles from the soil is twice as much than to the bark. Probably, we could relate these results to different intensity of the process of needle and bark self-cleaning from radionuclides. In the work (Von Fircks *et al.* 2002) it is mentioned that the foliage of willows (*Salix viminalis*) accumulated ^{90}Sr more significantly than the other rings. It is probable that this phenomenon is characteristic not only of deciduous trees, and thus we could justify a higher transfer of ^{90}Sr to the needles. In addition, it is important to remember that radionuclide transfer to the needles and bark is also significantly impacted by a direct transfer of radionuclides from the atmosphere. Making an analysis of radionuclide transfer coefficients from the soil to certain rings of Scots pine (*Pinus sylvestris* L.), it is possible to notice that transfer to the bark and needles is significantly higher than to pinewood. Such a transfer result could be related to the self-cleaning of different pine rings from contaminants as well as to a higher importance of atmospheric pollution to the bark and needles than to the wood.

4. Conclusions

1. The main ways of plant contamination with ^{90}Sr and ^{137}Cs is radioactive pollution from the atmosphere and radionuclide transfer to the wood from the soil via the plant root system.
2. ^{90}Sr transfer coefficient from the soil to pinewood at a height of 1 m from the stump in 1955–2000 varied from $0.2 \cdot 10^{-4} \pm 0.1 \cdot 10^{-4}$ m^2/kg to $4.0 \cdot 10^{-4} \pm 0.6$ m^2/kg, and the mean value at the bottom of the trunk within the entire growth period made $1.4 \cdot 10^{-4} \pm 0.3 \cdot 10^{-4}$ m^2/kg.
3. At the beginning of Scots pine growth ^{90}Sr transfer to the wood is higher than the average one $(0.2 \pm 0.1) \cdot 10^{-4}$ m^2/kg. The highest ^{90}Sr transfer is determined in the wood, formed within 1970–1975. From 1975, the exponential decrease of ^{90}Sr soil-to-wood transfer is observed.
4. ^{137}Cs transfer coefficient from the soil to Scots pinewood varies from $(4.0 \pm 1.0) \cdot 10^{-4}$ m^2/kg to $(8.0\pm2.0) \cdot 10^{-4}$ m^2/kg, and the mean coefficient value is $(7.0\pm1.0) \cdot 10^{-4}$ m^2/kg.
5. The determined ^{137}Cs transfer coefficients are approximately 5 times higher than ^{90}Sr soil-to-wood transfer coefficients.
6. In the variation of ^{137}Cs transfer coefficients, an increase is observed every 10 years.
7. Transfer coefficients of the radionuclides under study from the soil to the bark and needles of Scots pine (*Pinus sylvestris* L.) are higher than to the pinewood.
8. While analysing IARI relation with radionuclide transfer coefficients, it was noticed that from 1955 to 1965 ^{90}Sr transfer was of a decreasing character, and from 1965 to 1976 it had a tendency to increase. The variation of ^{137}Cs coefficients was inverse to the variation of ^{90}Sr coefficients. From 1982 to 1994 ^{90}Sr coefficient decreased, and that of ^{137}Cs, though insignificantly, but increased.

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^{137}Cs IR ^{90}Sr PERNAŠOS IŠ PRIESMĖLIO DIRVOŽEMIO Į PUŠIES (*Pinus sylvestris* L.) SANDUS TYRIMAS

I. Pliopaitė Bataitienė, D. Butkus

S a n t r a u k a

Dirbtinės kilmės radionuklidai daugiausiai į aplinką pasklido dėl vykdytų branduolinių sprogdinimų, avarijų atominėse elektrinėse ir tebepatenka veikiant branduolinei pramonei. Patekę į aplinką, radionuklidai globaliai pasklinda ir veikia visus aplinkos komponentus bei juose kaupiasi. Vienas iš tokių aplinkos komponentų yra medis. Jis tarsi istorijos metraštis fiksuoja buvusią užtaršą ir buvusias klimatines sąlygas. Vienas iš būdų perskaityti šį metraštį – radionuklidų pernašos iš dirvožemio į medį faktorių ar koeficientų vertinimas. Darbe pateikiami ir analizuojami eksperimentiniai radionuklidų pernašos iš dirvožemio į paprastąją pušį (*Pinus sylvestris* L.) duomenys. Nagrinėjama ^{90}Sr ir ^{137}Cs pernaša iš 0–20 cm dirvožemio sluoksnio į paprastosios pušies medieną. Tirti parinkta paprastoji pušis (*Pinus sylvestris* L.). Jos augavietė yra Alytaus apskrityje miškingoje teritorijoje, kuri patenka į didesnės radioaktyviosios užtaršos zoną. Šioje augavietėje vyrauja priesmėlio tipo dirvožemiai. Nustatyta, kad ^{90}Sr pernašos į tiriamąją pušies medieną faktorius kinta nuo 0,005±0,002 iki 0,315±0,002, o pernašos koeficientai – nuo $(0,2 \pm 0,1)\cdot10^{-4}$ iki $(4,0\pm0,6)\cdot10^{-4}$ m^2/kg. ^{137}Cs pernašos iš dirvožemio į paprastosios pušies medieną koeficientai kinta nuo $(4,0\pm1,0)\cdot10^{-4}$ m^2/kg iki $(8,0\pm2,0)\cdot10^{-4}$ m^2/kg. Nustatyti ^{137}Cs pernašos vidutiniai koeficientai yra apytiksliai 5 kartus didesni nei vidutiniai ^{90}Sr pernašos iš dirvožemio į medieną koeficientai.

Reikšminiai žodžiai: ^{90}Sr, ^{137}Cs, radionuklidų savitasis aktyvumas, radionuklidų pernaša, paprastoji pušis *Pinus sylvestris* L., priesmėlio dirvožemis.

ИССЛЕДОВАНИЕ ПЕРЕНОСА ^{137}Cs И ^{90}Sr ИЗ ПОЧВЫ В КОМПОНЕНТЫ СОСНЫ (*Pinus sylvestris* L.)

И. Плепайте Батайтене, Д. Буткус

Р е з ю м е

Искусственные радионуклиды в окружающей среде появились вследствие ядерных взрывов, аварий на атомных электростанциях и выпадений при работе ядерной промышленности. Радионуклиды распространяются в атмосфере и накапливаются в компонентах окружающей среды. Одним из компонентов окружающей среды являются деревья. Они как бы записывают историю случившегося загрязнения окружающей среды и климатические условия прошлого. Один из способов прочтения такого ежегодника – изучение факторов или коэффициентов переноса радионуклидов из почвы в деревья. В статье анализируются экспериментальные данные о поступлении радионуклидов в сосну (*Pinus sylvestris* L.) из почвы. Изучается перенос ^{90}Sr и ^{137}Cs из слоя почвы толщиной 0–20 см в компоненты сосны. Местом роста анализируемой сосны (*Pinus sylvestris* L.) послужила лесная местность в Алитусском районе, в которой после Чернобыльской катастрофы зафиксировано повышенное радиоактивное загрязнение. На этом участке доминирует супесь. Было установлено, что фактор передачи ^{90}Sr из почвы в сосну варьирует от 0,005±0,002 до 0,315±0,002, а коэффициенты – от $(0,2 \pm 0,1) \cdot 10^{-4}$ до $(4,0\pm0,6) \cdot 10^{-4}$ m^2/кg. Коэффициент переноса ^{137}Cs из почвы в древесину сосны варьирует от $(4,0\pm1,0) \cdot 10^{-4}$ m^2/kg до $(8.0\pm2.0) \cdot 10^{-4}$ m^2/кg. Установлено, что средний коэффициент переноса ^{137}Cs из почвы в древесину сосны (*Pinus sylvestris* L.) примерно в 5 раз больше, чем коэффициент переноса ^{90}Sr.

Ключевые слова: ^{90}Sr, ^{137}Cs, удельная активность, миграция радионуклидов, сосна *Pinus sylvestris L*., супесчаная почва.

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