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## *Accumulation of $^{137}$Cs, $^{90}$Sr, and $^{241}$Am in Forest Plants after Wildfires in the Chernobyl Exclusion Zone*

*Soils and the tissues organs of plants (birch leaves, above-ground parts of smallreed, matgrass, mosses, and lichens) were sampled from sites after significant wildfires and from non-burnt reference sites in the Belarussian sector of the Chernobyl exclusion zone. Activity concentrations of $^{137}$Cs, $^{90}$Sr, $^{241}$Am in the samples were determined by gamma- and beta-spectrometry. In most cases, a tendency towards decreasing accumulation of $^{137}$Cs, $^{90}$Sr and $^{241}$Am in terrestrial plants growing on the sites exposed to fire was observed. Various factors influencing the accumulation of radionuclides by plants are discussed in the article.*

***Key words:*** *wildfires, radionuclides, plants, soil, Chernobyl exclusion zone.*

## НАКОПЛЕНИЕ $^{137}$Cs, $^{90}$Sr И $^{241}$Am В ЛЕСНЫХ РАСТЕНИЯХ ПОСЛЕ ПОЖАРОВ В ЧЕРНОБЫЛЬСКОЙ ЗОНЕ ОТЧУЖДЕНИЯ

*Произведен отбор почвы, тканей и органов растений (листья березы, надземные части вейника и белоуса, мхи, лишайники) после крупных лесных пожаров и с несгоревших контрольных участков в Белорусском секторе Чернобыльской зоны отчуждения. С использованием гамма- и бета-спектрометрии, радиохимических методов анализа определено содержание $^{137}$Cs, $^{90}$Sr, $^{241}$Am в отобранных пробах. В большинстве случаев обнаружена тенденция к снижению накопления $^{137}$Cs, $^{90}$Sr и $^{241}$Am в наземных растениях на участках, подвергнутых воздействию пожара. В статье обсуждены различные факторы, влияющие на накопление радионуклидов растениями.*

***Ключевые слова:*** *лесные пожары, радионуклиды, растения, почва, зона отчуждения ЧАЭС.*

## Introduction

Forest fires occur periodically in the territories contaminated by Chernobyl-origin radionuclides leading to additional releases of radioactive substances into the atmosphere and their dispersion 15 over considerable distances. The total area of the Belarusian sector of the Chernobyl NPP exclusion zone impacted by wildfires between 1991 and 2012 was 10440,1 hectares, of which 4871,6 hectares were forested land (46,7 %). There have been – 342 fire incidents recorded during that 18 time. The average extent of the fires was approximately 24,3 hectares, the overall proportion of crown fires being insignificant (1,5 %). A significant proportion of the total number of fire events occurred on former agricultural lands and in former settlements. During the same period, some 920 buildings were destroyed due to the spread of fire from the former fields adjacent to them. For the period 2013–2016, 22 fires were recorded, the largest of which occurred in 2015. The impact of wildfires on the soil and understory ecosystem components of areas contaminated by radionuclides areas has two main aspects. First, the biological aspect related to the physiological characteristics of vegetation under post-fire conditions. Second, the radioecological aspect taking into account the peculiarities of changes in the physical and chemical properties of radionuclides and soil in the ecosystem as a result of thermal effects, and, as a consequence, effects on parameters governing accumulation and migration of radionuclides in forest vegetation.

Fire has a rapid effect on plants through changes of habitat imposed both directly or indirectly. Only in the case of fires on peat bogs, which may last weeks or even months, do the direct thermal effect persist with time. Some plants have developed adaptations that enable survival of the thermal effects, some species requiring repeated fire events for reproduction [1]. Some plant communities require periodic burnings to ensure favorable conditions for the germination of seeds, as well as the concomitant improvement in lighting conditions, availability of mineral elements (through intense ammonification and nitrification) and the sharp decline in older plants which compete for nutrients. It is generally accepted that the biological characteristics of pine and larch forests are closely linked with fire events, pine and larch populations being reliant on natural fires for expansion into new territories due to the elimination of competitors [1–4].

Post-fire impacts on under-story plants are determined by the type of forest, type and intensity of the fire, because these are factors which significantly impact the natural regeneration of the plant community. As an example, resistance to the impacts of fire events is greater where the proportion of hardwood species in higher [2]. Shifts in the species composition of under-story plants after a fire are caused primarily by changes resulting from shifts in the competitive relationship between species. This is particularly evident in plant communities comprised of half-shrubs, grass and herbs. The predominance of grasses in natural herbaceous plant communities is due to their resilience to fire events, many communities being quickly superseded by woody plants in the absence of fire events [1]. The differences in how individual plant species are impacted by fire events are determined by the extent of damage to resting buds. Shrubs and bushes have resting buds located above the ground and are damaged by fire to a larger extent than grasses (hemicryptophytes, and especially geophytes). Among the grasses, species whose resting buds are located in the soil or species which form tufts, such that the buds of internal shoots are protected from the effects of fire by shoots on the periphery, are more resistant to thermal stresses, tussock-forming plants being particularly resistant to fire. It is possible that the emergence of this form of vegetation is related to the impact of the fire events [1; 3; 4].

According to the work of Ukrainian researchers, the destruction of the organic matter component of litter during forest fires, causes a release of radionuclides, most of the activity entering the upper soil horizons of the burnt area. In the immediate years following a fire, the activity of $^{137}$Cs and $^{90}$Sr in the 0–2 cm soil layer increased by 60–80 % compared with an increase of 10–20 % in forests not exposed to fire [3]. Litter has ten times more $^{137}$Cs and 3,5 times more $^{90}$Sr than grass cover, plutonium being mainly concentrated in the litter, its activity in litter being approximately 1 700 higher than in grasses [5]. Coniferous forests growing on poor sandy soils, dry and semi-dry pine forests are more flammable than other types of forest. The activity of $^{137}$Cs in the litter of coniferous forest can reach 40 % of the total activity of the radionuclide in the soil [6]. The release of such a large amount of radioactive species in the process of litter burnout can cause an abrupt change in the radioecological situation. An increase in the concentration of $^{137}$Cs in the newly grown grasses and bushes, mushrooms, berries and other forest components, as well as in the wild animals that feed on these forest products, can be expected [7; 8]. The matter of transfer of radionuclides from soil to forest plants as a result of fire events remains open. Whether or not transfer factors increase or decrease as a result of fire events has not been firmly established. The pyrogenic effect is the most significant factor regarding changes to many properties of the soil, including those essential for plant growth and fertility. This work is focused on addressing this question, aiming to assess the impact of forest fires on the transfer of $^{137}$Cs, $^{90}$Sr, and $^{241}$Am from soil to plants growing in ecosystems exposed to wildfires.

## Materials and methods

The transformation of forest ecosystems after exposure to fire is a long process. Full ecosystem restoration requires approximately 100–150 years depending on the conditions under which its development takes place. An ecosystem changed by fire should be considered as a secondary succession, being replaced by other plant communities due to changing environmental conditions over time. In this context, investigations must be carried out at sites with successional changes at similar stages of development.

The selection of sites in the Polesie State Radioecological Reserve (PSRER) for the current study was conducted in several stages. First, information about fires that had occurred in the Belarusian sector of the exclusion zone of Chernobyl NPP was collated and analyzed. This analysis took account of, amongst other parameters: the intensity of the fire; the time of year when the fire occurred; the number of years since the fire; the type of forest that destroyed by the fire; the distance of the fire from the Chernobyl nuclear power plant; the γ-dose rate at the site and the radionuclides' activity in the soil of the affected area. Next, an inspection of the potential study sites was carried out. Visual inspection of the areas was conducted, the fire area was evaluated and the type of terrain and soil of the affected areas was determined. The impact of the fire on the soil was assessed, the γ-dose rate across each site was measured and representative soil samples extracted to obtain preliminary information regarding contamination of the sites by $^{137}$Cs, $^{90}$Sr, and $^{241}$Am. Adjoining sites with similar soil and plant characteristics were selected, to be used as reference sites not having been affected by the fire. Six wildfire sites and six reference sites were selected for the investigations. The locations of the sites relative to the CNPP are depicted in Figure 1.

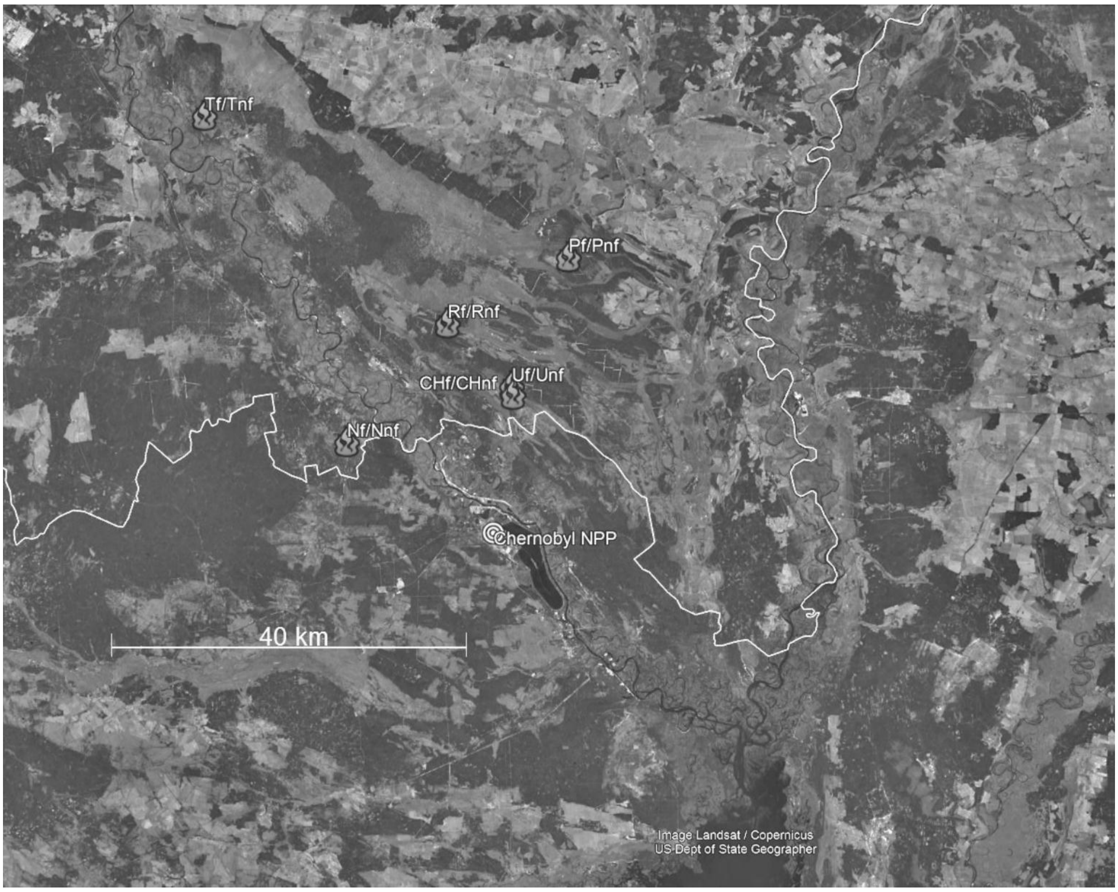

**Figure 1. – The locations of the sites relative to the Chernobyl NPP**

Paired samples of soil and plants were taken for analysis and determination of soil-to-plant transfer factors ($C_f$) and aggregated transfer factors ($T_{ag}$) during the period from June to September 2012. The size of the experimental plots in this case were of the order of (5–10) × (5–10) m. Soil sampling was carried out to a depth of 20 cm in 3-fold replicates. Plants taller than 1–2 cm were sampled on the same plots. Only leaves were sampled for birch. The plant samples were cleaned from foreign matter inclusions and dried at a temperature of 105 °C to constant weight. An electric mill of the type IKA M20 was used for homogenization of plant samples. Measurement of the radionuclides $^{137}$Cs, $^{90}$Sr in samples was performed by a gamma-beta-spectrometer of type «MKS AT-1315» (Atomtex, Belarus) and by semiconductor gamma-spectrometry (Canberra Industries, Inc., USA) ($^{137}$Cs, $^{241}$Am). Activity concentrations of the radionuclides in the plants are presented in Bq/kg dry weight; the measurement uncertainty being within 20 % for all cases. Soil-to-plant transfer factors were calculated as the quotient of the radionuclide activity concentration in dry plants to its activity concentration in the upper 20-cm soil layer (Eqn. 1), and the aggregated transfer factor was calculated as the ratio of the radionuclide activity concentration in dry plants to the soil surface contamination density of the radionuclide (Eqn. 2).

$$\text{Concentration Ratio } CR = \text{activity concentration in plant (Bq kg}^{-1}\text{) / activity concentration in soil (Bq kg}^{-1}\text{)} \quad (1)$$

$$\text{Aggregated Transfer Factor } TF\ (\text{m}^2\ \text{kg}^{-1}) = \text{mass activity density (Bq kg}^{-1}\text{) / unit area activity density (Bq m}^{-2}\text{).} \quad (2)$$

All the sites had a sod-podzolic type of soil. Relevant parameters of the sites are presented in Table 1.

Table 1. – Parameters for the test site near former settlement (f. s.)

| Name of site | Burnt/ unburnt | Date of fire | Coordinates | Fire area, ha | Fire type and ground cover | Dose rate, µSv/h | |
|---|---|---|---|---|---|---|---|
| | | | | | | on the soil surface | at height 1 m |
| CHf (Chemkov) | Burnt | 02.05. 1992 | 51$^0$ 30' 52.4 "<br>30$^0$ 07′ 48.4″ | 200 | Ground fire. Pine, Birch (undergrowth), lichens, smallreed | 9,39–9,49 | 7,11–7,22 |
| CHnf (Chemkov) | Unburnt | – | 51$^0$ 30′ 48.6″<br>30$^0$ 07′ 48.1″ | – | Pine, Birch (undergrowth), lichens, smallreed | 9,35–9,41 | 6,29–6,36 |
| Rf (Radin) | Burnt | 05.07. 2004 | 51$^0$ 35′ 18.5″<br>30$^0$ 01′ 21.8″ | 4 | Ground and crown fires. Pine, birch (undergrowth, sporadic trees), mosses, lichens, smallreed | 6,70–6,78 | 6,56–6,91 |
| Rnf (Radin) | Unburnt | – | 51$^0$ 35′ 17.5″<br>30$^0$ 01′ 12.5″ | – | Pine, birch (undergrowth, sporadic trees), mosses, lichens, smallreed | 8,77–8,83 | 6,70–6,78 |
| Uf (Ulasy) | Burnt | 06.08. 2005 | 51$^0$ 31′ 18.4″<br>30$^0$ 07′ 40.4″ | 14 | Ground fire. Pine (mature), Birch (undergrowth), matgrass, smallreed | 6,37–6,42 | 5,14–5,32 |
| Unf (Ulasy) | Unburnt | – | 51$^0$ 31′ 18.9″<br>30$^0$ 07′ 37.8″ | – | Pine (mature), Birch (undergrowth), matgrass, smallreed | 6,71–7,01 | 5,07–5,27 |
| Pf (Puchin) | Burnt | 01.08. 2002 | 51$^0$ 39′ 09.7″<br>30$^0$ 13′ 16.8″ | 450 | Ground and crown fires. Pine, Birch (undergrowth), matgrass, smallreed, mosses | 1,4 | 1,0 |
| Pnf (Puchin) | Unburnt | – | 51$^0$ 39′ 01.7″<br>30$^0$ 13′ 16.0″ | – | Pine, birch (undergrowth), matgrass, smallreed, mosses | 1,2 | 1,1 |
| Tf (Tulgovichi) | Burnt | 21.04. 2003 | 51$^0$ 48′ 10.6″<br>29$^0$ 37′ 21.5″ | 2109 | Ground and quick crown fires. Pine, birch (undergrowth), smallreed, mosses | – | – |
| Tnf (Tulgovichi) | Unburnt | – | 51$^0$ 47′ 32.9″<br>29$^0$ 38′ 39.4″ | – | Pine, birch (undergrowth), smallreed, mosses | – | – |
| Nf (Narovlya) | Burnt | 8.06. 2011 | 51$^0$ 28′ 01.9″<br>29$^0$ 51′ 33.7″ | 10,4 | Ground fire. Pine, birch, clubawngrass, heather | 0,25 | 0,13 |
| Nnf (Narovlya) | Unburnt | – | 51$^0$ 28′ 10.5″<br>29$^0$ 51′ 21.9″ | – | Pine, birch, clubawngrass, heather | 0,21 | 0,17 |

*Standard methods of biological statistics were used for analysis of obtained data.*

## Results and discussion

Wildfires periodically arise in the Chernobyl exclusion zone leading to significant financial losses, burning of vegetation and the loss of wildlife. Radioactive substances are released into the atmosphere with convective flows of air, and may be dispersed off-site. Weather conditions have a significant influence on the risk of fire events, the risk of a fire event occurring increasing with increasing air temperature and decreasing amount of precipitations. The period of highest fire risk is from April to October. Wind, depending on its speed, affects the transfer of the burning material from the main combustion site to different distances.

Analysis of soil contamination densities at the experimental plots indicates that the previously reported [2; 5; 6] tendency towards increasing contamination in the upper 20-cm soil layer on a fire site compared to unburnt reference sites is not always observed (Table 2), the situation being opposite for $^{137}$Cs at sites Uf and Pf. All burned sites have a lower level of $^{241}$Am soil contamination by than areas not affected by the fires. However, soil at all fire sites has level of contamination with $^{90}$Sr higher than unburnt sites (Table 2).

Table 2. – Density of soil contamination at the test sites, kBq/m$^2$

| Site | ā±Δ | | |
|---|---|---|---|
| | $^{137}$Cs | $^{90}$Sr | $^{241}$Am |
| CHf burned site | 5 810 ± 774 | 231 ± 33 | 21,8 ± 4,5 |
| CHnf reference site | 5 110 ± 380 | 201 ± 22 | 26,9 ± 2,0 |
| Rf burned site | 4 660 ± 472 | 83 ± 9 | 9,3 ± 0,9 |
| Rnf reference site | 4 570 ± 251 | 60 ± 10 | 10,3 ± 0,9 |
| Uf burned site | 5 050 ± 721 | 206 ± 22 | 20,6 ± 2,2 |
| Unf reference site | 5 300 ± 777 | 179 ± 14 | 21,6 ± 1,9 |
| Pf burned site | 9 370 ± 3 117 | – | – |
| Pnf reference site | 16 990 ± 5 742 | – | – |
| Tf burned site | 5 270 ± 1 343 | – | – |
| Tnf reference site | 2 060 ± 524 | – | – |
| Nf burned site | 1 010 ± 252 | – | – |
| Nnf reference site | 524 ± 134 | – | – |

Such observed differences in the behavior of $^{137}$Cs, $^{90}$Sr, and $^{241}$Am in soil may possibly be explained by localization of these radionuclides in the various components of the soil-plant complex and the thermal effects of forest fires on the upper layers of the soil, on understory plants, and litter [9; 10]. Soil properties are changed as a result of such exposure, and pyrolisation of the organic components by the fire. Organic matter content decreases and the level of generalization increases due to the burning of humus and litter. Soil microorganisms and soil fauna, which have a primary role in humification processes, disappear or change structure significantly. The stock of soil humus, humic and fulvic acids and humates, that are centers for accumulating $^{137}$Cs and $^{90}$Sr, [11] reduce. The behavior of transuranic elements may change as a result of the transformation of the chemical composition of the soil absorbing complex with which plutonium isotopes and $^{241}$Am have a high propensity to form complexes.

Chemical changes in the soil environment necessarily affect the mobility and bioavailability of radionuclides for plant communities although transuranic elements are not prone to accumulate in significant quantities in plants, unlike $^{137}$Cs and $^{90}$Sr. This means that the burnup of the vegetation does not lead to additional contamination of soil with $^{241}$Am unlike in the case of $^{137}$Cs and $^{90}$Sr.

A change in the physicochemical properties of the upper 20 cm of the soil as a result of thermal impact was confirmed at the investigated sites by a comparative analysis of the average density of soil from fire sites – 1,51 ± 0,05 g/cm$^3$ and 1,42 ± 0,08 g/cm$^3$ on areas not exposed to fire. However, this was not observed for all sites – at experimental sites Rf and Rnf and Nf and Nnf the situation is opposite (Figure 2).

Partial changes of the crystal structure of soil minerals can occur as a result of high temperature. Ash and products of incomplete combustion formed on fire sites can lead to an increase in the rate of soil generalization and an increasing content of potassium and calcium which are chemical analogs of $^{137}$Cs and $^{90}$Sr correspondingly. Potassium and calcium, in high concentrations, can replace the $^{137}$Cs and $^{90}$Sr previously fixed on the soil microaggregates and displace them in an ionic form to the soil solution. This can significantly change the parameters of accumulation of radionuclides in plants.

Succession processes at the time of sampling were at approximately the same stages at all investigated sites. The composition of leafy plant cover had a high level of similarity and regrowing trees were mostly of the same age, which made it possible to compare the sites with each other. The following species were selected for detailed study: silver birch (*Betula pendula* Roth), moss – Schreber’s big red stem moss (*Pleurozium Schreberi* Brid), lichen – Hypogymnia inflated (*Hypogymnia physodes* (L.) Nyl.), Small reed (*Calamagrostis epigeios* (L.) Roth), Mat grass (*Nardus stricta* L.), Clubawn grass (*Corynephorus canescens* (L.) P.Beauv), Heather (*Calluna vulgaris* (L.).

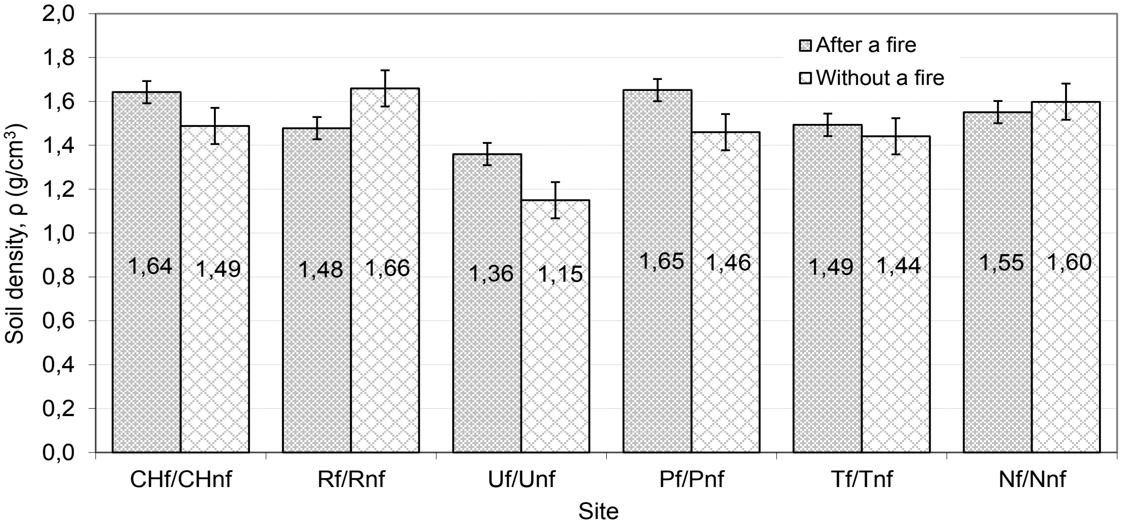

**Figure 2. – Soil density on the burned and reference sites, g/cm$^3$**

Analysis of soil-to-plant transfer of radionuclides at the experimental sites did not indicate significant differences in $C_f$ between burned and reference areas. Although an analysis of averaged $C_f$ indicated a tendency to reduction of $^{137}$Cs and $^{90}$Sr accumulation in various species and plant parts for areas exposed to fire; this appeared to be associated with changes in the structure and chemical composition of the soil as described above. A fall in the total soil fertility, reduction of the organic component together with an increased concentration of mineral elements (including element-analogs) can cause this effect. However, for $^{241}$Am the opposite trend was revealed, i. e. there was a tendency towards increased accumulation for areas exposed to fire. Different plant species exhibited significant differences in the accumulation of radionuclides. Lichens and Schreber's big red stem moss had the minimum soil-to-plant transfer factors for $^{137}$Cs on the burned sites, clubawn grass and mat grass having minimal levels of accumulation of the radionuclide. The accumulation of radionuclide in the phytomass of small reed and mat grass was 2,7 and 1,6 times higher respectively on the areas subjected to pyrogenic transformation, in contrast to the reference sites, where there was no fire. Soil-to-plant transfer factors for $^{90}$Sr were in most cases several times higher relative to those for other radionuclides. The only exception was lichen, which yielded the lowest $^{90}$Sr accumulation on all sites. Also, there was a significant difference in accumulation $^{90}$Sr by lichen in comparison with $^{137}$Cs (3,2 – 3,7 times less). Accumulation of $^{90}$Sr was higher for the reference areas in most cases, but small reeds and mat grass showed a significant increase of the radionuclide concentration on the burned sites. The maximum $C_f$ for $^{90}$Sr was in birch leaves on all study sites.

The activity concentrations of $^{241}$Am were reliably determined in samples only for mosses and lichens. Patterns of $^{241}$Am accumulation in these species are related to their biological characteristics, mosses and lichens being different taxonomic groups of organisms, with different processes of mineral nutrition that determine their ability to accumulate radionuclides. Lichens on the investigated sites had higher soil-to-plant transfer factors for $^{241}$Am as compared with mosses on all sites. $C_f$ for lichens was 2,0 and 2,6 times higher than for the mosses on burned sites and reference sites respectively. The established regularities of radionuclide accumulation in the studied plants were related with their biological properties, plants belonging to different taxons, having different modes of mineral nutrition that define their ability for radionuclide accumulation. In general, radionuclide accumulation by plants and their constituent parts in areas exposed to transformation by fire depend on many factors. Fires significantly affect the state of the forest stand, understory plants and the interrelations in ecosystems and processes occurring in it – carbon, nutrients, water cycles and forest productivity in the strict sequence of succession. Stable ground fires destroy the $A_0$ horizon – the litter of soddy podzolic soils, increasing the content of mobile calcium, magnesium, potassium, and phosphorus in the upper 10-cm soil layer and shifting the pH from the acidic range (4,2–5,0) to the slightly acidic or neutral range (5,7–7,0) while not significantly altering the total content of nutrients (phosphorus and potassium). Topsoil moisture is always higher on previously burnt sites than in the adjoined unaffected areas due to the lack of water loss by plant transpiration. $T_{ag}$ values exhibited a high variability on the experimental sites (Table 3) as indicated by previous work in relation to forest ecosystems [3; 5–8; 10].

Table 3. – Transfer factor ($T_{ag}$) of the radionuclides from soil to plants (specified uncertainty is related to the instrumental measurements of radionuclides in the samples), $m^2\ kg^{-1}$

| Object | $(T_{ag} \pm \Delta) \times 10^3$, $m^2\ kg^{-1}$ | | | | | |
|---|---|---|---|---|---|---|
| | $^{137}Cs$ | | $^{90}Sr$ | | $^{241}Am$ | |
| | without a fire | after a fire | without a fire | after a fire | without a fire | after a fire |
| Chemkov | | | | | | |
| Birch leaves | 38,0 ± 7,6 | 6,0 ± 1,2 | 252,2 ± 50,4 | 80,6 ± 16,1 | –* | –* |
| Lichen | 11,8 ± 2,4 | 19,4 ± 3,9 | 12,9 ± 2,6 | 2,6 ± 0,5 | 0,9 ± 0,2 | 3,6 ± 0,7 |
| Smallreed | – | 1,2 ± 0,2 | – | 86,4 ± 17,3 | –* | –* |
| Radin | | | | | | |
| Birch leaves | 15,5 ± 3,1 | 7,8 ± 1,6 | 545,2 ± 109,0 | 336,3 ± 67,3 | –* | –* |
| Moss | 36,7 ± 7,3 | 44,2 ± 8,8 | 100,9 ± 20,2 | 100,9 ± 20,2 | –* | 1,5 ± 0,3 |
| Lichen | 34,7 ± 7,8 | 14,2 ± 2,5 | 2,8 ± 0,6 | 6,9 ± 1,2 | –* | 2,0 ± 0,4 |
| Smallreed | 6,7 ± 1,3 | 4,4 ± 0,9 | 113,3 ± 22,7 | 149,2 ± 29,8 | –* | –* |
| Ulasy | | | | | | |
| Birch leaves | 6,4 ± 1,3 | 13,2 ± 2,6 | 129,9 ± 26,0 | 228,9 ± 45,8 | –* | –* |
| Moss | 48,9 ± 9,8 | 13,4 ± 2,7 | 141,1 ± 28,2 | 45,4 ± 9,1 | 0,5 ± 0,1 | –* |
| Smallreed | 13,8 ± 2,8 | 46,4 ± 9,3 | 43,3 ± 8,7 | 94,0 ± 18,8 | –* | –* |
| Matgrass | 5,3 ± 1,1 | 3,3 ± 0,7 | 27,4 ± 5,5 | 42,5 ± 8,5 | –* | –* |
| Puchin | | | | | | |
| Birch leaves | 0,17 ± 0,03 | 0,24 ± 0,05 | – | – | – | – |
| Moss | 0,26 ± 0,05 | 3,07 ± 0,61 | – | – | – | – |
| Smallreed | 0,19 ± 0,04 | 0,52 ± 0,10 | – | – | – | – |
| Tulgovichi | | | | | | |
| Birch leaves | 0,78 ± 0,16 | 0,71 ± 0,14 | – | – | – | – |
| Moss | 5,46 ± 1,09 | 7,28 ± 1,46 | – | – | – | – |
| Smallreed | 0,69 ± 0,14 | 1,08 ± 0,22 | – | – | – | – |
| Narovlya | | | | | | |
| Birch leaves | 3,36 ± 0,67 | 1,01 ± 0,20 | – | – | – | – |
| Clubawngrass | 1,92 ± 0,38 | 3,14 ± 0,63 | – | – | – | – |
| Heather | 62,1 ± 12,4 | 5,69 ± 1,14 | – | – | – | – |

** – $T_{ag}$ has not calculated due to activity concentration of the radionuclides in the samples are below the lower limit of detection.*

The analysis of the mean values of radionuclide $T_{ags}$ shows similar patterns as in the case of $C_f$. Mainly, the highest magnitudes of $^{137}$Cs and $^{90}$Sr $T_{ag}$ are typical for plants growing on the unburnt reference sites not exposed to thermal stress due to fires, although in some cases this dependence is not evident. The $T_{ag}$ for $^{241}$Am for the studied plant species on the burned sites was significantly higher than in reference sites: 3,2 times more for mosses, 3,1 times more for lichens (Figure 3).

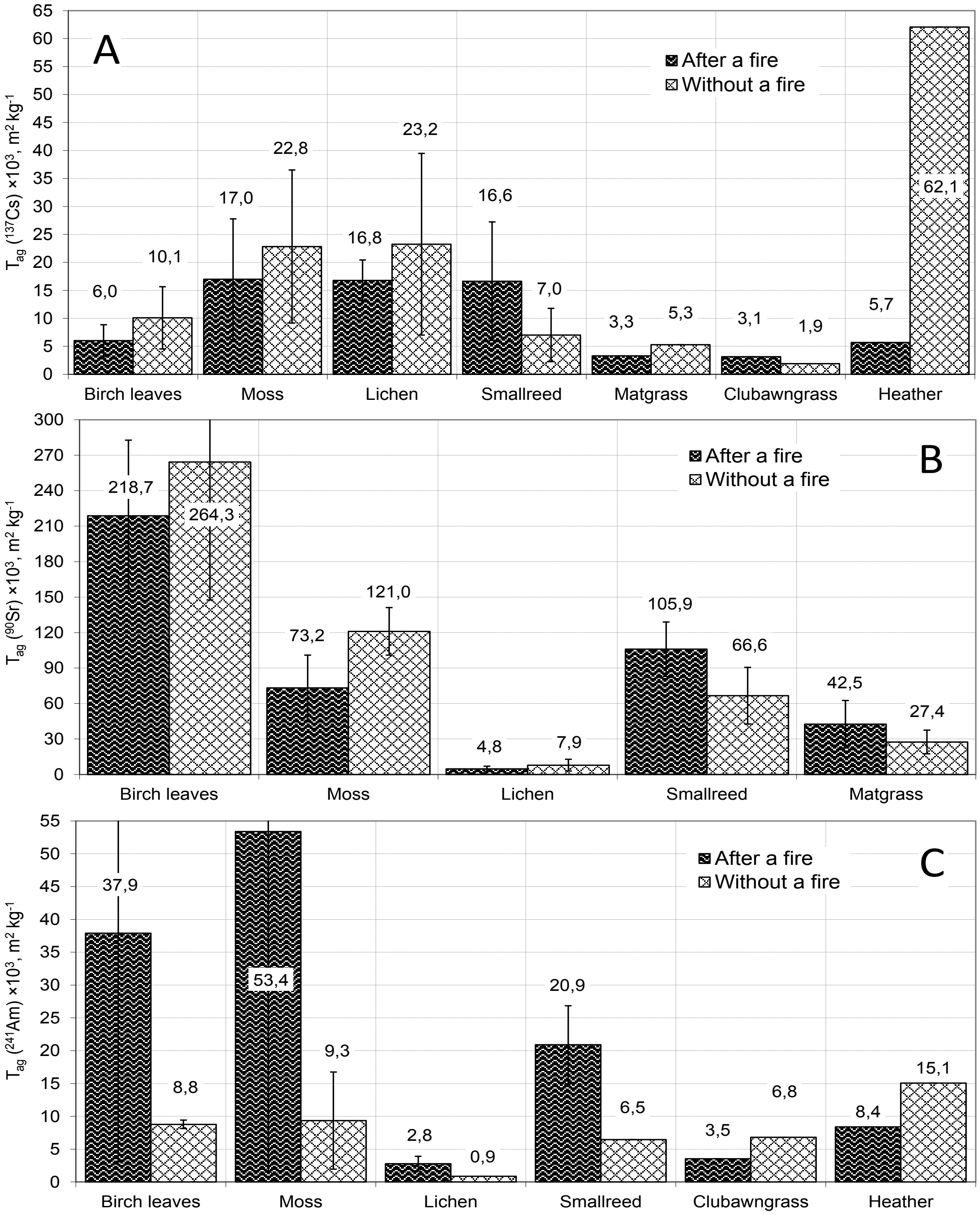

**Figure 3. – Mean values of transfer factors of $^{137}$Cs (A), $^{90}$Sr (B), and $^{241}$Am (C) from the soil into different plant species on the fire sites and sites not exposed to fire**

The analysis of the aggregated transfer factors for different species of plants shows the same patterns as soil-to-plant transfer factors. Schreber's big red stem moss, lichen, and small reed have maximum $T_{ag}$ of $^{137}$Cs on areas where a fire occurs; mat grass and clubawn grass exhibit the least $T_{ag}$ for the burnt sites. Heather has maximal levels of $^{137}$Cs accumulation on the sites not affected by fires. The maximum transfer of $^{90}$Sr from soil to plants is typical for birch leaves on the areas exposed to fire and on reference sites. Small reed displays the highest values of $C_f$ and $T_{ag}$ for $^{137}$Cs and $^{90}$Sr on all areas subjected to pyrogenic transformation due to fire. The highest transfer of $^{241}$Am is into lichen phytomass both on the burnt and in the reference sites. Also, on the unburnt reference sites, the transfer of $^{241}$Am in the moss and lichen is more than three times lower than on the burned forest sites.

All the investigated burnt sites are former pine forests with a small proportion of hardwoods (up to 10 %). The change in the moisture regime, the increase in incident light, the increasing of soil surface temperatures in the summer and the changes in the chemical composition of the soil horizons on these sites lead to the formation of other forest-growing conditions that have both a positive and negative impact on the succession changes and influence the presence of a new viable coniferous undergrowth. This enables the possibility of developing any scenario of secondary succession, and not necessarily one leading to restoration of the original pine forest species assemblage.

Analysis of the incident light levels in the cone of the noonday shadow and outside the shadow spot shows that the average illumination in the first case is 45 % lower, and the soil surface temperature is proportionally lower, as a result, pine shoots usually survive and form characteristic growth buds in the cone of the noonday shadow of old trees. Consequently, the forest-growing conditions are significantly improved due to the moderate shading of the surface. On burned sites the shading from the tree canopy is absent, shade being only partially provided by grassy vegetation which cover the soil on the burned sites for 2–3 years. Such species are represented mainly by deep root and rhizome species which retain growth buds after a fire, as well as by anemochores species, the seeds of which can transfer by air over distances of tens of kilometers. Herbaceous vegetation on the burned sites is a competitor to the tree shoots. The grass covers more quickly than the trees strengthen the surface from deflation, which is especially crucial for the burnings on contaminated territories [12].

As the uppermost layers of the soil are subjected to more pyrogenic changes [10], it would be logical to assume that changes in the parameters of radionuclides transfer to plants will be governed primarily by the processes taking place in these layers. However, the analysis of $T_{ag}$ values carried out for the upper 4-cm layer of the soil did not show any significant deviations from the earlier observed behavior of the radionuclides in the soil-plants system for the 20-cm layer. The only peculiarity was slightly higher values of the aggregated transfer factors, especially for $^{90}$Sr (up to 3 times), which is due to its greater capacity to migrate deep into the soil horizons and to be fixed in the underlying layers of the soil. Comparing the radionuclides accumulation parameters obtained in our studies with the data presented in the relevant IAEA document [13] indicates higher or close to the maximum level of values $C_f$ and $T_{ag}$ in most cases (Table 4) for this study.

This observation may be due to the peculiarities of Chernobyl fallout. Firstly, some of the Chernobyl radionuclides were included in fuel particles that are not readily available for root absorption by plants. This factor leads to underestimation of the transfer factor. Secondly, a high degree of surface contamination could cause overestimation of the calculated transfer factors for vegetation.

Analysis of changes in $T_{ag}$ on the burned sites over time shows that the highest accumulation of radionuclides by plants occurred 8–11 years later the fire event. By the second decade post-fire, the transfer of the radionuclides is decreasing (Figure 4).

Low values of $T_{ag}$ characterize the initial stage of secondary succession. Possibly, it is associated with a sharp decline in soil fertility, and changes of its physicochemical properties, biogenic changes that occur in the soil and plant associations.

Table 4. – Comparison of the experimental data from PSRER with IAEA data (IAEA, 2010)

| Object<br>$T_{ag}$, m$^2$ kg$^{-1}$<br>$C_f$, dimensionless | PSRER | | IAEA data | | |
|---|---|---|---|---|---|
| | Without a fire | After a fire | Minimum | Maximum | Page in [13] |
| $^{137}$Cs | | | | | |
| Birch leaves*, $T_{ag}$ | $3{,}4\times10^{-3}$ | $2{,}7\times10^{-3}$ | $2{,}8\times10^{-3}$ | $3{,}0\times10^{-2}$ | p. 101, tab. 38 |
| Moss, $C_f$ | 6,2 | 5,0 | $1{,}0\times10^{-2}$ | $9{,}9\times10^{-1}$ | p. 48, tab. 17 |
| Lichen, $C_f$ | 7,5 | 5,3 | $1.0\times10^{-2}$ | $9{,}9\times10^{-1}$ | p. 48, tab. 17 |
| Smallreed, $C_f$ | 1,7 | 4,5 | $1{,}0\times10^{-2}$ | $9{,}9\times10^{-1}$ | p. 48, tab. 17 |
| Matgrass, $C_f$ | 1,3 | $8{,}5\times10^{-1}$ | $1{,}0\times10^{-2}$ | $9{,}9\times10^{-1}$ | p. 48, tab. 17 |
| Clubawngrass, $C_f$ | $6{,}1\times10^{-1}$ | $9{,}8\times10^{-1}$ | $1{,}0\times10^{-2}$ | $9{,}9\times10^{-1}$ | p. 48, tab. 17 |
| Heather, $C_f$ | $2{,}0\times10$ | 1,8 | $1{,}0\times10^{-2}$ | $9{,}9\times10^{-1}$ | p. 48, tab. 17 |
| $^{90}$Sr | | | | | |
| Birch leaves*, $T_{ag}$ | $2{,}1\times10^{-1}$ | $1{,}9\times10^{-1}$ | $4{,}3\times10^{-3}$ | $7{,}8\times10^{-2}$ | p. 101, tab. 39 |
| Moss, $C_f$ | $3{,}2\times10$ | $2{,}1\times10$ | $2{,}6\times10^{-1}$ | 2,8 | p. 58, tab. 17 |
| Lichen, $C_f$ | 2,4 | 1,4 | $2{,}6\times10^{-1}$ | 2,8 | p. 58, tab. 17 |
| Smallreed, $C_f$ | $1{,}9\times10$ | $3{,}1\times10$ | $2{,}6\times10^{-1}$ | 2,8 | p. 58, tab. 17 |
| Matgrass, $C_f$ | 6,5 | $1{,}1\times10$ | $2{,}6\times10^{-1}$ | 2,8 | p. 58, tab. 17 |
| $^{241}$Am | | | | | |
| Moss, $C_f$ | $1{,}0\times10^{-1}$ | $4{,}4\times10^{-1}$ | $4{,}2\times10^{-4}$ | $2{,}6\times10^{-1}$ | p. 43, tab. 17 |
| Lichen, $C_f$ | $2{,}6\times10^{-1}$ | $8{,}8\times10^{-1}$ | $4{,}2\times10^{-4}$ | $2{,}6\times10^{-1}$ | p. 43, tab. 17 |

** – average geometric values are given for birch leaves, the average arithmetic are given for all other cases.*

It is most probable that the above-described features of $^{137}$Cs, $^{90}$Sr, and $^{241}$Am accumulation in different plant species are related to peculiarities of the species morphology and physiology along with the physicochemical properties of radionuclides and the structure of the soil-absorbing complex at different sites.

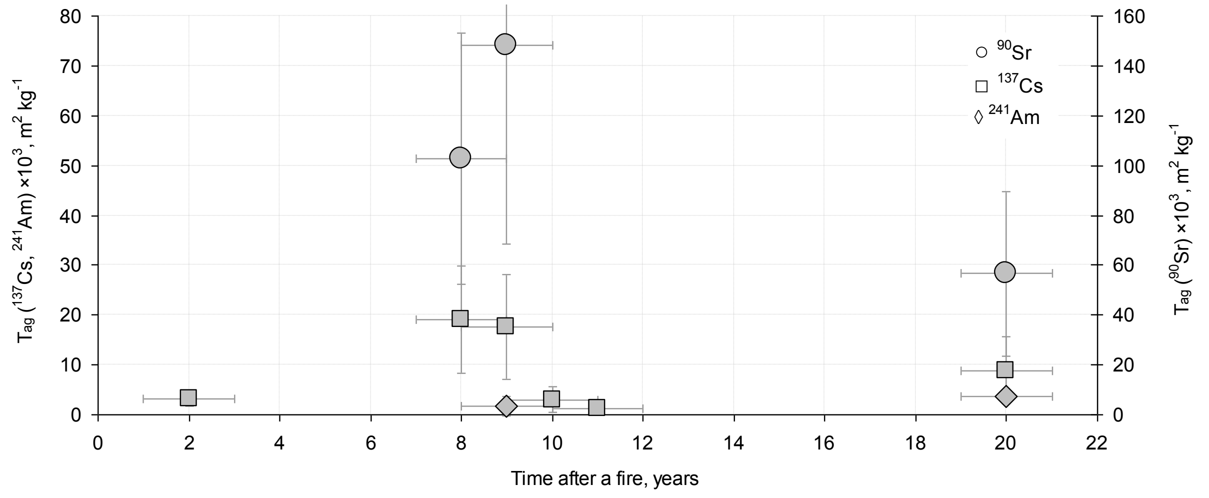

**Figure 4. – $T_{ag}$ dynamics of the radionuclides on burned sites**

The highest accumulation of $^{90}$Sr by birch leaves is explained by the structure of the root system and the depth of penetration of the roots, as well as by the higher tendency for accumulation of radionuclides by the physiologically active leaf parenchyma. Many species of birches are pioneer species on harvested and burnt sites and wastelands, where stands comprised totally of birch are present. Extensive birch stands populate abandoned lands, burned and forest-cleared sites, their shoots, at early stages of development, are very sensitive to the conditions of the environment. First of all, most birch species, especially at a very young age, need abundant sunlight. Even a slight competitive pressure from weeds is fatal for them, and their seeds germinate only lying on the surface of the soil. Birch has the fastest rate of growth among most of the forest tree species and fallen birch leaves in the process of decomposing secrete substances that are growth inhibitors for some plants [14–16].

Conditions on the burned sites, where competition from other species is least, are the best for tender birch seeds. The seeds settle first on the places cleared by recent fires, they successfully take root, and have here a high affinity for the accumulation of calcium. **S**imultaneously, $^{90}$Sr is an element-analog of Ca, and the plants absorb the radionuclide in large quantities. The branched and robust root system of a young birch can penetrate to a great depth which is inaccessible for most other forest plant species. In addition, $^{90}$Sr, currently possessing greater mobility in the ecosystems of the exclusion zone, it able to penetrate deeper into the soil than other studied radionuclides.

Schreber's big red stem moss has the greatest affinity for accumulation of $^{137}$Cs among all the studied plants, and it absorbs other radionuclides to a considerable extent. This species has a vast ecological range, growing in all forest types, on nutrient-poor sandy soil in oligotrophic forest types and on humus-rich soil in eutrophic forest types. Mosses can intensively accumulate radionuclides and heavy metals from the environment [17]. Mosses, unlike angiosperms, characterized by lower levels of light saturation and optimum temperatures for photosynthesis, more extended periods of illumination for optimal photosynthesis, an earlier beginning of photosynthesis in spring and a later end of photosynthesis in autumn [18]. These advantages in photosynthetic activity allow mosses to dominate over flowering plants in the regard of accumulation radionuclides, as well as elements of nutrition. Absorbing nutrients from the upper soil layer, mosses are more demanding with respect to fertility; explaining the lower accumulation of $^{137}$Cs and $^{90}$Sr by them on the depleted soils of burned sites.

Lichen exhibits peculiarities in $^{241}$Am accumulation from the soil: it has the most significant transfer factor in areas exposed to wildfire (Fig. 3), most likely it is related to its anatomy. Lichens are a form of symbiosis between fungi with algae, and are often considered as a means of fungi feeding. The body of lichens (thallus) consists of fungi hyphae, between which are found green or blue-green algae or cyanobacteria (photobiont). which feeds both itself and the fungi component of the lichen. Lichens often play a role as pioneer species, preparing the substrate for other plants [19; 20]. The largest lichen $C_f$ and $T_{ag}$ of $^{241}$Am on the burned sites can be explained by the fact that the photobiont in these conditions (no competitors, increased illumination) is more metabolically active and the processes of growth and absorption of nutrients by the mycelium are consequently more active. Also, mosses and lichens can absorb radionuclides with their aerial parts to a greater extent than other groups of plants.

A peculiarity observed for the small reed is more rapid absorption of $^{137}$Cs and $^{90}$Sr from soil at all sites where fires had occurred. This may explained by the lesser demands imposed by the plant on the organic component of the soil and soil moisture. Small reed grows on dry soils, often in pine forests or in dry meadows, felled and burnt areas, or on marginal land. Forming extensive thickets, any shading suppresses the development of its generative organs and generally weakens the vitality of the plant. Rhizomes, growing in the horizontal direction, penetrate to a depth governed largely by the soil and moisture regime [21; 22]. Another feature of small reed is the resistance of its seeds to desiccation and elevated environmental temperatures, allowing it to be one of the first species to repopulate burnt areas. Furthermore, the germination rate of its seeds considerably increases with increasing ambient temperature.

Matgrass exhibits relatively low transfer factors for $^{137}$Cs, factors for $^{90}$Sr being somewhat higher in this study. A peculiarity of this species is its low demand on nutrients, thriving on poor soils. It dominates in meadows of calcium poor podzolic and peaty soils. It often dominates in the grass cover of low-grass meadows, with sparse species composition [23; 24]. Matgrass does not tolerate significant flooding and shading but responds well to soil compaction, which is characteristic of burnt land. Features of its root system allow it to grow successfully on acid soils likely contributing to the lower soil-to-plant transfer for this species. In this study, clubawn grass accumulates $^{137}$Cs to a lesser than other plants. A small graminea plant of 10–30 cm height, it is not demanding on nutrients and grows on poor soils. The plant is soddy, havingshort ascending rhizomes, that form a dense sod, the leaves being very narrow and bristle-shaped. The main habitat for the growth of the species are forest and forest-steppe, being found in dry and thinned pine forests, on felled and fallow lands, usually on sandy soils. The radionuclides transfer factors for this species observed in this study are in good agreement with literature values.

Heather exhibited the greatest $T_{ag}$ of $^{137}$Cs among all the studied plants. Heather, together with some species from the genus Erica, forms specific plant communities – large thickets or heaths. A thin layer of acidic soil is usually formed under the thickets of heather. It has specific properties: dark gray color, mixed with white sand, loose, light and poor with nitrogen, potassium and phosphorus. Heather grows in dry pine forests, on burnt sites, on barren sands and sphagnum marshes – the places with the poorest and most acidic soils. Heather exists symbiotically with fungi like most representatives of the heather family, their hyphae helping the plant to extract nutrients from very poor soils, which is associated with a higher accumulation of radionuclides in virgin lands.

## Conclusion

The behavior of radionuclides in soil and plants leads to the so-called biogenic fractionation, which manifests itself in a different radionuclide composition of contaminated soil and the plants growing on it. The distribution of radionuclides in organs of plants is specific and depends on the functions of corresponding elements in the plant, its availability in soil, biological peculiarities of the plant, etc. Uptake of $^{137}$Cs, $^{90}$Sr, and $^{241}$Am from soil into plants has a quite complex dependence upon numerous factors. As establishing all of the dependences is unrealistic, it is necessary to choose a set of indicators.

As a rule, the lowest growing plants (mosses, lichens, mushrooms) exhibit the highest activity concentrations, followed by herbaceous species, shrubs, undergrowth, and young trees. The lowest activity is exhibited by trees – the upper level of the forest stand. This is due to the peculiarities of biology and plant structure – high activities of radionuclides being accumulated in organs and tissues of plants that have a high rate of metabolism and a relatively high percentage of protein. Lignified organs and tissues that play a transportational function to accumulate radionuclides in other organs and tissues. The mechanism of assimilation of radionuclides by the roots of plants is similar to the mechanisms of absorption of essential nutrients – macro- and trace elements. A certain similarity is observed in the absorption and distribution in plants of $^{90}$Sr, $^{137}$Cs and their chemical analogs – Ca and K, therefore the content of these radionuclides in biological objects is sometimes expressed in the relationship with their chemical analogs, in the so-called strontium and cesium units.

Sorption properties of the soil also determine the absorption of radionuclides by plants. Thus, $^{137}$Cs is absorbed in more significant amounts than $^{90}$Sr when entering in roots from water solution, but it accumulated to a lesser extent from the soils. The analysis of $C_f$ and $T_{ag}$ of radionuclides from the soil in different species and parts of plants showed in most cases a tendency to decrease the accumulation of $^{137}$Cs, $^{90}$Sr, and $^{241}$Am in areas exposed to fire. Decreasing the soil pH, its fertility, organic matter content together with the increasing concentration of some mineral elements (including element-analogues) can cause this effect. The maximum $C_f$ and $T_{ag}$ of $^{137}$Cs in our study were associated with Schreber's big red stem moss and the minimum values of these coefficients with matgrass. The maximum transfer of $^{90}$Sr from soil to plants is typical for birch leaves on the areas exposed to fire and on reference sites. Lichens exhibit the highest transfer factors for $^{241}$Am on both burned and reference sites. Small reed, in all cases, exhibits the highest $C_f$ and $T_{ag}$ in areas subjected to pyrogenic transformation.

In general, $^{90}$Sr shows the highest transfer from the soil to vegetation on former fire sites in all the investigated plots, which is associated with its physicochemical properties and high mobility in the soil-plant system. Dynamic processes in the development of secondary successions after a fire do not reveal a sharp jump in the parameters of the transfer of radionuclides to vegetation. The highest accumulation of radionuclides by plants was observed 8–11 years after the wildfire. A decrease in transfer of $^{137}$Cs and $^{90}$Sr from the soil to the plants over time has been observed. However, $^{241}$Am exhibits a very low transfer into plants, and its transfer factor does not vary significantly.

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