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Dynamic equilibrium of radiocesium with stable cesium within the soil–mushroom system in Turkish pine forest

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The $^{137}Cs/^{133}Cs$ ratios observed in mushroom samples and in organic layers shows that ^{137}Cs is well mixed with stable Cs within the biological cycle in the studied pine forest.

Abstract

Mushrooms and soils collected from pine forests in İzmir, Turkey were measured for radiocesium and stable Cs in 2002. The ranges of ^{137}Cs and stable Cs concentrations in mushrooms were 9.84 ± 1.67 to 401 ± 3.85 Bq kg^{-1} dry weight and 0.040 ± 0.004 to 11.3 ± 1.09 mg kg^{-1} dry weight, respectively. The concentrations of ^{137}Cs and stable Cs in soils were 0.29 ± 0.18 to 161 ± 1.12 Bq kg^{-1} dry weight and 0.14 ± 0.004 to 1.44 ± 0.045 mg kg^{-1} dry weight, respectively. Even though different species were included, the concentration ratios of ^{137}Cs to stable Cs were fairly constant for samples collected at the same forest site, and were in the same order of magnitude as the ^{137}Cs to stable Cs ratios for the organic soil layers. The soil-to-mushroom transfer factors of ^{137}Cs and stable Cs were in the range of 0.19–3.15 and 0.17–12.3, respectively. The transfer factors of ^{137}Cs were significantly correlated to those of stable Cs.

Keywords: Radiocesium; Mushroom; Transfer factors; Forest ecosystem; Dynamic equilibrium

1. Introduction

Large areas of forests in the world have been contaminated by radiocesium (^{137}Cs: $T_{1/2} = 30.1$ years, ^{134}Cs: $T_{1/2} = 2.1$ years) originating from nuclear weapons testing and/or nuclear accidents. Many studies on the distribution and transfer of radiocesium in forest ecosystems were carried out after the Chernobyl accident in 1986 (Mietelski et al., 1994; Yoshida and Muramatsu, 1994b; Blagoeva and Zikovsky, 1995; Gaso et al., 1996; Lee et al., 1997; Papastefanou et al., 1999; Rühm et al., 1999; Gri et al., 2000; Shenber, 2001; Outola et al., 2003; Pokarzhevskii et al., 2003). Since 1986, a number of authors have reported considerable levels of radiocesium contamination in different species of fungi not only in European countries, such as Austria, Sweden, Italy, Germany, Poland, France, England, Spain and Ukraine, but also in Japan, Taiwan and Mexico (Yoshida and Muramatsu, 1994a; Calmet et al., 1998; Gaso et al., 1998; Kirchner and Daillant, 1998; Korobova et al., 1998; Tsukada et al., 1998; Wang et al., 1998; Barnett et al., 1999; McGee et al., 2000; Kalac, 2001; Mascanzoni, 2001; Ko et al., 2003; Baeza et al., 2004). More than a decade after deposition on Chernobyl fallout, radiocesium is still localized mainly in the organic soil layers, and radiocesium contamination of forest products is still high in contrast to agricultural products (Nimis, 1996; Kruyts and Delvaux, 2002; Yoshida et al., 2004; Kuwahara et al., 2005). Fungal and microbiological activities are likely to contribute substantially to the long-term retention of radionuclides in organic layers of forest soil (Rafferty et al., 1997, 2000; Steiner et al., 2002; Kuwahara et al., 2005) and radiocesium is

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expected to be stored predominantly within fungal biomass below ground (Nikolova et al., 1997, 2000). In consequence of these findings, studies are required on the long-term behavior of ^{137}Cs in the forest ecosystem. It is suggested that much information can be obtained from studies of the behavior of stable Cs in the forest ecosystem and the ratio between stable Cs and ^{137}Cs (Yoshida et al., 2000).

In this study, the concentrations of ^{137}Cs originating from the Chernobyl accident and global fallout and stable Cs were determined for mushroom and soil samples collected from forests. Data on radiocesium concentrations in mushrooms and soils of the İzmir forest region have not previously been reported and the present study is the first systematic effort to provide data on this aspect.

2. Concept of transfer factors

Knowledge about the long-term behavior of ^{137}Cs in the forest ecosystem is needed in order to estimate the radiation dose to humans using forest products and to predict the future contamination levels of forest products from the affected areas. Several models for radionuclide transport in forest ecosystem have been developed (Steiner et al., 2000; Absalom et al., 2001), but there is still high uncertainty on model predictions (Gillett and Crout, 2000). One of the most uncertain parameters commonly used in forest models are transfer factors. In view of the limited knowledge about the mechanisms and processes involved in the mobilization, translocation and uptake of stable elements and their radioactive isotopes, the concept of transfer factors is a popular approach to quantify the transfer of radionuclides from soil to fungal fruit-bodies (Steiner et al., 2002). Transfer factors have been recommended for dose assessment by many international agencies IAEA (IAEA, 1996), IUR (IUR, 1992) and ICRP (ICRP, 1979) because of its conceptual simplicity and common use in radioecology (The International Atomic Energy Agency (1982, 1989, 1994), The International Commission On Radiological Protection (1979) and International Union of Radioecologists (1992)). In general, transfer factors are expressed as parameters representing the contamination of fungal fruit bodies divided by parameters representing the contamination of soil. The contamination of fungal fruit bodies is usually expressed as the amount of radioactivity per unit weight, either on a dry weight (Bq kg^{-1} dry weight) or a fresh weight basis (Bq kg^{-1} fresh weight). The contamination of soil is usually expressed as the amount of radioactivity per surface area (Bq m^{-2}) or per unit dry weight (Bq kg^{-1} dry weight).

The concept of transfer factors was originally developed for agricultural systems, which are characterized by a more or less homogeneous distribution in soil of the investigated radionuclide. The situation, however, is different in the case of forest ecosystems, where soil is characterized by layers or so-called horizons with different radiocesium activity levels. Here, it is difficult to decide which part of the soil should be used for the calculation of the transfer factor.

In principle, this problem can be solved by expressing transfer factors in terms of the radiocesium activity in mushrooms (Bq kg^{-1}) relative to its area-related deposition in soil (Bq m^{-2}). This definition gives a measure of the transfer independent of the variations that radionuclides exhibit throughout the soil profile. However, one has to be aware of the dynamics of the radiocesium transport within soil. Contamination of upper layers is expected to decrease, while contamination of deeper layers is expected to increase, due to the vertical transport of radiocesium. As a consequence, transfer factors defined relative to area-related soil deposition become time-dependent, since plants or fungi have a distinct root or mycelium location. It is therefore not surprising that these transfer factors for fungi vary by about two orders of magnitude, even for the same fungal species (Linkov et al., 2000).

Transfer factors referring to standardized soil depths are defined as the ratio of the activity concentration in fungal fruit body (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration in soil (Bq kg^{-1} dry weight) within the uppermost layer of a standardized thickness. Transfer factors referring to the concentration within standardized soil depth are of limited usefulness in the case of soil with a multi-layered structure and a vertical profile of concentration. Averaging the radionuclide concentration over a standardized soil depth, irrespective of the location of the mycelium or the fine roots, might lead to a large variation and a time dependence of transfer factors.

In view of these arguments it is suggested to calculate transfer factors using the radiocesium activities in Bq kg^{-1} dry weight of that soil layer, from which radiocesium is taken up (Rühm et al., 1998). Transfer factors, defined as the ratio of the activity concentration in fungal fruit bodies (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration of the specific soil layer exploited by the mycelium (Bq kg^{-1} dry weight), proved to be useful, especially in connection with dynamic radioecological models. This definition of transfer factors was proposed in the late 1980s.

Unfortunately, it is very difficult to localize fungal mycelia in situ species specifically. An useful method was developed by Rühm et al. (1997) in order to determine the mycelium location. This method is based on the idea that the ^{137}Cs/^{134}Cs ratio in fungal fruit bodies should reflect the isotopic ratio of that soil horizon from which radiocesium is predominantly taken up.

The methodology described above is very difficult to apply now, since ^{134}Cs is a short-lived radionuclide. Current research projects deal with the question of whether the ratio ^{137}Cs/^{133}Cs can alternatively be used to localize fungal mycelia. This approach is based on the assumption that the physico-chemical properties of stable ^{133}Cs and radioactive ^{137}Cs are similar.

However, it is known from agricultural systems that transfer factors for stable cesium are lower by about one order of magnitude than those for ^{137}Cs. This finding has mainly been attributed to differences in bioavailability between ^{137}Cs and stable ^{133}Cs in soil (Rühm et al., 1999). Minerals which can exist in the agricultural soils will normally contain stable ^{133}Cs. The portion of stable cesium that is enclosed within mineral grains cannot be reached by roots and therefore is not available for uptake. Accordingly the transfer factor for