Factors Influencing the Soil to Plant Transfer of Radiocaesium

J. Guillén, A. Baeza, A. Salas, J.G. Muñoz-Muñoz, and A. Muñoz-Serrano

Abstract Radiocaesium isotopes are among the long-lived radionuclides which were released in largest quantities into the environment as a consequence of the atmospheric nuclear weapon tests and accidents involving nuclear material (Chernobyl and Fukushima, etc.). Its transfer to plants, especially those for human or animal food, can be a major pathway for human intake and, therefore, have a significant radiological impact. There are many factors that can affect the radiocaesium transfer to plants, which are reviewed in this chapter, such as the considered plant species, its habitat, climatic conditions, type of soil (clay content, physico-chemical characteristics, organic matter, use of amendments, etc.).

Keywords Plant • Transfer factor • Speciation • Organic matter • Clay • Soil

1 Introduction

The occurrence of radiocaesium is ubiquitous in the environment, as a result of the global fallout caused by the atmospheric nuclear weapon tests carried out in the 1950–1960s (UNSCEAR 2000). Accidents involving nuclear material, such as Chernobyl and Fukushima Dai-ichi also released large quantities of radiocaesium and, as a result, vast areas were contaminated (UNSCEAR 2000; IAEA 2006, 2014). Radiocaesium is one of the long-lived radionuclides that are highly significant from the radiological protection point of view. Due to the chemical similarities with potassium, it enters the food chain pathway and may cause a health hazard to humans. Plant consumption, either as direct foodstuff or indirectly as animal foodstuff, is a major pathway for radiocaesium incorporation. In fact, its content in foodstuff has been controlled since the Chernobyl accident happened, with temporary permissive limits for different foodstuff after an emergency (IAEA 2006; EU 2009; CA 2011; Hamada and Ogino 2012). One of the main interests of the study of the

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soil to plant transfer processes of radiocaesium is its quantification and later use to predict its content in plants for human and/or animal consumption. This knowledge is necessary in order to carry out remedial actions to decrease the radiocaesium incorporated by men, reducing the potential health hazard. In this chapter, the most usual ways to quantify the radiocaesium soil-to-plant transfer are presented, along with its problems and factors that influence its great range of variation.

2 Quantification of the Transfer Process

The concept of transfer factor (usually TF, CR or F_v) is the most usual and straightforward approach to quantify this process. It is usually defined as the ratio between the radionuclide content in the plant or plant compartment and its content in the soil (see Eq. 1) (IAEA 2010).

$$\mathrm{TF,CR,F_v} = \frac{\mathrm{Bq / kg\ d.w.\ plant}}{\mathrm{Bq / kg\ d.w.\ soil}} \tag{1}$$

The radionuclide content in plant is often expressed as dry weight basis, in an attempt to decrease the data variability. In case of fruits or when data are used to estimate internal dose rates, F_v values are usually given as fresh weight. These data can be converted to dry weight basis, using dry matter content reference values, such as those presented in IAEA TRS 472 (IAEA 2010). Calculation of the transfer factor values, F_v depends on the depth of soil considered, which is especially important in cases in which the distribution of radionuclides is inhomogeneous in depth. In order to reduce the influence of this variable, the International Union of Radioecology (IUR) recommended a standardized root location in soil. This approach assumes that all roots and all radionuclides present in the rooting zone are in that soil layer. The soil depth is 10 cm for grass, and 20 cm for all other crops, including trees (IUR 1992). The use of standard soil depth for agricultural systems is reasonable since ploughing is usually part of soil preparation for growing crops. Thus, the radionuclide content of this soil layer becomes homogeneous. The concept of transfer factor also implies, by definition, that the transfer process proportional to the radionuclide content in the soil, and that the soil-plant system is in equilibrium or quasi-equilibrium. This condition is achieved in most cases, assuming that the radionuclide flow from soil to plants is negligible compared to the total amount of radionuclides present in soil (IAEA 2010).

Figure 1 shows the range of variation of ^{137}Cs transfer factors worldwide for different plants/crops. The soil-plant transfer factors for radiocaesium present a large range of variation, about 4–5 orders of magnitude, reflecting the fact that transfer processes are complex and affected by many variables. Wild products, such as mushrooms or berries, usually have high F_v values, followed by grass, non-leafy vegetables and stems of cereals. This high range of variation for F_v implies that they

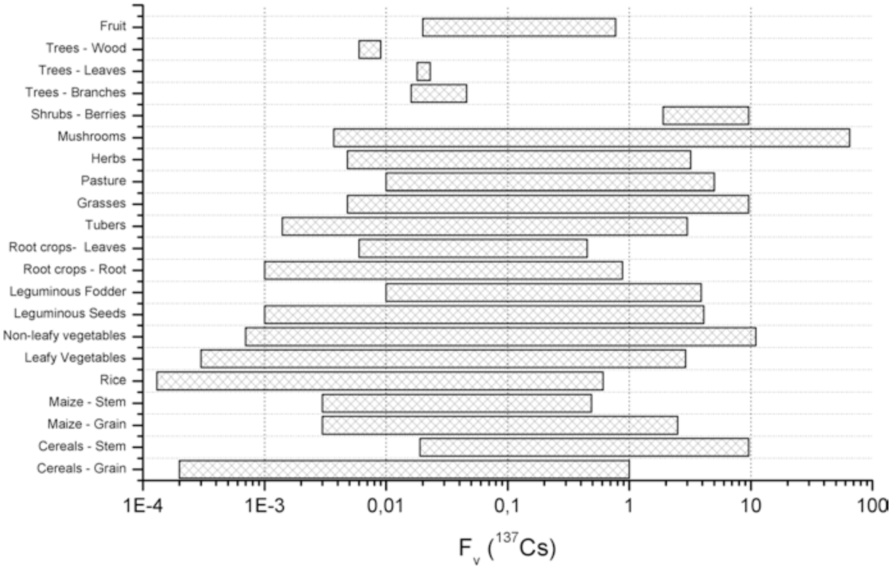

Fig. 1 Worldwide range of variation of soil-to-plant transfer factor F_v for ^{137}Cs. Data adopted from Ban-nai and Muramatsu (2002), Baeza et al. (2005), Kuwahara et al. (2005), Al-Oudat et al. (2006), Gaso et al. (2007), Handl et al. (2008), Moran-Hunter and O'Dea (2008), IAEA (2010), Zhiyanski et al. (2010), Karadeniz and Yaprak (2011), Velasco et al. (2012), Kobayashi et al. (2014), Yamashita et al. (2014), Godyń et al. (2016)

should be considered valid at local level, due to a great number of factors. As cereals are a wide extended crop, the FAO/IAEA/IUR workgroup proposed to consider it as reference crop in an attempt to reduce variability, taking into account the ratios between different crops and cereals (Frissel et al. 2002)

In the case of a radioactive fallout event, the aggregated transfer factor, T_{agg}, is also frequently used to estimate the transfer, because it relates the activity in plant compartments with that deposited on soil (see Eq. 2). Thus, they are used to predict the transfer of radionuclides to plants, essential for stakeholders to manage emergencies.

$$T_{agg}\left(m^2kg^{-1}\right) = \frac{\text{Bq / kg d.w. plant}}{\text{Bq / m}^2\text{deposited on soil}} \tag{2}$$

The T_{agg} values are also used in natural and semi-natural ecosystems, such as forests, in which the distribution of anthropogenic radionuclides is inhomogeneous in depth. This is especially important for radiocaesium, because after radioactive fallout, it is deposited on the surface layer of soil and then it migrates downwards to deeper layers. Figure 2 shows the range of variation of T_{agg} for ^{137}Cs for different plants in forest ecosystems worldwide. As it occurred for F_v values, the range of

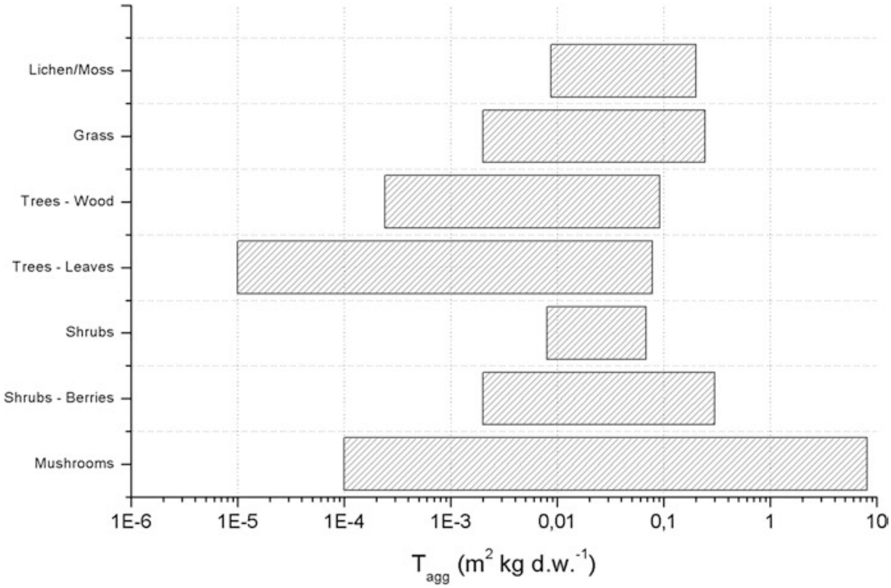

Fig. 2 Worldwide range of variation of soil-to-plant aggregated transfer factor, T_{agg}, for ^{137}Cs. Data adopted from Strandberg (1994), Fesenko et al. (2001b), Kaduka et al. (2006), IAEA (2010), Karadeniz and Yaprak (2011), Nakai et al. (2015)

variation of T_{agg} values is about 5–6 orders of magnitude. Mushrooms presented again the highest T_{agg} values.

Another important integrating transfer factor for the ecosystems is the geochemical transfer factor, defined as the ratio between the radionuclide total activity in vegetation collected from the certain area, expressed as Bq/m^2, and its deposition in the same area, expressed as Bq/m^2.

$$T_{\text{geo}} = \frac{\text{Bq / m}^2 \text{in plant biomass}}{\text{Bq / m}^2 \text{in soil}} \tag{3}$$

The F_v values can also be used in forest ecosystems, especially when the source term of the deposition of radionuclides occurred long time ago, as it is the case of areas in which the main deposition event was the global fallout from atmospheric nuclear tests in the 1950–1960s (UNSCEAR 2000). However, in this case it is very important to define precisely the depth of the soil layer used in its calculation. For wild grass, as it is a Reference Animal and Plant (RAP) used to analyze the dose rate to wildlife (ICRP 2008), the soil depth used is standardized to 10 cm. This depth of soil has also been considered in software to assess the radiological impact of radionuclides in non-human biota, such as ERICA (Brown et al. 2008). In the case of mushrooms, some authors took into account the radiocaesium content of the soil layer in which mycelium was located. This location was performed by using different techniques: mechanical isolation of fungal mycelium directly from the soil (Nikolova et al. 2000); the determination in the soil of chemical compounds present in the cell walls of the mycelium (Baeza et al. 2005); and the comparison of the ratio $^{137}Cs/^{134}Cs$ in different layers of soil and in the fruit bodies (Rühm et al. 1997). Regarding trees, a high percentage, 54–70 %, of fine roots of pine (*Pinus sylvestris*) and birch (*Betula pendula*) were detected at 40 cm depth in soil (Fesenko et al. 2001a). Therefore, the depth of the soil layer should be given when calculating F_v values in forest ecosystems.

Other radioisotopes of caesium also occur in the environment, such as ^{134}Cs or stable ^{133}Cs. The source term for both ^{137}Cs and ^{134}Cs is usually the same, and their ratio, $^{134}Cs/^{137}Cs$, depends on the characteristics of the fallout event. Differences in the transfer process due to isotopic effect are almost negligible, being the F_v values for ^{134}Cs and ^{137}Cs the same within measuring uncertainties (Kobayashi et al. 2014). In fact, ^{134}Cs is used in transfer experiments under controlled conditions (Carini and Lombi 1997). The stable isotope of caesium, ^{133}Cs, has been considered as a key element to predict long-term behaviour of radiocaesium in the environment, i.e. the transfer of ^{137}Cs and stable caesium is expected to be the same long time after the fallout event. The correlation of the ^{137}Cs and stable caesium is considered as an indication of equilibrium in forest ecosystems (Yoshida et al. 2004; Kuwahara et al. 2005)

There are also other ways to quantify the transfer processes, such as compartmental models (Kichner 1998; Absalom et al. 1999, 2001; Baeza et al. 2001). These models are capable of determining the effect of several variables affecting the radiocaesium transfer processes: influence of clay, organic matter, bioavailability of radiocaesium, exchangeable potassium, etc. However, they require a high degree of knowledge of the compartments involved in the transfer, and are difficult to implement for screening purposes.

3 Factors Affecting the Caesium Soil-to-Plant Transfer

The fact that the soil-to-plant transfer factors for radiocaesium vary widely over several orders of magnitude implies that there are factors influencing the transfer processes other than its content in soil and plant compartment considered. Climatic conditions can affect the transfer processes, since they control major variables affecting the plant growth, as temperature, water regime, humidity, etc. Temperate environments a generally located between the Tropics and the Polar Regions, in which the temperatures are relatively moderate, and changes between winter and summer are also moderate. Tropical environments are typically found in the Tropics, with mean monthly temperatures about 18 °C, and seasonal variations dominated by precipitation. In these environments, there is a rapid decomposition of almost all organic materials deposited on surface soil, and high mineral weathering rates in soil. Subtropical climates are roughly located in areas between the Tropics and the

38th parallel in each hemisphere. Mediterranean climate is a particular variety of subtropical climate characterized by warm to hot, dry summers and mild to cool, wet winters.

Table 1 shows the range of variation of radiocaesium F_v values for several plant groups in different climates (IAEA 2010). In the case of cereal grain and root crop plant groups, these ranges do not vary greatly, and are about the same order of magnitude temperate, subtropical and tropical climates. In the case of root crops in tropical climate, the lower range can be due to the limited number of samples considered. Regarding grass plant group, tropical environments seem to present a wider range of variation. Thus, the direct influence of climatic conditions on the radiocaesium transfer seems to be minimal, but its indirect effect, through changes in soil and crop properties can be significant (IAEA 2010). However, in Mediterranean environments, in which there is an alternation between hot dry seasons (mainly summer) and cold wet seasons (mainly winter), the radiocaesium transfer to grass showed a seasonal dependence reflecting the variation of nutrient availability in each season (Baeza et al. 2001).

Soil characteristics play a crucial role on radiocaesium transfer. As a way of example, Table 2 shows the range of variation of radiocaesium F_v for several plant groups in different soil groups, based on their texture, cation exchange capacity and organic matter content. Clay soils show lower F_v values than other soil groups. This is because clay content in soil acts as a sink for radiocaesium (Ohnuki 1994). It is found usually that the higher the clay content the higher the radiocesium content in it (Apostolakis et al. 1991), because its sorption on clay minerals (Ohnuki 1994; Atun et al. 1996; Kruyts and Delvaux 2002; Stauton et al. 2002). This sorption can

Table 1 Range of variation of radiocaesium transfer factors, F_v for several plant groups in different climatic conditions

Plant group	Environment		
	Temperate	Subtropical	Tropical
Cereal-grain	2×10^{-4}–0.9	1×10^{-3}–2.6×10^{-2}	6×10^{-2}–1.0
Root crops-root	1×10^{-3}–0.88	1.4×10^{-3}–0.23	0.13–0.81
Grass	4.8×10^{-3}–0.99	6×10^{-3}–3.7	1.5×10^{-4}–13

Data adopted from IAEA (2010)

Table 2 Range of variation of radiocaesium transfer factors, F_v for several plant groups in different soil groups

Plant group	Soil group			
	Sand	Loam	Clay	Organic
Cereals/grain	2×10^{-3}–0.66	8×10^{-4}–0.20	2×10^{-4}–9×10^{-2}	1×10^{-2}–0.73
Root crop/root	8×10^{-3}–0.40	1×10^{-4}–0.16	5×10^{-3}–6×10^{-2}	1.6×10^{-2}–0.88
Grass	1×10^{-2}–4.8	1×10^{-2}–2.6	1×10^{-2}–1.2	0.3–5.0

Data adopted from IAEA (2010)

be carried out in two different ways: sorption sites on regular exchange sites (RES) and on frayed edge sites (FES). These sorption sites have different properties. RES are reversible and non-specific; while sorption on FES is irreversible and specific, due to the small hydration energy for caesium ions. The energy required for caesium desorption from FES was found to be so large that desorption was energetically unfavorable (Stauton et al. 2002). The FES selectivity for monovalent cations decrease in the order: $Cs^+ > NH_4^+ > Rb^+ > K^+ > Na^+ > Li^+$ (Rigol et al. 2002; Stauton et al. 2002). The K^+ content in soil can cause the collapse of the expanded interlayers (Rigol et al. 2002). In this case the caesium binded inside the interlayers is blocked and unavailable for transfer processes. Climate can also affect the type and exchange capacity of clays. The occurrences of low exchange capacity clays, such as kaolinite, are more common in the Tropics than in temperate climates, due to the high mineral weathering rate (IAEA 2010).

Soil organic matter content can also influence the radiocaesium transfer, since it can influence its association with soil components. The organic matter compounds comprise humin and humic substances present in soil. The humic acids are macromolecules with carboxylic and phenolic functional groups. In the pH range 3–6, the humic substances chemistry is dominated by carboxylic acid groups (Lofts et al. 2002). The addition of humic substances to mineral clays reduced its radiocaesium adsorption (Dumat et al. 1997; Dumat and Stauton 1999). This reduction was not due to the direct sorption of caesium on humic substances. In fact, its association with humic and fulvic acids is very low, about 5 % (Dumat and Stauton 1999; Amano et al. 1999; Vinichuk et al. 2005; Guillén et al. 2015). The presence of organic matter can produce a dilution effect on FES, increasing the availability of radiocaesium (Kruyts and Delvaux 2002). The association of organic compounds can also impede the collapse of interlayers in clay minerals, maintaining them open for reversible exchange sites (Rigol et al. 2002). However, even a small content of clay, about 1–6 %, can bind caesium effectively (Forsberg et al. 2001; Lofts et al. 2002; Rigol et al. 2002). Only in soils with more than 95 % content of organic matter and no clay, the adsorption occur in non-specific sites (Rigol et al. 2002).

Microflora can also retain effectively the radiocaesium present in the soil. This association can be assessed by killing all living organisms present in the soil by autoclaving, γ-ray sterilization, fungicides, or chloroform fumigation (Brückman and Wolters 1994; Guillitte et al. 1994; Stemmer et al. 2005). The caesium retention by microflora was about the 1–56 % (Brückman and Wolters 1994; Guillitte et al. 1994; Stemmer et al. 2005). After these treatments, it was observed an increase of the labile caesium (Stemmer et al. 2005). The soil organic matter and its mineralization potential controlled the radiocaesium transfer in soddy-podzolic and peat bog soils, since the mineralization is accompanied by the release of ^{137}Cs and mineral nitrogen (Tulina et al. 2010).

Fungal material present in soil also participated greatly in the radiocaesium cycling in forests. The fungal mycelium in soil can retain about 0.1–32 % of the total inventory in soil (Olsen et al. 1990; Vinichuk and Johansson 2003, Vinichuk et al. 2005). The increase of labile radiocaesium after the elimination of microflora may be attributed to its release from fungal mycelium. About 42–83 % of the radiocaesium associated with fungal mycelium can be extracted with distilled water (Vinichuk et al. 2005). Larger amounts of caesium were found in white spots on fungal mycelium, mainly associated with polyphosphates (Landeweert et al. 2001). The importance of fungal mycelium can be due to their ability to exude organic acids, such as citric and oxalic acid, which can complex metals in the surrounding (Gadd 1999). The concentration of these acids in bulk soil is usually low, but high in the microenvironments near the hyphaes (Landeweert et al. 2001). The occurrence of mycorrhizal fungi living in symbiotic relationship with plant hosts can also influence the radiocaesium transfer. They usually present higher radiocaesium content than other fungi with saprophyte or parasitic nutritional mechanisms (Guillitte et al. 1994; Kammerer et al. 1994; Yoshida and Muramatsu 1994). This suggested that mycorrhizic fungi act as a "filter" for the host plant, accumulating non essential elements such as caesium (Guillitte et al. 1994; Kammerer et al. 1994). Laboratory experiences growing Norway spruce (*Picea abis*) seedlings inoculated with the ectomycorrhizal fungus *Hebeloma crustiliniforme* and without it showed a reduction in the uptake of ^{134}Cs, and a higher accumulation in the hypae (Brunner et al. 1996; Riesen and Bunne 1996). The arbuscular mycorrhizal colonization of several species of grass by *Glomus mossea* also resulted in reduction of caesium content in shoots and root (Berreck and Haselwandter 2001). However, this does not reflect the whole tendencies. According to the previous research, the uptake of caesium by plants in the presence of arbuscular mycorrhiza was either lower (Berreck and Haselwandter 2001), similar (Rosén et al. 2005) or higher (Entry et al. 1996) as in nonmycorrhizal plant species.

In the discussion of the previous factors affecting the radiocaesium transfer, the concept of bioavailability has been mentioned, as the fraction of the radionuclide pool in soil that it is able to be transferred to a plant compartment. Its empirical determination is based on sequential extraction procedures in which soil is attacked by a series of reagents with increasing replacement/extraction power. One of the main handicaps of this approach to the transfer is the fact that there is no unified procedure/reagent to carry out (Guillén et al. 2014). There is in the literature a great number of modifications on these procedures varying reagents, their concentrations, contact times, and order of application. This fact implies that sometimes the comparison of the results from different sequential extraction methods can be complex, because although the extractants are usually designed to attack a single geochemical phase, they are not completely specific (Schultz et al. 1998). The bioavailable fraction can be subdivided into two different sub-fractions: readily available (water soluble and exchangeable) and potentially available (associated with oxides and carbonates). However, the number of reagents usually involved in the bioavailable fraction is quite standard, at least for the readily available fraction. Water soluble fraction present the weakest attachment with soil particles, as the reagent used is distilled water. Radiocaesium associated with this fraction is usually minimal and sometimes it is omitted from some sequential procedures (Riise et al.

1990). Exchangeable fraction comprises the radionuclides associated with ion exchange sites. The NH_4AcO 1 M at pH 7 is the reagent usually selected for this fraction, and is considered to be a robust extractant for acidic and neutral soils, but not for alkaline soils, although it can be buffered to extend its range of application (Kennedy et al. 1997). Its validity for radiocaesium is based on the fact that NH_4^+, along with K^+, are competitors with Cs^+ in soils. Other reagents based on divalent ions, such as $MgCl_2$ or $CaCl_2$, are also able to desorb caesium from exchange sites, but are considered to be less effective in clay interlayers (Rigol et al. 2002). The exchangeable fraction (extracted with NH_4OAc) of radiocaesium in soils is in the range 1.8–29 % of the total content of soil (Riise et al. 1990; Bunzl et al. 1997; Lee and Lee 2000; Forsberg et al. 2001; Vinichuk et al. 2005). The exchangeable fraction presented a dependence of the layer of soil considered, increasing with the depth of that layer (Bunzl et al. 1997). This fraction was not constant with time, but it decreased while increasing the lapsus of time since the deposition of radiocaesium occurred (Cheshire and Shand 1991; Krouglov et al. 1998; Baeza et al. 1999; Forsberg et al. 2001), which is usually known as ageing effect, due to the irreversible sorption of caesium. This ageing effect caused the reduction of the transfer factors to rice after Fukushima accident (Fujimara et al. 2015). The definition of the potentially available fraction is more diffuse, and only considered in one of the sequential extraction procedure used in agriculture testing (Pavlotskaya 1974). The use of dilute inorganic acids, HCl 1 M, after the exchangeable fraction can remove cations from exchange complexes in soil, and also dissolve oxides, hydroxides, carbonates and some alkaline earth compounds. It has been considered potentially available to plants (Fesenko et al. 2001a). Other reagents can be considered partially equivalent to this potentially bioavailable fraction. The radionuclides associated with the carbonated fraction can be extracted with sodium acetate in acetic acid (Tessier et al. 1979) or with NH_4OAc after the extraction of the exchangeable fraction with $MgCl_2$ (Schultz et al. 1998). The reducible fraction, also named in some procedures bound to Fe and Mn oxides, are obtained by the application of a reducing extractant, generally $NH_2OH \cdot HCl$ (Tessier et al. 1979; Riise et al. 1990; Schultz et al. 1998).

Finally, the definition of transfer factor does not take into account the stage of maturity of the plants (see Eqs. 1 and 2). They are usually derived from activities in soil and plant compartments at the end of the growing period, which may lead to high values (IAEA 2010). In this sense, the transfer factors can be considered as integrators of the whole growth period, providing a mean value of the period. However, there are variations due to the different nutritional requirements in each stage of development. In the case of fruits, the transfer factor was found to be maximum at the initial stages and decrease as fruit developed, being lowest at the maturation period (Velasco et al. 2012). Experiences of ^{134}Cs contamination of *Pleurotus eryngii* fungus species at different stages of development showed an increase of the transfer of ^{134}Cs as the contamination occurred closer to maturity stage (Baeza et al. 2002).

4 Modification of the Radiocaesium Transfer

The modifications of the radiocaesium transfer processes are a key factor to controlling its accumulation in plant and thus remediate contaminated sites. The modification can be carried out in two opposite ways: enhancement and inhibition. Phytoremediation uses the first one. Its objective is to locate hyperaccumulators and thus maximize the soil-to-plant transfer and therefore remove radiocaesium from soil (Cook et al. 2009; Jagetiya et al. 2014). In this case, the management of the plants after harvest has also to be considered. Other remediation techniques are focused on reducing the transfer. Ploughing was considered to reduce the radiocaesium content in soil after a deposition by mixing the surface soil with that at different depth with lower content, thus diluting the deposited radiocaesium. In areas affected by the Chernobyl accident, the combination of ploughing and reseeding was effective to reduce the radiocaesium transfer (Camps et al. 2004). However, ploughing in areas affected by Fukushima accident did not result in reducing the transfer factors (Kobayashi et al. 2014). This might be as consequence of the quantity of radiocaesium deposited, and not being able to dilute it effectively.

The radiocaesium transfer can also be modified by the application of inorganic fertilizers and soil amendments. The final effect depends on the fertilizer composition. Potassium based fertilizer are used to reduce the radiocaesium transfer, by saturating the soil with an additional supply of nutrients chemically analogous to the radionuclides (Nisbet et al. 1993). There is a critical threshold in the concentration of potassium in soil solution. At lower concentrations, the root uptake mechanism is unable to distinguish between alkaline elements Cs^+, Rb^+, and K^+ (Nisbet et al. 1993; Shaw 1993; Zhu et al. 2000). Above ca. 20 μM, wheat root uptakes K^+ preferentially to Cs^+ (Shaw 1993). These fertilizers can reduce the radiocaesium transfer about 40–60 %, being more successful the lower K^+ available in soil (Jacob et al. 2009; Rosén et al. 2011). The reduction effects are lasting and observable during long periods of time, 10–34 years after fertilization (Kaunisto et al. 2002; Robison et al. 2009; Rosén et al. 2011). The addition of fertilizers that supply NH_4^+ can also modify the content of ^{137}Cs in the soil solution, but in the opposite way. It increased the transfer by a factor of 3–4, due to its competition with Cs^+ for exchange sites (Nisbet et al. 1993). The application of NH_4^+ and manure can also reduce the uptake of ^{137}Cs, probably due to the release of potassium and other ions from the manure when NH_4^+ is applied (Fuhrmann et al. 2003).

5 Conclusion

The knowledge of the factors affecting the radiocaesium soil-to-plant transfer is one of the key factors to manage the consequences of nuclear accidents and the health hazard involved in plant consumption, either directly or indirectly via animal foodstuff. Transfer factors defined as the ratio between what is in the plant and what is in the soil are the most usual way to quantify the transfer processes. However, their uses have some difficulties. The range of variability for transfer factors for the same plant groups is great, about 4–5 orders of magnitude, which limits their applicability. The definition of soil compartment is also not perfectly defined. In the case of agricultural crops, some international standards were proposed to limit this variability, mainly the depth of soil considered. But in the case of natural or semi-natural environments, such as forest, it is not so well established. There are also some factors responsible for the variation of the radiocaesium transfer factors. Climate contributes to it, but indirectly, as it conditions the type of plants able to grow and the type and characteristics of soil. The latter is the main factor affecting the radiocaesium transfer. Soil clay content is a major factor, as caesium can be adsorbed irreversibly on them and, therefore unable to be transferred to plant. Organic matter present in soil and its mineralization potential can modify this adsorption, making radiocaesium more available to transfer. The bioavailability concept is linked to the transfer process as the fraction of the total radiocaesium that can be effectively be transferred to plant. The methods for its evaluation are based on sequential extraction techniques, and it can be classified into readily available (water soluble and exchangeable fractions) and potentially available (extractable with diluted inorganic acids). The bioavailable fraction can be modified by the addition of fertilizers and soil amendments. The transfer factor is reduced when potassium based fertilizer or amendments are added to the soil, so that there is more potassium bioavailable. When it surpasses a concentration threshold, potassium is taken preferentially to caesium up. The addition of NH_4^+ has the opposite effect, increasing the transfer.

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