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Ecology of Soils Polluted with Radio Nuclides: A Review

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ABSTRACT

The purpose of this study was the review of radio nuclides transfers to plant/soil by mycorrhizas, soil animals and soil microbes. Radio iodine 125 is as a surrogate of I-129 which has complex ecosystems because they can involve diverse reaction with soil type, plant species and vegetative. High variability is observed in radio iodine migration and redistribution patterns in contaminated soils. In the long term, the soil compartment represents the major reservoir of radio iodine which can give rise to long-term plant and hence food contamination. The contamination of various specific plant products has commonly been quantified using the transfer factor which integrates various environmental parameters including soil and plant type, root distribution as well as nature and vertical distribution of the deposits. Long lasting availability of some radio iodine was to be source of much transfer in plant/soil ecosystems in soils. Transport of radioiodine to lower layers proceeds very slowly. The experiment indicated that I 125 was mobile only within the saturated/low redox zone at the base of the soil column and accumulated in the zone of transition between anoxic and oxic soil conditions. Maximum contamination of tree components with radio iodine was associated with hydro morphic areas with thick humus layers. The rate of contaminant was being determined by the dynamics of radio iodine in aquatic ecosystems. A model had been developed by applying the selection of plant varieties used for animal feeding, the plants' growing cycles and harvests, the animal feeding practices and the human consumption rates for iodine radioactivity in soil, plants and animal products.

Key words: Radio nuclides, radio iodine, transfer factor, microbes, cycling, soil type

INTRODUCTION

The behavior of radio isotopes in the soil/plant system, especially radio- iodine would be necessary researches. An understanding of this behavior is essential if we are to be able to predict the food chain and ultimately, the human health impact of radio iodine, accidental or otherwise. Research on the effect of the radio iodine requires an interdisciplinary approach involving an understanding of both the physicochemical and biological properties of the soil environment (Hosseini and Jayapalan, 2007).

Dynamics of radio iodine in the soil: The movement of pollutant of radio iodine in soil will depend both on isotopes involved and their concentration as well as the physic-chemical nature of the recipient soil (Liu and van Gunten, 1988). Of particular concern of radio nuclides were I-125 as a surrogate for I-129 which is a beta and gamma radiation emitter with a half life of about 60 days. I-29 is also a beta emitter with a half-life of about 1000000 years. Radio- iodine is products of nuclear fission reactions and is therefore regularly released into the environment as a result of weapons testing, nuclear power production and nuclear fuel reprocessing (Hou *et al*., 2009). In April 1986, however, a massive release of these and other radio nuclides occurred as a result of reactor accident at chernobyl data obtained from the regions near chernobyl showed no increased risk of other types of malignancy (leukaemia, Hodgkin's and non Hodgkin's lymphoma) in 1986-1996 (Ma *et al*., 2008; Cardis *et al*., 2005). The efficiency through which iodine is transferred through this pathway is important in ascertaining the risk of radioiodine exposures in the general human population from continuous or accidental releases of 131-I and 129-I, especially in children Some regions of European country such as England received total radio nuclides with levels of radio iodine being particularly high (Zonenberg *et al*., 2006).

In terrestrial plants, iodine can be taken up through the roots, mainly as iodide and to a lesser extent, as iodate or iodine. The average iodine concentration in terrestrial plants is 0.42 $\mu g\ g^{-1}$. The uptake is dependent on soil conditions and the use of fertilizers distribution of iodine and iodide varies throughout the plant (Coughtrey and Thorne, 1983). The uptake of iodine into terrestrial plants in combination with deposition of iodine onto the surfaces of plants plays an important role in the transfer of iodine through the soil-plant-cow-milk pathway (Bowen and Cawse, 1964).

Radio-iodine, a monovalent, alkali metal cation, absorbs some kind of halogen such as chlorides ions. Movement of radio iodine through the soil will not largely depend on the cation exchange capacity, the base composition/saturation of station exchange site and the charge and hydrated ion radius of radionuclide. Soil thermal energy can affect radio iodine adsorption in the soil (Ashworth and Shaw, 2006a). Knowledge of physical property as temperature variation effect on radio-iodine adsorption of soils is particularly essential for estimating iodide group, especially I-129 transferto fluvial systems and for successfully measuring radio-iodine enthalpy and entropy in soil studies (Hosseini and Qureshi, 2010; Seki *et al*., 1988).

A large proportion (-80%) of the iodine released through the flue gases is deposited onto the surrounding soils by both wet and dry deposition, adding approximately 4×105 kg iodine $year^{-1}$ to soils globally. When the various chemical forms of iodine enter into the soil, these species are initially retained by their absorbance onto soil components in equilibrium with their solubility in soil solution. Mobility of iodine into the soil is greatest when the soil is saturated with water. The drier the soil, the thinner the water films within the soil, thus limiting the flow rate of water through the soil. In addition to the aforementioned direct deposition of particulate deposition of iodine onto plant surfaces, there is also evidence of the uptake of inorganic iodine into the plant through the roots. Iodide is more readily taken up into plant roots than is iodate or Iodine; the uptake is dependent on the concentration of iodine in the soil, the properties of the soil and the use of fertilizers. The use of fertilizers, however, will not result in an appreciable increase in the iodine content of soil. Yet, the use of iodine-containing herbicides, such as ioxynil/ioxynil octanoate (recommended application of 0.5 kg ha^{-1}) and the fungicide benodanil (recommended application of 1.1 kg ha^{-1}), can increase iodine content of soil about 0.17 and 0.21 $\mu g\ g^{-1}$ to a depth of 15 cm, respectively (Hosseini and Bagheri, 2008).

The experiment indicated that I 125 was mobile only within the saturated/low redox zone at the base of the soil column and accumulated in the zone of transition between anoxic and oxic soil conditions (Ashworth *et al*., 2003). The soils, with CEC'S and generally based saturation, will not tend to favor exchange between protons and radio nuclides. Field survey of organic soil show that

I-125 which should be less strongly adsorbed to soil particles in comparison to radio caesium which remain quite tightly held in the surface horizons with a half value depth, the depth above which more than half of the isotope remains, after nearly 6 months of about 3 cm. Inputs of some competing cations will tend to cause mobilization of the radio nuclides and render them available for biological uptake and cycling. Inputs of ammonium from animals waste and input calcium from liming will tend to desorbs radio nuclides from exchange sites. Both these forms of inputs of cations commonly result from farming activities and will have a major dynamic bearing on radio nuclides of soils under either pastoral and arable farming. Of course, arable farming may have greater effect on radio nuclides movement (Hosseini, 2008).

Soils rich in iron oxide had high affinity for iodate. No correlation between iodate adsorption and cation exchange capacity and soil pH was found (Dai *et al.*, 2004). In peat soil where CEC is dominantly of organic origin, radio iodine will be held as insoluble organic as well as simply by charge (Seki *et al.*, 1988). In more mineral soils where CEC is clay based, radio nuclides will be held by charge only. Desertion of radio iodine in sub-sample soil of humus is more than coarse sand clay and clay soil. Adsorption of radio iodine in clay soil as case is measured at larger amounts than in coarse sand - clay soil (Hosseini, 2007). This is because they become trapped within the clay lattice preventing subs equating ion exchange with soil solution. This type of interlayer fixation will be particularly strong in illicit clay minerals, micas and biotitic with vermiculite and montmorilloites being less effective.

Prediction of the fate of radio nuclides in soil is essential ultimately to predict the behavior of radio nuclides at the foot chain and ecosystem level. Most modeling of radio nuclides dynamics in soil relies on a weak empirical approach, based on measurement of solid-liquid distribution coefficient (K_d values) and transfer factors between soil and vegetation. Soil-to-plant Transfer Factors (TF) which are defined as the grams iodine per kilogram wet or dry weight of plant material divided by the grams of iodine per kilograms dry soil, typically range between 0.001(Hosseini *et al.*, 2009). Values of these coefficient obtained under laboratory conditions, however, have little or no field value and progress must be made toward obtaining in situ values for these coefficients based on predictions from soil properties that are much more readily and reliably measured in the laboratory (Anderson and Domsch, 1988).

CYCLING OF RADIO NUCLIDES BY THE SOIL BIOTA

Radio iodine is bioactive and is taking up by components of soil biota, particularly if they are limits by these nutrients. Uptake will tend to be greater in peaty rather than mineral soils particularly because of the greater exposure of the exchange sites in the peat soils. Uptake will also tend to decrease as the clay content of the soil increases, as the capacity for clay fixation of radio nuclides increases (Hosseini and Qureshi, 2010). Comparing the concentrations of I-125 and I-131in blanket weed which is a food source of the swan and river water samples collected from above, below and in such effluents, showed that radio-iodine enters the river via these routes (Howe and Hunt, 1984). Uptake of I-125 by the ryegrass was found to be low (Ashworth *et al.*, 2003). The moorland vegetation on the widespread acid, peaty soils responds well to potassium. This would therefore suggest that radio nuclides that are analogous of potassium will be readily taken up by moorland species such as heather (Martin *et al.*, 1988).

Even vegetation types that are not so responsive to potassium will still require it as a macronutrient and so will take up significant levels of radio nuclides analogues. Transfer factor of radio iodine was found in some grass of unimproved pastures. it was shown that such interactions were highly dependent on I-concentrations added to soil suspensions, contact time with the soil and organic carbon content, resulting in an empirical particle-water partition coefficient (K_d) that was an inverse power function of the added I-concentration (Schwehr *et al.*, 2009).

As with plant uptake of soil nutrients, uptake and translocation and hence release in to above-ground food chains of radio nuclide is essential, being greatest in the spring and early summer. Of course, uptake is also dependent upon plant species as well as on a whole variety of other plant variables, particularly the stage of development of plant (Dai *et al.*, 2004). Because of the high biological activity of radio iodine, plant roots will almost certainly provide a major pathway for radio nuclides movement in the soil profile. Uptake, translocation and then root decomposition/turnover will ensure a ready cycling of radio nuclides within rooting depth of most soils. As the rooting depth migrate through soil during the growing season, so too wills the zone of cycling of the radio nuclides. Dense root systems such as under grassland vegetation will tend to be particularly effective in meditating this radio nuclides cycling data obtained for radioactive iodine (a key radionuclide in the consideration of radioactive waste disposal) are presented and indicate that soil moisture content, particularly in conjunction with soil redox potential (through water-logging of the soil), has a marked effect on measured K_d values (Ashworth and Shaw, 2006b).

The period over which radio iodine cycling may be maintained in the soil/plant system will depend on a number of factors but particularly on soil type. Soils which facilitate removal of the radio nuclide from exchange sites more readily than many clay soils, may have active cycling for many years after initial deposition (Muramatsu *et al.*, 1993).

The cycling of radio nuclides by plants will, to some extent, be a function of the climate conditions under which the plants are growing. Plants at a higher altitude will tend to take up greater amounts and this is probably more related to the greater rainfall experienced by the plants than any differences in plant physiology (Ashworth and Shaw, 2006a).

Radio nuclides in herbage grazed by higher animals will partly be retained by the animal but the bulk will be returned to the soil in animal wastes. The rapid break down of these wastes will further encourage the ready cycling of the radio nuclides in the soil/plant system (Hosseini and Bagheri, 2008).

Because of the high rates of radio iodine uptake by some grasses, the growth and removal of above ground parts of such grasses may have potential as part of a further strategy to rehabilitant contaminated soil after radio nuclide deposition (Hosseini, 2008).

Mycorrhizas: It was proved for all analyzed soils that the biodegradation activity of the total biomass was caused by only a small part of the present microbial strains. Not only bacteria but also fungi and yeast have to be taken into consideration in a pesticide biodegradation (Tykva, 1998). Mycorrhizal fungal infection, particularly of the VAM type can stimulate plant metal uptake in soils where phosphorous and the metals in question are sparingly available. Infection by VAM can enhance plant uptake of vast range of trace and minor elements (Killham, 1985) as well as calcium (Tinker, 1978) and potassium (Bowen, 1984). There is very strong circumstantial evidence; therefore, that VAM infection will enhance plant uptake and cycling of radio nuclides if they are deposited on an ecosystem as a result of accidental release. On this bases, the high phosphate statues of fertilized arable soils may well minimize radio nuclide uptake by crops from contaminated soil because high phosphate levels inhibit mycorrhizal development (Killham, 1985).

Soil microbes: Free living microbes in the soil may be important agents of both movement and retention of radio nuclides. Microbial cells will immobilize the radio nuclides by both uptake and simple surface adsorption. Uptake of radio iodine may be particularly marked if the cells are potassium limited (Behrens, 1986). Surface adsorption will reflect the surface chemistry of the cells. Microbial cells will also release immobilized radio nuclides back in to the soil through action exchange of surfaced adsorb radio nuclide and by osmoregulatory efflux of those in the cell cytoplasm, as well as by the mineralization reaction of organically complex nuclides, on death of the cell. Microbes will also be involved in mobilization of radio nuclides in the soil itself through the breakdown of insoluble organic chalets holding the radio nuclides (Tsukada *et al.*, 2008).

As with plants, the potential for uptake and cycling of radio nuclides by soil microbes and by animals, will vary greatly from one organism to another. Free-living saprophytic fungi are strong accumulators of radio nuclides. Presumably, this accumulation is the result of mobilization of the radio nuclides during decomposition of organic matter and subsequent uptake is analogous to that of mineralized potassium. Radio nuclide accumulate ion in the mycelium and fruiting bodies of radio iodine has important ecological ramification involvements in term of food chain transport, because of the fundamental involvement of the ectomyrrhizedal. Basidiomycotina in the nutrition of their tree host and also because the fruiting bodies represent a major food source for a number of soil animals and microbes as well as above ground animals such as deer.

Fungi in association with blue green algae in the lichen symbiosis, are strong accumulators of radio nuclides such as radio iodine on the soil surface and, as with fungal fruiting bodies, food chain transport of radio nuclides cycling results because lichen are an important source of food for above-ground animals particularly deer (Hosseini, 2010).

Soil animals: The contribution of the soil fauna is a key aspect of radio nuclide cycling in his environment. The presence of soil animals can double the loss of radio iodine (Bors and Martens, 1992) from contaminated litter, even though the fauna are expending only 1% of the total decompose energy budget of the soil (Scheu and Rillig 1966; Calmon *et al.*, 2009). The presence of soil animals can also markedly change the time sequence of mineralization/ immobilization of radio nuclides from contaminated plant residues, particularly through the temporary immobilization (Malone and Reickle, 1973).

Probably the main role of soil animals, particularly earthworms, in term of radio nuclides dynamics is through their major contribution to the vertical and lateral mixing of soil. Because soil animals such as springtail graze fungal hyphen, particularly hose associated with mycorrhizas redistribution of grazed material with potentially high concentration of radio nuclides must play an important role in the cycling of those pollutant. Soil animals are also prey to many higher animals and so have a key function in the food chain transport of the radio nuclides (Slavik *et al.*, 2001).

Modeling of radionuclide movement in the soil/plant system: The soil solid-liquid distribution coefficient (K_d) value is of great significance in understanding and modelling the environmental behaviour of soil contaminants. Research in the movement and cycling of pollutant radio nuclides in the soil/plant system will eventually lead to the construction of reliable models to enable effective production of ecosystem fate of radio nuclides. A simple predicting mechanistic model predicting the fate of radio iodine in soil has been developed by Handl *et al.* (1990) and Tournassat *et al.* (2005) and intended to expose area where we lack the fundamental knowledge of radio iodine dynamics, rather than predict movement in real situation, largely because of the current scarcity of available data. Measurements of iodine radioactivity in soil, plants and animal products are compared with the results of the adapted model. Data from various locations with dry, mixed or predominantly wet deposition show in general good agreement.

CONCLUSIONS

Much of the cycling of radio nuclides in ecosystem is centered in the soil where plant root s microbes and animals are actively involved in the bio-mobilization and Tran's action of radionuclides. The cycling of radio nuclides in the soil/plant system must be understood to assess fully the impact of pollutant radio nuclides on human health.

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