

The European Programme BORIS: Involvement of Biological Components in the Transfer of Radionuclides to Plants

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Abstract. The ability to predict the consequences of an accidental release of radioactive nuclides relies mainly on the level of understanding of the mechanisms involved in radioactive nuclides interactions with different components of agricultural and natural ecosystems and their formalisation into predictive models. Numerous studies and databases about contaminated agricultural and natural areas have been obtained but their use to enhance our prediction ability has been largely limited by their unresolved variability. Such variability seems to stem from an incomplete knowledge about radioactive nuclide interactions with the soil matrix, soil moisture, biological elements in the soil and additional pollutants, which may be found in such soils.

In this project, we investigated mainly the role of the biological elements (plants, mycorrhiza, microbes) in: radioactive nuclide sorption/desorption in soils and radioactive nuclide uptake/release by plants. Because of the importance of the chemical nature of the involved radioactive nuclides, we followed the bioavailability of three radioactive nuclides: caesium, strontium, and technetium. The role of one additional non-radioactive pollutant (copper) has been scrutinised.

Role of microorganisms (K_d for caesium and strontium in organic soils is much greater in the presence of microorganisms than in their absence), plant physiology (changes in plant physiology affect radionuclide uptake by plants), the presence of mycorrhizal fungi (interferes with the uptake of radionuclides by plants), have been demonstrated.

Knowledge acquired from these experiments has been incorporated into two mechanistic models CHEMFAST (a soil Column Heuristic Model of radionuclide Fixation and Solution Transport) and BIORUR specifically modelling radioactive nuclide sorption/desorption from soil matrices and radioactive nuclide uptake by/release from plants. These mechanistic models will be incorporated into an assessment model to enhance its prediction ability.

1. Introduction

Numerous studies and databases about contaminated agricultural and natural areas have been obtained from previous contamination events including nuclear accidents. The use of these previous data to enhance our prediction ability has been largely limited by their unresolved variability. Such variability seems to proceed from a lack of knowledge about radionuclide interactions with the soil matrix, soil moisture, biological elements in the soil and additional pollutants, which may be found in such soils [1].

The first objective of the BORIS programme is to improve our understanding of the mechanisms governing the transfer of radionuclides to plants. To do so, we apprehended the role of abiotic components: soil elements and structure and of biotic components: soil microorganisms, plants, mycorrhiza, and the association of these three biological components.

The second objective of the BORIS programme is to improve existing predictive models of radionuclide interaction with soils by incorporating the knowledge acquired from the experimental results. This will be a three-step procedure: (1) specifying a conceptual and compartment model based initially on the existing knowledge and on previously developed models; (2) developing mechanistic models of the processes occurring between the soil solid phase (mineral, organic, microbial) and the soil solution, and of the processes occurring between the plant system (including eventual mycorrhiza) and the soil solution, and validating these mechanistic models with the experimental results; and (3) integrating the results obtained into an improved compartment model with reduced uncertainties of predictions compared to those of previous models.

Because of the size limit of this paper, only one major result will be presented for each part of this programme.

2. Role of abiotic components

For the sake of homogeneity defined as a major factor of experimental works by the modellers, a considerable effort was put into identifying, locating and sampling a suitable agricultural soil for use in both the full scale column experiments and in the associated laboratory experiments. It was decided to use a soil located in England because the consortium member with the heaviest need of soil was the Imperial College. In consultation with other members of the consortium a silty loam (Batcombe series) from southern England was selected by the Imperial College. The results of its analysis are indicated in the following table 1:

Table 1. Characteristics of the Batcombe soil

Gravimetric analysis		Soil mineralogy		Major components		Major ions with HF method	
< 2µm	211	Quartz	64%	pH water	6,5	Fe (g/kg)	16
		Plagioclase,		CaCO ₃ (g/kg)	<1	Al (g/kg)	26,8
2-20 µm	230	Microcline trace				Cu (mg/kg)	16,3
		Clay fraction		P ₂ O ₅ (Olsen)	0,062	Ni (mg/kg)	16,7
20-50 µm	226	Kaolinite	77%	g/kg		Mn (mg/kg)	1112
50-200 µm	237	Illite/mica	18%	Organic C g/kg	32,19	Si total (g/kg)	333
200-2000 µm	95	Chlorite	5%	Organic N g/kg	2,87		

2.1. Specific role of soil components on the radionuclide bioavailability

We have studied the adsorption and desorption of Cs and Sr on synthetic soils consisting of individual solid reference materials, or simple binary or ternary mixtures of model soil components. Because of the very small adsorption of Tc on soils and soil components, it was not included in this study. Five reference minerals that are typical soil constituents were selected for this study: Wyoming montmorillonite, Le Puy illite, St Austell kaolinite, Calcium carbonate, Quartz. After purification and size separation, their adsorption properties were studied in dilute suspension, with and without organic and or mineral coatings. The organic coatings were obtained by contact with three soil extracted humic substances (fulvic acid, humic acid and a neutral organic fraction) or a protein (bovine serum albumin, BSA) or citrate, a typical exudate of root and associated micro-organisms, often accumulated in the rhizosphere.

Adsorption and desorption measurements were carried out in dilute suspension of the synthetic soils (1 g dm⁻³ for the clays, 10 g dm⁻³ for quartz and calcium carbonate).

No adsorption of ⁸⁵Sr on quartz was detected, adsorption on calcium carbonate was small (Kd~25 dm³ kg⁻¹) and adsorption on the reference clays was larger and varied little between clays (Kd~1000 dm³ kg⁻¹). There was no significant effect of either inorganic or organic coatings on adsorption. The desorption Kd was 4-5 times greater than the adsorption Kd, indicating some degree of reversibility.

No adsorption of ¹³⁷Cs on quartz was detected, and the adsorption on calcium carbonate was small (Kd~25 dm³ kg⁻¹). In agreement with other published data, adsorption was greatest on illite (Kd~1.5x10⁵ dm³ kg⁻¹). Surprisingly, there was relatively little difference between the montmorillonite and kaolinite (Kd~4000 dm³ kg⁻¹) despite the smaller cation exchange capacity and expected lesser selectivity of Cs over Ca. Neither inorganic nor organic coatings had no effect on the adsorption of Cs on calcium carbonate. In general, organic coatings decreased Cs adsorption. The extent of the effect decreased in the order illite>montmorillonite>kaolinite for the clays and in the order Humic substance>BSA>citrate for the nature of the coating. In contrast, inorganic coatings had no effect on Cs adsorption on either illite or kaolinite, but caused a 2-fold decrease for montmorillonite. Mixed organic + inorganic coatings had more complicated effects. In all cases the Cs Kd measured by desorption was greater than for adsorption, suggesting a strong irreversibility. However the trends in Kd (adsorption) and Kd (desorption) were similar, suggesting that the extent of fixation was unaffected by the coatings. It is implied that there are two forms of adsorbed Cs, one form that is fully exchangeable, that re-equilibrates with solution rapidly upon dilution, and the other that is completely irreversibly adsorbed, and so the concentration remains the same during the adsorption and desorption steps.

In conclusion, the adsorption properties of Sr on mineral soil components are simple, and not influenced by inorganic or organic coatings. Reference clay minerals are therefore good models to predict and explain the mobility of Sr in soil systems. In contrast, the adsorption of Cs depends in a complex manner on inorganic and inorganic coatings. Care must be taken in extrapolating adsorption data obtained using reference minerals to soils.

2.2. Role of soil structure on radionuclide bioavailability.

The selected Batcombe soil sampled at Swan Bottom, UK was characterized (See Table 1). Cs, Tc and Sr incubation experiments were performed in different physico-chemical conditions. Results are expressed in figures 1. It shows the strong affinity of Cs for the solid phase, the low affinity of Tc and its dependence upon the presence of microorganisms, and the quasi-absence of effects of changing conditions on Sr affinity for soil matrix.

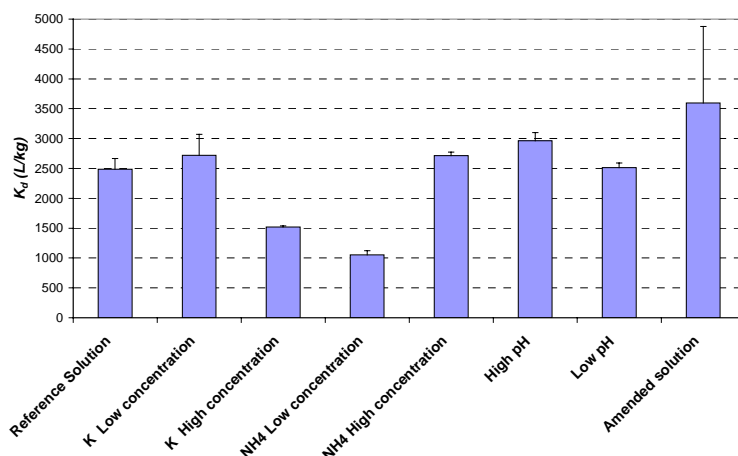


FIG.1a. Distribution coefficient of Cs in different physico-chemical conditions.

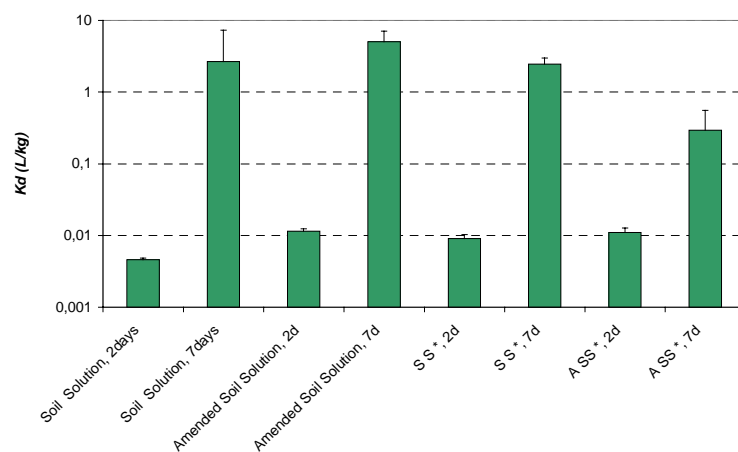


FIG.1b. Distribution coefficient of Tc in different physico-chemical conditions. Experimental conditions: Soil solution (SS); Amended soil solution (ASS); γ -sterilization (*).

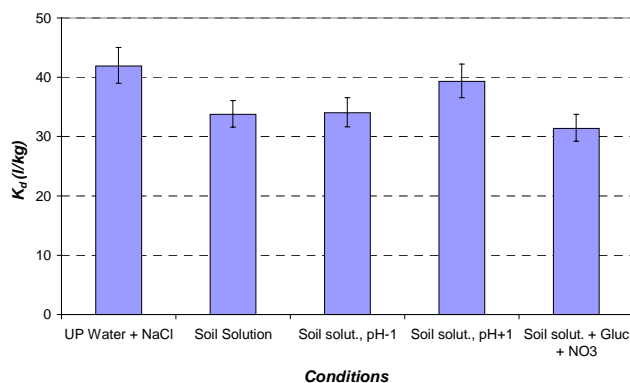


FIG.1c. Distribution coefficient of Sr in different physico-chemical conditions (Contact time = 48 hours; $[Sr_{Total}]_{Initial} = 10^{-6} M$).

3. Role of biotic components

3.1 Role of soil microorganisms in radionuclide bioavailability in soils

3.1.1 Preparation of a mineral-free organic matter enriched in microorganisms

Deciduous leaves were collected, were washed, air dried and shredded. The mixed leaf litter was decomposed over a 6-month period, then producing a mineral free organic matter. To ensure that the starting organic material would be similar in all experiments, the mulched litter was stored at -20°C in 500 g aliquots. Sufficient amounts of the material are then taken out from the freezer and thawed for 24 h prior to use in the experiments. A preliminary experiment to determine the most effective method of soil sterilisation was done and autoclaving was observed to be the method of choice. In order to have the same underlying soil matrix, for direct comparison of biotic and abiotic components, two sets of autoclaved material will be used. One sample will then be re-inoculated with a soil extract solution (in sterile distilled water - SDW) and the other will be amended with an equal quantity of SDW.

3.1.2. Effects of added clay minerals on uptake of Cs and S.

To check the role of microbes on radionuclide behaviour [2], experimental protocols were developed to determine the effect of added clay minerals (montmorillonite and illite) in litter on the uptake of Sr-85 and Cs-137. The distribution ratio, K_d , for ^{85}Sr and ^{137}Cs (calculated as the ratio of the activity concentration per g of dry solid to the activity concentration per g of leachate) in abiotic versus biotic microcosms is shown in Figure 2.

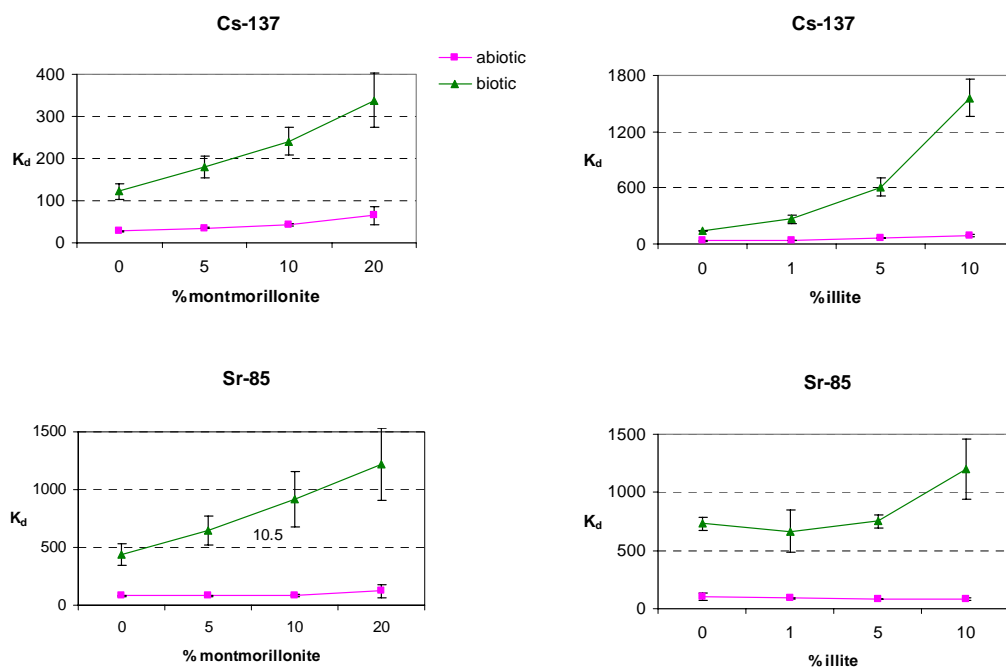


FIG.2. Calculated distribution ratio for ^{137}Cs and ^{85}Sr in biotic and abiotic microcosms following 14 days incubation, for different amendments of montmorillonite and illite to an organic matrix.

Following the 14-day incubation period, bioavailability measurements were performed on the spiked materials from each microcosm. This was done by extraction using, in sequence, deionised water, NH_4Cl and CaCl_2 to determine the fraction of bound ^{137}C and ^{85}Sr that can be readily displaced by these reagents. A 'dialysis bag' technique was used: known weights of the spiked material from each microcosm were packed into dialysis bags and extracted for 24 hrs (end-over-end shaking), using a ratio of 1 g dry equivalent of material to 30 g of each extractant solution. The percentage of ^{85}Sr and ^{137}Cs extracted was determined from the measured activity concentrations in each extractant solution and the initial activity of each nuclide added into the dialysis bags. The 'bioavailable' fractions of ^{137}Cs and ^{85}Sr followed similar trends in montmorillonite and illite amended microcosms, results for illite microcosms are shown in Figure 3.

3.1.3. Conclusions

The following observations were noted from the results:

- ✓ K_d biotic $>$ K_d abiotic suggesting the possible role of microbes in the sorption of both radionuclides

- ✓ High degree of variability observed in the biotic K_d values
- ✓ There are slight differences in the extraction behaviour of both nuclides in abiotic vs. biotic systems:
 - readily exchangeable fraction (H_2O): $^{137}Cs > ^{85}Sr$
 - the major fraction of both nuclides was extracted by NH_4Cl
- ✓ 95% of both nuclides were in the extractable fraction (sum of $H_2O + NH_4Cl + CaCl_2$ extracts)
- ✓ K_d for ^{137}Cs increased with increasing amounts of clay, this was more pronounced in the biotic system.
- ✓ K_d for ^{137}Cs was higher in the presence of illite than montmorillonite
- ✓ K_d for ^{85}Sr was similar in presence of both types of clay

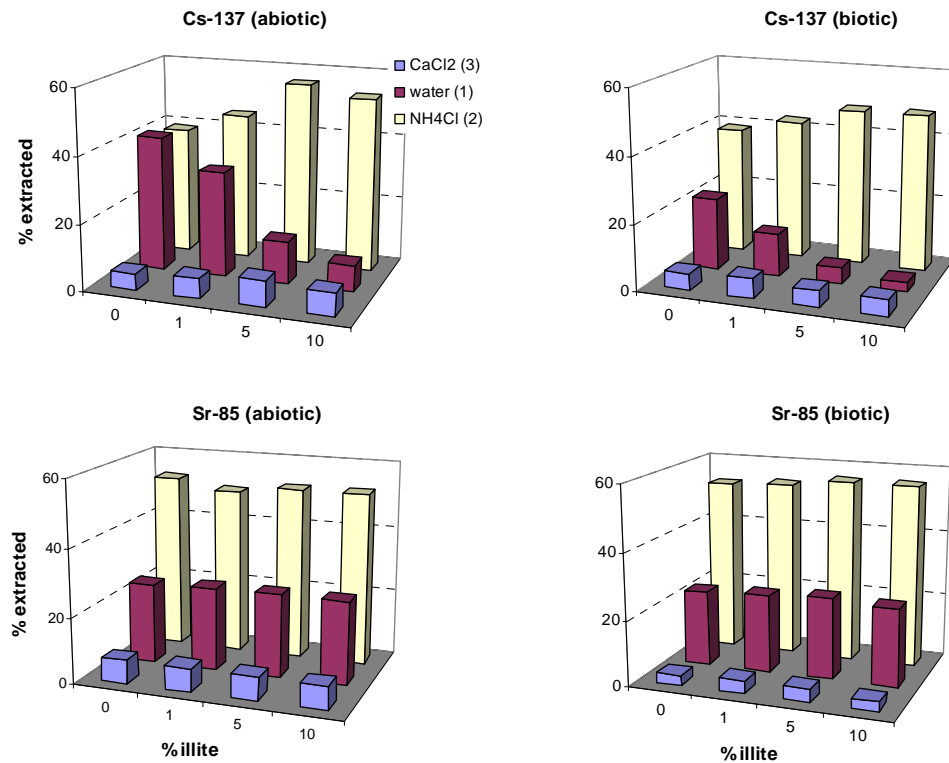


FIG.3. Percentage of ^{137}Cs or ^{85}Sr extracted from the organic matrix in biotic and abiotic microcosms following 14 days incubation with different amendments of illite. Extraction using three stages of extraction solutions; water (1), NH_4Cl (2) and $CaCl_2$ (3).

In order to determine the sterility of the abiotic samples and to assess the microbial populations in the biotic samples, samples were inoculated on specific growth media. The results show that the abiotic samples were sterile at the start of each experiment and remained sterile through the 14 days incubation period. We also confirmed that the biotic samples contained a variety of soil microorganisms throughout the experiment.

3.2. Identifying Plant physiological mechanisms involved in radionuclide transfer to plants.

Soil to plant transfer is not determined only by the concentration of radionuclides in the soil solution [3]. Therefore, a more mechanistic approach should replace the use of empirically obtained transfer factors. It has been established [4] that the K^+ , Ca^{2+} and NO_3^- concentrations in the root medium largely affect the root uptake of Cs^+ , Sr^{2+} and TcO_4^- respectively.

3.2.1. The effect of plant demand (age) and nutrient supply.

In this experiment we studied the effect of plant demand by analysing Cs uptake at two different ages based on the hypothesis that young plants are high nutrient demanding while old plants are low nutrient demanding. We designed a two-factor experiment with the factor plant age (7 days and 25 days old plants) and factor supply (0.2 mM K and 2 mM K).

K uptake was significantly higher in young plants (Figure 4) with respect to old plants. In addition some effect of K starvation appeared after 24 hours in contact with the 0.2 mM K supply medium. Old plants, with low demand

presented lower Cs uptake stimulation though there was still a clear effect of K-starvation on Cs root uptake in agreement with results found in the previous experiment. In plants with high demand (young plants), the higher Cs uptake under 0.2 mM K treatment highlighted the low discrimination between Cs and K when the high-K affinity transport system is dominating.

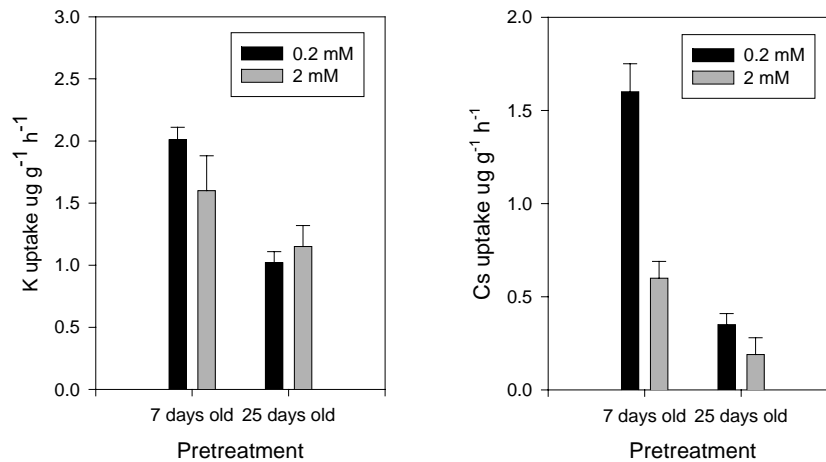


FIG. 4. K and Cs root uptake at two different plant ages (7 and 25 days old plants) after 24 hour growing in two contrasted K supply (0.2 and 2 mM K) solution culture.

3.2.2. The effect of growth stimulation

We have analysed also the effect of plant demand by assaying two growth rates. The high growth rate consisted on a 16 hours day/8 hours night cycle, 26°C temperature and 80 % of relative humidity. The low growth rate consisted on an 8 hours day/16 hours night cycle, 23°C temperature and 80 % relative humidity. In both cases the growth medium contained 2 mM K.

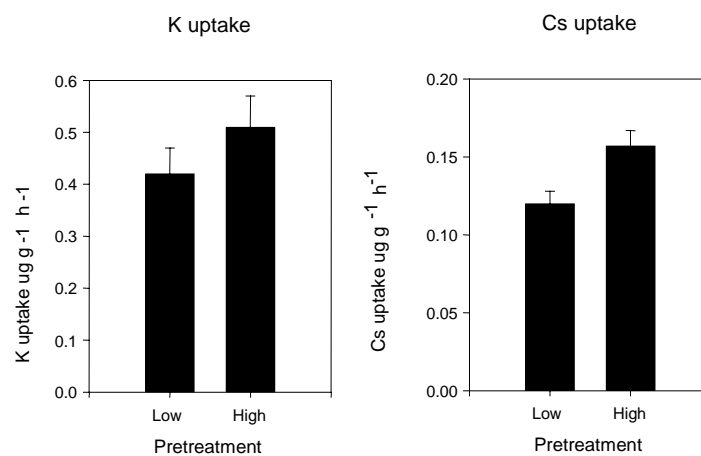


FIG. 5. K and Cs uptake at low and high plant growth rate treatments.

Figure.5. shows that both the uptake of K and Cs increased in plants with higher growth rate probably because plant nutrient requirements for growing was higher under these conditions. Ca and Sr determinations are in progress.

3.2.3. Conclusions

The results obtained in the various experiments evidenced that increasing K demand for plant growth produced higher Cs uptake by sunflower and the effect was magnified at low K supply.

On the contrary (not shown), decreasing plant demand by suppressing the new sprouts significantly reduced plant uptake of both Cs and Sr.

Nutrient starvation also increased the uptake of both Cs and Sr (data not shown) although for Sr the effect is less pronounced. Cs affinity increased under K-starving conditions. After starving, the addition of K to the growth medium rapidly reduced the uptake of Cs showing the inhibitory effect of K addition on Cs uptake.

It has been described elsewhere that plant regulation of Cs uptake is mediated by the expression of high affinity K transport system and the transition between carrier and channel-mediated K transport systems. For Ca and Sr,

the transport regulation is not as well documented as for K although some carrier mediated transport for Ca in conditions of low Ca in the medium has been described. For Sr the main pathway of uptake is through mass flow. In our experiments we have forced the plant to change the transport systems, hence producing contrasted Cs and Sr uptake patterns. According to our results, the uptake of caesium and strontium have been modulated by the plant to keep a target flux of either K or Ca. Therefore we confirm the hypothesis of plant as a key regulator of soil-to-plant transfer of Cs and Sr.

3.3. Role of mycorrhizal fungi in radionuclide bioavailability in soils

The aim of this experimental work is to determine how ectomycorrhizal infection can modify the uptake and translocation of radionuclides [5]. One set of experiments is presented in this paper dealing with the effect of ectomycorrhizal symbiosis on the radionuclide uptake and the role of copper (additional pollutant) on that uptake.

Rhizopogon roseolus was grown in association with *Pinus pinaster*. A first batch of 2-month old plants was set up in rhizotrons containing 4 types of soil: (1) untreated Batcombe soil, (2) Batcombe soil plus radionuclides (^{95m}Tc , ^{137}Cs and ^{85}Sr added respectively at a rate of 0.13, 10 and 2 MBq/kg of dry soil), (3) soil + radionuclides + Cu^{++} 10 mg/kg and (4) soil + radionuclides + Cu^{++} 100 mg/kg. The soil was laid in each side of the rhizotrons, and 6 plants per treatment were set up, giving a total of $7 \times 6 = 42$ mycorrhizal plants and $4 \times 6 = 24$ non-mycorrhizal plants. Plants were watered weakly with a sterile nutritive solution containing 0.5 mM $\text{Ca}(\text{NO}_3)_2$, MgSO_4 1 mM, Sequestrène 0.5 ml/l and thiamine 100 $\mu\text{g/l}$, pH 5.5. Plants were harvested after 4 months of growth. Photography was taken for each plant that was then cut into shoots, stem and roots. The soil was also harvested and dried at 50°C for 5 days and stored before analysis. Plants were weighed before extraction of ergosterol in roots and acid hydrolysis carried out on the whole plant tissues.

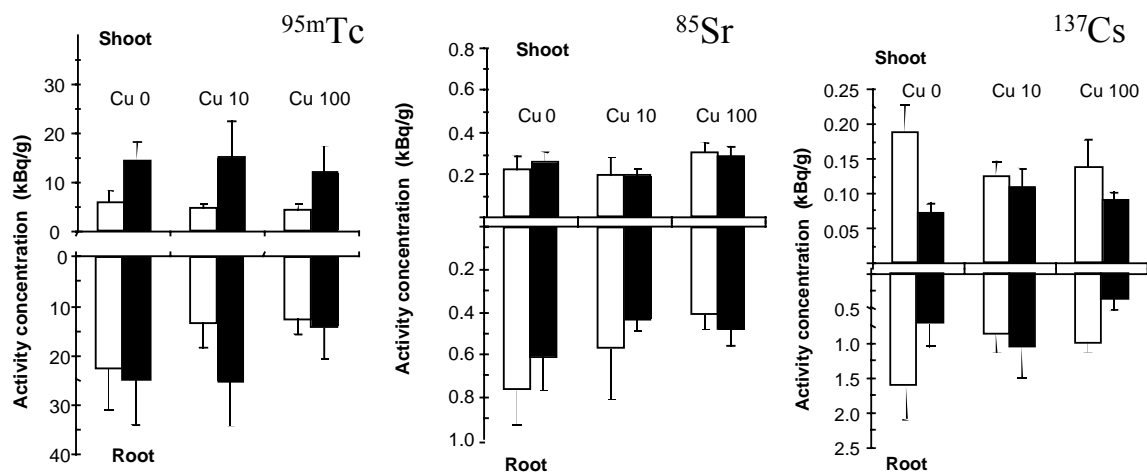


FIG.6. Activity concentration (kBq/g fw) shoots and roots of *Pinus pinaster* plants, whether associated with the ectomycorrhizal fungus (white boxes) *Rhizopogon roseolus* or non mycorrhizal (black boxes).

Two-month old plants were grown for 4 months in rhizotrons containing a thin layer of Batcombe soil not amended (Soil) or amended with ^{95m}Tc , ^{85}Sr and ^{137}Cs (RN) or radionuclides and copper at 10 mg kg^{-1} (RN, Cu 10) or 100 mg kg^{-1} of soil (RN, Cu 100). Data are means \pm standard deviation ($n=6$).

At the time of harvest (after 4 months of growth in rhizotrons), macroscopic examination of fungal development indicated that the colonization of plant compartment by *R. roseolus* hyphae was good. However, despite this good fungal growth, hyphae did not yet cross the nylon mesh and did not colonize the fungal compartment. Therefore, rhizotrons (control and mycorrhizal ones) containing different soil treatments in the plant compartment were harvested only. The remaining rhizotrons containing soil treatments in the fungal compartment were let to grow in the growth cabinet for further time. Shoots or roots fresh weights of control plants were not significantly modified by the addition of copper to the soil (data not shown). Mycorrhizal plants presented always higher fresh weights of biomass than control plants, although the data were significantly different only when the soil was amended with radionuclides and copper at 100 mg kg^{-1} of soil. Radionuclides concentrations in plant tissues were measured in HCl solution. The results (Fig. 6) showed that mycorrhizal effects on radionuclides in plants varied with the radionuclide type and the Cu level. Indeed, the presence of the ectomycorrhizal fungus decreased the accumulation of ^{95m}Tc in the shoots of mycorrhizal plants, did not modify ^{85}Sr accumulation in roots or shoots of mycorrhizal plants and increased greatly ^{137}Cs accumulation in roots and shoots of mycorrhizal plants when the soil did not contain any copper.

3.4. Global role of biological components in radionuclide bioavailability in soils

Such role is scrutinized by implementing soil column experiments to determine *in situ* K_d values for Cs, Sr and Tc in Batcombe soil type (see table 1). A good source of Batcombe series soil was located (at Swann Bottom, Buckinghamshire) and sufficient soil collected to allow experimentation in a soil column system.

3.4.1. Experimental design

The column experiment (see Figure 7) employed soil columns of 50 cm height and 15 cm diameter into which homogeneously contaminated soils were carefully packed to provide initially uniform soil physical, chemical and biological conditions. After contamination with ^{137}Cs , ^{85}Sr and $^{95\text{m}}\text{Tc}$ the selected soils were packed to a constant bulk density of 1.1 g cm^{-3} . Sterilisation of half the columns was achieved by fumigation with CH_3Br , a non-specific sterilant commonly used in the horticulture industry. In each of the columns a water table was then established 5 cm from the base and the column was either planted with the experimental plant species of interest (wheat), or left unplanted.

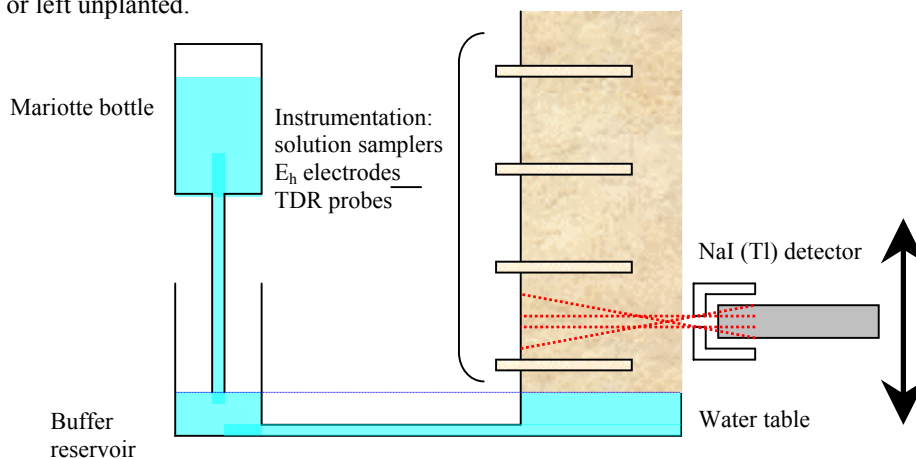


FIG.7. *Experimental column design. In situ instrumentation includes redox potential (E_h) electrodes, hollow fibre soil solution samplers and time domain reflectometer (TDR) probes for soil moisture determinations.*

Soil solution samples were collected on nine occasions within this six-month period, using hollow fibre samplers, which had been tested for their suitability in the first few months of the BORIS project. Soil solutions were analysed for radionuclide activity concentrations using a well-type Na(Tl)I detector and stable element chemistry (cations and anions) using ICP-Atomic Emission Spectrometry. TDR and redox measurements were taken at approximately fortnightly intervals using *in situ* probes located as shown in Figure 7. *In situ gamma* measurements were taken at two monthly intervals.

Summary of Results of Column Experiment 1

The results of primary importance are the *in situ* K_d values for ^{137}Cs , ^{85}Sr and $^{95\text{m}}\text{Tc}$. Ranges of K_d values for one of the eight experimental columns (Column 1) were 0.69 – 171 for $^{95\text{m}}\text{Tc}$, 1.96 – 238 for ^{85}Sr and 39.7 – 9431 for ^{137}Cs .

These *in situ* K_d values are within the expected ranges for each of these radionuclides. However, the variability within these ranges, as shown in Figure 8, was completely systematic, with a monotonic increase in K_d from lowest to highest values from the dry surface of the column to the saturated region below the water table at 45 – 50 cm depth. This reflects the observation that soil solution ion concentrations of both radionuclides and stable elements declined monotonically from surface to the saturated deeper part of the soil columns. Figure 3 shows results of stable strontium determinations in soil solution at several sampling intervals during experiment 1.

Other factors, which showed a systematic variation with soil depth, were soil moisture content (increased from top to bottom), soil redox potential (decreased from top to bottom) and soil pH (increased from top to bottom). Each of the eight columns has yielded 40 to 50 estimates of *in situ* K_d for each radionuclide and this, together with the supporting data concerning stable element concentrations and soil physico-chemical parameters, provides a substantial database, which will be used to determine the key parameters controlling *in situ* K_d for Cs, Sr and Tc.

In conclusion, solution concentrations of major ions and radionuclides show a decrease from maximum values in the dry soil at the column surfaces to the saturated lower column regions. This is mirrored by a monotonic

increase in *in situ* K_d values of radionuclides from minima in dry surface soil to saturated soil at and below the water table.

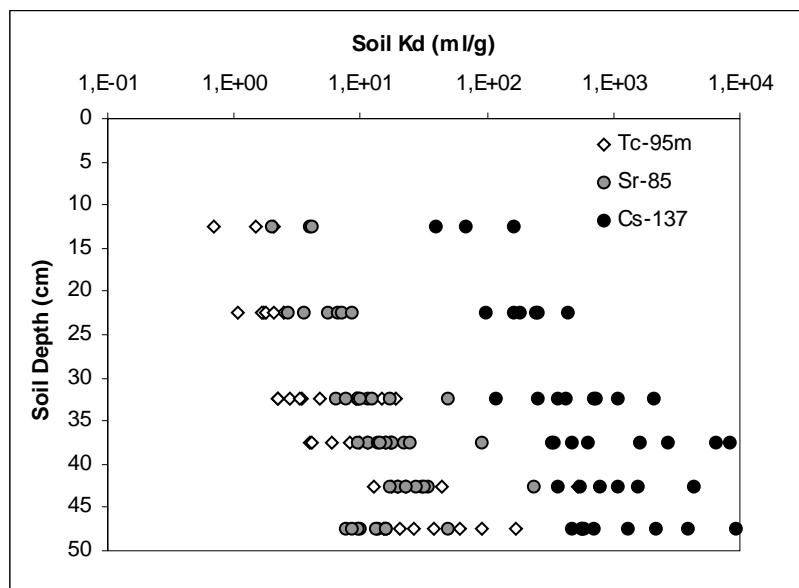


FIG.8. *In situ* solid-liquid K_d values determined for one of the experimental columns over a six-month period.

4. Improving predictive models

The usual approach for evaluating radioactive nuclide uptake by plants from soil is by using concentration ratios, CR, which is defined as the ratio of radioactive nuclide concentration in plants (Bq/kg dry or wet weight) to that in soil at a specified depth (Bq/kg dry or wet weight). This is the approach recommended by the IAEA to use in models of radioactive nuclide uptake by agricultural plants. A review shows that the radiocaesium CR for agricultural plants can vary by a factor of up to 25 even for plants grown on the same site. Different types of soils lead to a variation by a factor of 50. A variation from two to three orders of magnitude of the CR from soil to forest plants has been reported. Such high variability puts constraints upon the applicability of CR for assessments of radioactive nuclide transfer from soil to plants. In this project we explore a possible solution to this problem by describing the bioavailability of radioactive nuclides in soils in mechanistic terms [6,7].

4.1. Soil/soil solution partition of radionuclides CHEMFAST biogeochemical model

The CHEMFAST model is a Column Heuristic Model of radionuclide Fixation and Solution Transport. Heuristic models are based on progressive learning of system behaviour – in the BORIS programme learning is based on experiments.

The CHEMFAST model addresses vertical transport of radionuclides in the solution phase by solving the advection dispersion using an implicit finite difference technique. This has been found to reproduce the migration behaviour of non-sorbed radioactive solutes such as $^{36}\text{Cl}^-$ and $^3\text{H}\text{H}\text{O}$. Sorption and desorption between the solid and liquid phases are represented as kinetic processes which are first order with respect to time but which vary in a non-linear manner with respect to solution concentration.

Concentration-dependent sorption-desorption of radionuclides is based on the Langmuir formulation. The rate of sorption to a surface should be proportional to a driving force times the area of the surface sites available for sorption. The effective driving force is the concentration in the liquid phase.

4.2. Soil solution/plant transfer of radionuclides BIORUR model

The main concept is that radionuclides move in plants and ecosystems through the same pathways as other elements and the activities of these pathways vary between scenarios in a way which can be explained (and predicted) by known rules of nutrient cycling. In this sense, the algorithms of BIORUR do not pretend to describe in mechanistic terms all the involved process because this would be extremely complex. Instead, the algorithms of BIORUR focus at the difference between the behaviour of a radionuclide and that of its nutrient analogue in the most relevant pathways, in order to translate the flux of nutrients in a given scenario to the corresponding flux of radionuclides.

BIORUR considers the soil composed by a number of soil layers, which in turn are composed by three compartments: 1) soil solution and mineral soil, 2) organic matter and 3) roots. The pathways between these compartments are root uptake, mineralization of organic matter and mycorrhizal transfer. Similarly, plants are composed by four compartments, 1) roots, 2) functional leaves, 3) growing parts and 4) storage pool, and the pathways between plant compartments are either xylem flow, or phloem flow. The algorithms of BIORUR describe the flux of radionuclides between compartments in terms of flux of nutrients and a selectivity coefficient for radionuclide compared to nutrient, where the flux of nutrient is an input parameter, which characterizes the scenario, while the selectivity coefficient is assumed a constant property of the pathway, valid for any scenario.

4.3. Development and validation of a predictive model

A literature review on the use of the concept of bioavailability in studies of the soil-to-plant transfer of radionuclides and other contaminants, as well as in studies of nutrients cycling in the soil-plant system, was carried out. Alternative operational definitions of bioavailability were proposed and how to use this concept in the assessment model. This issue was considered in detail and a complete strategy and plan for the integration of the mechanistic model and the experimental data in the assessment model was formulated. Alternative formulations of the assessment model were developed at the conceptual and mathematical level, i.e. without including parameterisation.

Acknowledgements

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