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Quantification of Radionuclide Transfer in Terrestrial and Freshwater Environments for Radiological Assessments

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FOREWORD

For more than thirty years, the IAEA has published a set of documents aimed at the limitation of the radiation exposure of the population from various nuclear activities. In particular, in 1994 the IAEA published Technical Reports Series No. 364, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments. Over the years, it has proved to be a valuable reference for radioecologists, modellers and authorities in Member States, and has been quoted in numerous impact assessments.

Technical Reports Series No. 364 was based on a review of available data up to the end of 1992. However, a number of high quality critical reviews have been produced in recent years for some of the transfer parameter values which merit consideration. Thus, it was assumed that there is sufficient new information available to warrant reconsideration of a significant proportion of the values given in Technical Reports Series No. 364 and to initiate an updating of Technical Reports Series No. 364 in the framework of the IAEA EMRAS (**E**nvironmental **M**odelling for **RA**diation **S**afety) project. It is expected that the revision of Technical Reports Series No. 364 will initiate further updating of related IAEA publications, and international and national radiological models.

The present IAEA-TECDOC is intended to be a support to the update of Technical Reports Series No. 364, overcoming the limitations of the former, and comprises both revised transfer parameter values, as well as missing data, key transfer processes, concepts and models that were found to be important for radiation safety.

The publication was prepared by members of the EMRAS project Working Group 1, Theme 1 Revision of IAEA Technical Reports Series No. 364. The group was chaired initially by P. Santucci, followed by P. Calmon. The IAEA wishes to express its gratitude to M. Thorne, A. Gondin da Fonseca and B.Varga for their kind assistance in editing and preparation of the document for publishing, and.to all the contributors to the IAEA-TECDOC and those who assisted in its drafting and review. The IAEA officer responsible for this publication was S. Fesenko of the Department of Nuclear Sciences and Applications.

CONTENTS

INTRODUCTION

AGRICULTURAL ECOSYSTEMS: FOLIAR UPTAKE

AGRICULTURAL ECOSYSTEMS: RADIONUCLIDE MOBILITY IN SOIL

AGRICULTURAL ECOSYSTEMS: ROOT UPTAKE

AGRICULTURAL ECOSYSTEMS:TRANSFRE TO ANIMALS

AGRICULTURAL ECOSYSTEMS: TRANFER TO FRUITS

NATURAL AND SEMI NATURAL ECOSYSTEMS

RADIONUCLIDE TRANSFERS IN FRESHWATER ECOSYSTEM

FRESHWATER ECOSYSTEMS

MISCELLANEOUS TOPICS

INTRODUCTION

QUANTIFICATION OF RADIONUCLIDE TRANSFER IN TERRESTRIAL AND FRESHWATER ENVIRONMENTS: A SUMMARY

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1. BACKGROUND

Exposure may occur as a result of incidental releases and discharges from all stages of the nuclear fuel cycle [1]. This includes uranium mines, fuel fabrication facilities, reactors, reprocessing plants and low, intermediate and high level radioactive waste repository sites. Each of these releases involves mainly fission and neutron activation products as well as actinides [2]. Uranium mining and fuel fabrication also involve radionuclides of the uranium, actinium and thorium decay series. In addition, some industries such as mineral sands production and processing, phosphate ore processing, coal burning and oil and gas production can generate naturally occurring radioactive materials (*NORMs*) as waste. Other sources of human exposure include nuclear weapon testing and radiation accidents or incidents [2].

Any radiological assessments are based on an analysis of radionuclide transport patterns and processes governing radionuclide behaviour in the environment [3]. Therefore, it is an aim of the IAEA to support Member States to enhance their capability for evaluating radionuclide transport, geochemical and biochemical enrichment, and radiation effects on human beings and other biota [1].

Radionuclides released during routine operations of nuclear installations are dispersed in the environment and often difficult and expensive to measure directly because of very low radionuclide concentrations in the Environment. As a consequence, environmental radiological impacts usually are assessed by modelling both for operating nuclear facilities and during licensing. Radioecological models also have to be used for predicting future impacts, e.g. in consequence assessments of nuclear waste repositories or in emergency response planning.

Thus, many radiation protection models need to predict transfers of a large number of radionuclides. This requires information on transfer of many less mobile radionuclides, which do not usually comprise an important component of discharges or dose. Over the years, Technical Reports Series No. 364 provided an important source of such information, and is one of the key cited sources for many models. It is thus essential that such information is kept up-to-date and that any relevant recent literature is included, especially considering the paucity of existing data sources. This, in itself, is a strong argument for revision if the information given is now known to be incorrect, inadequate (given new information available since its preparation) and incomplete.

A few years ago a revision of the IAEA Safety Series No. 57 [4] was undertaken by the IAEA with the purpose of providing simple methods for calculating doses arising from radioactive discharges into the environment for risk assessment evaluations (screening of radioactive discharges). Within the revised document, some of the default parameter values, used in the models for calculating the transfer of radionuclides in food chains to humans and contributing to ingestion dose were those given in Technical Reports Series No. 364. In the new Safety Reports Series No. 19 only conservative values are considered [5]. In contrast, in Technical Reports Series No. 364 the expected values are based on different criteria and are not intended to be conservative. Current contradictions between the two documents have been resolved in the revision of these documents and updating Technical Reports Series No. 364.

The scope of Technical Reports Series No. 364 had three major limitations. First, the parameters given in the document were mainly limited to temperate environments. Second, it gave parameters mainly for equilibrium conditions, and, therefore, they often cannot be applied for environmental assessments in accidental situations or to situations with variable release of radionuclides into the environment. And finally, Technical Reports Series No. 364 did not provide information on transfers of some radionuclides that are of importance in waste management practice.

A specific task of the Technical Reports Series No. 364 revision was to provide reference values for the most commonly used transfer parameters in radiological assessment models. However, some important details and recommendations on how to use these parameters were often omitted that undermined the making of relevant choices of necessary parameters. This problem is being resolved by publishing of two separate but well-linked documents i.e. the Technical Reports Series document focused on the reference information intended for radiological assessment and the IAEA-TECDOC intended for justification of radioecological information used to derive reference values, radioecological concepts and models facilitating use of reference values in specific situations.

Thus, the present IAEA-TECDOC is intended to be a supportive document for the updated Technical Reports Series No. 364 overcoming the limitations of the former document, and comprises both revised transfer parameter values, as well as missing data, key transfer processes, concepts and models that were found to be important for radiation safety.

The scope of the current document covers both main radioecological concepts on mechanisms governing radionuclide behaviour in the environment and the parameters derived from numerous radioecological studies.

2. OBJECTIVES

The main objective of this document is to provide radioecological concepts, models, parameters and data for assessing site-specific past, present and potential future radiation exposures of humans and other biota in terrestrial and freshwater environments in different climate conditions.

3. SCOPE

The IAEA Technical Document is primarily intended to provide the Member States with an overview of radioecological concepts, models and parameters for radioecological assessment of both routine discharges of radionuclides to the environment and accidental releases.

The document is also intended for further updating and facilitating of use of the recently published IAEA documents related to the assessment of the radiological impact of radioactive discharges as described in IAEA Safety Report Series No. 19 Generic Models for Use in Assessing the Impact of Discharges of Radioactive Substances to the Environment [5], IAEA Safety Standards Series No. WS-G-2.3 Regulatory Control of Radioactive Discharges to the Environment and other related documents [6]. The document can be also used as background documentation for other relevant activities such as training in radioecology and radiation protection.

4. DATA COLLECTION AND ANALYSIS

International databases of bibliographical references (such as INIS, Scopus, JCR, SpringerLink and Academic Search Premier, INIST, Current Contents, *etc.*) and some national databases (such as IRSN bibliographical database) were consulted by using relevant key words. Such a bibliographical study did not aim at exhaustiveness and was limited to published documents within the international scientific community and, depending on their accessibility, to reports from different scientific institutions. That inventory of existing knowledge was drawn upon with a view to consulting original publications in order to use all the information that they contained, rather than relying on summaries of such information.

During the second step, databases were elaborated, where necessary. These databases associated each value for a given transfer parameter with a number of descriptors. The bibliographical references accepted were also included in the database.

The data have been analysed with the objectives of (i) estimating a geometric mean and/or an arithmetic mean for a given parameter, and (ii) obtaining an indication of the extent of uncertainty about these estimates. The estimation of these values and the extent of uncertainty about every such value have been carried out by applying statistical analysis, when possible. In the ideal case, if adequate data were available, both a geometric (GM) and an arithmetic means (AM) were derived.

When the number of observations was 2, only the arithmetical mean (AM) was calculated, and the single observation is presented when number of data was equal to 1. The uncertainties assigned to geometric mean were estimated by the geometrical standard deviation (GSD), although the standard deviations (SD) are also calculated as the uncertainty assigned to arithmetic mean. Ranges with minimum and maximum values are also given as well as the number of data (N) and number of references (#ref) used for a value evaluation. In some cases, the expected values are given without a statement of uncertainty or a range because of the limited data available. The recommended values in such cases should be used with caution.

Geometric mean is a type of mean or average, which indicates the central tendency or typical or expected value of a set of numbers:

$$GM = (\prod_{1}^{N} x_i)^{1/N} \qquad (2)$$

Geometric standard deviation (GSD) is the exponent of the standard deviation of the natural logarithms of the individual values. GSD can also be calculated based on the expression:

$$GSD = \exp(\sqrt{\frac{(\sum_{1}^{N}(\ln(x_i) - \ln(GM))^2}{N}})$$

Arithmetic mean:

$$AM = \frac{1}{N} \cdot \sum_{1}^{N} x_i \qquad (3)$$

Standard deviation:

$$SD = (\frac{1}{N-1} \cdot \sum_{1}^{N} (x_i - AM)^2)^{1/2}$$

Thus, standardized tables reporting transfer parameter values are provided where possible giving the following information for each radionuclide: number of entries (N); geometric mean (GM); geometric standard deviation (GSD); arithmetic mean (AM), standard deviation (SD), minimum (MIN) and maximum (MAX) values and number of references from which entries were extracted (# Ref).

5. SUMMARY OF THE DOCUMENT

The document consists of twentry seven papers unified in nine topics (sections) according to traditional presentation of radioecological data for environmental assessments. Some of radioecological concepts, definitions, units and some important references as well as classifications of soil and plants used in the document are described in the introductory section.

Four papers included into the second section are dedicated to data for, and models of, foliar uptake. The main aspects and processes governing radionuclide mobility in soil including the K_d radionuclide vertical migration concepts and data, are described in the next two papers, composing the section on radionuclide mobility in soil.

Data on root uptake for various environments: temperate, tropical and subtropical, transfer to rice and data for some time dependent contamination scenaries are presented and discussed in the section on root uptake in agricultural ecosystem while the next section is dedicated to radionuclide transfer to animals and animal products. The data on radionuclide transfer in agricultural ecosystems are being completed by the paper on radionuclide transfer to fruits.

Parameters for modelling radionuclide transfer, including data and evaluation of processes governing radionuclide transfer in forest ecosystems (trees, berries, mushrooms, game), Arctic and Alpine environment compose the section on natural and semi-natural ecosystems.

Section on freshwater ecosystems comprises five papers devoted to transport of radionuclides in freshwater ecosystems including contamination routes, physical processes and radionuclide accumulation by freshwater biota. Specific activity models for the relevant radionuclides (^{3}H, ^{14}C, ^{36}Cl), food processing and use of analogue approach for radiological assessments are covered by the section on miscellanius topics. Most papers consist appendexis which provide references used for data evaluation and examples of data evaluation.

REFERENCES

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RADIOECOLOGICAL DEFINITIONS, SOIL, PLANT CLASSIFICATIONS AND REFERENCE ECOLOGICAL DATA FOR RADIOLOGICAL ASSESSMENTS

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Abstract

The paper gives an overveiew of radioecological concepts, reference information on plants and soils used in the project. Main classification schemes applied to summarize the data on radionuclide transfer to plants are also described. Generic quantities and units used in the current document are also given. They are mainly defined according to the the ICRU 65 report.

1. DEFINITIONS, QUANTITIES AND UNITS

In 1997, a report committee was initiated and established by the International Commission on Radiation Units and Measurements (ICRU) to prepare a report on statistical aspects and sampling strategies when measuring radionuclides in the environment.

Two reports are being produced: one on the quantities and units used that is already available [1], and the second on statistical aspects and sampling strategies that is at the final stage of preparation.

The first report (ICRU report 65) contained a list of units and quantities frequently used in the field of radioecology, and tried to harmonise the units derived from various different disciplines such as ecology, chemistry, medicine or physics. First symbols, definitions and descriptions, their dependencies and a list of previously used symbols are given. In addition, appendices give generic, helpful information for the 'unexperienced in the specific field' on soil chemistry, soil classifications, taxonomy of species in common and Latin names in terrestrial and aquatic environments.

The current document mainly follows the recommendations given in the above ICRU report [1]. However, the classification schemes for soils and plants deviate from the recommendations of the ICRU report in a few places, because of specific requirements from the data presentations and their use for radiological assessments. Generic quantities and units used across the entire document are given below (Table 1). As mentioned earlier, they are mainly (with some exceptions discussed below) defined as in the ICRU report 65 [1].

The definitions of specific terms are given in the introductions to the stand-alone papers. Only some of the generic quantities (transfer parameters) that are in wide use are covered by the ICRU report. Those generic parameters that were not defined by the ICRU are also specified in the papers where they are used.

Soil-to-plant transfer factors were defined on a dry weight basis, for both plants and soil to reduce uncertainty. In those cases in which the transfer factor (concentration ratio) values or the plant concentrations, the feed or to berries reported in the literature were expressed relative to fresh weight, the fresh weight/dry weight conversion factors given by Tables 2-4 were applied.

In some estimations given in the document the carcass weighs and meat fractions are in use (Table 5). These fractions were derived by subtraction of the weight of bones and other not edible parts from carcass weights (Table 5).

Thus, all transfer factor values were expressed on a dry weight plant and dry weight soil basis before further analysis and interpretation. It is emphasised that actual fresh weight to dry weight ratios are likely to have varied somewhat around the adopted values, so this is an additional source of uncertainty in the analysis. In some specific situations, fresh weight is used in assessment calculations and in these situations the fresh weight/dry weight conversion factors (see Appendix 1) can be applied to scale the dry weight based values given in this report.

The International Union of Radioecology (IUR) decided, in 1982, to standardize the depth of soil in defining transfer factor values [7, 8]. Instead of the real rooting depth, a standardized depth of soil was adopted. All roots and all activity present in the actual rooting zone are assumed to be present in the standardized zone. For grass, this value is 10 cm and for all other crops (fruit trees included) the value is 20 cm.Transfer factors are commonly used to quantify radionuclide uptake from soil by plants. However, the definition based on the ratio of the activity concentration in plants (Bq kg^{-1} dry weight) to that in soil within the layer of a standardized thickness (Bq kg^{-1} dry weight), is not appropriate for forests and some other natural and semi-natural ecosystems. The reasons are the multi-layered character of the soils and the high inhomogeneity of distribution of root systems (or mycelia) over the soil profile. Therefore, aggregated transfer factors (T_{ag}) are used in the current document as an alternative to quantify radionuclide availability to various types of natural or semi-natural vegetation, animals or other products. T_{ag} is defined as the ratio of the radionuclide activity concentration in plant or any other natural or semi-natural product (Bq kg^{-1} fresh or dry weight, depending on the product) divided by the total deposition on the soil (Bq m^{-2}). The T_{ag} concept is also used for assessments of radionuclide transfers to game.

TABLE 1. QUANTITIES AND UNITS USED

Symbol used by the ICRU 65	Symbol used in IAEA publications	Symbol used in the current document	Name	Definition[1]	Unit[1]	Specification[1]	Comments[1].
C_{ag}	T_{ag}	T_{ag}	Aggregated transfer factor	C_{ag} is the mass activity density (Bq kg^{-1}) in a specified object per unit area activity density, A_a (Bq m^{-2}) in the soil.	m^2 kg^{-1}	Time, location, pathway, object, wet or dry mass basis, deposition type etc.	Developed to deal with natural and semi-natural transfers from soil to biota. Can be soil to animal, soil to plant etc. It will encompass many processes including food-chain transfers, root uptake, soil adhesion, direct soil ingestion, etc. It may have lower variation in the case of transfers from soil because it may overcome the effects of the differences in soil bulk density between organic and mineral soils. Previous symbols *TAG, TF,* T_{ag}
C_r	B_v, F_v, TF	F_v	Concentration ratio	C_r is the ratio of the activity density of radionuclide in the receptor compartment to that in the donor compartment.	Dimensionless or kg kg^{1}	Depending on compartments specified, radionuclide, receptor and source compartment, wet or dry weight of for each compartment, degree of equilibrium etc.	Most often used to describe plant uptake from the soil, or activity concentrations in aquatic organisms in relation to activity concentrations in water. The intention here is to include such ratios as feed to plant tissue/soil, animal tissue/feed, fish muscle/water etc. The terms *bioaccumulation*, *bioconcentration* and *biomagnification* are related, but are used with very specific connotations that should be carefully specified. The recommended symbol C_r is a change from past usage, but it accomplishes the desire to have a single primary character and to achieve compatibility with other symbols in the ICRU 65 report. Previous symbols *CR, TF,* B_v, *CF, BCG,* B_r

TABLE 1. QUANTITIES AND UNITS USED (Cont.)

Symbol used by the ICRU 65	Symbol used in IAEA publications	Symbol used in the current document	Name	Definition[1]	Unit[1]	Specification[1]	Comments[1].
C_{fr}	F_m, F_f	F_m, F_f	Feed transfer coefficient	C_{fr} is the mass or volumetric activity density in the receptor tissue or product of an animal (Bq kg^{-1} wet mass or Bq L^{-1}) divided by the transfer rate (in Bq d^{-1})	d kg^{-1} or d L^{-1}, where d is the time in days	Radionuclide, animal, specific tissues or animal product; feed type, contamination source; degree of equilibrium between activity density in product and intake rate	The coefficient is used to estimate the activity density in an animal tissue or product that is eaten by humans (i.e. milk, meat, eggs). The tissue or product activity density is assumed to be at steady state with respect to the intake rate. The intake rate in this case refers strictly to intake by ingestion into mouth to be continued.
K_d	K_d	K_d	Distribution coefficient	The distribution coefficient is the ratio of the mass activity density (A_m in Bq kg^{-1}) on the specified solid phase (usually on a dry mass basis) to the volumetric activity density (A_v in Bq L^{-1}) in the specified liquid phase.	L kg^{-1}	Radionuclide; solid and liquid media; adsorption and desorption basis; laboratory or field conditions; linearity of solid versus liquid concentrations; attainment of equilibrium; ratio of solid to liquid mass, chemical form of radionuclide.	The strict definition is at equilibrium (steady-state equilibrium and reversible processes), but the quantity is often used where complete equilibrium cannot be proven. Additional subscripts may be assigned to clarify the conditions specified, such as whether sorption or desorption is used for the measurement of the K_d. Sorption and desorption K_d values at equilibrium will usually differ. Many methods are used to measure the K_d, and these differences lead to variability and interpretation ambiguities. It is particularly important to describe in detail the method of separating the solid from the solution. If filtration is used, the pore size of the filter should be specified. An operationally defined or fully empirical K_d, for which the two concentrations are measured in field samples, is often preferred to minimise bias caused by the method, but these values often include radionuclides which are not reversibly bound, and for which equilibrium is not necessarily attained. K_d values are sensitive to the composition of the solution (e.g. competing ions, ionic strength, temperature, pH, redox potential), to properties of solids (e.g. clay content and type, particle size, mineralogy) and to the chemical form of the radionuclide. Previous symbols: R_d, CR

TABLE 1. QUANTITIES AND UNITS USED (Cont.)

Symbol used by the ICRU 65	Symbol used in IAEA publications	Symbol used in the current document	Name	Definition[1]	Unit[1]	Specification[1]	Comments[1].
α	α , f	α	Interception coefficient	The interception coefficient, α, is the ratio of the initial mass activity density on the plant (A_m in Bq kg^{-1}) per unit areal activity density ($A_.$ in Bq m^{-2}) on the terrestrial surface (soil plus vegetation).	m^2 kg^{-1}	Radionuclide; plant species and part; wet or dry mass basis; washed or unwashed; wet or dry deposition	This quantity is used in the same sense as the C_{ag}, except that is an initial condition, measured before processes such as weathering, leaching and absorption, which alter the activity density on the plant. The term is well defined only for dry deposition. It is often used in the expression for the fraction of the activity density or deposition (A_α) initially intercepted by plant foliage ($1-e^{-\alpha\beta}$), which was described by Chamberlain. The value of the interception coefficient depends on the surface characteristics, meteorological conditions, biomass density, leaf area index, species, plant part, time, etc. Previous symbol: μ
K	K_s	K_s	Resuspension ratio	The resuspension ratio is the ratio of the volumetric activity density, A_v, measured in air or water (Bq m^{-3}) to the areal activity density. A_α, measured on the soil or sediment surface (Bq m^{-2}).	m^{-1}	Receptor compartment; donor compartment (in particular, the depth of the soil or sediment layer assumed to be subject to resuspension); process involved; degree of time-averaging.	The resuspension ratio is intended to represent the transfer from a surface into a volume. It is most often used for the transfer of material from a soil surface into air, but the transfer can be from sediment to water, from water to air, and from other surfaces (e.g. asphalt) to air. The processes are extremely time-dependent due to episodic weather changes, disturbance events, depletion losses, and movement into the soil. The temporal fluctuations can be dampened with time-averaging. When soil or sediment is the donor compartment, it is important to specify the assumed thickness of the compartment, since contamination deeper than a few mm is usually not available for immediate resuspension. In addition to weather and disturbance events, K is extremely sensitive to nature of the surface and to the characteristics of the actual material being resuspended. Previous symbols: *RF, RR*

TABLE 1. QUANTITIES AND UNITS USED (Cont.)

Symbol used by the ICRU 65	Symbol used in IAEA publications	Symbol used in the current document	Name	Definition[1]	Unit[1]	Specification[1]	Comments[1]
f_{tr}	f_{tr}	f_{tr}	Translocation ratio, translocation factor, translocation coefficient	The translocation ratio is the mass activity density (A_m in Bq kg^{-1}) in one tissue, typically an edible tissue, divided by the mass activity density (A_m in Bq kg^{-1}) in another tissue of the same plant or crop. It is also can be calculated as the mass activity density in the edible tissue (Bq kg^{-1}) in another tissue of the same plant or crop. It is also can be calculated as the mass activity density in the edible tissue (Bq kg^{-1}) divided by the activity contained on the mass foliage covering a square metre of land surface (Bq m^{-2})	Dimensionless, m^2 kg^{-1}	Radionuclide; kind of plant; specific tissues; type of contamination; contamination history; wet or dry mass.	This quantity is often used to estimate the activity density in an unmeasured tissue from another tissue in the same plant which has been measured. The quantity may be used for fruits, seeds, specific edible vegetables or the bole of trees. It varies with radionuclide and chemical form, time, species, growth stage, nutrition status, etc. Previous symbols: T_i, TF, TLF, TLF_α
f_{fp}	F_r	F_r	Food processing retention factor	The food processing retention factor is the ratio of the mass activity density (A_m) of a given food item when ready for consumption to the activity density before processing and preparation.	Dimensionless	Radionuclide; food; type and conditions of the food processing and preparation.	Examples of food processing and preparation include cheese making, washing, scrubbing, peeling, boiling, baking and frying. Conditions used to determine sample masses before and after processing and preparation must be specified for proper interpretation. This quantity varies according to the radionuclide, the way it is bound to the food, the type of food, and the varied conditions of preparation (e.g. temperature, time, etc.).

TABLE 1. QUANTITIES AND UNITS USED (Cont.)

Symbol used by the ICRU 65	Symbol used in IAEA publications	Symbol used in the current document	Name	Definition[1]	Unit[1]	Specification[1]	Comments[1]
$T_{1/2}$	$T_{1/2}$	$T_{1/2}$	Half life	The half life is (ln2)/λ, where λ is the decay constant (s^{-1})	s	Radionuclide	The half life is specific to each radionuclide and can be visualised as the time required for the activity as a consequence of radioactive decay.
f_a	f_1	f_1	Absorbed fraction	The absorbed fraction is the fraction of the ingestion or inhalation intake by animal or a human that is transferred to a specified receptor tissue.	Dimension-less	Radionuclide; route of intake (pathway), receptor tissue.	This quantity is related to the f_1 factor in the metabolic model for human dosimetry, where the receptor tissue is blood, and to blood-tissue transfer fractions. The method of determination is very important; for example, an \`apparent\` f is computed from information on the whole-body intake and excretion of the radionuclide. *A* \`true\` f is measured in a metabolic study that involves injection of a tracer, and allows for the determination of endogenous faecal excretion (i.e. direct transfer from blood to the intestine). The absorbed fraction depends on metabolic factors (e.g. age, lactation state, physiological condition), chemical and physical speciation of the radionuclide, competing ions, gut contents and chemical conditions, etc. Previous symbols: f_1, α, A_α, At

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[1] According to ICRU Report 65 [1].

TABLE 2. DRY MATTER CONTENTS IN VARIOUS PLANTS, % [2, 3]

Crop	Dry matter content (%)
Spring vetch, seeds	86.9
Vegetative mass	24.3
Winter vetch, seeds	87.9
Vegetative mass	22.2
Field pea, seeds	85.1
Vegetative mass	16.7
Garden pea, seeds	83.0
Vegetative mass	15.8
Grass pea vine, seeds	86.4
Vegetative mass	20.9
Soya, seeds	88.6
Vegetative mass	26.0
Lupin yellow, seeds	85.2
Vegetative mass	13.6
Lupin blue, seeds	85.5
Vegetative mass	18.0
Seradella, vegetative mass	22.4
Broadbeans, seeds	88.0
Vegetative mass	18.3
Bean (field, kidney and French), vegetative mass	27.7
Lentil, vegetative mass	24.9
Winter rye, grain	87.0
Vegetative mass	22.6
Wheat, grain	88.0
Vegetative mass	18.2
Oats, grain	86.7
Vegetative mass	27.6
Barley, grain	87.0
Vegetative mass	33.8
Maize (corn), grain	85.2
Vegetative mass	18.8

TABLE 2. DRY MATTER CONTENTS IN VARIUS PLANTS, % (Cont.)

Crop	Dry matter content (%)
Sudan grass, seeds	90.0
Vegetative mass	20.0
Sorghum, grain	87.0
Vegetative mass	25.0
Annual ryegrass, vegetative mass	20.0
Millet, grain	88.0
Vegetative mass	23.0
Alfalfa, vegetative mass	26.0
Sickle alfalfa, vegetative mass	33.0
Bastard Lucerne, vegetative mass	23.0
Red clover, vegetative mass	22.0
Ladino clover, vegetative mass	26.0
Sainfoin	23.0
White sweetclover, vegetative mass	22.0
Yellow sweetclover, vegetative mass	22.0
Fussian brome grass, vegetative mass	20.8
Slender wheat grass , vegetative mass	34.0
Couch grass, vegetative mass	37.0
Standard crested grass, vegetative mass	39.0
Timothy grass, vegetative mass	25.5
Meadow fescue, vegetative mass	20.0
Cock’s foot grass, vegetative mass	22.0
Meadow grass, vegetative mass	22.0
Cabbage	12.0
Lettuce	8.0
Leek	11.0
Onion (aboveground part)	11.0
Spinach	8.0
Celery	6.0
Cauliflower	11.0
Kohlrabi	6.0
Tomato	6.0
Cucumber	5.0
Pumpkin (english)	7.5
Vegetable marrow (English)	9.0
Zuchini	5.0
Beetroot (red beet)	16.0
Sugar beet	22.0

TABLE 2. DRY MATTER CONTENTS IN VARIUS PLANTS, % (Cont.)

Crop	Dry matter content (%)
Radish	9.0
Carrot	14.0
Potato	21.0
Turnip (Swede)	12.0
Jerusalem artichoke	22.0
Tapioca	38.0
Raspberry	16.0
Water-melon	7.0

TABLE 3. DRY MATTER CONTENTS IN VARIUS FEED, % (www.agriknowledge.co.uk)

Feed	Dry matter content (%)
Concentrate feed	0.88
Grass silage	0.26
Pasture	0.20
Grass hay	0.86
Lucerne hay	0.86
Lucerne silage	0.34
Corn silage	0.25

TABLE 4. DRY MATTER CONTENT OF SOME WILD BERRIES, %

English name	Latin name	N	AM	SD	Max	Min
Blueberry	*Vaccinium myrtillus*	307	13.2	1.9	21	8.6
Lingonberry	*Vaccinium vitis-idaea*	254	14.1	1.3	18.8	11.3
Cranberry	*Vaccinium oxycoccus*	16	10.8	0.9	12.1	9.3
Bog bilberry	*Vaccinium uliginosum*	6	12.1	1.1	13.5	10.5
Black crowberry	*Empetrum nigrum*	1	7.4	-	-	-
Cloudberry	*Rubus chamaemorus*	26	14.0	1.6	18	9
Wild raspberry	*Rubus idaeus*	21	17.3	1.8	21.9	14.4
Wild strawberry	*Fragaria vesca*	1	15.4	-	-	-

TABLE 5. CARCASS WEIGHT AND MEAT FRACTION FOR GAME ANIMALS [4-6]

Species of animal	Carcass weight, kg	Fraction of meat in carcass weight
Moose, adult	1.9×10^2	0.80
Moose, calf	8.3×10^1	0.78
White-tailed deer	5.0×10^1	0.78
Fallow deer	3.3×10^1	0.78
Roe deer	1.8×10^1 [a]	0.78
Brown hare	2.4	0.90
Arctic hare	1.8	0.90
Capercaillie	1.9	0.90
Black grouse	6.6×10^{-1}	0.90
Hazel grouse	2.4×10^{-1}	0.90
Willow grouse	3.6×10^{-1}	0.90
Partridge	2.4×10^{-1}	0.90
Pheasant	6.9×10^{-1}	0.90
Goose	2.3	0.90
Eider	1.3	0.90
Long-tailed duck	3.8×10^{-1}	0.90
Mallard	6.6×10^{-1}	0.90
Goldeneye	4.5×10^{-1}	0.90
Teal	1.8×10^{-1}	0.90

[a] Roe deer gains more weight in Northern than in Central Europe.

The concept of T_{ag} is adopted as a reasonable empirical measure to normalize radionuclide accumulation in forest and semi-natural products regardless of variations in the vertical radionuclide distribution and availability in the soil profile, which greatly depends on the site.

Because of the multi-layered character of soils and of great variations in humus thickness in natural and semi-natural ecosystems such as forests, a transfer factor that averages over the total deposit in soil without consideration for the soil layers most exploited by roots, can exhibit a strong time-dependence. Some observations report on the time-dependence of the migration in soil and movement of radionuclides between the various environmental components.

The long-term, time-dependent behaviour of radionuclides is often quantified by reference to the ecological half-life, $T_{1/2}^{eco}$, which is an integral parameter that lumps all processes except radioactive decay that cause a reduction of activity in a specific medium. The processes involved in determining the value of the ecological half-life are specific to the medium considered, e.g. for the reduction of activity in game, losses of radionuclides from the root layer of the soil, fixation to soil particles and uptake by plants are the most relevant processes. If radioactive decay – characterized by a physical half-life T_r–is included in the reduction of the content or concentration of a particular radionuclide in a system, then the effective half-life $T_{1/2}^{eff}$ is given by:

$$\frac{1}{T_{1/2}^{eff}} = \frac{1}{T_{1/2}^{eco}} + \frac{1}{T_r} \qquad (1)$$

2. PLANT AND SOIL CLASSIFICATIONS

Based on the analyses of available information on radionuclide transfer to plants [7-10], fourteen plant groups have been identified (Table 2). Assignment of individual plants to these groups is given in Appendix 1 while plant compartments are shown in Table 3. The transfer of radionuclides through the food chain varies considerably depending on soil properties. In the FAO/UNESCO soil classification, there are 28 units and 125 sub-units [11]. F_v values are not available for all units or sub-units, even for the most extensively studied radionuclides. Therefore, a more broadly based classification is adopted here that permits some distinction on the basis of texture and organic matter content, while ensuring that a reasonable amount of data are available for each category.

TABLE 6. PLANT GROUPS AND COMPARTMENTS

Plant group	Plant compartment
Cereals	Grain, seeds and pods
	Stems and shoots
Maize	Grain, seeds and pods
	Stems and shoots
Rice	Grain, seeds and pods
	Stems and shoots
Leafy Vegetables	Leaves
Non-leafy Vegetables	Fruits, heads, berries, buds
Leguminous-Vegetables	Seeds and pods
Root Crops	Roots
Tubers	Tubers
Fruits	Fruits, heads, berries, buds
Grasses (cultivated species)	Stems and shoots
Fodder Leguminous (cultivated species)	Stems and shoots
Pasture (species mixture – natural or cultivated)	Stems and shoots
Herbs	Leaves; grain, seeds and pods; fruits, heads, berries, buds
Other Crops	Grain, seeds and pods; leaves; stems and shoots; fruits, heads, berries, buds; roots; tubers

For this document, four soil groups were defined: sand, loam, clay and organic. Soils were grouped according to the sand and clay mineral percentages referred to the mineral matter, and the organic matter (OM) content in the soil. This defined the ‘texture/OM’ criterion, which is similar to the criterion followed in the former Technical Reports Series No. 364. For the mineral soils, three groups were created according to the sand and clay percentages referred to the mineral matter [6]: ‘Sand group’: sand fraction ≥65 %; clay fraction <18 %; ‘Clay group’: clay fraction ≥35 %; ‘Loam group’: rest of cases. A soil was included in the ‘Organic group’ if the organic matter content was ≥20 %. Finally, an ‘Unspecified soil group’ was created for soils without characterization data, or for mineral soils with unknown sand and clay contents.

Unspecified group was created for soils without characterization data, or for mineral soils with unknown sand and clay contents.

Average characteristics of the soils in the different groups are given in Table 7 and more details of the typical textures of the mineral soil classes are given in Tables 8 and 9, based on two different texture classification schemes. These general texture soil groups are presented for different specific soil classifications [1, 11, 12].

TABLE 7. USUAL RANGES OF VALUES FOR VARIOUS SOIL PARAMETERS FOR THE SOIL GROUPS ADOPTED

Soil group	pH	% OM	CEC $cmol_c/kg$	Sand content in the mineral matter fraction	Clay content in the mineral matter fraction
Sand	3.5-6.5	0.5-3.0	3.0-15.0	≥65	<18%
Loam	4.0-6.0	2.0-6.5	5.0-25.0	65-82	18-35
Clay	5.0-8.0	3.5-10.0	20.0-70.0	-	≥35
Organic	3.0-5.0	≥20	20.0-200.0	-	-

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TABLE 8. FAO SOIL CLASSIFICATIONS BASED ON TEXTURE [11]

SOIL GROUP										
SAND			LOAM				CLAY			
	Content %			Content %				content %		
Texture	Clay (<0.002mm)	Sand (>0.5mm)	Texture	Clay (<0.002mm)	Silt (0.02-0.5mm)	Sand (>0.5mm)	Texture	Clay (<0.002mm)	Silt (0.002-0.5mm)	Sand (>0.5mm)
COARSE			MEDIUM				FINE			
Sand	<18	>65	Sandy loam	18-35		65-82	Clay	>35		
Loamy sand			Loam				Silty clay			
Sandy loam			Sandy clay loam				Sandy clay			
			Silt loam				Clay loam			
							Silty clay loam			

TABLE 9. RUSSIAN SOIL CLASSIFICATIONS BASED ON TEXTURE [12]

SOIL GROUP								
SAND			LOAM			CLAY		
Texture	Physical clay (<0.01mm)	Clay (<0.001mm)	Texture	Physical clay (<0.01mm)	Clay (<0.001mm)	Texture	Physical clay (<0.01mm)	Clay (<0.001mm)
Light loams sandy	20-30	<20	Medium loams slimy	30-40	>20-35	Heavy clay slimy	>80	>35
Light loams coarse silty	20-30		Medium loams silty			Heavy clay silty	>80	
Loamy sand sandy	10-20		Medium loams coarse silty			Medium & light clays slimy	60-80	
Loamy sand coarse silty	10-20					Medium & light clays silty	60-80	
Sands sandy	0-10					Medium & light clays coarse silty	60-80	
Sandy coarse silty	0-10	<10				Heavy loams slimy	40-60	
						Heavy loams silty	40-60	
						Heavy loams coarse silty	40-60	

Physical sand > 0,01 mm; clay <0.001 mm; silt 0.001-0,05 mm; sand 0.5-1.0 mm.

APPENDIX

TABLE A1. PLANT GROUPS, WITH COMMON AND LATIN NAMES OF ASSOCIATED CROPS

Plant group	Common name	Latin name
Cereals	Rye	*Secale cereale L. subsp. cereale*
	Wheat	*Triticum aestivum L. non. cons. Subsp. aestivum*
	Oats	*Avena sativa L.*
	Barley	*Hordeum vulgare L.subsp. vulgare*
	Maize (corn)	*Zea mays L. subsp. mays*
	Sorghum	*Sorghum bicolor (L.) Moench*
	Millet	*Panicum L.*
	Buckwheat	*Fagopyrum esculentum*
	Foxtail millet, Italian millet	*Setaria italica L.*
Maize	Maize (corn)	*Zea mays L. subsp. mays*
Rice	Rice	*Oryza sativa L.*
Leafy vegetables	Hiroshimana (Pot herb, mustard)	*Brassica rapa L..*
	Kikuna (chop suey green)	*Chrysanthemum coronarium L. var. Spatiosum L.H. Bailey*
	Mizuna (green)	*Brassica rapa L. subsp. nipposinica (L.H. Bailey) Hanelt (Mizuna Group)*
	Burdock (great burdock)	*Arcitum lappa L.*
	Asparagus	*Asparagus officinalis L.*
	Purslane	*Portulaca oleracea L.*
	Cabbage, flowering	*Brassica rapa L. var. parachinensis (L.H. Bailey) Hanelt*
	Chinese spinach	*Amaranthus tricolor L.*
	Cauliflower	*Brassica oleracea L. var. botrytis L.*
	Cabbage	*Brassica oleracae L. var. capiatata L.*
	Pak-choi, Chinese cabbage	*Brassica rapa L. chinensis (L.) Henelt).*
Leafy vegetables	Kale	*Brassica oleracea L. var. viridis L.*
	Kohlrabi	*Brassica oleracea L. var. gonylodes L.*
	Lettuce	*Lactuca sativa L.*
	Leek	*Allium porrum L.*
	Swiss Chard	*Beta vulgaris L. Subsp cicla (L.) W.D.J. Koch var. flavescens (Lat). Lat&DC*
	Spinach	*Spinacia oleracea L.*
	Celery	*Apium graveiolus L. var. dulce (Mill.) Pers.*
	Chinese lettuce	*Lactuca sativa L. var. angustana L.H. Bailey*
	Sorrel	

TABLE A1. PLANT GROUPS, WITH COMMON AND LATIN NAMES OF ASSOCIATED CROPS (Cont.)

Plant group	Common name	Latin name
Non-Leafy vegetables	Tomato	*Lycopersicon esculentum Mill.*
	Lady's finger (gumbo, okra)	*Abelmoschus esculentus (L.) Moench*
	Eggplant, (brinjal)	*Solanum melongena L.*
	Bottle gourd	*Lagenaria siceraria (Molina) Standl.*
	Pepper, banana pepper	*Capsicum annuum L. var. annuum*
	Amaranthus (Cherra??)	*Amaranthus L. spp.*
	Red chili (pepper)	*Capsicum frutescens L.*
	Eggplant	*Solanum melongena L.*
	Cucumber	*Cucumis sativus L. var sativus*
	Squash (American)	*Cucurbita pepo L.*
	Pumpkin (English)	*Cucurbita pepo L.*
	Vegetable marrow	*Cucurbita pepo L.*
	Zuchini	*Cucurbita pepo L.*
	Onion	*Allium cepa L.*
	Garlic	*Allium sativum L.*
	American artichoke	*Helianthus tuberosus L.*
	Pepper	*Capsicum annuum L. var. annuum*
Leguminous-vegetables	Peas (garden pea, field pea)	*Pisum sativum L.*
	Chickpea, garbanzo	*Cider arietinum L.*
	Hyacinth-bean	*Lablab purpureus (L.) Sweet subsp. purpureus*
	Soybean; soya	*Glycine max (L.) Merr.*
	Soya (wild soybean)	*Glycine max (L.) Merr. (=Glycine hispida L.)*
	Bean (field, kidney, French, etc.)	*Phaseolus vulgaris L. cultivars*
	Lentil	*Lens culinaris Medik. subsp. culinaris (Ervum lens L.)*
	Asiatic haricot bean (Mung-bean)	*Phaseolus aurens Roxb. = Vigna radiate (L.) R. Wilczek*
	Horse-beans	*Vicia faba L, var. equina Pers.*
Root crops	Beet, beetroot, red beet/ Mangold	*Beta vulgaris L. subsp. vulgaris (Crassa Group)*
	Sugarbeet	*Beta vulgaris L. subsp. vulgaris*
	Turnip (Swede)	*Brassica napus L. var. napobrassica (L.) Rchb.*
	Radish	*Raphanus sativus L.*
	Carrot	*Daucus carota L. subsp. Sativas (Hoffm.) Arcang.*
	Manioc, manihot; cassava, yucca, tapioca	*Manihot esculenta Crantz* *Manihot ultissima*

TABLE A1. PLANT GROUPS, WITH COMMON AND LATIN NAMES OF ASSOCIATED CROPS (Cont.)

Plant group	Common name	Latin name
Tubers	Potato	*Solanum tuberosum L. subsp.tuberosum*
	Yam	*Dioscorea L. spp.*
	Arrowhead	*Sagittaria sagittifolia L. subsp. Leucopetala (Miq.) Hartog*
	Sweet potato	*Ipomoea batatas L.*
Fruits	Apple	*Malus domestica Borkh.*
	Date palm	*Phoenix dactylifera L.*
	Banana	*Musa L. spp.*
	Papaya	*Carica papaya L.*
	Pear	*Pyrus L. spp.*
	Cherry	*Prunus L. spp.*
	Apricot	*Prunus armeniaca L.*
	Peach	*Prunus persica (L..) Batsch var. Persica*
	Prunes or plums	*Prunus domestica L.*
	Strawberry	*Fragaria* ^{x}*ananassa Duchesne*
	Black currant	*Ribes nigrum L..*
	Red currant	*Ribes rubrum L.*
	Gooseberry	*Ribes uva-crispa L.*
	Raspberry	*Rubus ideaus L.*
	Blackberry	*Rubus L. spp.*
	Melon	*Cucumis melo L.*
	Water-melon	*Citrullus lanatus (Thunb.) Matsum. & Nakai*
	Lemon	*Citrus limon (L.) Burm.*
	Orange	*Citrus sinensis (L.) Osbeck*
	Grapefruit	*Citrus paradisi Macfad.*
	Mandarin	*Citrus reticulate Blanco*
	Avocado	*Persea Americana Mill. var. americana*
	Mango	*Mangifera indica L.*
	Grapes	*Vitis L. spp.*
	Olive	*Olea europaea L. subsp. europaea*
	Blueberry	*Vaccinium L. spp.*
	Pineapple	*Ananas comosus (L.) Merr.*
	Pomegranate	*Punica granatum L.*

TABLE A1. PLANT GROUPS, WITH COMMON AND LATIN NAMES OF ASSOCIATED CROPS (Cont.)

Plant group	Common name	Latin name
Grasses (cultivated species)	Sudan grass	*Sorghum sudanensis (Piper) Sterf.*
	Perennial Ryegrass	*Lolium perenne L.*
	Annual Ryegrass	*Lolium multiflorum Lam. Var. Westerwoldicum*
	Bromegrass (smooth brome)	*Bromus inermis (Leyss.) Holib.*
	Smooth bromegrass	*Bromus racemosus L.*
	Quack grass, couch grass	*Elytrigia repens (L.) Desv. Ex Nevski.*
	Siberian crested wheatgrass	*Agropyron fragile (Roth)P. Candargy subsp. sibircum (Willd.)Melderis*
	Standard crested wheatgrass	*Agropyrum desertorum Fisch. Ex Link) Schult.*
	Fairway crested wheatgrass	*Agropyrum cristatum (L.) Gaertn.*
	Timothy grass	*Phleum pratense L.*
	Meadow fescue	*Festuca pratensis Huds.*
	Red fescue	*Festuca rubra L.*
	Redtop (Am) creeping bent grass (Eur)	*Agrostis gigantean Roth (American) or Agrostis stolonifera L. (European)*
	Orchard grass, cocksfoot	*Dactylis glomerata L.*
	Bluegrass, meadow grass	*Poa annua L.*
	Bluegrass, meadow grass	*Poa steppe (Kryl.) Roshev.*
	'Grass'	*Gramineae*
	Reed grass	*Calamagrostis Adans. Spp.*
	Sedge	*Carex L. spp.*
	Sheep fescue	*Festuca ovina L.*
Fodder Leguminous (cultivated species)	Spring vetch (common vetch)	*Vicia sativa L.*
	Leucaena	*Leucaena leucocephala (Lam.) de Wit*
	Desmodium	*Desmodium Desv. spp.*
	Winter vetch (hairy vetch)	*Vicia villosa Roth.*
	Peas (field pea)	*Pisum sativum L. subsp. sativum var. arvense (L.) Poir.*
	Grass peavine, grass pea	*Lathyrus sativus L.*
	Lupin yellow	*Lupinus luteus L.*
	Lupin (blue lupin)	*Lupinus angustifolius L.*
	Seradella	*Ornytorus sativus L. Ornithopus satious Brot.*
	Bean (faba-bean; broad-bean)	*Vicia faba L.*
	Clover (crimson clover)	*Trifolium incarnatum L..*
	Alfalfa	*Medicago lupulina L.*
	Alfalfa blue	*Medicago sativa L.*
	Alfalfa yellow	*Medicago sativa L. falcate (L.)*

TABLE A1. PLANT GROUPS, WITH COMMON AND LATIN NAMES OF ASSOCIATED CROPS (Cont.)

Plant group	Common name	Latin name
	Alfalfa hybrid	*Medicago sativa L. varia (Martyn)*
	Clover red	*Trifolium pratense L.*
	Clover (hybrid clover)	*Trifolium hybridum L.*
	Clover white	*Trifolium repens L.*
	Esparsetter (animal forage)	*Onobrychis Mill.*
	Sweet-clover white	*Melilotus albus Medik.*
	Sweet-clover yellow	*Melilotus officinalis Lam.*
Pasture (species mixture)	Grass-leguminous mixture (festuca+ timothy-clover, oats-clover….)	
	Natural grasses mixture	
	Undefined mixture	
	Canadian thistle	*Cirsium arvense (L.) Scop.*
Herbs	White mustard	*Sinapis alba L.*
	Basil, sweet basil	*Ocimum basilicum L.*
	Nigundi	*Vitex negundo L.*
	Coriander, cilantro	*Coriandrum salivum L.*
	Parsley	*Petroselinum crispum (Mill.) Nyman ex A.W. Hill*
	Spearmint	*Mentha spicata L.*
	Dill	*Anethum graveolens L.*
	African spider-flower	*Cleome gynadra L.*
	Milkweed, crownplant, (giant-milkweed)	*Calotropis gigantea (L.) Dryand. ex W. T. Aiton*
	Cassia	*Cassia tora L.*
	Seaside clerodendrum, (tubbeflower, Turk's-turban)	*Clerodendrum indicum (L.) Kuntze*
	Wild indigo, fish poison	*Tephrosia purpurea (L.) Pers.* *T. sinapou (Buc'hoz) A. Chev.*
	Hogweed (red hogweed, red spiderling)	*Boerhavia L.*
	Indian and leaf mustard	*Brassica juncea L.*
	Tea	*Camella sinensis L.*
	Thyme	*Thymus L.*

TABLE A1. PLANT GROUPS, WITH COMMON AND LATIN NAMES OF ASSOCIATED CROPS (Cont.)

Plant group	Common name	Latin name
Other crops	Rape (winter rape)	*Brassica napus L.*
	Margosa	*Azadirachta indica A. Juss.*
	Walnut	*Juglans regia L.*
	Canola, rape	*Brassica napus L. napus*
	Sunflower	*Helianthus annuus L.*
	Peanut	*Arachis hypogaea L.*
	Flax	*Linum usitatissiumum L.*
	Tobacco	*Nicotiana tabacum L.*

AGRICULTURAL ECOSYSTEMS: FOLIAR UPTAKE

Radionuclides may be released from nuclear facilities to the terrestrial environment as particulates or gases. The deposition of radionuclides on vegetation and soil represents the starting point for their transfer in the terrestrial environment and in food chains. There are two principal deposition processes for the removal of pollutants from the atmosphere: Dry deposition is the direct transfer to and absorption of gases and particles by natural surfaces such as vegetation, whereas wet deposition is the transport of a substance from the atmosphere to the ground within snow, hail or rain.

Interception is defined as the fraction of a radionuclide deposited by dry and wet deposition that is initially retained by the vegetation. Although the activity retained is subsequently removed by weathering to the soil and, the fraction that is initially intercepted is a very important quantity in any radioecological model. This is because direct deposition may cause relatively high activity concentrations in feed and foods.

Once deposited on vegetation, radionuclides are lost from plants due to removal by wind and rain, either through leaching or by cuticular abrasion. The increase of biomass during growth does not cause a loss of activity; but it does lead to a decrease in activity concentration due to effective dilution. Since growth is subject to seasonal variations, the post-deposition decrease of the activity concentration of plants depends on the season.

Translocation describes the systemic transport of radionuclides in the plant subsequent to foliar uptake. Translocation has no or very little influence on the long-term-fate of radioactivity in the environment, since it describes only the distribution of radionuclides within the plant subsequent to foliar deposition and absorption by the leaves. However, for estimating radionuclide concentrations in foods and for the assessment of doses to man, the systemic transport of radionuclides is a key issue. It is especially important for plants from which only specific parts are used as food or feed, such as cereals and potatoes. For plants that are used whole, such as leafy vegetables or maize silage, translocation is relevant only in that it may reduce the amount of activity that is lost by weathering processes.

INTERCEPTION

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Absract

The interception of dry and wet deposits represents the link between the atmospheric dispersion of radionuclides released to the atmosphere and their transport in foodchains. In this section, the processes involved in interception and their dependence on the characteristics of the deposit and the environmental conditions are discussed. For dry deposition, the most important factor is the particle size spectrum which depends on the characteristics of the release and the distance to the release point. Furthermore, the development of the vegetation, whether the plants are dry or wetted, as well as the actual weather conditions have a considerable impact on the dry deposition of radionuclides. For iodine isotopes the chemical form has a dominating influence on the deposition. The interception of wet deposited is the result of a complex interaction of the vegetative development of the plant canopy, the amount of rainfall, and the chemical form of radionuclides. Due to the dependence on plant development, interception of both dry and wet deposits is subject to pronounced seasonality.

1. INTRODUCTION

Interception defines the fraction of radioactivity deposited by wet and dry deposition processes that is initially retained by the plant. This section describes the factors that control interception and discusses existing approaches to estimating interception in radioecological models.

There are several possible ways to quantify the interception of deposited radionuclides. The simplest is the interception fraction *f*, which is defined as the ratio of the activity initially retained by the standing vegetation A_i immediately subsequent to the deposition event to the total activity deposited A_t:

$$f = \frac{A_i}{A_t} \quad (1)$$

The interception fraction is dependent on the stage of development of the plant. To take account of this, in some experiments and models, the interception fraction is normalized to the standing biomass *B* (dry mass). This quantity is denoted as the mass interception fraction f_B:

$$f_B = \frac{f}{B} \quad (2)$$

Since the leaf area represents the main interface between atmosphere and vegetation, the interception fraction *f* is sometimes normalised to the leaf area index (*LAI*), which is defined as the ratio of the (single-sided) leaf area to the soil area:

$$f_{LAI} = \frac{f}{LAI} \quad (3)$$

2. WET DEPOSITION

Radionuclides in air may be washed out by precipitation. The vegetation retains a fraction of radionuclides deposited with rain, whereas the rest falls through the canopy to the ground. The activity retained is subsequently transferred to the soil by weathering and is only temporarily present on the surface of the vegetation. Nevertheless, the fraction that is initially intercepted is a very important quantity in radioecological models. This is because direct deposition can result in relatively high activity concentrations in feed and foods.

2.1. Factors governing interception of wet deposition

The interception of radionuclides is the result of the interaction of various factors, including the stage of development of the plant, the capacity of the canopy to retain water, elemental properties of the radionuclide, and the amount of rain during a rainfall event and the intensity of the precipitation.

2.1.1. Stage of development of the plant

For a specific deposition event, the interception fraction increases with the degree of development of the plant canopy, as this is associated with a larger contact surface between the falling rain and the plant surface. There are two principal approaches to representing plant development, the standing biomass per unit area and the leaf area per unit area (leaf area index: *LAI*).

The advantage of the use of standing biomass is easy determination. The sample has simply to be dried and the mass determined. However, the biomass does not really represent the key characteristics of the rain/plant interface, which are better represented by the *LAI*. Whereas in the early stages of growth, a good relationship between biomass and leaf area can be observed, the correlation decreases towards the end of growth. Then, the biomass continues to increase, due to growth of storage organs such as seeds or tubers, whereas the leaf area decreases substantially, due to dying off of the foliage. So, interception as a function of *LAI* may be reasonably consistent across a range of crop types, whereas interception as a function of standing biomass is likely to be less consistent, requiring the use of crop-specific parameterizations. The disadvantage of the use of *LAI* is a complicated determination that requires specific optical devices, which are usually not available.

The relationship between interception fraction and *LAI* is shown in Fig. 1, where the interception fraction for ^{137}Cs for various species of cereals is plotted against the *LAI* at the time of application [1]. The activity was applied at different stages of development for an artificial rainfall amount of 1 mm. The interception fraction increases in approximate proportion to the leaf area index, though there is some indication of levelling off of the intercepted fraction at large values of the *LAI*.

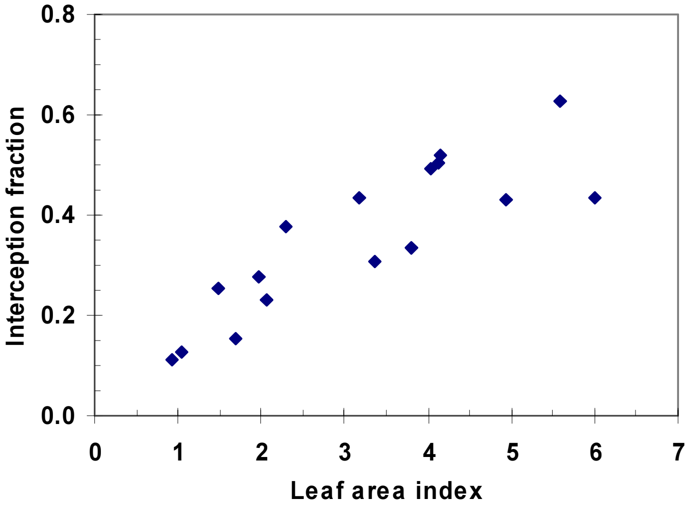

FIG.1. Relationship between the interception fraction for ^{137}Cs by barley and the leaf area index for a simulated rainfall amount of 1 mm [1].

2.1.2. Water storage capacity

The interception of rain by vegetation is closely linked to the water storage capacity of a plant canopy (S). The interception increases during a rainfall event until the interception capacity is achieved and the weight of more rain overcomes the surface tension holding the water on the plants.

The water storage capacity of a plant depends on the surface characteristics of the leaves and the leaf angles. Since interception is an important quantity in assessing the water balance of a given area, the water storage capacity of vegetation has been determined in a number of investigations. It is quantified in terms of the thickness of a water film that covers the foliage. A summary of storage water capacities is given in Table 1. Values reported for the water storage capacity are 0.18-0.38 mm for prairie grass [2], 0.15-0.2 mm for maize [3], 0.35 mm for clover [4]; [5]) and ca. 0.2 for grass [5].

TABLE 1. SUMMARY OF INTERCEPTION MEASUREMENTS FOR WET DEPOSITS

Plant	Water storage capacity	Unit	Type of experiment	Reference
Prairie grass	$(1{,}8\text{-}3.8) \times 10^{-1}$	mm	Field	[2]
Maize	$(1.5\text{-}2) \times 10^{-1}$	mm	Field	[3]
Clover	3.5×10^{-1}	mm	Laboratory	[4-5]
Grass	2×10^{-1}	mm	Laboratory	[5]
Beans	$(1\text{-}4) \times 10^{-1}$	mL/g [a]	Laboratory	[6]
Wheat	$(3\text{-}6) \times 10^{-1}$	mL/g	Laboratory	[6]
Wheat	$(1\text{-}3) \times 10^{-1}$	mm	Laboratory	[6]
Grass	7×10^{-1}	mm	Field	[7]
Crops	3×10^{-1}	mm	Field	[7]
Various plants	$(1\text{-}5) \times 10^{-1}$	mm	Field	[8]
Pine	2.5×10^{-1}	mm	Field	[9]
Wheat	$(1\text{-}3) \times 10^{-1}$	mm	Field	[10]
Boreal forest	1.3-2	mm [b]	Field	[11]

[a] mL water per g leaf ; **[b]** The value gives the total water storage on the canopy il.

Kinnersley et al. [6] determined the water storage capacity of beans and wheat in terms of [mL water per g plant tissue]. There was some dependence on the rainfall intensity; typical values were 0.3 mL/g (0.1-0.4 mL g^{-1}) and 0.5 mL/g (0.3-0.6 mL g^{-1}) for broad beans and wheat, respectively. The differences between broad beans and wheat might have been caused by differences in leaf angles and the specific ratio of mass per unit leaf area. Taking into account the relationship between leaf area and biomass as given in Vandecasteele et al. [12] for wheat, the findings of Kinnersley et al. [6] are consistent with a water storage capacity for wheat in the order of 0.1-0.3 mm.

In the compilation of Breuer et al. [7], mean interception capacities are reported for grass and crop canopies of 1.9 mm and 2.6 mm, respectively. Taking into account the corresponding leaf area indices, water storage capacities per unit leaf area of 0.7 mm and 0.3 mm, respectively, can be estimated.

Keim et al. reported that the leaf area index is the best predictor of storage [8]. For unusual high rainfall intensities of 20 mm h^{-1}, he found storage capacities of 0.1 to 0.5 mm. It is interesting to note that the storage on coniferous species is higher than for broad-leaved species. The storage capacities increase slightly with rainfall intensity. Toba and Ohta [11] observed average interception fractions for water on coniferous trees in Siberia of about 0.29

and 0.36 for total amounts of rainfall of 59 and 49 mm that fell during 13 and 9 rainfall events, respectively, at mean intensities of 0.6 mm h^{-1}. In the same study, interception fractions for water were determined at Japanese sites of 0.13 to 0.24 with average amounts of precipitation per rainfall event of 10-50 mm.

In measurements on *Pinus Koraiensis Nakai*, Anzhi et al. have determined water storage capacities of about 0.25 mm for rainfall intensities of about 50 mm h^{-1} [9]. The capacity dropped continuously to 0.15 mm, when the intensity increased to 150 mm h^{-1}. This intensity is extraordinarily high for field conditions. Furthermore, Anzhi et al. [9] studied the relationship between water storage capacity, LAI, and rainfall intensity. They found a linear relationship between water storage capacity and *LAI*. The water storage capacity was much less dependent on the rainfall intensity. For a given *LAI*, the storage capacity dropped by 20 %, when the rainfall intensity increased by a factor of 3.

Anzhi et al. [9] also investigated the water storage as a function of the wetness of the canopy. Their experimental design is appropriate to estimation of the interception of intermittent rain. It was found that the interception of water on wet canopies decreases as the wetness of the canopy increases.

During sprinkler irrigation experiments with winter wheat in China, Kang et al. [10] found a very good correlation between *LAI* and water storage. The estimated amounts of water stored by the canopy were 0.2 mm, 0.6 mm, 0.9 mm, and 0.6 mm for *LAI* values of 2.3, 2.8, 6.6, and 6.7, respectively. A trend to lower water storage was observed when the wind speed increased.

2.1.3. Element properties

The interception of a wet-deposited radionuclide is controlled by the storage capacity of water and the interaction of the radionuclide with the leaf surface, which strongly depends on the chemical form of the deposit. The differences in interception between different elements are due to their different valencies. As plant surfaces are negatively charged, they have properties of a cation exchanger [13]. Therefore, the initial retention of anions such as iodide is less than for polyvalent cations, which seem to be very effectively retained on the plant surface. In Table 2, the mass interception factor (f_B) values, as measured in Neuherberg (Germany) and corrected for radioactive decay, are given for the period of May 1-6, 1986, for ^{106}Ru, ^{131}I, ^{137}Cs and ^{140}Ba [14]. Since most of the deposition occurred during a heavy shower on April, 30 and May 1, dry deposition of ^{106}Ru, ^{137}Cs, and ^{140}Ba did not contribute significantly to the total deposition [15] The mass interception factors increase in the order ^{106}Ru, ^{131}I, ^{137}Cs, ^{140}Ba, with these radionuclides having been deposited during the same rainfall event. The highest values were observed for ^{140}Ba, which behaves similarly to strontium. Barium is a bivalent cation, and seems to be more strongly retained on the negatively charged plant surface than the monovalent caesium cation. The mass interception factor is smaller for iodine, which was probably deposited as a monovalent anion on the plant surface.

The mass interception factors for ^{106}Ru are slightly lower than those for ^{131}I. However, accounting for the contribution of dry deposition due to the presence of elemental iodine, this may indicate that ruthenium behaved similarly to iodine in respect of the interception of a wet deposit. However, since the information about the chemical form of ruthenium is very limited and speculative, it cannot be determined whether this observation is typical for ruthenium or specific for the Chernobyl accident. In the following days, the differences in the mass interception factors were less than on the first day, due to the increasing contribution of dry deposition in that period during which only small amounts of precipitation fell.

These observations agree very well with those reported by Hoffman et al. [16] who measured the mass interception factors for a variety of anions and cations on the leaves of several tree species (Fig. 2) during a simulated shower of 8.5 mm. The mass interception factors for cations are approximately a factor of 3 to 5 higher than for anions such as iodide or sulphate. Compared with the effects of valence, the data of [16] suggest that plant properties are of less importance.

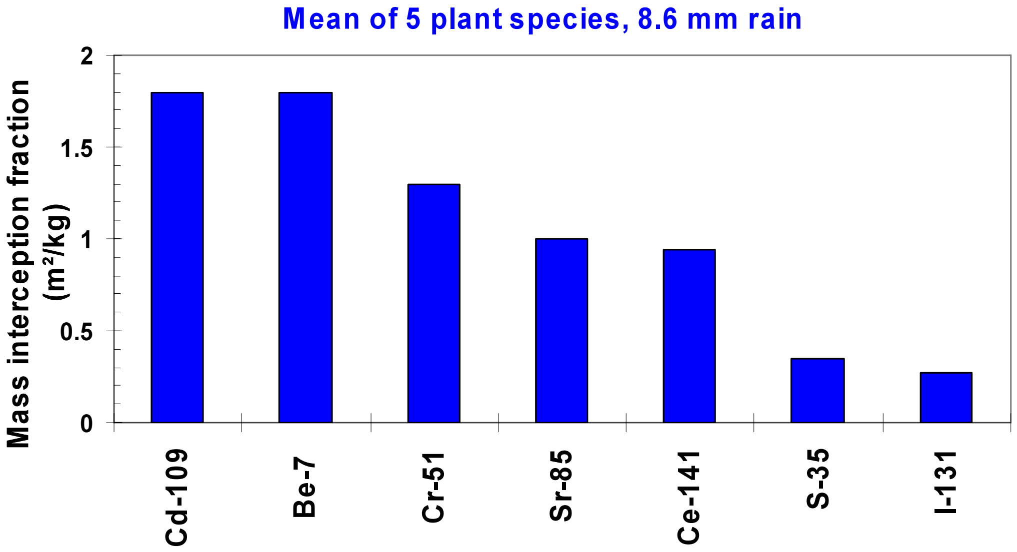

FIG. 2. Comparison of the mass interception fractions for radionuclides in cationic and anionic forms [16].

2.1.4. Amount of rain

Since the capacity of plant canopy to retain water is limited, the interception fraction decreases in general with increasing amount of rainfall in a rainfall event.

The effect of the amount of rainfall applied on the mass interception factors is obvious from Fig. 3, where the interception factor values for ^{137}Cs are plotted against the amounts of rainfall for two rainfall intensities [6]. The data presented demonstrate that the interception factor is inversely proportional to the amount of rainfall. Although the rainfall intensity varied by a factor of 3, its impact on the interception was of minor importance.

2.2. Measured values for wet deposition

As discussed above, the interception fraction is the result of a complex interaction of plant development, rainfall and radionuclide chemical properties. So, dependent on the circumstances, the interception fraction varies from close to zero to almost one. Experimentally determined interception fractions are summarized in Table 2 by element, crop, stage of development and amount of rainfall. Unfortunately, there is no generally agreed quantity to express interception. Often the interception fraction f is used, which is defined as the fraction of the deposited material that is initially retained by vegetation.

TABLE 2. SUMMARY OF INTERCEPTION MEASUREMENTS FOR WET DEPOSITION

Element/ conditions	Crop	Standing biomass	Amount of rainfall (mm)	Interception fraction	Mass interception fraction ($m^2 kg^{-1}$)	Reference
Chernobyl						
^{140}Ba	Grass		5.6		1.7	[11-14]
^{137}Cs					1.1	
^{131}I					7×10^{-1}	
^{106}Ru					4.8×10^{-1}	
Simulated Rain						
^{131}I	Grass		1		4.3	[5-17]
	Clover				8.7	
	Grass		2		1.6	
	Clover				4.1	
	Grass		4		1.1	
	Clover				2.5	
^{90}Sr	Grass		1		7.6	
	Clover				8.2	
	Grass		2		5.1	
	Clover				8.0	
	Grass		4		4.8	
	Clover				8.2	
^{137}Cs	Wheat	n.a.	4×10^{-1} [b]	3×10^{-2} [c]	1.4	[6]
			7×10^{-1}	7.4×10^{-2}	3.5	
			1.5	2.9×10^{-2}	1.4	
			4.4	2.4×10^{-2}	1.2	
			8.9	1.4×10^{-2}	5×10^{-1}	
	Beans	n.a.	3.4×10^{-1}	5.9×10^{-2}	2.1	
			6.8×10^{-1}	3.1×10^{-2}	1.1	
			1.4	3.9×10^{-2}	1.4	
			4.1	1×10^{-2}	4×10^{-1}	
			8.2	1.3×10^{-2}	5×10^{-1}	
	Grass	n.a.	4.5×10^{-1}	1.8×10^{-1}	4.6	
			9×10^{-1}	2.1×10^{-1}	5.5	
			1.8	1.1×10^{-1}	2.8	
			5.4	3.6×10^{-2}	9×10^{-1}	
			1.1×10^{1}	2.7×10^{-2}	7×10^{-1}	
Pure water[a]	Grass		1		6.2	
	Clover				1.1×10^{1}	
	Grass		2		4.3	
	Clover				5.9	
	Grass		4		1.8	

	Clover				4.0	
^{7}Be	Pasture		1		3.2±0.91	[18]
			10		1.4±0.86	
^{131}I			1		2.3±0.48	
			10		0.26±0.13	
^{7}Be	Mean of 5 species	n.a.	8.5	1.8		[16]
^{109}Cd				1.8		
^{144}Ce				9.4×10^{-1}		
^{51}Cr				1.3		
^{131}I				2.7×10^{-1}		
^{35}S				3.5×10^{-1}		
^{85}Sr				1		
Simulated very fine drizzle, no water run-off from the foliage						
Mixture of radionuclides	Rice	8×10^{-2}	$(3\text{-}4) \times 10^{-2}$	4.8×10^{-1}	6.0	[19-21]
		3.9×10^{-1}		7.9×10^{-1}	2.1	
		9.3×10^{-1}		8.8×10^{-1}	9.5×10^{-1}	
		1.0		8.7×10^{-1}	8.4×10^{-1}	
		1.7		9.4×10^{-1}	5.5×10^{-1}	
		1.9		9.4×10^{-1}	4.9×10^{-1}	
	Soybean	2×10^{-2}		3.4×10^{-1}	1.7×10^{1}	
		1.3×10^{-1}		8.3×10^{-1}	6.6	
		4.4×10^{-1}		9.3×10^{-1}	2.1	
		7.4×10^{-1}		8.8×10^{-1}	1.2	
		7.9×10^{-1}		8.4×10^{-1}	1.1	
		6.3×10^{-1}		4.5×10^{-1}	7.1×10^{-1}	
	Chinese cabbage	1×10^{-2}		1.6×10^{-1}	3.0×10^{1}	
		3×10^{-2}		5.9×10^{-1}	1.9×10^{1}	
		1.0×10^{-1}		7.7×10^{-1}	7.8	
		1.5×10^{-1}		8.3×10^{-1}	5.6	
		2.9×10^{-1}		8.7×10^{-1}	3.0	
	Radish	1×10^{-2}		1.8×10^{-1}	1.6×10^{1}	
		4×10^{-2}		6.7×10^{-1}	1.5×10^{1}	
		9×10^{-2}		8.2×10^{-1}	9.3	
		1.5×10^{-1}		8.6×10^{-1}	5.9	
		1.7×10^{-1}		8.6×10^{-1}	5.2	

[a]Retention of radionuclide-free water [b]Rainfall intensity: 4.4 mm/h; [c] *LAIF*: Interception fraction per unit leaf area.

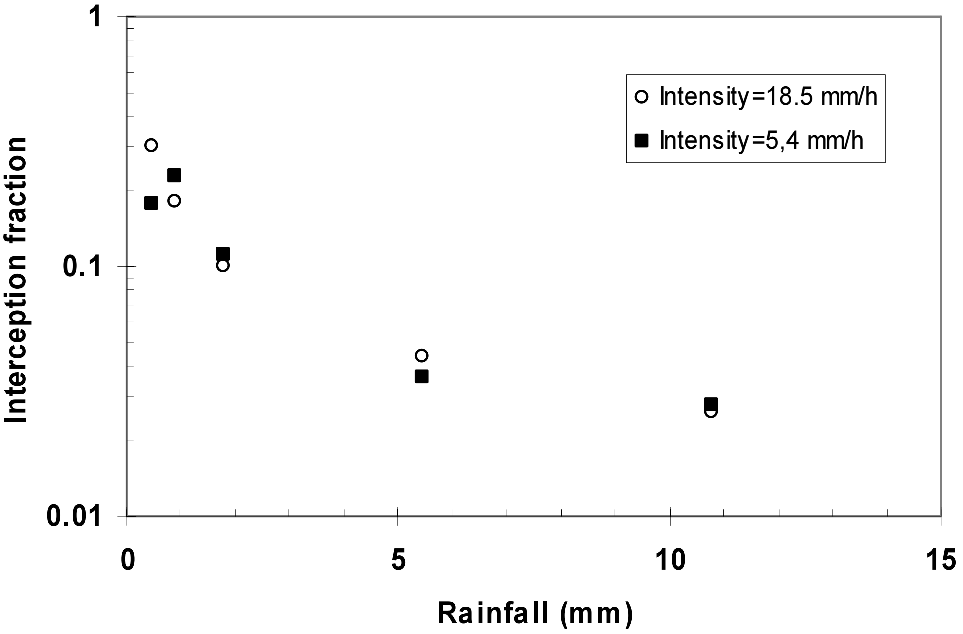

FIG. 3. Interception of ^{137}Cs by grass as function of the amount of rainfall and rainfall intensity [6].

To account for the plant development, in some experiments the interception fraction was normalized to the standing biomass B to obtain the mass interception factor f / B (m^2 kg^{-1}), which facilitates the comparison of experiments performed with different values of biomass. However, normalizing the interception fraction is only justified as long as the exposed leaf area increases in proportion to the standing biomass. This is only the case in the first part of the growth. In the second part of the growing period the biomass still increases, whereas the leaf area decreases [22]. Furthermore, in all estimations of f or f_B, it should be carefully checked whether the deposition is dry or wet, since the deposition mode has a large influence on the interception fraction, as is discussed below. The values are therefore only warranted if the deposition scenario under which f or f_B is to be applied is similar to the situation for which these parameters were derived.

In Table 2, only measurements designed to determine interception fractions were included. There are a number of experiments in which radionuclides dissolved in water were sprayed on leafs of plants with the primary objective of determining post-deposition retention and systemic transport. In those experiments, the application technique is not consistent with natural rainfall, since very low amounts of water and very fine sprays were used to optimise the wetting of the plant. Also, the spray often was targeted to the plants to minimise initial losses of radionuclides.

3. DRY DEPOSITION

3.1. Definitions

Absolute rates of dry deposition and interception of dry deposits by vegetation need to be considered together. Dry deposition, quantified through use of a deposition velocity, is dependent on the characteristics of the surface, which is often parameterised by the surface roughness [23], which, in general, increases with the development of the plant canopy. This means that the deposition increases with the development of the canopy.

Chamberlain [24] defined the interception fraction f (eq. 1) of dry deposition in terms of a dependence on the standing biomass B (kg/m², dry mass) and the empirically derived mass interception coefficient α :

$$f = 1 - \exp(-\alpha \cdot B) \tag{4}$$

The mass interception fraction f_B (eq. 2) is then derived to take into account the dependence of the interception fraction on the biomass B using:

$$f_B = \frac{1 - \exp(-\alpha \cdot B)}{B}. \tag{5}$$

For small standing biomass, there is little difference between f_B and α . However, the validity of the equations above is limited to that part of the growing period during which the exposed leaf area increases in proportion to the standing biomass. This is only the case in the first part of the growing period; in the second part the biomass continues to increase, whereas the leaf area decreases [12, 25].

3.2. Factors governing interception of dry deposition

The interception fraction f for dry deposition depends on the standing biomass, the chemical form and the particle size of the deposit.

The mass interception coefficient appears to be similar for small (up to a few micrometers diameter) particles and elemental iodine. With increasing particle size, α decreases considerably. This is probably due to the effect that larger particles roll off the plant surface more easily than smaller ones. It was found that in those experiments in which the vegetation was moist or wet, the observed absorption coefficients were considerably higher, which is probably due to the enhanced sticking effect. This is confirmed by the observations reported in Pinder et al. [26] who found lower interception fractions on plants with waxy leaf surfaces. However, with the exception of the near-field of the release point, for the purpose of dose assessment, particles with a diameter up to a few micrometers are most relevant, since the depletion of larger particles from a radioactive cloud is very effective.

3.3. Measured values for dry deposition

As for wet deposits, the interception of dry deposits is the result of a complex interaction of various factors, such as the yield, the particle size, the crop, the chemical form and the characteristics of the surface, e.g. whether it is wet or dry.

The results from experiments or monitoring measurements are summarised in Table 3. To facilitate comparisons, the results are given in terms of the mass interception coefficient α .

4. MODELS FOR ESTIMATING INTERCEPTION OF DEPOSITION FOR SINGLE EVENTS

The classical model for interception of dry deposition is that described by Chamberlain [24] and set out above. Kinnersley et al. [27] suggested that due to the sedimentation characteristic of rain drops, this approach can also be applied to model interception of both, dry and wet deposits. To account for the correlation of f with B, the mass interception fraction f_B for wet deposition is calculated in analogy to equation 5.

TABLE 3. SUMMARY OF INTERCEPTION MEASUREMENTS OBTAINED FROM FIELD EXPERIMENTS FOR DRY DEPOSITS

Deposited material	Diameter (μm)	Crop	Interception fraction (*f*) Interception coefficient [$m^2\ kg^{-1}$] (α)	Reference
			Mean±SD	
Lycodium spores	32	Grass	3.1±0.15 (α)	[24]
		Wheat, dry	3.2±0.5 (α)	[24]
		Wheat, moist	9.6±3.7 (α)	[24]
Quartz particles	44-88	Grass	2.7±0.3 (α)	[28]
Sand particles	40-63	Grass, dry	0.44±0.15 (α)	[29]
		Grass, wet	0.88±0.13 (α)	[29]
	63-100	Grass, dry	0.23±0.07 (α)	[29]
		Grass, wet	0.69±0.16 (α)	[29]
	100-200	Grass, dry	0.24±0.07 (α)	[29]
		Grass, wet	0.46±0.11 (α)	[29]
^{238}Pu particles	≈1	Corn	3.6±0.05 (α)	[30]
^{131}I vapour		Grass	2.8±0.1 (α)	[30]
^{212}Pb vapour		Artificial grass	13 (α)	[31]
^{85}Sr, ^{133}Ba, ^{137}Cs	3.5	Beans 30 d [a]	1-1.2 (α)	[32]
		45 d	1.1 (α)	[32]
		65 d	0.85-0.93 (α)	[32]
^{85}Sr, ^{133}Ba, ^{137}Cs, ^{123}Te		85 d	0.3 (α)	[32] [32]
^{85}Sr, ^{133}Ba, ^{137}Cs, ^{123}Te	3.5	Grass	0.84±0.06 (f) 3.27±1.15 (α)	[32] [32]
^{137}Cs, ^{85}Sr		Wheat	$f = 1 - \exp(-0.316 \cdot LAI)$ $f = 0.85 \cdot (1 - \exp(-13.1 \cdot B))^{b}$	[12] [12]
Spherical porous silica particles	4	Lettuce	0.71±0.1 (f)	[33]
	10		0.88±0.07 (f)	[33]
	18		0.88±0.08 (f)	[33]
	22		0.81±0.23 (f)	[33]
	4	Wheat	0.56±0.29 (f)	[33]
	22		0.65±0.13 (f)	[33]
Spherical porous silica particles	4	Wheat	1.6 (f)	[34]
	22		1.2 (f)	[34]
Uranium particles (wind tunnel)	0.82	Spruce (LAI=3.1)	0.97 (f)	[35]
^{90}Sr, ^{137}Cs		Rice	0.04-0.12 (f / LAI), n=6	[36]
		Wheat	0.05-0.09 (f / LAI), n=2	[36] [36]
		Carrot	0.1-0.3 (f / LAI), n=2	[36]
		Cabbage	0.18-0.2 (f / LAI), n=2	[36]
		Tomato	0.08-0.9 (f / LAI), n=2	[36]

[a] Days after sowing, [b] *B*–Yield (kg m^{-2}, dry mass).

Most of the work has been done for pasture grass and little for other crop types [27]. Vandecasteele et al. [12] concluded that the approach of Chamberlain [24] is only adequate for plants that are in the vegetative period, when a good correlation exists between biomass and leaf area.

However, when canopies switch from vegetative to generative development, biomass continues to increases, whereas the leaf area remains constant and subsequently declines. Therefore, Vandecasteele et al. [12] have proposed a modified model as:

$$f = 1 - e^{-\beta \cdot LAI} \tag{6}$$

and

$$f_{LAI} = \frac{1 - e^{-\beta \cdot LAI}}{LAI} \tag{7}$$

where β is the interception coefficient referred to LAI, and is dimensionless.

Figs. 4 and 5 compare the dependence of the interception fraction f of particles by wheat plants on the biomass (Fig. 5) and on the leaf area (Fíg. 6) [12]. In both cases, a good correlation has been found.

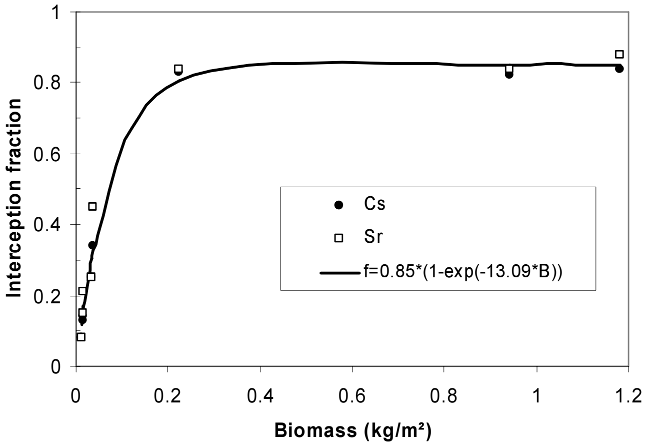

FIG. 4. Relationship between the interception fraction f of ^{137}Cs and ^{90}Sr particles on wheat and the above-ground biomass (DW) [12].

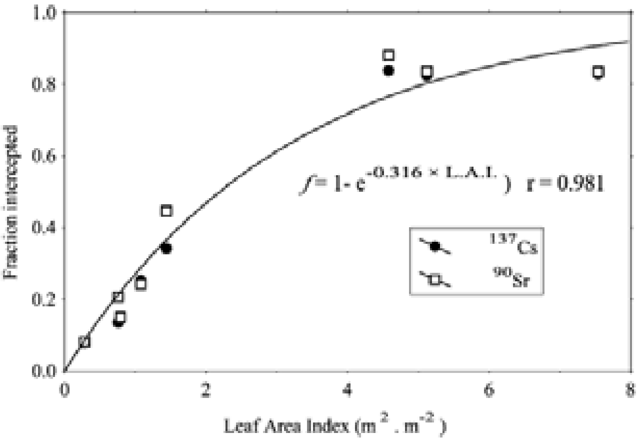

FIG. 5. Relationship between the interception fraction f for ^{137}Cs and ^{90}Sr particles on wheat and the leaf area index, LAI [12].

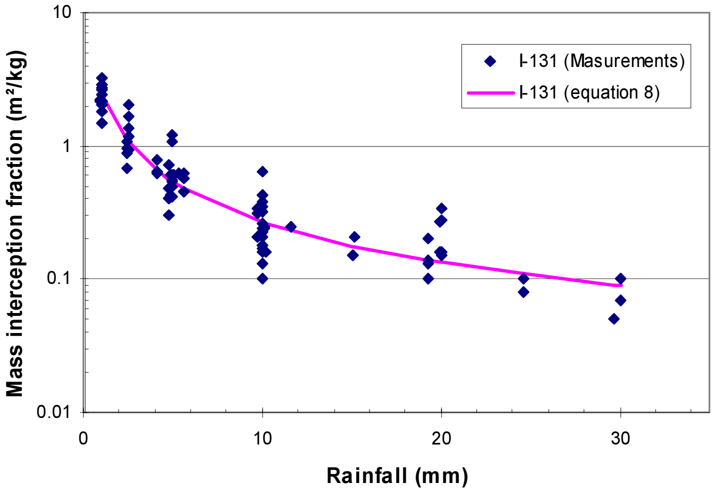

FIG. 6. Comparison of measured mass interception fractions for ^{131}I *on grass [16] and the model approach represented [37].*

Kinnersley et al. [27] stated that, for wet deposition, α varies with vegetation type, amount of rainfall and element. These dependencies are taken into account in the approach described in Müller and Pröhl [36] who modelled the interception fraction for wet deposits as a function of the leaf area index *LAI*, the storage capacity of the plant *S*, an element-dependent factor *k* that quantifies the ability of the element to be attached to the leaves and the total amount of rainfall *R* that falls during a single event:

$$f = \min(1; \frac{LAI \cdot k \cdot S}{R}\left[1 - e^{-\frac{\ln(2)}{3\cdot k\cdot\cdot S}\cdot R}\right]) \qquad (8)$$

For *k*, values of 0.5, 1 and 2 are assumed for anions (iodide, sulphate), monovalent cations (e.g. Cs) and polyvalent cations, respectively. For the water storage capacity, 0.2 mm is assumed for grass, cereals and corn, and 0.3 mm for all other crops.

The approach shown in equation 8 is consistent with the findings of Kinnersley et al. [6], who carried out interception experiments for grass, broad beans, and wheat. The interception decreased with increasing rainfall, since the plant surface approached a saturation level of contamination. For the interception fraction, the total amount of rainfall was more important than the rainfall intensity. The water storage capacity for beans and wheat was found to approach saturation as total rainfall increased in a manner similar to that of the measured contamination levels. The findings obtained confirmed that interception of wet deposits can be modelled appropriately by a water storage capacity term and a 'chemical term' derived from the affinity of a particular chemical form (or valence state) for a particular plant surface.

Good agreement of this approach (equation 8) is found with the interception of ^{131}I (anions) measured by Hoffman et al. (1995) [16] in field experiments (Fig. 6). However, for rainfall events above 10 mm, equation 8 seems to under-predict the interception of the cationic ^{7}Be (Fig. 7) that was used in the same experiments. Due to the negative charge of the plant surface, cations are apparently intercepted more effectively than would be expected from the retention of the water film.

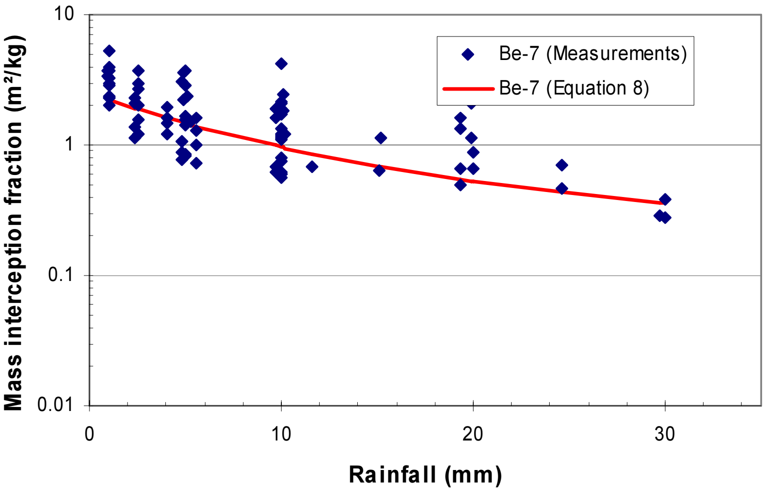

FIG. 7. Comparison of measured mass interception fractions for ^{7}Be on grass [16] and the model approach represented [37].

5. CONCLUSIONS

For the interception of dry and wet deposits by vegetation, the development of the plant canopy is a key factor. The biomass density or the leaf area index may be used to quantify the development of the plant. During vegetative growth, both approaches are equally appropriate, whereas, during the generative phase, the leaf area index is a more adequate basis for interception modelling. In this phase, the biomass increases whereas the leaf area declines. Variations in the degree of interception of both dry and wet deposits are less if it is normalised to the standing biomass or to the leaf area index.

For dry deposition, the particle size is the other key parameter. Interception is more effective for small particles and reactive gases. Interception of wet-deposited radionuclides is a result of the complex interaction of the chemical form of the element, the development of the plant, and the amount of rainfall. It has been found that rainfall intensity appears to be of minor importance in determining interception.

The existing data show the interception of both dry and wet deposits depends on the chemical form of the deposit and its interaction with the plant surface and the canopy structure. A deeper knowledge about the processes involved would considerably improve the predictive power of the models applied so far.

Specific data for interception are available only for a few elements, such as caesium, strontium and iodine. Single estimates are available for a few other elements. This means, for the majority of elements, interception is derived from analogy based on the assumed chemical form and valence of the element.

In case of long-term precipitation and irrigation, there are indications that the interception of wet deposited polyvalent cationic radionuclides may be a factor 2 or 3 larger than given by the approach based on water storage capacity and leaf area index. Due to the negative charge of the plant surface, the radionuclide retention of cations by the leaf is enhanced. The existing data are not sufficient to provide a reliable quantification of such effects.

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WEATHERING

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Abstract

Weathering is a process of clearance of contamination from plants by rain or irrigation, surface abrasion, tissue ageing, leaf fall and many other processes. Available data summarised for different elements and plant groups are given in the paper. It was found that the magnitude of the weathering depends on many factors such as solubility of radionuclides, strength of adsorption to the plant surface, degree of penetration into the inner flesh, leachability from the interior and biological factors. The results have shown limited differences between cationic species (Mn, Co, Sr, Ru, Cs) for most plant species, but shown that the weathering rate is strongly dependent on plant characteristics as well as plant growth stage at the time of deposition.

Weathering is a process of clearance of contamination from plants. Weathering includes contributions from the wash-off of previously intercepted material by rain or irrigation, surface abrasion and leaf bending from the action of the wind, resuspension, tissue ageing, leaf fall or herbivore grazing, addition of new tissue (growth dilution), volatilization or evaporation.

The magnitude of the weathering loss of a radionuclide depends on many factors including its solubility, strength of adsorption to the plant surface, degree of penetration into the inner flesh and leachability from the interior. Biological factors such as the structure of the epidermis, plant senescence and defoliation, and shedding of old epicuticular wax also play a part in the weathering process. It can be inferred that a very complex interaction of those factors may be the cause of the observed difference in weathering loss among radionuclides, plant species and their growth stages.

Weathering is normally described in radioecological models by a single exponential function characterised by a first-order rate constant λ_w or a weathering half life T_w.

$$\lambda_w = \frac{Ln(2)}{T_w} \tag{1}$$

Results from numerous studies showed limited differences between cationic species (Mn, Co, Sr, Ru, Cs) for most plant species, but showed T_w to be dependent on plant characteristics as well as plant growth stage at the time of deposition [1]. Available data summarised for differing elements and plant groups are given in Table 1 and 2.

TABLE 1. VARIATIONS IN WEATHERING RATE FOR DIFFERENT ELEMENTS AND PLANTS[1]

Element	Plant group	T_w (days)	N	Range (days)	References
Cs	Cereal	35	1		[2]
Cs	Grass	10	4	7.9 to 11.1	[3-5]
I	Grass	13	9	8.3 to 29	[3; 5-9]
I	Rice	14	1		[8]
Mn-Ce	Cereals	30	1		[2]
Pu	Cereals	12	1		[11]
Pu	Fruits	43	1		[12]
Sr	Grass	24	4	12.9 to 49	[9; 10]
Sr	Cereals	21	1		[2]

[1] Including the effect of growth dilution.

TABLE 2. VARIATIONS IN WEATHERING RATE FOR DIFFERENT ELEMENTS, PLANTS AND PLANT GROWTH STAGES [1, 12, 13-15]

Element	Plant group	T_w (days)	Days until harvest	References
Co	Rice	43.2	112	[14]
		33.7	84	
		40.5	62	
		31.5	49	
		27.3	35	
		14.1	17	
Co	Roots crops (radish)	28.7	55	[14]
		25.0	42	
		17.1	31	
		12.6	20	
		9.4	12	
Co	Leafy vegetables (Chinese cabbage)	15.6	51	[13]
		15.3	39	
		14.9	28	
		12.9	17	
		10.0	7	
Cs	Rice	35.6	112	[14]
		44.1	84	
		42.1	62	
		49.4	49	
		39.2	35	
		32.1	17	
Cs	Leguminous vegetables (Soybean)	31.5	105	[1]
		58.9	84	
		52.6	66	
		28.0	49	
		14.5	34	
		6.2	17	
Cs	Leafy vegetables (Chinese cabbage)	32.0	51	[15]
		34.5	39	
		31.1	28	
		22.4	17	
		11.5	7	
Cs	Root crops (radish)	36.7	55	[14]
		34.3	42	
		25.1	31	
		21.1	20	
		16.0	12	

TABLE 2. VARIATIONS IN WEATHERING RATE FOR DIFFERENT ELEMENTS, PLANTS AND PLANT GROWTH STAGES [1, 12, 13-15] (Cont.)

Element	Plant group	T_w (days)	Days until harvest	References
I	Rice	45.3	99	[13]
		38.3	85	
		17.2	64	
		12.1	57	
		13.1	52	
		10.8	43	
		7.8	40	
		6.4	29	
		2.4	9	
Mn-Ce	Rice	47.2	112	[14]
		44.5	84	
		42.3	62	
		32.8	49	
		27.9	35	
		15.8	17	
Mn-Ce	Leafy vegetables (Chinese cabbage)	16.5	51	[13]
		19.9	39	
		18.9	28	
		15.0	17	
		9.3	7	
Mn-Ce	Root crops (radish)	19.9	55	[14]
		25.2	42	
		19.2	31	
		14.9	20	
		11.2	12	
Ru	Rice	52.7	112	[14]
		49.7	84	
		40.9	62	
		46.4	49	
		35.0	35	
		24.4	17	
Ru	Leafy vegetables (Chinese cabbage)	18.8	51	[15]
		29.3	39	
		31.6	28	
		25.8	17	
		16.7	7	
Ru	Roots crops (radish)	20.1	55	[14]
		20.7	42	
		22.8	31	
		21.6	20	
		26.1	12	
Sr	Root crops (Radish)	21.0	55	[14]
		20.4	42	
Sr	Soybeen	15.6	31	[1]
		10.6	20	
		8.6	12	
Sr	Rice	46.9	112	[14]
		40.2	84	
		39.6	62	
		31.8	49	
		27.1	35	
		13.7	17	

Sr	Leguminous vegetables (Soybean)	12.5	105	[1]
		11.4	84	
		9.7	66	
		6.5	49	
		5.3	34	
		4.4	17	
Sr	Leafy vegetables (Chinese cabbage)	19.1	51	[15]
		22.1	39	
		16.7	28	
		14.4	17	
		9.0	7	

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TRANSLOCATION

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Abstract

Translocation is the process leading to the redistribution of radionuclides from the contaminated aerial parts of a plant to the other parts that have not been contaminated directly. Translocation factors presented in this paper were derived from 52 publications corresponding either to the original documents published by the data providers or to compilations. As a result of statistical analysis of the raw data, the mean and ranges in variation of the translocation factors were eastimated for 19 radionuclides. Most information was for caesium and strontium. For cereals, data are given for different stages of the plants development.

1. DEFINITIONS

Translocation is the phenomenon leading to the redistribution of a chemical substance, once it has been deposited on the aerial parts of a plant, to the other parts that have not been contaminated directly. The direct contamination of plants by radionuclides or toxic elements and their transfer from the foliage to edible parts depends on many physical, chemical and biological factors [1-3]. Physical factors include characteristics of the deposition regime and of the contaminants (rain duration, size of particles) and of the plant (foliage layout, leaf size and cuticular structure). Chemical factors include the speciation of the pollutant, water composition and cuticle composition [4-6]. Biological factors are mainly associated with the vegetative cycle at the time of the foliar deposit [7, 8-15]. Translocation partly reflects the mobility of an element within the plant.

The translocation factor represents the ratio between the transferred activity in the organs of the plant that were not contaminated directly and the deposited activity on the plant itself. Its expression and, therefore its meaning, vary among authors. Three most widespread definitions include:

1. f_{tr1} is the ratio between the concentration in the edible part at harvest time ($Bq \cdot kg^{-1}$) and the concentration in the foliage or in the whole plant at harvest time ($Bq \cdot kg^{-1}$), [3, 16, 17];
2. f_{tr2} (dimensionless or $m^2.kg^{-1}$) is the ratio between the concentration in the edible part at harvest time ($Bq \cdot kg^{-1}$) and the concentration in the foliage ($Bq \cdot kg^{-1}$) [17] or the activity per square metre of foliage at the time of deposit ($Bq \cdot m^{-2}$) [18, 19];
3. f_{tr3} is the ratio between the activity of the edible part within 1 m^2 of crops at harvest time ($Bq\ m^{-2}$) and the foliage activity of 1 m^2 of crops at the time of deposit ($Bq\ m^{-2}$) [17 and 20]. The value may be expressed as a percentage (%) [7, 3, 8-10].

In this review, the translocation factor is expressed in accordance with the f_{tr3} definition and in the form of a percentage. When the percentage was not mentioned directly in a cited publication, it was calculated on the basis of available data (biomass at deposit time and crop yield at harvest time, interception factor, etc.).

In contrast with other transfer parameters, particularly with regard to soil-plant transfers, no experimental method has been standardised so far. Hence, there are as many experimental protocols as there are experiments, and results remain very heterogeneous. Variants concern mainly the contamination mode and the plant-growth stage of the crops at the time of foliar deposit. With regard to the contamination mode, the main variants include:

- simulations of sprinkling irrigation or contaminating rain at various timescales and intensities over the whole vegetation cover with or without soil protection, followed or not by non-contaminated rainfalls. That operating mode is the most realistic for investigation purposes;

- sprays of contaminated solution over the foliage, followed or not after drying by a non-contaminated rain deposit;
- foliar contamination by using a deposit of dry or wet aerosols, followed or not after drying by a non-contaminated rain deposit;
- deposit of droplets over part of or all the plant foliage with a view to detecting translocation and mobility mechanisms within the plant. However, the method is unable to determine a translocation factor, as defined in this study.

With respect to the plant-growth stage at the time of deposit, only a few authors have been able to describe it precisely. They often limited themselves to mentioning the time between sowing and contamination or between contamination and harvesting, rather than the indication of a specific physiological stage. Comparing the results of different authors becomes, therefore, all the more difficult.

Whatever the contamination mode, the translocation-factor values drawn from the literature have a dual origin: open-field experiments or greenhouse tests.

2. DATABASE FOR TRANSLOCATION FACTOR VALUES

Translocation-factor values were drawn from 48 publications corresponding either to the original documents published by the data providers or to compilations by Carini et al. [10, 11, 20] when it was impossible to consult the initial source publications, because they were unavailable (e.g. internal reports of some organisations).

The majority of collected and usable translocation-factor values for the purposes of this study come from relatively old documents published before the publication in 1994 of IAEA's Handbook of Parameter Values for Radionuclide Transfers in Temperate Environment [21]. All later data originate from the following sources:

- three publications [22, 17, 24] dealing with cereal grains;
- three publications [22, 25, 26] dealing with root vegetables;
- one publication [26] dealing with leaf vegetables;
- eight publications [15, 25, 27-32] dealing with fruits.

The following descriptive parameters have been selected for the translocation-factor database:

TABLE 1. STRUCTURE OF THE DATABASE USED FOR THE ASSESSMENT OF TRANSLOCATION FACTOR VALUES

Item	Information given
Plant species	Cultivated plant species and, if appropriate, variety
Element	Name of element under study, atomic number and, if possible, chemical form of the element
Translocation factor	Value obtained for the edible part in percentage and, if appropriate, in $m^2 \cdot kg^{-1}$
Distribution at harvest time	Percentage of the activity contained in the edible part in relation to the whole plant
Transfer factor	Ratio between the specific activity of edible organs at harvest time ($Bq \cdot kg^{-1}$) and the initial surface activity ($Bq.m^{-2}$) of the total deposit (soil + plant) or the specific activity of the irrigation water ($Bq \cdot l^{-1}$)
Yield	Yield of edible parts at harvest time ($kg \cdot m^{-2}$ or $kg \cdot plant^{-1}$)
Type of contamination	Deposit modality of the element on the vegetation cover, sprinkling irrigation, rain, aerosols, spray
Time between foliar deposit and harvesting	Number of days between contamination and harvesting of the edible part of the plant
Duration of the plant cycle	Number of days between sowing and harvesting
Type of test	Open-field experimentation or greenhouse test
Species	Cultivated plant species and, if appropriate, variety
Consideration given to the value	Relevancy assessment of the value in relation to this study
Comments	All additional useful information
Data supplier	Original publication of the data supplier
Bibliographical reference of data	Publication(s) from which data was drawn

The list of descriptive parameters relies in principle on acquired information concerning foliar-transfer mechanisms. Its purpose is to assess the best relevant data. Commonly, all parameters are rarely available.

After examining the entire set of 682 values included in the database, 537 were selected as source data. The appreciation criteria for selecting the translocation-factor values to be used as the basis to determining a reference value and a corresponding range of potential variations rely mostly on the contamination mode of the whole plant. The selected values are those of a deposit that closely simulates a sprinkling irrigation. Hence, all values resulting from specific contamination processes (drop deposit on leaves or fruit), which are very distant from conditions likely to occur in agricultural ecosystems, were discarded. Other values were rejected either because they did not seem to apply only to translocation (direct contamination of a large fraction on the edible part) or because they were expressed in inappropriate units that could not be converted due to a lack of information on intermediate quantities.

The translocation-factor values covered in this study, with all isotopes taken into account, are distributed as follows: 66% for grains, 19% for fruit and 15% for root vegetables.

The majority of experiments dealt with caesium and strontium (Fig. 1). Some other radionuclides, such as radioisotopes of manganese, cobalt, ruthenium and cerium, were addressed in a few investigations and provided between 20 and 40 translocation-factor values, with all plant types taken into account. With regard to the other elements, their processing is more complicated, due to the small amount of data. Translocation-factor values for grain relate to a relatively large range of elements compared with other crops.

For leaf vegetables and grass, translocation is not considered because of contamination mainly by direct deposition.

3. TRANSLOCATION FACTORS VALUES

3.1. Cereals

With all isotopes taken into account, most translocation-factor values found in the literature concern wheat and barley. Most studies dealt with caesium and strontium, but a few experiments were carried out on manganese and cobalt (Fig. 2).

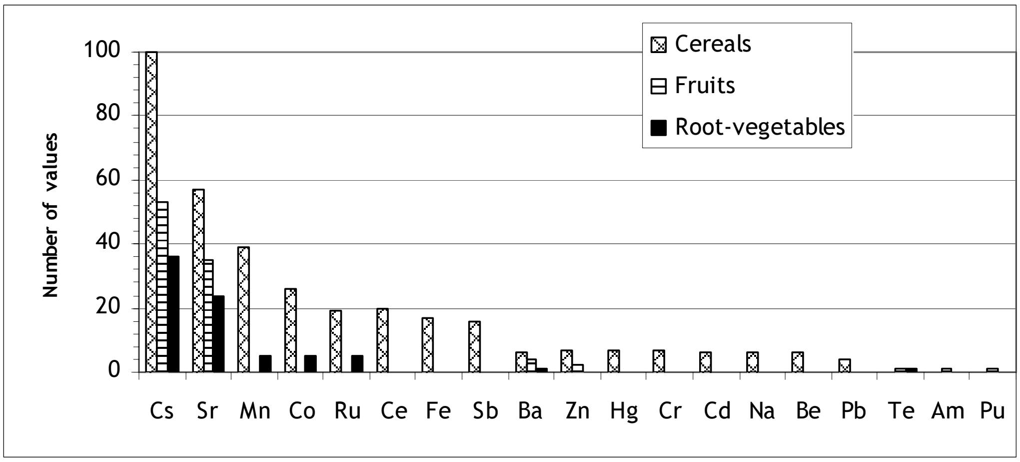

FIG. 1. Distribution of the selected translocation-factor values (%) in relation to radionuclides for the different types of plants.

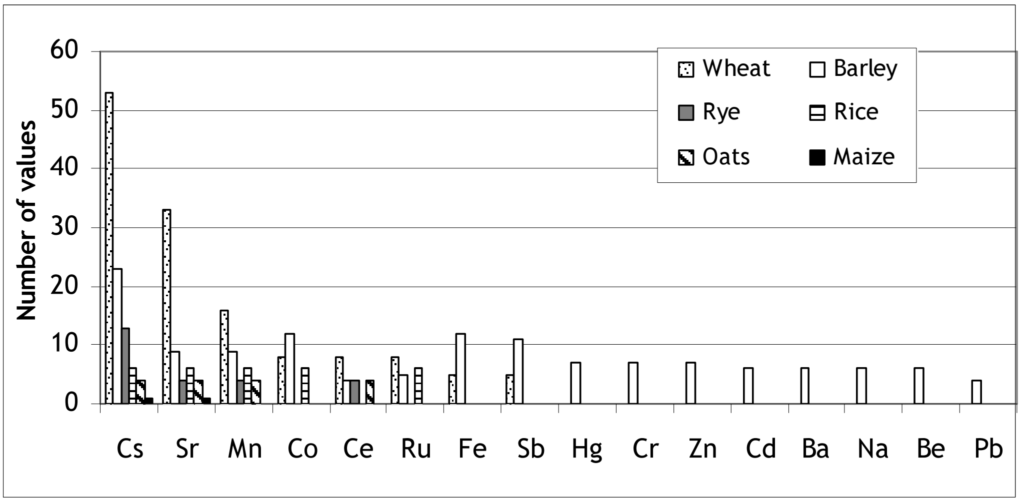

FIG. 2. Distribution of the selected translocation-factor values (%) in relation to radionuclides for the different types of grains.

After processing the selected values, it was possible to divide the vegetative cycle of the different crops under investigation into stages corresponding to the different physiological steps of plant growth, namely: leaf development-tillering, stem elongation, earing-flowering, grain growth and ripening. Experimental values for translocation factors were aggregated in relation to the vegetation stage corresponding to the state of the plant at the time of foliar contamination. The aggregation was applied to the associations of grain types, barley, wheat and rye on the one hand, and rice on the other.

The data were processed according to the following sequence:

- for every experiment, translocation-factor values were distributed in accordance with the five vegetative stages;
- data were plotted on a graph, with an abscissa representing the vegetative cycle and the ordinate showing the translocation factors expressed as percentages according to a logarithmic scale,
- statistical analysis was performed in order to determine the best estimation of the translocation factor for a vegetative stage (geometric mean of the data) and a 95% confidence range.

3.1.1. Caesium

The value-distribution diagrams show that the maximum translocation value is reached when foliar contamination occurs about 45 days before harvesting and therefore corresponds to the earing-flowering stage of the plants (Fig. 3).

As a result of statistical analysis of raw data, the recommended values (geometric mean for each plant growth stage) and the range of variation for the caesium translocation factors (whatever the vegetative stage) are given in Table 2**.**

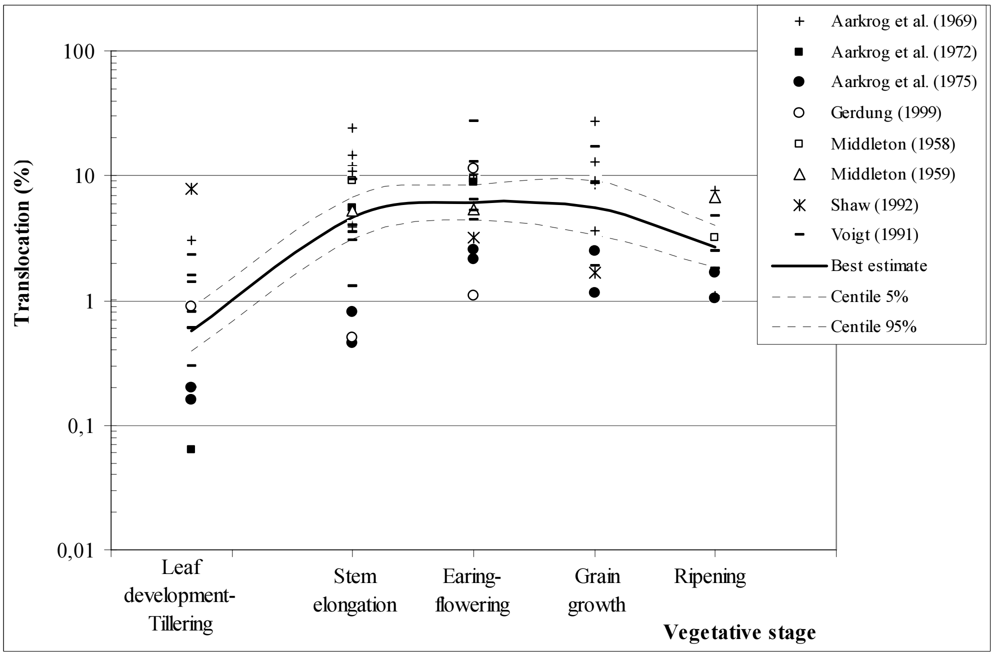

FIG. 3. Experimental values of caesium translocation factor for wheat, barley and rye in relation to the plant-growth stage at the time of foliar contamination. The continuous line shows the best estimate of translocation factor and the 95% confidence range is also shown (dotted lines).

TABLE 2. TRANSLOCATION FACTOR (f_{tr}) VALUES OF CAESIUM FOR CEREALS, %

Elment	N	Plant growth stage	GM	Min	Max	References
		Wheat, barley and rye (grains)				
Cs	21	Leaf development-tillering	6×10^{-1}	6×10^{-2}	7.9	[7, 3, 8, 9, 12, 14, 16, 33, 34]
	21	Stem elongation	4.6	4.5×10^{-1}	24	
	15	Earing-flowering	6.1	1.1	27	
	11	Grain growth	5.5	1.1	27	
	11	Ripening	2.7	1.1	7.7	
		Rice				
Cs	2	Leaf development-tillering	2.3	1.2	3.4	[35, 36]
	1	Stem elongation	4.3			
	1	Earing-flowering	8.4			
	1	Grain growth	11			
	1	Ripening	2.2			

3.1.2. *Strontium*

The translocation-factor values of strontium to wheat, barley and rye are shown in Fig. 4. Depending on the time between foliar contamination and harvesting, the values range from 0.007% to 8.5%. The value-distribution diagram shows that the maximum translocation value is reached when foliar contamination occurs about 30 days before harvesting and, therefore, corresponds to the grain growth stage of the plants.

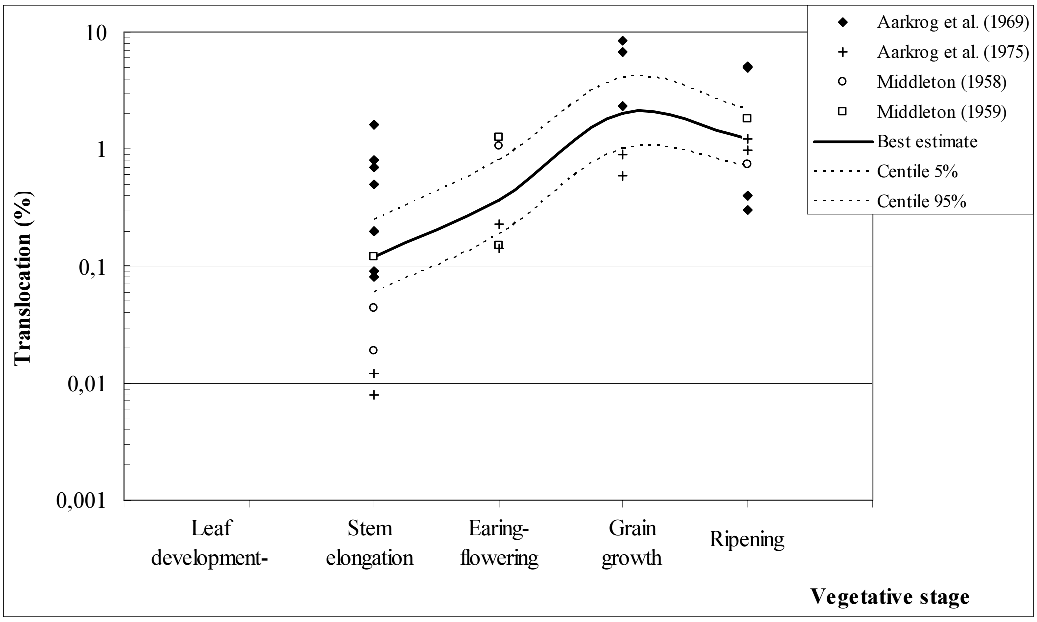

FIG. 4. Experimental values of strontium translocation factor for wheat, barley and rye in relation to the plant-growth stage at the time of foliar contamination. The continuous line is the best estimation of the translocation factor and the 95% confidence range is also shown (dotted line).

As a result of statistical analysis of the raw data, the recommended values (geometric mean for each plant growth stage) and the range of variation for the strontium translocation factors (whatever the vegetative stage) are reported in Table 3.

3.1.2. Other elements

Due to the scarcity of data for the elements other than caesium and strontium, the available translocation factors values are given together in Table 4.

3.2 Root vegetables and tubers

Eighty-nine values of translocation factors taken from 12 publications have been used for assessing translocation factors for root-vegetables. Among them, 76 were selected to be used in this review. The greater majority of the experiments concerned caesium and strontium and essentially potatoes and radishes. Radioisotopes of some other elements (manganese, cobalt, ruthenium, tellurium and barium) have been investigated, but their analysis and interpretation is limited due to the small amount of data (Fig. 5).

TABLE 3. STRONTIUM TRANSLOCATION FACTOR (f_{tr}) VALUES FOR CEREAL GRAINS, %.

Element	Plant growth stage	N	GM	Min	Max	References
			Wheat, barley and rye (grains)			
Sr	Leaf development-tillering	2	0^1			[7,9,12,13,37]
	Stem elongation	13	1×10^{-1}	8×10^{-3}	1.6	
	Earing-flowering	5	4×10^{-1}	1×10^{-1}	1.3	
	Grain growth	6	2.0	6×10^{-1}	8.5	
	Ripening	8	1.2	3×10^{-1}	5.1	
			Rice			
Sr	Leaf development-tillering	2	2×10^{-2}	2.1×10^{-2}	2.4×10^{-2}	[33,35]
	Stem elongation	1	2×10^{-2}			
	Earing –flowering	1	6×10^{-1}			
	Grain growth	1	1.3			
	Ripening	1	1			

[1] Below the detection limits.

TABLE 4. TRANSLOCATION FACTOR (f_{tr}) VALUES FOR ELEMENTS OTHER THAN CAESIUM AND STRONTIUM, CEREALS, %

Element	Plant growth stage	N	GM	Min	Max	References
			Wheat, barley and rye (grains)			
Ba	n.d	6	2×10^{-1}	1×10^{-3}	4.3	[8,37]
Be	n.d	6	2×10^{-1}	1×10^{-3}	2.7	[8,37]
Cd	n.d	6	7×10^{-1}	2.5×10^{-2}	3.8	[8,37]
Ce	Stem elongation	8	1×10^{-1}	2×10^{-2}	8×10^{-1}	[7,37]
	Grain growth	4	6×10^{-1}	1×10^{-1}	7.8	
	Ripening	4	1.3	3×10^{-1}	6.0	
Co	Leaf development-tillering	5	5×10^{-1}	6×10^{-2}	3.4	[6,9,16,37]
	Stem elongation	3	1.0	2.4×10^{-1}	4.6	
	Earing-flowering	4	2.0	3×10^{-1}	18.0	
	Grain growth	4	2.8	3×10^{-1}	29.0	
	Ripening	3	1.5	5×10^{-1}	6.6	
Cr	n.d	7	1.0	2×10^{-2}	7.4	[6,37]
Fe	Leaf development-tillering	4	8×10^{-1}	6.5×10^{-2}	1.2	[6,9,37]
	Stem elongation	3	1.0	5.7×10^{-1}	1.5	
	Earing-flowering	3	1.9	1.3	2.6	
	Grain growth	3	2.7	1.0	7.5	
	Ripening	3	1.5	3.5×10^{-1}	9.2	
Hg	n.d	6	5×10^{-1}	1×10^{-2}	8	[6,37]
Mn	Leaf development-tillering	3	3×10^{-1}			[7, 9,16, 37]
	Stem elongation	8	2.1			
	Earing-flowering	6	2.3			
	Grain growth	6	2.0			
	Ripening	6	1.0			

TABLE 4. TRANSLOCATION FACTOR (f_{tr}) VALUES FOR ELEMENTS OTHER THAN CAESIUM AND STRONTIUM, CEREALS, % (cont)

Element	Plant growth stage	N	GM	Min	Max	References
			Wheat, barley and rye (grains)			
Na	n.d	6	2.0	1.7×10^{-1}	7	[8,37]
Pb	n.d	3	2.0	2×10^{-1}	8.2	[6,37]
Ru	n.d	8	1.1×10^{-1}	4×10^{-2}	1.2	[9,37]
Sb	Leaf development-tillering	5	2×10^{-2}	2×10^{-3}	6×10^{-1}	[8,9,37]
	Stem elongation	3	1×10^{-1}	3.4×10^{-2}	1.0	
Sb	Earing-flowering	3	1.2	3×10^{-1}	5.2	
	Grain growth	3	2.2	1.0	7.5	
	Ripening	2	6×10^{-1}	3×10^{-1}	1.3	
Zn	n.d.*	6	15.8	7.6	32	[6,37]
			Rice			
Co	Leaf development-tillering	2	2×10^{-1}	6×10^{-2}	2×10^{-1}	[35]
	Stem elongation	1	1.6			
	Earing-flowering	1	4			
	Grain growth	1	6.6			
	Ripening	1	8×10^{-1}			
Mn	Leaf development-tillering	2	5×10^{-2}	4×10^{-2}	5.2×10^{-2}	[35]
	Stem elongation	1	3×10^{-2}			
	Earing-flowering	1	6×10^{-1}			
	Grain growth	1	1.6			
	Ripening	1	7×10^{-1}			
Ru	Leaf development-tillering	2	5×10^{-3}	5×10^{-3}	6×10^{-3}	[33,35]
	Stem elongation	1	2×10^{-2}			
	Earing-flowering	1	1.2×10^{-1}			
	Grain growth	1	3.8×10^{-1}			
	Ripening	1	3.5×10^{-1}			

*Not defined.

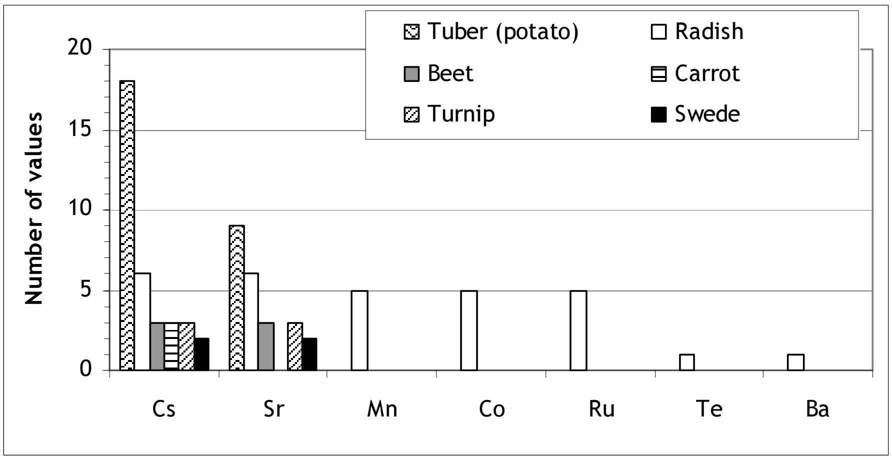

FIG. 5. Distribution of translocation-factor values for various elements for the different types of root vegetables, %.

As a result of statistical analysis of raw data, the recommended values (geometric mean whatever the plant growth stage) and the ranges of variation for strontium translocation factors (whatever the vegetative stage) are presented in Table 5.

3.4. Fruits

Ninety seven values of translocation factors to fruits have been derived from the literature and have been selected for use in this review [1,3,10-12,15,18,24,25,27-32,38,40-46]. The great majority of the experiments dealt with caesium and strontium (Fig. 6).

Radioisotopes of some other elements (barium, zinc, tellurium, americium and plutonium) have been investigated, but interpretation of the experimental results is difficult due to the small amount of data.

As a result of statistical analysis of the raw data, the recommended values (geometric mean whatever the plant growth stage) and the ranges of variation for the translocation factors (whatever the vegetative stage) are given in Table 6.

4. CONCLUSIONS

This review gives the state of knowledge about the translocation factor values defined as the ratio between total activity in the edible parts of the plant and activity retained on foliage (percentage). The analysis of the scientific literature concerning this parameter of transfer highlighted an important lack of knowledge. Especially the interaction of translocation and plant development is poorly investigated for many crops and radionuclides. Furthermore, there are practically no data on translocation factors in chronic contamination situations simulating sprinkling irrigations distributed throughout the vegetative cycle of a crop.

Also, the majority of the available data relate to caesium and strontium. Radioisotopes of some other elements (manganese, cobalt, iron, ruthenium, antimony, cerium, barium, zinc, tellurium, mercury, chromium, sodium, cadmium, beryllium and lead) have been investigated, but the data are far from sufficient to obtain reliable values for all the plant types.

TABLE 5. TRANSLOCATION FACTORS FOR ROOT VEGETABLES AND TUBERS, %

Element	GM	Plant growth stage	N	Min	Max	References
			Root vegetables			
Ba	2.2	n.d.	1			[24, 26]
Co	8	n.d.	5	4.9	12	[35]
Cs	4.6	n.d.	17	7×10^{-1}	13	[12, 13, 15, 24-26, 38, 39]
Mn	2.4×10^{-1}	n.d.	5	2×10^{-1}	4×10^{-1}	[35]
Ru	1.5×10^{-1}	n.d.	5	1×10^{-1}	4×10^{-1}	[35]
Sr	5×10^{-1}	n.d.	14	2×10^{-1}	1.6	[12, 13, 15, 24-26]
Te	8×10^{-1}	n.d.	1			[24, 26]
			Tubers			
Cs	11.6	n.d.	23	1.3	4.6×10^{1}	[12-14, 18, 38]
Sr	1×10^{-1}	n.d.	9	2×10^{-2}	5×10^{-1}	[12-14, 18, 38]

n.d. not detactable.

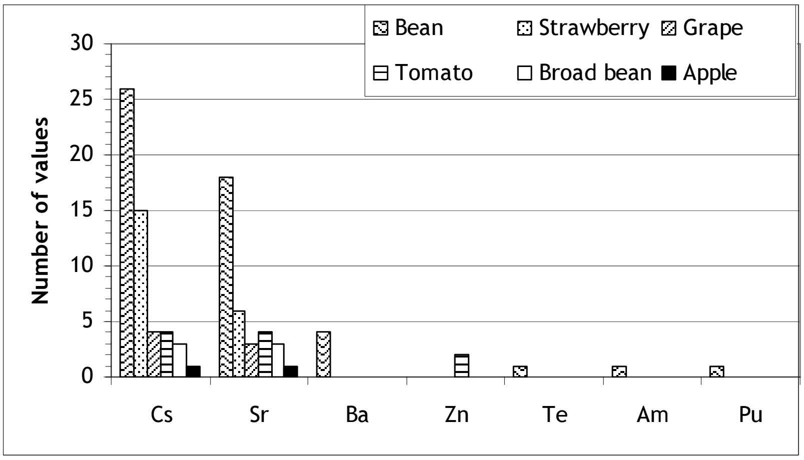

FIG. 6. Distribution of translocation-factor values for various elements for the different types of fruit, %.

TABLE 6. TRANSLOCATION FACTORS (f_{tr}) VALUES FOR FRUITS, %

Element	GM*	Types of fruit	N	Min	Max.	References
Am	5×10^{-4}	Beans	1			[40]
Ba	1.3×10^{-1}	Beans	4	4×10^{-2}	1.6	[15;24]
Cs	4.6	Apples, beans, grapes, tomatoes, strawberries	53	1×10^{-1}	29	[3,11,12,15,18,24,25,27-32,38,41-46]
Pu	3×10^{-4}	Beans	1			[40]
Sr	4.4×10^{-1}	Apples, beans, grapes, tomatoes, strawberries	35	1×10^{-2}	12.1	[3,11,12,15,18,24,25,27-32,38,41-46]
Zn	4.3	Tomatoes	2	2.6	7	[27]

*plant growth stage not defined

Many radioecological models use values based on poorly justified extrapolations or chemical analogies. This method is arbitrary insofar as the concept of analogy refers only to the chemical properties of the elements e. g. chemical analogy does not necessarily imply the same behaviour inside the plant, as shown for Ca and Sr [47] and Cs, K and Rb [48]. Besides, it does not take into account the various physiological and physicochemical mechanisms inside the plant which govern the translocation processes.

Moreover, some authors do not make a distinction between plant types and recommend a single default value whatever the element and plant type. Data given without growth stage indication should be used with caution indicating the wide range of associated uncertainty. In order to supplement the current lack of knowledge on translocation, it seems essential to undertake further research through experimental investigations.

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RESUSPENSION

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Abstract

The paper provides an extended review of the data on resuspension. Processes and definitions as well as factors governing an extent of the resuspension were considered in details. Resuspension models for evaluation of this process in different (urban, rural, arid) environmental conditions are considered. Data have been compiled to derive resuspension factors to update the values given in Technical Reports Series No. 364. Most of this information is given for Cs (Chernobyl observation) and Pu, however, some data were also derived for iodine, uranium and radium. In spite of the some improvements made after publishing Technical Reports Series No. 364 it was found that the variability of measured resuspension factors and rates is very high and a lot of gaps in resuspention modelling still remain.

1. PROCESSES AND FACTORS GOVERNING RESUSPENSION

Resuspension occurs when the wind exerts a force exceeding the adherence of particles to the surface material. The forces in action are the weight of the particle, the adherence, as well as the aerodynamic loads related to the flow of wind.

In general, three types of process are used to describe the spread of contaminants deposited on surface soil [1]: surface creep, saltation and suspension.

Surface creep relates to large, heavy particles (500 to 1000 µm diameter), which roll and slip on soil without air elevation. This process enables snow and sand migration.

Saltation relates to particles in the 100 to 500 µm diameter range. In this process, particles alternate periods with and without soil contact. The main process of transport takes place between 0.1 and 1 meter above the soil. When particles being transported by saltation impact the ground, they lift small particles: as in sandblasting [2]. The saltation process results in horizontal transport and the sandblasting process induces vertical transport.

Suspension relates to small particles (< 100 µm diameter), which are not subject to multiple contacts with the soil, as occurs in saltation. In this process, particles leave the ground and are entrained in the air flow. Chamberlain [2] considered that particles with sizes ranging from 5 to 30 µm are carried long distances in the atmosphere following resuspension.

Another process for resuspension is the mixed effect of wind and rain on particle detachment. Rain splash transport of soil particles in windless conditions has been studied in detail [3-10]. The overall result of these studies is that the contribution of rain splash transport is very small when compared with overland flow transport. Nevertheless, it is still a short distance process studied mainly in relation to horizontal dispersal. Vertical spread has received much less attention. Studies show that rain splash is not a predominant process for atmospheric dispersion of particles due to resuspension.

2. RESUSPENSION CHARACTERISATION

The suspension (or resuspension) of contaminated soil/sediment into the atmosphere is characterized following to two main approaches.

The resuspension factor approach is based on the ratio between the volumetric air concentration and the soil contamination. The resuspension factor has the dimension of reciprocal length; an expression of this parameter is:

$$K_S(t) = \frac{C_V(t)}{C_{S,0}} \quad (m^{-1}), \tag{1}$$

where $K_S(t)$ is the resuspension factor, m^{-1}; $C_V(t)$ is the contaminant concentration above the contaminated soil, Bq m^{-3} and $C_{S,0}$ is the initial soil contamination, Bq m^{-2}.

The resuspension rate approach is based on the ratio between particle flow density and soil contamination. In this case, the resuspension rate has the dimension of reciprocal time and can be given as follows:

$$R_S(t) = \frac{F_{part}(t)}{C_{S,0}} \tag{2}$$

Where $R_S(t)$ is the resuspension rate, s^{-1}; $F_{part}(t)$ is the particle flow density above the contaminated soil, Bq m^{-2} s^{-1}; $C_{S,0}$ is the initial soil contamination, Bq m^{-2}.

Garger [11] suggested a relationship between these two characteristic factors of resuspension:

$$R_S(t) = K_S(t).v_d \tag{3}$$

Where $R_S(t)$ is the resuspension rate, s^{-1}; $K_S(t)$ is the resuspension factor, m^{-1}; v_d is the deposition velocity, m s^{-1}.

This definition implies that deposition and resuspension are determined by the same factors. It is not clear why this should be the case.

The resuspension factor approach makes it possible to obtain directly the concentration in the air, whereas the rate of resuspension approach makes it possible to obtain a flux of mass or activity, which is more convenient to use in the conservation equations of pollutant transport in mechanistic models.

3. MEASURED VALUES OF IN-SITU RESUSPENSION

The resuspension of radionuclides from accidentally contaminated sites has been documented by *in situ* measurements for plutonium contamination and for caesium from the Chernobyl accident (Table 1).

Sehmel studied the resuspension of submicronic particles from a slightly rough soil [12]. The resuspension rates (R) lay between 4.0 10^{-5} hr^{-1} and 4.0 10^{-6} hr^{-1}. A DOE Handbook [13] recommends use of the value of 4.0 10^{-5} hr^{-1} to evaluate the rate of resuspension of particles deposited and exposed to standard ambient conditions (273.15 K, 101.325 kPa).

In the first days and first months that follow an accident, the values of resuspension factor (K) available in the literature [14– 17] generally range between 10^{-5} m^{-1} in residential areas, on a site undergoing cleanup operations and on an arid site and 10^{-6} m^{-1} on a rural site (see Table 1). The resuspension factor decreases to 1.0 10^{-8} m^{-1} or 1.0 10^{-9} m^{-1} after 3 to 4 years.

It is also observed that in humid or semi-humid climates, resuspension is generally more important in urban conditions than in rural. However, this might not be the case in desert or semi-desert conditions environments.

TABLE 1. IN-SITU RESUSPENSION FACTOR VALUES K_s (m^{-1}), cited according to DOE Handbook, 1994 [13]

Element	K, measured	Location	Authors	References
^{137}Cs	2.0×10^{-9}	Munich, semi-rural	Rosner, 2001	[23]
^{137}Cs	1.0×10^{-6} to 1.0×10^{-5}	Hannover, urban	Hollander, 1994	[19]
^{137}Cs	3.6×10^{-10}	Denmark urban	Fogh, 1999	[18]
^{137}Cs	1.0×10^{-9}	Kiev, urban	Nair, 1997	[21]
^{137}Cs	4.3×10^{-10}	Chernobyl, urban	Garger, 1997	[11, 17]
^{137}Cs	1.6×10^{-10}	Chernobyl, forest fire	Kasparov, 2000	[20]
^{131}I	1.0×10^{-7} to 1.0×10^{-5}	Nevada	Anspaugh, 1970[1]	[29]
^{131}I	2.0×10^{-6} to 1.0×10^{-5}	Nuclear test site	Stewart, 1967[1]	[16]
^{131}I	4.0×10^{-6} to 5.0×10^{-5}	Nuclear test site, cleanup operations	Stewart, 1967[1]	[16]
^{40}K	3.2×10^{-9}	Munich, semi rural	Rosner, 2001	[23]
Pu	3.0×10^{-10} to 3.0×10^{-9}	Nevada	Anspaugh, 1975[1]	[14]
^{239,240}Pu	1.2×10^{-9}	Munich, semi-rural	Rosner, 1997, 2001	[22, 23]
^{238}Pu	5.0×10^{-8}	New York	Bennett, 1976[1]	[30]
^{238}Pu	5.0×10^{-9}	UK	Bennett, 1976[1]	[30]
Pu	1.4×10^{-9} to 7.8×10^{-6}	Palomares accident, Spain	Iranzo, 1987[1]	[31]
Pu	1.2×10^{-10} to 8.7×10^{-7}	Palomares, Spain– cultivation	Iranzo, 1987[1]	[31]
Pu	5.0×10^{-8} to 1.0×10^{-6}	Contaminated field, tractor traffic	Milham, 1976[1]	[32]
Pu	7.9×10^{-10}	Contaminated field, fertilization	Milham, 1976[1]	[32]
Pu	5.0×10^{-6}	Contaminated field, ploughing	Milham, 1976[1]	[32]
Pu	1.8×10^{-8} to 7.9×10^{-6}	Contaminated field, planting	Milham, 1976[1]	[32]
Pu	1.8×10^{-5}	Contaminated field, traffic	Milham, 1976[1]	
Pu	7.0×10^{-6}	Nevada, contaminated field (rural)	Langham, 1971[1]	[15]
Pu	7.0×10^{-5}	Nevada, traffic	Langham, 1971[1]	[15]
Pu	1.0×10^{-9} to 1.0×10^{-5}	Kentucky	Sehmel, 1973[1]	[33]
Pu	5.6×10^{-8}	Contam. waste treatment	Myers, 1976[1]	[34]
Pu	3.0×10^{-7} to 7.0×10^{-4}	Maralinga test site	Stewart, 1967[1]	[16]
Ra	1.0×10^{-9} to 1.2×10^{-9}	Munich, semi rural	Rosner, 2001	[23]
^{238}U	1.0×10^{-4}	Maralinga test site, 8 hours after deposition	Stewart, 1967[1]	[16]
^{238}U	3.0×10^{-4}	Maralinga test site	Stewart, 1967[1]	[16]
^{238}U	3.2×10^{-9}	Munich, semi rural	Rosner, 2001	[23]

4. RESUSPENSION MODELLING

The development of a resuspension model is not easy task because of the number of processes of different significance which are involved. Overall, these result in dependences on the material (particle size, shape and adherence), surface type (roughness, humidity), time lapsed since deposition and intensity of mechanical actions (soil processing).

As with measurements, resuspension models can be distinguished according to the environmental context. It is recommended that models tested on the data collected after the accident at Chernobyl should be used in the context of accidental releases to air. However, other types of model may be more appropriate in other contexts, e.g. in assessing the radiological impacts from contaminated land on sites that currently or formerly handled or processed radioactive materials [35].

For rural conditions, the model suggested by Garland ([24-26]):

$$K_s(t) = 1.2 \cdot 10^{-6} t^{-1} \quad m^{-1} \tag{4}$$

where *t* is the time in days since deposition , obtained one of the best scores in the IAEA model intercomparison exercise in the framework of the IAEA BIOMOVS II programme [27, 28].

In this and subsequent models discussed in this section, the model formulations are not independent of the units in which time is expressed. Generally, time has units of days unless otherwise stated. Garland [25] advised that this formula should be applied to deposits older than 1 day.

For the urban environment, the Linsley model [28] provided the best results in the intercomparison exercises:

$$K_s(t) = 10^{-6} \exp(-0.01 \cdot t) + 10^{-9} \quad m^{-1} \tag{5}$$

This expression yields a resuspension factor that lies within the range of those estimated in *in situ* experiments. However, this expression tends to over-estimate short-term concentrations and to underestimate the long-term values. Moreover, exponential decrease with time is difficult to justify because it is rarely measured in experiments.

For arid and desert conditions, it is recommended that the model of [15] should be used. This gives values that are intermediate between those observed for urban and rural environments in the long term. The model form is:

$$K_s(t) = 10^{-6} \exp\left(-0.15\sqrt{t}\right) + 10^{-9} \quad m^{-1} \tag{6}$$

5. CONCLUSIONS

Values and models selected here are adapted for evaluation of the radiological impacts of resuspension of radionuclides deposited in the natural environment following accidental releases to air. However, the variability of measured resuspension factors and rates are very high and accuracies of these models predictions are rather low.

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AGRICULTURAL ECOSYSTEMS: RADIONUCLIDE MOBILITY IN SOIL

SOIL-RADIONUCLIDE INTERACTIONS

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Abstract

Radionuclide mobility in agricultural systems, and thus in the food chain, is strongly affected by the extension of its sorption in soils, which can be estimated by the quantification of the solid-liquid distribution coefficient (K_d). This parameter may vary within various orders of magnitude depending on the radionuclide and soil combination, but also on the experimental method used for its determination. Here, a new K_d database was built up with around 2900 records of 67 elements, which allows us to calculate the best estimates for the K_d values of a number of radionuclides for various soil types. Best estimates are derived from geometric means calculated from grouping soils by texture and organic matter content and, when possible, also using the main soil cofactors governing soil-radionuclide interaction, concretely for radiocaesium, radiostrontium, uranium, radioiodine, and a few heavy-metal radionuclides. The use of the cofactor approach permits, in most cases, decreasing the variability of the ranges of K_d values associated with a soil type. Additionally, some hints are given in this section on the sorption dynamics of radiostrontium and radiocaesium, based on rate constants values and changes with time in the reversibly sorbed fraction.

1. DISTRIBUTION OF RADIONUCLIDES BETWEEN SOLID AND LIQUID PHASES

1.1. The solid-liquid distribution coefficient concept

The chemical form and speciation of radionuclides strongly affects their movement through environmental media and uptake by biota. Specifically, the way that a radionuclide is bound to solids (soils in the terrestrial environment; sediments in aquatic systems) eventually controls the amount of radionuclide in solution, which directly influences the fraction of radionuclide that may be incorporated by organisms.

Dissolved radionuclide ions can bind to solid surfaces by a number of processes often classified under the broad term of sorption. Although significant progress has been made to describe sorption in heterogeneous solids as a weighted result of sorption on homogeneous surfaces, models for the description of radionuclide sorption are still mostly based on empirical solid-liquid distribution coefficient (K_d) values. This approach is the simplest sorption model available and is the ratio of the concentration of radionuclide sorbed on a specified solid to the radionuclide concentration in a specified liquid phase at equilibrium (K_d, L/kg). The K_d -based model does not assume any knowledge of sorption mechanisms, nor does it contain a term to quantify the capacity and selectivity of the sorption sites or the competition with other ions to fill the sorption sites. Such competitive effects can only be taken into account by the empirical selection of different K_d values applicable in different environmental contexts.

The simple K_d -based model relies on the hypothesis that the radionuclide on the solid phase is in equilibrium with the radionuclide in solution, and thus can exchange with it. However, the elapsed time since the incorporation of the radionuclide is known to affect the quantification of K_d, since a fraction of the incorporated radionuclide may become fixed by the solid phase (an aging effect related to sorption dynamics). Although in most cases no specific comments are made on this issue in published papers, consideration of this process has led to the definition and reporting of various types of K_d in the literature. Whereas the *labile* or *exchangeable* K_d refers to the initial sorption process where the radionuclide is reversibly sorbed, due to an ion-exchange based mechanism for most radionuclides, terms such as *total* K_d are used when there may be radionuclide irreversibly sorbed to the solid phase. Therefore, and as the radionuclide speciation in the solid phase may change with time, an estimation of the changes in the reversibility of the sorption in the short and medium term is also required in any experimental approach designed to derive information on sorption dynamics.

Most laboratory tests are, in principle, designed to obtain the so-called exchangeable K_d (K_d^{exch}). The additional advantage of using this approach is that the value of K_d^{exch} can be easily predicted on the basis of soil characteristics, such as binding capacity of the soil (number and selectivity of sorption sites), and the composition of the soil solution (concentration of sorption-competitive ions present in the solution). However, a K_d deduced from a laboratory test cannot be unequivocally considered to be a K_d^{exch}, since the nature of the sorption process for a given radionuclide may lead to a *quasi* instantaneous irreversible sorption, and, in other cases, long contact times, for instance, may result in a fraction of the radionuclide activity becoming irreversible sorbed, and thus no longer participating in the soil – soil solution equilibrium. Besides this, the large number of approaches used to quantify K_d values, and the contrasting experimental conditions applied in each case, lead to wide ranges of K_d values being obtained for similar soil and radionuclide combinations. The variation in approaches adopted often makes it difficult to compare among K_d values derived from laboratory experiments.

In this document, generally, the term K_d is utilized to describe the radionuclide distribution coefficient, although other terms are used, as required in specific cases.

1.2. Experimental methods used to estimate the K_d values

Most common approaches derive K_d values from field-contaminated soils, and from sorption and mass transport experiments at a laboratory scale with initially non-contaminated soils.

1.2.1. K_d values from field contaminated soils

The K_d values can be quantified from the radionuclide concentration in the soil solid phase divided by the concentration of the radionuclide in the soil solution obtained from the contaminated soil [1]. This approach is reliable when the level of contamination is high enough to disregard the uncertainty in obtaining and measuring a representative sample of the soil solution.

This approach may lead to K_d values higher than those resulting from a laboratory sorption test, because the radionuclide quantified in the solid phase of the contaminated soil may include sorbed radionuclide not available for exchange with the soil solution due to the time elapsed since the radionuclide incorporation. Therefore, this approach is not recommended for quantification of K_d^{exch}.

1.2.2. Laboratory sorption experiments

Among the laboratory studies, the approach applied most often is to undertake sorption experiments on non-contaminated soils, mainly using batch methods. Experimental conditions may be extremely different from one experiment to another, since a harmonized procedure has not been established, although recommended methods are available from several organizations [2-3].

Sorption experiments are conducted at various radionuclide activity and (more pertinently) mass concentrations, in different hydro chemical and mineralogical contexts. Experimental conditions, such as the composition of the contact solution, contact (shaking) time, volume/mass ratios, and filtration of the resulting solution, may differ. Regarding filtration, the absence or presence of this step, as well as the pore size of the filter can be of a major significance for those radionuclides exhibiting an association with colloids, since colloids may be mobile and in batch experiments the colloid fraction may either be associated with the solution or solid phase depending on the filtration employed. As the competitive effect of major ions has been widely described, especially when dealing with specific sorption sites, it is recommended that sorption experiments should be performed that simulate as closely as possible field conditions of interest, e.g. by reproducing the pH and ionic composition of the soil solution in the sorption medium [4-5].

Care is needed to avoid undertaking experiments at higher concentrations than could be expected after a radioactive release. In particular, excessively high mass concentrations may arise in simulations of radionuclide sorption using stable or very long-lived isotopes of the element or analogue elements. It is important to ensure that at no time in the experiment do solution concentrations exceed the solubility limit for the radionuclide or stable element being studied.

1.2.3. Laboratory mass transport experiments

Another experimental approach is to derive K_d values from the diffusion pattern of a radionuclide in compacted soils, in column or diffusion cells [6].

In a porous medium like soils, the radionuclide diffusion process differs from diffusion in free water. An effective diffusion coefficient (D_e; m^2/s) must, therefore, be defined. Only the pores that contribute to the transport of the dissolved radionuclide species have to be considered (the diffusion-accessible porosity, Φ), although in most cases (mainly when the relative saturation tends to one and for cationic radionuclides) to use the total porosity (ε) is an adequate approximation. In the case of radionuclides with significant sorption, an apparent diffusion coefficient (D_a; m^2/s) can be calculated from the diffusion profile into the sample. The latter diffusion coefficient takes into account the retardation of the radionuclide due to interactions with the porous material. It can be written:

$$D_a = \frac{D_e}{f_{ret}} \tag{1}$$

where f_{ret} is the Retardation Factor. If we hypothesize a linear sorption pattern, with a constant K_d in the range of concentrations studied, the f_{ret} can be defined as:

$$f_{ret} = 1 + (\frac{\rho}{\varepsilon}) \cdot K_d \tag{2}$$

where ρ is the dry bulk density of the soil.

Scarce data comparing K_d values from batch and diffusion experiments are available to date, and conclusions on whether the batch sorption methods over- or under-estimated K_d values are still contradictory [7-8]. However, overall more cases are described where the K_d values derived from diffusion experiments were lower than those derived from batch experiments than *vice versa* [9-10], mostly due to in the experimental conditions adopted in the batch studies, such as the volume/mass ratio and contact time.

1.3. Mechanistic approach to prediction of K_d values

The increased knowledge of interaction mechanisms between certain radionuclides and solid materials allows review of K_d values in terms of a more fundamental description of underlying processes. In this section, the mechanisms responsible for radionuclide sorption are described for a number of radionuclides, thus introducing the concept of *cofactors* influencing soil-radionuclide interactions. An advantage of using cofactors for grouping soils is that the variability of K_d values may decrease considerably with respect to the variability observed when the classification is based solely on sand, clay, and organic matter contents.

1.3.1. Cofactors for radiostrontium

The solid-liquid partitioning of a number of radionuclides (*RN*), such as radiostrontium, may be better understood by reference to the partitioning of an analogue (sorption competitive) ion (*AN*), characterized by similar sorption behaviour. In this approach:

$$K_d(RN) = K_d(AN) \cdot K_c(\frac{RN}{AN}) \tag{3}$$

where the $K_d(RN)$ is calculated by a linear amplification of the $K_d(AN)$ by a factor equal to the RN-AN selectivity coefficient at the sorption sites $K_c(\frac{RN}{AN})$.

Regarding radiostrontium, $K_d(Sr)$ can be predicted from the ratio of the Ca and Mg in the exchangeable complex in soil solids (in cmol$_c$/kg) to the sum of the concentrations of Ca and Mg in the soil solution (in cmol$_c$/L) [11-12], amplified by the trace selectivity coefficient Sr-to-Ca and Sr-to-Mg, K_c (Sr/Ca-Mg), which corresponds to equation (4):

$$K_d(Sr) = K_c\,(Sr\,/\,Ca - Mg)\,\frac{Ca_{exch} + Mg_{exch}}{Ca_{ss} + Mg_{ss}} \tag{4}$$

As the K_c (Sr/Ca-Mg) is reported to be close to 1 [13], in most cases similar trace selectivity coefficients Sr-to-Ca and Sr-to-Mg may be assumed to derive a simpler model. Therefore, equation 4 can be simplified to:

$$K_d(Sr) = \frac{Ca_{exch} + Mg_{exch}}{Ca_{ss} + Mg_{ss}} \tag{5}$$

If data on exchangeable cations are not available, the ratio of the cationic exchange capacity (CEC, in cmol$_c$/kg, which is usually quantified in routine soil analyses), to the sum of the concentrations of Ca and Mg in the soil solution can be used as a satisfactory approach to estimate the K_d (Sr), especially when dealing with soils with a saturated exchange complex [11].

Another approach to the prediction of $K_d(Sr)$ is based on correlating the $K_d(Sr)$ to other soil properties that are also easily measured in routine studies. An example of this is to relate the $K_d(Sr)$ to the Cation Distribution Ratio (CDR), defined as the value of the cationic exchange capacity (CEC, cmol$_c$/kg) divided by the electrical conductivity (*EC*, mS/cm) in the soil solution [14]. This correlation is easily explained by the fact that the electrical conductivity is controlled by the concentrations of major cations in the soil solution, especially Ca and Mg:

$$EC\ (mS\ cm^{-1}) \approx 1000\ [Na_{ss} + K_{ss} + NH_4{}^{+}{}_{ss} + Mg_{ss} + Ca_{ss}]\ (cmol_c/L) \quad (6)$$

The resultant regression equation for the set of soils examined is [14]:

$$K_d\ (Sr)\ (L/kg) = 2.1\ CDR\ (L/kg) \quad (7)$$

1.3.2. Cofactors for radiocaesium

One approach to prediction of the value of $K_d(Cs)$ is based on the application of the Radiocaesium Interception Potential (RIP) concept. The RIP value estimates the capacity of a given soil to specifically sorb Cs. The most common protocol to determine the RIP is based on pre-equilibrating the samples with a solution containing 100 mmol/L of Ca and 0.5 mmol/L of K (m_K). After pre-equilibrating the samples, these are equilibrated with the same solution, but labelled with radiocaesium. The distribution coefficients ($K_d(Cs)$) are obtained by measuring the radiocaesium activity in the supernatant, before and after the equilibration. The calculated product $K_d(Cs) \times m_K$ defines the RIP value (in mmol/kg). Details can be found elsewhere [15].

The RIP value relates to the content and selectivity of expandable clays, especially illite and other 2:1 phyllosilicates, in which Frayed Edge Sites (FES), which are specific sites for Cs sorption, are present [16]. Other exchange sites are of little relevance for Cs sorption [17-19], except when dealing with soils with extremely low clay content (*e.g.* organic matter content over 90%; highly sandy podzols), in which Cs may also be sorbed at other, less specific sites [20].

As Cs sorption is controlled by the specific FES, the Cs solid-liquid distribution coefficient at these sites ($K_d^{FES}(Cs)$) accounts for more than 80 % of the total sorption process [19]. The $K_d^{FES}(Cs)$ can be predicted by dividing the RIP value by the sum of K and NH_4^+ concentrations in the soil solution, the latter amplified by the NH_4-to-K trace selectivity coefficient in the FES ($K_c^{FES}(NH_4/K)$) [16]. This parameter, which can be easily quantified by laboratory experiments, ranges from 4 to 8 for soils in which specific sites control Cs sorption quantitatively, and down to 2 in those soils where sorption at regular exchange sites may be significant [20-21].

For a more accurate prediction of the value of K_d (Cs), a second term must be added to account for Cs sorption at regular exchange sites ($K_d^{RES}(Cs)$) by dividing the sum of the exchangeable K and NH_4^+ by the sum of K and NH_4^+ concentrations in the soil solution (in mmol/L), assuming a selectivity coefficient NH_4/K of approximately 1 at these sites. The equation derived may be written as follows:

$$K_d(Cs) = K_d^{FES}(Cs) + K_d^{RES}(Cs) = \frac{RIP}{K_{ss} + K_c^{FES}(NH_4/K)\cdot NH^+_{4\,ss}} + \frac{K_{exch} + NH^+_{4\,exch}}{K_{ss} + NH^+_{4\,ss}} \quad (8)$$

For the case of highly saline soils, near to marshlands, with high Na concentrations in the soil solution, equation 8 may be slightly modified to include the potential competitive role of Na and its effect on the quantification of $K_d{}^{FES}$. The equation may be rewritten as follows:

$$K_d(Cs) = K_d^{FES}(Cs) + K_d^{RES}(Cs) =$$

$$= \frac{RIP}{K_{ss} + K_c^{FES}(NH_4 / K)\cdot NH_{4\,ss}^{+} + K_c^{FES}(Na / K)\cdot Na_{ss}} + \frac{K_{exch} + NH_{4\,exch}^{+}}{K_{ss} + NH_{4\,ss}^{+}} \qquad (9)$$

where the K_c^{FES} (Na/K) term is the Na-to-K trace selectivity coefficient in the FES. As this coefficient takes values of around 0.02 [15], the role of Na will have a significant effect only in those contexts in which an unusually high Na concentration occurs.

Equations (8) and (9) may be simplified by considering that Na and NH_4^+ concentrations are generally much lower than K concentrations, as is the case for most agricultural systems with mineral soils. As the value of K_d^{FES} (Cs) is much larger than the value of K_d^{RES} (Cs*),* K_d (Cs) is reasonably well predicted by equation 10, except for those soil types (upland, peat soils; soils affected by flooding) in which $NH_4^+{}_{ss}$ can be significant:

$$K_d(Cs) = \frac{RIP}{K_{ss}} \qquad (10)$$

A major limitation of this approach to predicting the value of K_d (Cs) is the fact that a K_d value must be obtained to quantify the *RIP* value. To date, attempts to predict the *RIP* value from soil properties have been only partially successful. Waegeneers et al. showed that the *RIP* value depended not only on the clay content, but also on the type of clay and geological origin of the soil [22]. After performing a stepwise regression analysis, the clay content alone accounted for up to the 71% of the variance of the *RIP* in the most favourable set of soils, whereas for another set of soils it explained only 13% of the variance. Regarding the K_d database compiled for this IAEA-TECDOC (see below), the *RIP* and the clay content had a correlation coefficient near to 0.7.

1.3.3. Cofactors for uranium and radium[1]

Uranium occurs in the valences +3, +4, +5 and +6. In soils, the valences +4 and +6 are the most important. U (IV) dominates at $E_h < 200\,\text{mV}$ [23], which is typical for waterlogged to wet soils. U (IV) tends to strongly bind to organic matter and to precipitate, and it is, therefore, immobile. The most oxidized state for U in nature is U (VI). At pH below 5, U (VI) is present as the uranyl ion, UO_2^{2+}. At a higher pH, the uranyl ion hydrolyzes, forming a number of aqueous hydroxide complexes, according to the general hydrolysis reaction:

$$UO_2^{2+} + yH_20 \Leftrightarrow UO_2(OH)_y + yH^+$$

The hydrolysed species (*e.g.* UO_2OH^+, $UO_2(OH)_2^0$ and $(UO_2)_2(OH)_2^{2+}$) often dominate U(VI) speciation in the absence of dissolved inorganic ligands (carbonate, fluoride, sulphate and phosphate). In the presence of dissolved carbonates, U(VI) forms strong carbonate complexes, such as $(UO_2)_2CO_3(OH)_3^-$, UO_2CO_3, $UO_2(CO_3)^{2-}$, $UO_2(CO_3)_3^{4-}$. At the pH range of 6 to 10, uranium is largely partitioned into three stable complexes: the acid biphosphate, bicarbonate and tricarbonate [24]. The oxidized uranyl ion phosphate, sulphate and carbonate complexes are soluble and readily transported.

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[1] The section was written by H. VANDENHOVE, SCK-CEN, Belgian Nuclear Research Centre, Mol, Belgium.

The above description of the uranium speciation under oxidizing conditions points out the importance of pH on uranium behaviour. EPA [25] performed an extensive review of K_d (U) values for soils, crushed rock material and single-mineral phases, which indicated that pH and dissolved carbonate concentrations are the two most important factors influencing the sorption behaviour of U(VI). Dissolved carbonate species increase uranium availability through the formation of strong anionic carbonate complexes, especially in alkaline conditions. The complexity of these reactions requires application of geochemical reaction codes and surface complexation models as the best approaches to predicting values of K_d (U).

Since so many factors affect K_d (U) (pH, dissolved carbonates, amorphous Fe content, soil mineralogy, CEC, soil organic matter content), K_d (U) exhibits large variability. However, K_d(U) values show a specific trend in relation to the pH. In general, the sorption of uranium by soils is low at pH values less than 3, increases rapidly with increasing pH from 3 to 5, reaching a maximum in the pH range from 5 to 7 and then decreases with increasing pH at pH values greater than 7 [25].

One important source of variability in the relationship between K_d (U) and pH is the heterogeneity in soil mineralogy. Soils containing larger percentages of iron oxide minerals and mineral coatings and/or clay minerals will exhibit higher sorption than soils dominated by quartz and feldspar minerals.

Echevarria et al. explored the effect of pH on the sorption of uranium in French soils [26]. They deduced a linear relationship for soils ranging in pH from 5.8–8.8 [$\log K_d = -1.25 \times \text{pH} + 10.9$, R^2=0.89]. When including *K_d (U)*–pH values for Canadian soils in their regression analysis, the same influence on K_d (U) was found [$\log K_d = -1.29 \times \text{pH} + 11.0$, R^2=0.76]. Vandenhove et al. [27] explored the effect of soil properties on uranium availability for eighteen soils, and a similar linear decrease of $\log K_d$ with pH was observed for soils with pH ≥6 [$\log K_d = -1.18 \times \text{pH} + 10.8$, R^2=0.65], which was explained by the increased amount of soluble uranyl-carbonate complexes at high pH. For soils with pH<6, these latter authors suggested exploring the possibility of relating the K_d (U) to organic matter (OM, %) [K_d (U) = 1963 × OM – 5432, R^2=0.78] or to concentration of amorphous iron in soil (mg kg^{-1}) [K_d (U) = 1.02 × $Fe_{amorphous}$ + 1691, R^2=0.88]. Considering all soils (complete pH range), these relationships remained significant [K_d (U) = 1591 × OM – 3362, R^2=0.70; K_d (U) = 1.08 $Fe_{amorphous}$ + 2783, R^2=0.88].

Radium occurs in nature as a divalent cation. It has a high affinity for the regular exchange sites of the soil, as shown by the fact that organic matter sorbs about ten times as much radium as clay [28]. It also co-precipitates with barium and strontium to form insoluble sulphates. Due to its alkaline character, it is not easily complexed. There has been limited research on defining K_d (Ra). In geochemical equilibrium models, often data for Ba, which may act as analogue, are recommended for assessing the behaviour of radium. High Ca levels in the soil solution or exchangeable phase and low organic matter and clay content are conducive to higher radium availability. Overall, not enough data have been assembled so far to mathematically relate K_d (Ra) to any of these soil properties. Vandenhove et al., exploring the effect of soil properties on radium availability [29] in a small-scale study covering 8 soils, concluded that K_d (Ra) could be predicted by CEC [K_d (Ra) = 0.71 × CEC – 0.64, R^2=0.91] and soil organic matter content (%) [K_d (Ra) = 27 × OM–27, R^2=0.83].

1.3.4. Cofactors for other radionuclides

Other soil properties are as significant as mineral and organic matter contents in governing soil-radionuclide interactions for a large number of radionuclides. These other properties, alone or combined with textural information, can be used as cofactors for classification of soils in order to reduce the variability in the ranges of K_d values.

As for U, pH strongly affects the sorption of heavy metal radionuclides. Chemical speciation may also affect the K_d values of several radionuclides, since different species (*e.g.* oxidized-reduced species; oxyanions) may have contrasting sorption behaviour. As an example of this, a major effect of speciation on K_d values has been observed for selenium. Whereas selenate shows K_d values close to zero, for selenite they are up to a few thousand L kg^{-1} [30]. For iodine, the effect of the speciation (iodide and iodate) on the K_d should also be initially taken into account, as well as the content of organic matter and Fe and Al oxides, microbial activity and the water regime of the soil [31].

1.4. K_d database and ranges of K_d values

Estimates of K_d values for soils grouped on the basis of the texture and organic matter content criterion as well as on cofactors are given in Tables 1-16 and in Figs. 1-3. Data come from field and laboratory experiments, with various contamination sources, considering mainly the scenario of soils contaminated by radionuclides, and from references mostly from 1990 onwards, including the former Technical Reports Series No. 364 and related reports [32-34], reviewed papers, and grey literature (PhD theses; reports). Around 80 references (see Appendix) have been finally accepted to elaborate the K_d database. In most cases, data from experiments using other materials (*e.g.* sediments; pure soil phases such as clays or Fe-Mn-Al oxides; rock materials) or stable elements have not been considered. Data from radioisotopes of the same radioelement have been pooled. From around 2900 records for 67 elements, caesium and strontium have the highest number of observations. A few elements have more than 100 entries each, such as iodine, uranium, cobalt, potassium, antimony and selenium.

Comparing the new database with the former Technical Reports Series No. 364, no new data are available for radionuclides of a few elements (Ac, Br, Ho, Pa, Rb, Si and Sm), and data presented in the tables originate from the former Technical Reports Series No. 364. For a number of elements (Ag, Be, Bi, Hf, Mo, P, Pd, Sn, and Ta) although some new data are available, most originate also from the former Technical Reports Series No. 364. In contrast, data on elements not covered in the former Technical Reports Series No. 364 (As, Ba, Cl, Cu, Dy, Ga, H, Hg, In, Ir, K, La, Lu, Mg, Na, Pm, Pt, Rh, Sc, Tb, Te, Tm, V, and Y) have been included. However, for a few cases, data come from a single reference.

The K_d values, expressed in L kg^{-1}, have been grouped according to the organic matter content and sand and clay percentages of the mineral matter content (texture/OM criterion). This criterion for soil classification is described in earlier.[1]Besides presenting the K_d data according to the texture/OM criterion, the K_d values are also grouped for a limited number of radionuclides according to the cofactor approach, using specific soil properties (radiostrontium and radiocaesium), soil pH (uranium and heavy metal radionuclides with a sufficiently large number of observations) and speciation data and water regime (radioiodine).

Although AM and SD values are given, GM and GSD are preferred to describe K_d data, since the log-transformed K_d values are typically normally distributed. Exploratory analysis, based on box-and-whisker plots, has been applied to exclude potential outliers and thus to decrease

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[1] See paper by Fesenko et al. ‘Radioecological Definitions, Soil, Plant Classifications and Reference Ecological Data For Radiological Assessments’ in this publication.

data variability. A potential outlier is identified when it is beyond three times the interquartile ranges defined by the box-and-whisker plots.

1.4.1. Strontium

Whereas grouping the K_d (Sr) according to soil CEC values is a slightly worse approach than the soil texture/OM criterion (since there is not a consistent relationship between the CEC value and K_d (Sr)), grouping according to the CEC/$(Ca+Mg)_{ss}$ ratio leads to a good estimation of the K_d (Sr) and to the construction of ranges with lower variability (Tables 1 and 2, Fig. 1).

TABLE 1 K_d (Sr) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L kg^{-1})

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	255	5.2×10^1	6	2.0×10^2	5.4×10^2	4.0×10^{-1}	6.5×10^3	28
Sand	65	2.2×10^1	6	1.1×10^2	3.2×10^2	4.0×10^{-1}	2.4×10^3	19
Loam	120	5.7×10^1	5	1.6×10^2	2.9×10^2	2.0	2.5×10^3	12
Clay	19	9.5×10^1	4	1.9×10^2	2.0×10^2	9.0	7.5×10^2	5
Organic	37	1.1×10^2	6	4.9×10^2	1.2×10^3	3.0	6.5×10^3	10
Unspecified	14	7.3×10^1	3	1.1×10^2	9.0×10^1	8.0	2.7×10^2	8

TABLE 2 K_d (Sr) FOR SOILS GROUPED ACCORDING TO THE COFACTOR CRITERION (L kg^{-1})

Soil group	N	GM	GSD	AM	SD	Min	Max
CEC < 10	50	2.1×10^1	4	5.0×10^1	6.0×10^1	4.0×10^{-1}	3.3×10^2
10 < CEC < 20	44	2.0×10^2	2	2.3×10^2	1.0×10^2	3.3×10^1	4.6×10^2
20 < CEC < 50	82	6.2×10^1	6	2.9×10^2	8.6×10^2	1.0	6.5×10^3
CEC > 50	25	9.4×10^1	6	2.9×10^2	4.1×10^2	5.0	1.8×10^3
$CEC/M_{ss} < 15$	25	4.2	2	5.4	3	4.0×10^{-1}	1.5×10^1
$15 < CEC/M_{ss} < 150$	28	2.2×10^1	3	3.3×10^1	3.0×10^1	4.0	1.1×10^2
$150 < CEC/M_{ss} < 500$	18	1.7×10^2	2	1.8×10^2	6.0×10^1	7.7×10^1	2.7×10^2
$CEC/M_{ss} > 500$	25	3.2×10^2	2	4.1×10^2	3.5×10^2	8.1×10^1	1.8×10^3

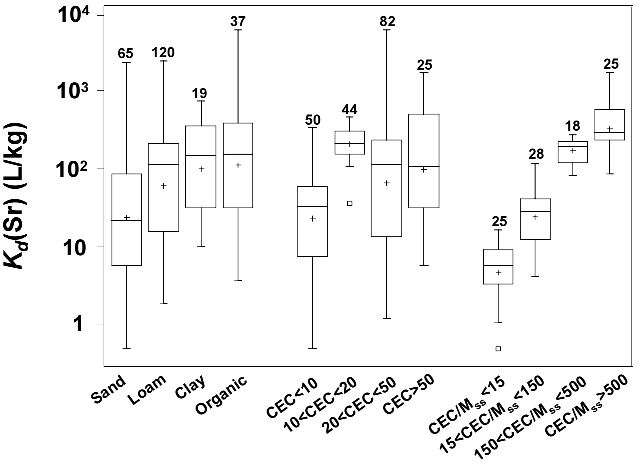


FIG. 1. Box-and-whisker plots of K_d (Sr) for soils grouped according to the texture/OM and cofactor criteria. The box encloses the middle 50% of the distribution of values, and the median is represented as a horizontal line inside the box. Vertical lines extend to the limits of the 1.5 interquartile ranges. Other symbols represent GM (+) and points at >1.5 interquartile ranges (□).

1.4.2. Caesium

As expected, the K_d (Cs) increases consistently with the clay content, and exhibits the lowest GM value in the organic soils. The K_d (Cs) can also be estimated for soils grouped according to their RIP values, and with respect to the ratio RIP/K_{ss}, which leads to a good prediction of the K_d (Cs) and to ranges of values of K_d (Cs) with the lowest variability (Tables 3 and 4, Fig. 2).

TABLE 3 K_d (Cs) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	469	1.2×10^3	7	6.1×10^3	2.1×10^4	4.3	3.8×10^5	32
Sand	114	5.3×10^2	6	2.2×10^3	5.0×10^3	1.0×10^1	3.5×10^4	19
Loam	191	3.5×10^3	4	7.2×10^3	9.9×10^3	3.9×10^1	5.5×10^4	17
Clay	36	5.5×10^3	4	2.2×10^4	6.7×10^4	5.7×10^2	3.8×10^5	9
Organic	108	2.7×10^2	7	3.0×10^3	1.2×10^4	4.3	9.5×10^4	14
Unspecified	20	1.7×10^3	5	6.7×10^3	1.5×10^4	4.0×10^1	5.5×10^4	8

TABLE 4 K_d (Cs) FOR SOILS GROUPED ACCORDING TO THE COFACTOR CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max
RIP < 150	47	7.4×10^1	2	1.1×10^2	1.3×10^2	1.0×10^1	7.3×10^2
150 < RIP < 1000	78	3.2×10^2	6	1.8×10^3	5.2×10^3	1.0×10^1	3.4×10^4
1000 < RIP < 2500	72	2.4×10^3	4	7.2×10^3	1.5×10^4	6.2×10^1	9.5×10^4
RIP > 2500	60	7.2×10^3	4	2.1×10^4	5.2×10^4	2.2×10^2	3.8×10^5
RIP/K_{ss} < 100	37	8.5×10^1	3	1.5×10^2	1.6×10^2	1.0×10^1	7.0×10^2
100 < RIP/K_{ss} < 1000	85	2.4×10^2	5	7.9×10^2	1.3×10^3	2.0×10^1	5.8×10^3
1000 < RIP/K_{ss} < 10000	78	2.0×10^3	4	4.5×10^3	6.7×10^3	6.2×10^1	3.4×10^4
RIP/K_{ss} > 10000	57	9.9×10^3	4	2.6×10^4	5.5×10^4	2.2×10^2	3.8×10^5

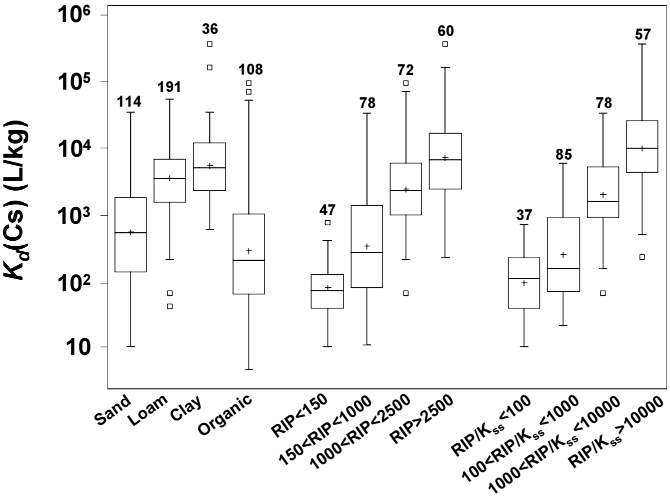


FIG. 2. Box-and-whisker plots of K_d (Cs) for soils grouped according to the texture/OM and cofactor criteria. The box encloses the middle 50% of the distribution of values, and the median is represented as a horizontal line inside the box. Vertical lines extend to the limits of the 1.5 interquartile ranges. Other symbols represent GM (+) and points at >1.5 interquartile ranges (□).

1.4.3. Uranium

Estimates of $K_d(U)$ values for soils grouped based on the texture/organic matter content cretarion and pH are given in Tables 5-6 and in Fig. 3.

Whereas values of K_d (U) do not show any correlation with soil texture, the values of K_d (U) can also be grouped according to the pH of the soil, which leads to ranges of K_d (U) with lower variability and confirms the K_d-pH dependence. A significant 10-fold higher K_d (U) value is observed for the 5-7 pH range.

TABLE 5 K_d (U) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	178	2.0×10^2	1.2×10^1	2.0×10^2	6.7×10^3	7.0×10^{-1}	6.7×10^4	22
Sand	50	1.1×10^2	1.2×10^1	2.1×10^3	9.5×10^3	7.0×10^{-1}	6.7×10^4	8
Loam	84	3.1×10^2	1.2×10^1	2.5×10^3	6.3×10^3	9.0×10^{-1}	3.9×10^4	12
Clay	12	2.8×10^1	7	1.2×10^2	1.7×10^2	2.6	4.8×10^2	3
Organic	9	1.2×10^3	6	2.9×10^3	2.8×10^3	3.3×10^1	7.6×10^3	7
Unspecified	23	1.7×10^2	6	8.6×10^2	1.7×10^3	1.6×10^1	6.2×10^3	5

TABLE 6 K_d(U) FOR SOILS GROUPED ACCORDING TO THE pH (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max
$pH < 5$	36	7.1×10^1	1.1×10^1	5.4×10^2	1.2×10^3	7.0×10^{-1}	6.7×10^3
$5 \leq pH < 7$	78	7.4×10^2	8	4.0×10^3	9.7×10^3	2.6	6.7×10^4
$pH \geq 7$	60	6.5×10^1	8	4.4×10^2	1.1×10^3	9.0×10^{-1}	6.2×10^3

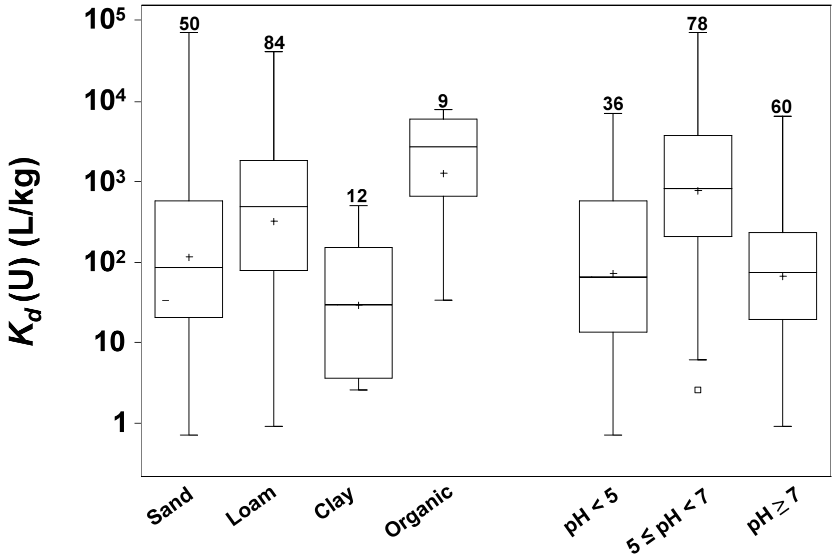


FIG. 3. Box-and-whisker plots of K_d (U) for soils grouped according to the texture/OM and pH criteria. The box encloses the middle 50%, and the median is represented as a horizontal line inside the box. Vertical lines extend to the point within 1.5 interquartile ranges. Other symbols represent GM (+) and points at > 1.5 interquartile ranges (□).

1.4.4. Iodine

K_d (I) values for soils grouped on the basis of the texture and organic matter content criterion as well as data on effect of watter content on K_d values for iodine are presented in Table 7.

The effect of organic matter and water regime can be more significant than the iodine species involved in the sorption, depending on the soil redox potential and factors such as drying temperature and sorption contact time. This indicates a complex dependency of K_d (I) on the organic matter and water content, microbial activity, and oxidizing-reducing conditions.

1.4.5. Heavy metals: Cd, Co, Ni, and Zn

K_d values for some heavy metals (Cd, Co, Ni, Zn) K_d values for soils grouped on the basis of the texture and organic matter content criterion as well as data on effect of pH (excluding organic soils) are given in Tables 8-15.

While there is not a straight relationship between the soil texture and the K_d value for the selected heavy metals, the K_d values of these heavy metal radionuclides in mineral soils show a clear dependence on pH.

2. CHANGES IN SOIL-RADIONUCLIDE INTERACTIONS WITH TIME: SORPTION DYNAMICS

Radionuclide-soil interactions exhibit changes with time. After an extended period since the contamination, radionuclides incorporated in the solid soil phase can occur in exchangeable and non-exchangeable forms. Whereas the exchangeable form includes radionuclide sorbed by the ion-exchange mechanism that is easily transferred to soil solution, the non-exchangeable form refers to radionuclide sorbed by irreversible mechanisms. The significance of the fixation process is controlled by the rate of transition between the mobile and fixed states.

TABLE 7 K_d (I) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION, SPECIATION, AND ORGANIC MATTER CONTENT (L/kg). ADDITIONAL EFFECT OF WATER CONTENT ON K_d (I)

	Soil group	N	GM	GSD	AM[1]	SD	Min	Max	# ref.
All data	All soils	250	6.9	5	2.5×10^1	7.0×10^1	1.0×10^{-2}	5.8×10^2	9
	Sand	48	4.1	7	1.3×10^1	2.0×10^1	1.0×10^{-2}	1.3×10^2	7
	Loam	129	8.0	4	2.5×10^1	7.0×10^1	2.0×10^{-1}	5.4×10^2	6
	Clay	19	1.1×10^1	5	3.1×10^1	4.0×10^1	1.0	1.8×10^2	5
	Organic	11	3.2×10^1	3	8.0×10^1	1.7×10^2	8.5	5.8×10^2	4
	Unspecified	43	4.2	6	1.8×10^1	6.0×10^1	1.0×10^{-1}	3.7×10^2	2
I^-	All soils	157	5.4	6	2.5×10^1	7.0×10^1	1.0×10^{-2}	5.8×10^2	6
	Sand	37	3.6	8	1.3×10^1	2.0×10^1	1.0×10^{-2}	1.3×10^2	5
	Loam	74	6.5	5	2.4×10^1	7.0×10^1	2.0×10^{-1}	5.3×10^2	4
	Clay	13	6.8	6	2.1×10^1	3.0×10^1	1.0	1.2×10^2	2
	Organic	9	3.6×10^1	4	9.3×10^1	1.8×10^2	8.5	5.8×10^2	3
	Unspecified	24	2.6	6	2.0×10^1	7.0×10^1	1.0×10^{-1}	3.7×10^2	2
IO_3^-	All soils	67	7.9	4	2.3×10^1	7.0×10^1	4.0×10^{-1}	5.4×10^2	2
	Sand	6	3.6	5	1.0×10^1	2.0×10^1	4.0×10^{-1}	4.1×10^1	1
	Loam	41	8.9	4	2.9×10^1	8.0×10^1	1.0	5.4×10^2	2

	Organic	1	-	-	1.3×10^{1}	-	-	-	1
	Unspecified	19	7.7	5	1.6×10^{1}	2.0×10^{1}	4.0×10^{-1}	5.8×10^{1}	2
All data	OM < 2	75	2.3	6	7.3	1.0×10^{1}	1.0×10^{-2}	5.7×10^{1}	
	$2 \leq$ OM < 5	106	9.1	3	2.0×10^{1}	5.4×10^{1}	6.0×10^{-1}	5.4×10^{2}	
	$5 \leq$ OM < 10	27	1.8×10^{1}	4	3.8×10^{1}	5.5×10^{1}	2.0	2.6×10^{2}	
	OM $\geq$ 10	19	3.4×10^{1}	3	8.7×10^{1}	1.7×10^{2}	8.5	5.8×10^{2}	
Effect of water content on the K_d [30, 35-36]									3
I^-	Sand-dry	1	-	-	2.8×10^{1}	-	-	-	
	Sand-wet	1	-	-	3.2×10^{1}	-	-	-	
	Unspecified-dry	9	1.9×10^{1}	6	7.5×10^{1}	1.5×10^{2}	8.0×10^{-1}	4.7×10^{2}	
	Unspecified-wet	9	5.5×10^{2}	6	1.4×10^{3}	2.2×10^{3}	8.0	7.0×10^{3}	
IO_3^-	Sand-dry	1	-	-	2.8×10^{1}	-	-	-	
	Sand-wet	1	-	-	3.5×10^{1}	-	-	-	
	Unspecified-dry	9	1.5×10^{1}	7	8.5×10^{1}	1.8×10^{2}	7.0×10^{-1}	5.5×10^{2}	
	Unspecified-wet	9	6.8×10^{2}	7	1.8×10^{3}	2.3×10^{3}	8.0	7.5×10^{3}	

[1] Single value is given in this column if N=1.

TABLE 8 K_d (Cd) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	61	1.5×10^{2}	9	7.7×10^{2}	1.3×10^{3}	2.0	7.0×10^{3}	11
Sand	30	1.1×10^{2}	8	4.2×10^{2}	5.3×10^{2}	2.0	1.8×10^{3}	5
Loam	5	1.0×10^{2}	7	4.1×10^{2}	7.3×10^{2}	9.2	1.7×10^{3}	4
Clay	4	1.3×10^{2}	1.5×10^{1}	8.2×10^{2}	1.3×10^{3}	6.9	2.7×10^{3}	3
Organic	13	6.5×10^{2}	6	1.7×10^{3}	2.2×10^{3}	9.6	7.0×10^{3}	6
Unspecified	9	6.8×10^{1}	1.5×10^{1}	7.3×10^{2}	1.4×10^{3}	6.2	4.4×10^{3}	4

TABLE 9 K_d (Cd) FOR SOILS GROUPED ACCORDING TO THE pH (EXCLUDING ORGANIC SOILS; L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max
pH < 5	8	1.1×10^{1}	3	1.9×10^{1}	2.0×10^{1}	2.0	6.4×10^{1}
$5 \leq$ pH < 6.5	11	1.8×10^{1}	4	5.4×10^{1}	9.0×10^{1}	6.2	2.5×10^{2}
pH $\geq$ 6.5	24	3.8×10^{2}	6	9.2×10^{2}	1.0×10^{2}	3.7	4.4×10^{3}

TABLE 10 K_d (Co) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	118	4.8×10^{2}	1.6×10^{1}	6.3×10^{3}	1.7×10^{4}	2.0	1.0×10^{5}	8
Sand	18	2.6×10^{2}	1.8×10^{1}	4.8×10^{3}	1.1×10^{4}	5.0	3.7×10^{4}	4
Loam	71	8.1×10^{2}	1.5×10^{1}	6.6×10^{3}	1.8×10^{4}	2.0	1.0×10^{5}	5
Clay	10	3.8×10^{3}	6	1.6×10^{4}	3.2×10^{4}	5.4×10^{2}	9.9×10^{4}	3
Organic	17	8.7×10^{1}	9	7.0×10^{2}	1.6×10^{3}	4.0	5.8×10^{3}	4
Unspecified	2	-	-	7.5×10^{3}	-	1.3×10^{2}	1.5×10^{4}	2

TABLE 11 K_d (Co) FOR SOILS GROUPED ACCORDING TO THE pH (EXCLUDING ORGANIC SOILS; L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max
pH < 5	21	1.2×10^{1}	5	3.5×10^{1}	5.0×10^{1}	2.0	1.5×10^{2}
5 ≤ pH < 6.5	50	1.9×10^{3}	5	9.5×10^{3}	2.0×10^{4}	2.9×10^{1}	9.9×10^{4}
pH ≥ 6.5	26	4.6×10^{3}	4	1.0×10^{4}	2.4×10^{4}	5.5×10^{2}	1.0×10^{5}

TABLE 12 K_d (Ni) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	64	2.8×10^{2}	7	9.5×10^{2}	1.4×10^{3}	3.0	7.2×10^{3}	12
Sand	26	1.3×10^{2}	1.0×10^{1}	9.4×10^{2}	1.8×10^{3}	3.0	7.2×10^{3}	4
Loam	14	1.8×10^{2}	5	4.1×10^{2}	4.1×10^{2}	7.7	1.2×10^{3}	6
Clay	12	9.3×10^{2}	2	1.2×10^{3}	9.4×10^{2}	2.5×10^{2}	3.2×10^{3}	5
Organic	8	1.1×10^{3}	2	1.4×10^{3}	1.5×10^{3}	4.1×10^{2}	5.0×10^{3}	3
Unspecified	4	4.8×10^{2}	8	1.1×10^{3}	9.6×10^{2}	2.2×10^{1}	2.3×10^{3}	3

TABLE 13 K_d (Ni) FOR SOILS GROUPED ACCORDING TO THE pH (EXCLUDING ORGANIC SOILS; L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max
pH < 5	10	1.4×10^{1}	2	1.8×10^{1}	1.0×10^{1}	3.0	4.8×10^{1}
5 ≤ pH < 6.5	11	5.8×10^{1}	4	1.6×10^{2}	3.0×10^{2}	7.0	1.1×10^{3}
pH ≥ 6.5	30	8.2×10^{2}	3	1.4×10^{3}	1.6×10^{3}	4.0×10^{1}	7.3×10^{3}

TABLE 14 K_d (Zn) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Soil group	N	GM	GSD	AM	SD	Min	Max	# ref.
All soils	92	9.5×10^{2}	1.1×10^{1}	5.2×10^{3}	1.6×10^{4}	9.0×10^{-1}	1.5×10^{5}	11
Sand	17	1.1×10^{2}	2.3×10^{1}	2.6×10^{3}	6.8×10^{3}	9.0×10^{-1}	2.8×10^{4}	6
Loam	48	2.4×10^{3}	4	7.6×10^{3}	2.2×10^{4}	2.1×10^{2}	1.5×10^{5}	4
Clay	8	2.4×10^{3}	2	3.2×10^{3}	2.4×10^{3}	4.8×10^{2}	6.9×10^{3}	3
Organic	12	5.6×10^{2}	8	1.9×10^{3}	2.5×10^{3}	9.7	7.6×10^{3}	6
Unspecified	7	2.4×10^{2}	3.5×10^{1}	2.6×10^{3}	2.8×10^{3}	4.6	6.2×10^{3}	5

TABLE 15 K_d (Zn) FOR SOILS GROUPED ACCORDING TO THE pH (EXCLUDING ORGANIC SOILS; L/kg)

	N	GM	GSD	AM	SD	Min	Max
pH < 5	9	8.2	8	6.0×10^{1}	1.1×10^{2}	0.9×10^{-1}	3.0×10^{2}
5 ≤ pH < 6.5	49	1.6×10^{3}	6	4.2×10^{3}	6.0×10^{3}	6.2	3.0×10^{4}
pH ≥ 6.5	17	4.3×10^{3}	4	1.4×10^{4}	3.6×10^{4}	4.4×10^{2}	1.5×10^{5}

1.4.6. Miscellany of elements

TABLE 16 K_d FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION (L/kg)

Element	Soil group	N	GM	GSD	AM/Value	SD	Min	Max	# ref.
Ac	All soils	4	1.7×10^3	3	2.4×10^3	2.0×10^3	4.5×10^2	5.4×10^3	1[a]
	Sand	1	-	-	4.5×10^2	-	-	-	1
	Loam	1	-	-	1.5×10^3	-	-	-	1
	Clay	1	-	-	2.4×10^3	-	-	-	1
	Organic	1	-	-	5.4×10^3	-	-	-	1
Ag	All soils	9	3.8×10^2	7	2.3×10^3	5.0×10^3	3.6×10^1	1.5×10^4	5
	Sand	3	1.3×10^2	5	2.7×10^2	3.7×10^2	3.6×10^1	7.0×10^2	3
	Loam	1	-	-	1.2×10^2	-	-	-	1[a]
	Clay	1	-	-	1.8×10^2	-	-	-	1[a]
	Organic	2	-	-	9.7×10^3	-	4.4×10^3	1.5×10^4	2
	Unspecified	2	-	-	2.6×10^2	-	1.2×10^2	4.0×10^2	2
Am	All soils	62	2.6×10^3	6	1.0×10^4	1.8×10^4	5.0×10^1	1.1×10^5	6
	Sand	17	1.0×10^3	7	5.1×10^3	1.0×10^4	6.7×10^1	3.7×10^4	4
	Loam	31	4.2×10^3	6	1.2×10^4	1.6×10^4	5.0×10^1	4.8×10^4	5
	Clay	1	-	-	8.1×10^3	-	-	-	1[a]
	Organic	13	2.5×10^3	5	1.1×10^4	3.0×10^4	2.1×10^2	1.1×10^5	4
As	All soils	7	5.5×10^2	5	1.2×10^3	1.2×10^3	2.5×10^1	3.0×10^3	3
	Sand	4	2.1×10^2	5	4.7×10^2	6.0×10^2	2.5×10^1	1.4×10^3	2
	Loam	1	-	-	1.0×10^3	-	-	-	1
	Unspecified	2	-	-	2.8×10^3	-	2.5×10^3	3.0×10^3	2
Ba	Loam	1	-	-	4.0×10^{-1}	-	-	-	1
Be	All soils	5	9.9×10^2	2	1.3×10^3	1.0×10^3	2.4×10^2	3.0×10^3	2
	Sand	1	-	-	2.4×10^2	-	-	-	1[a]
	Loam	1	-	-	8.1×10^2	-	-	-	1[a]
	Clay	1	-	-	1.3×10^3	-	-	-	1[a]
	Organic	1	-	-	3.0×10^3	-	-	-	1[a]
	Unspecified	1	-	-	1.3×10^3	-	-	-	1
Bi	All soils	6	4.8×10^2	2	6.1×10^2	4.7×10^2	1.2×10^2	1.5×10^3	2
	Sand	2	-	-	3.0×10^2	-	1.2×10^2	4.9×10^2	2[a,b]
	Loam	1	-	-	4.0×10^2	-	-	-	1[a]
	Clay	1	-	-	6.7×10^2	-	-	-	1[a]
	Organic	1	-	-	1.5×10^3	-	-	-	1[a]
	Unspecified	1	-	-	4.9×10^2	-	-	-	1[b]
Br	All soils	4	5.5×10^1	3	8.0×10^1	7.0×10^1	1.5×10^1	1.8×10^2	1[a]

	Sand	1	-	-	1.5×10^{1}	-	-	-	1
	Loam	1	-	-	4.9×10^{1}	-	-	-	1
	Clay	1	-	-	7.4×10^{1}	-	-	-	1
	Organic	1	-	-	1.8×10^{2}	-	-	-	1
Ca	All soils	34	8.0	3	1.7×10^{1}	4.0×10^{1}	7.0×10^{-1}	1.1×10^{2}	2
	Sand	7	3.0	4	6.6	1.0×10^{1}	7.0×10^{-1}	2.8×10^{1}	2
	Loam	21	8.3	3	1.4×10^{1}	2.0×10^{1}	2.0	8.9×10^{1}	2
	Clay	5	1.6×10^{1}	3	2.3×10^{1}	2.0×10^{1}	6.0	4.9×10^{1}	2
	Organic	1	-	-	1.1×10^{2}	-	-	-	1
Ce	All soils	11	1.2×10^{3}	5	3.7×10^{3}	5.9×10^{3}	1.2×10^{2}	2.0×10^{4}	5
	Sand	3	4.0×10^{2}	1	4.0×10^{2}	9.0×10^{1}	3.2×10^{2}	4.9×10^{2}	3
	Loam	4	3.0×10^{3}	3	4.1×10^{3}	3.1×10^{3}	6.5×10^{2}	8.1×10^{3}	4
	Clay	3	9.1×10^{2}	1.5×10^{1}	6.8×10^{3}	1.1×10^{4}	1.2×10^{2}	2.0×10^{4}	2
	Organic	1	-	-	3.0×10^{3}	-	-	-	1[a]
Cl	All soils	22	3.0×10^{-1}	3	5.0×10^{-1}	4.0×10^{-1}	4.0×10^{-2}	1.2	4
	Sand	3	5.0×10^{-1}	4	7.0×10^{-1}	5.0×10^{-1}	1.0×10^{-1}	1.1	2
	Loam	10	4.0×10^{-1}	3	5.0×10^{-1}	3.0×10^{-1}	4.0×10^{-2}	9.0×10^{-1}	3
	Clay	5	2.0×10^{-1}	3	4.0×10^{-1}	4.0×10^{-1}	6.0×10^{-2}	9.0×10^{-1}	4
	Organic	2	-	-	7.0×10^{-1}	-	1.0×10^{-1}	1.2	2
	Unspecified	2	-	-	2.0×10^{-1}	-	1.0×10^{-1}	2.0×10^{-1}	1
Cm	All soils	18	9.3×10^{3}	4	1.6×10^{4}	1.4×10^{4}	1.9×10^{2}	5.2×10^{4}	2
	Sand	5	3.4×10^{3}	1.4×10^{1}	1.0×10^{4}	1.3×10^{4}	1.9×10^{2}	3.1×10^{4}	2
	Loam	9	1.9×10^{4}	2	2.3×10^{4}	1.5×10^{4}	6.8×10^{3}	5.2×10^{4}	2
	Clay	1	-	-	5.4×10^{3}	-	-	-	1[a]
	Organic	3	7.4×10^{3}	2	7.9×10^{3}	3.6×10^{3}	5.1×10^{3}	1.2×10^{4}	2
Cr	All soils	31	4.0×10^{1}	2.0×10^{1}	7.3×10^{2}	1.7×10^{3}	1.0	7.9×10^{3}	6
	Sand	9	8.4	8	3.5×10^{1}	4.0×10^{1}	1.2	1.0×10^{2}	4
	Loam	9	4.5×10^{1}	2.3×10^{1}	4.2×10^{2}	5.7×10^{2}	1.0	1.6×10^{3}	3
	Clay	5	1.4×10^{1}	2.0×10^{1}	3.1×10^{2}	6.7×10^{2}	1.0	1.5×10^{3}	2
	Organic	6	1.6×10^{2}	1.0×10^{1}	7.2×10^{2}	1.1×10^{3}	8.3	2.9×10^{3}	3
	Unspecified	2	-	-	6.4×10^{3}	-	4.8×10^{3}	7.9×10^{3}	2
Cu	All soils	11	5.3×10^{2}	3	8.7×10^{2}	8.7×10^{2}	7.6×10^{1}	2.7×10^{3}	4
	Sand	2	-	-	2.3×10^{2}	-	1.3×10^{2}	3.3×10^{2}	1
	Loam	1	-	-	4.8×10^{2}	-	-	-	1
	Clay	2	-	-	2.1×10^{3}	-	1.4×10^{3}	2.7×10^{3}	1
	Organic	4	3.2×10^{2}	3	4.6×10^{2}	3.7×10^{2}	7.6×10^{1}	8.8×10^{2}	2
	Unspecified	2	-	-	1.3×10^{3}	-	5.0×10^{2}	2.1×10^{3}	2
Dy	All soils	2	-	-	1.5×10^{3}	-	8.2×10^{2}	2.1×10^{3}	1[b]
	Sand	1	-	-	8.2×10^{2}	-	-	-	1

	Unspecified	1	-	-	2.1×10^{3}	-	-	-	1
Fe	All soils	23	8.8×10^{2}	2	1.2×10^{3}	1.1×10^{3}	2.2×10^{2}	4.9×10^{3}	2
	Sand	4	3.2×10^{2}	1	3.3×10^{2}	9.0×10^{1}	2.2×10^{2}	4.2×10^{2}	2
	Loam	12	8.9×10^{2}	2	1.1×10^{3}	7.2×10^{2}	2.9×10^{2}	2.2×10^{3}	2
	Clay	4	1.6×10^{3}	1	1.7×10^{3}	5.5×10^{2}	1.2×10^{3}	2.2×10^{3}	1
	Organic	3	1.4×10^{3}	3	2.2×10^{3}	2.4×10^{3}	5.2×10^{2}	4.9×10^{3}	2
Ga	All soils	2	-	-	3.0×10^{2}	-	2.8×10^{2}	3.1×10^{2}	1[b]
	Sand	1	-	-	3.1×10^{2}	-	-	-	1
	Unspecified	1	-	-	2.8×10^{2}	-	-	-	1
H	Sand	1	-	-	1.0×10^{-1}	-	-	-	1
Hf	All soils	6	2.5×10^{3}	3	3.6×10^{3}	3.0×10^{3}	4.5×10^{2}	8.5×10^{3}	2
	Sand	2	-	-	1.9×10^{3}	-	4.5×10^{2}	3.3×10^{3}	2[a,b]
	Loam	1	-	-	1.5×10^{3}	-	-	-	1[a]
	Clay	1	-	-	2.4×10^{3}	-	-	-	1[a]
	Organic	1	-	-	5.4×10^{3}	-	-	-	1[a]
	Unspecified	1	-	-	8.5×10^{3}	-	-	-	1[b]
Hg	Unspecified	1	-	-	6.3×10^{3}	-	-	-	1
Ho	All soils	4	9.3×10^{2}	3	1.3×10^{3}	1.2×10^{3}	2.4×10^{2}	3.0×10^{3}	1[a]
	Sand	1	-	-	2.4×10^{2}	-	-	-	1
	Loam	1	-	-	8.1×10^{2}	-	-	-	1
	Clay	1	-	-	1.3×10^{3}	-	-	-	1
	Organic	1	-	-	3.0×10^{3}	-	-	-	1
In	All soils	2	-	-	4.8×10^{2}	-	2.4×10^{2}	7.3×10^{2}	1[b]
	Sand	1	-	-	2.4×10^{2}	-	-	-	1
	Unspecified	1	-	-	7.3×10^{2}	-	-	-	1
Ir	Unspecified	15	3.0	-	-	-	1	1.1×10^{1}	1
K	All soils	237	1.3×10^{1}	4	3.5×10^{1}	7.0×10^{1}	7.0×10^{-1}	9.1×10^{2}	9
	Sand	60	3.4	3	1.4×10^{1}	4.0×10^{1}	7.0×10^{-1}	1.8×10^{2}	8
	Loam	81	2.0×10^{1}	4	5.0×10^{1}	1.1×10^{2}	1.8	9.1×10^{2}	5
	Clay	12	4.3×10^{1}	3	7.7×10^{1}	9.0×10^{1}	9.3	2.9×10^{2}	2
	Organic	76	1.9×10^{1}	3	3.0×10^{1}	3.0×10^{1}	2.5	1.3×10^{2}	5
	Unspecified	8	1.1×10^{1}	3	2.9×10^{1}	6.0×10^{1}	4.3	1.8×10^{2}	2
La	Sand	1	-	-	5.3×10^{3}	-	-	-	1[b]
Lu	Sand	1	-	-	5.1×10^{3}	-	-	-	1[b]
Mg	All soils	30	3.8	3	8.0	1.0×10^{1}	4.0×10^{-1}	4.5×10^{1}	1
	Sand	6	1.3	4	3.5	6	4.0×10^{-1}	1.6×10^{1}	1
	Loam	20	4.8	3	8.8	1.0×10^{1}	9.0×10^{-1}	4.5×10^{1}	1
	Clay	4	6.8	3	1.1×10^{1}	1.0×10^{1}	2.1	2.9×10^{1}	1

Mn	All soils	83	1.2×10^{3}	9	9.8×10^{3}	1.8×10^{4}	3.6×10^{1}	7.9×10^{4}	4
	Sand	13	9.8×10^{2}	1.4×10^{1}	1.2×10^{4}	2.5×10^{4}	4.0×10^{1}	7.9×10^{4}	4
	Loam	56	1.1×10^{3}	8	7.7×10^{3}	1.5×10^{4}	6.0×10^{1}	7.7×10^{4}	3
	Clay	10	4.5×10^{3}	1.3×10^{1}	2.2×10^{4}	2.4×10^{4}	1.4×10^{2}	5.7×10^{4}	3
	Organic	3	1.6×10^{2}	4	2.5×10^{2}	2.3×10^{2}	3.6×10^{1}	4.9×10^{2}	2
	Unspecified	1	-	-	1.0×10^{4}	-	-	-	1
Mo	All soils	9	3.8×10^{1}	3	5.7×10^{1}	5.0×10^{1}	7.4	1.3×10^{2}	5
	Sand	2	-	-	4.5×10^{1}	-	7.4	8.2×10^{1}	2
	Loam	1	-	-	1.3×10^{2}	-	-	-	1[a]
	Clay	1	-	-	9.0×10^{1}	-	-	-	1[a]
	Organic	2	-	-	2.3×10^{1}	-	1.8×10^{1}	2.7×10^{1}	2
	Unspecified	3	3.7×10^{1}	3	5.3×10^{1}	5.0×10^{1}	1.3×10^{1}	1.1×10^{2}	3
Na	All soils	30	3.4	3	5.7	6	2.0×10^{-1}	2.6×10^{1}	1
	Sand	6	2.2	4	5.2	9	4.0×10^{-1}	2.3×10^{1}	1
	Loam	20	4.6	2	6.3	6	3.0×10^{-1}	2.6×10^{1}	1
	Clay	4	1.7	6	3.9	5	2.0×10^{-1}	1.1×10^{1}	1
Nb	All soils	11	1.5×10^{3}	4	2.9×10^{3}	2.7×10^{3}	1.6×10^{2}	8.4×10^{3}	3
	Sand	2	-	-	1.7×10^{2}	-	1.6×10^{2}	1.9×10^{2}	2
	Loam	5	2.5×10^{3}	3	3.6×10^{3}	3.0×10^{3}	5.4×10^{2}	8.4×10^{3}	2
	Clay	3	2.4×10^{3}	2	3.0×10^{3}	1.9×10^{3}	9.0×10^{2}	4.7×10^{3}	2
	Organic	1	-	-	2.0×10^{3}	-	-	-	1[a]
Np	All soils	26	3.6×10^{1}	6	1.6×10^{2}	3.2×10^{2}	1.3	1.2×10^{3}	3
	Sand	8	1.4×10^{1}	4	2.9×10^{1}	4.0×10^{1}	3.0	1.1×10^{2}	3
	Loam	12	2.3×10^{1}	4	4.2×10^{1}	4.0×10^{1}	1.3	1.2×10^{2}	3
	Clay	2	-	-	3.8×10^{1}	-	2.0×10^{1}	5.5×10^{1}	2
	Organic	4	8.1×10^{2}	1	8.5×10^{2}	2.9×10^{2}	5.0×10^{2}	1.2×10^{3}	3
P	All soils	6	8.7×10^{1}	5	2.2×10^{2}	3.0×10^{2}	9.0	7.6×10^{2}	2
	Sand	2	-	-	3.9×10^{2}	-	9.0	7.6×10^{2}	2
	Loam	2	-	-	2.0×10^{2}	-	3.0×10^{1}	3.8×10^{2}	2
	Clay	1	-	-	4.9×10^{1}	-	-	-	1[a]
	Organic	1	-	-	1.1×10^{2}	-	-	-	1[a]
Pa	All soils	4	2.0×10^{3}	3	2.9×10^{3}	2.6×10^{3}	5.4×10^{2}	6.6×10^{3}	1[a]
	Sand	1	-	-	5.4×10^{2}	-	-	-	1
	Loam	1	-	-	1.8×10^{3}	-	-	-	1
	Clay	1	-	-	2.7×10^{3}	-	-	-	1
	Organic	1	-	-	6.6×10^{3}	-	-	-	1
Pb	All soils	23	2.0×10^{3}	1.0×10^{1}	1.5×10^{4}	3.3×10^{4}	2.5×10^{1}	1.3×10^{5}	5
	Sand	9	2.2×10^{2}	4	4.0×10^{2}	4.3×10^{2}	2.5×10^{1}	1.3×10^{3}	2
	Loam	5	1.0×10^{4}	3	1.5×10^{4}	1.6×10^{4}	3.6×10^{3}	4.3×10^{4}	2
	Clay	2	-	-	6.6×10^{4}	-	5.4×10^{3}	1.3×10^{5}	2
	Organic	5	2.5×10^{3}	3	3.7×10^{3}	3.8×10^{3}	8.8×10^{2}	1.0×10^{4}	2

	Unspecified	2	-	-	5.9×10^{4}	-	1.6×10^{4}	1.0×10^{5}	2
Pd	All soils	6	1.8×10^{2}	2	2.4×10^{2}	2.2×10^{2}	5.5×10^{1}	6.7×10^{2}	2
	Sand	2	-	-	9.0×10^{1}	-	5.5×10^{1}	1.3×10^{2}	2[a,b]
	Loam	1	-	-	1.8×10^{2}	-	-	-	1[a]
	Clay	1	-	-	2.7×10^{2}	-	-	-	1[a]
	Organic	1	-	-	6.7×10^{2}	-	-	-	1[a]
	Unspecified	1	-	-	1.7×10^{2}	-	-	-	1[b]
Pm	All soils	2	-	-	4.5×10^{2}	-	4.5×10^{2}	4.5×10^{2}	1[b]
	Sand	1	-	-	4.5×10^{2}	-	-	-	1
	Unspecified	1	-	-	4.5×10^{2}	-	-	-	1
Po	All soils	44	2.1×10^{2}	5	5.6×10^{2}	1.1×10^{3}	1.2×10^{1}	7.0×10^{3}	2
	Sand	14	1.0×10^{2}	6	7.4×10^{2}	1.9×10^{3}	1.7×10^{1}	7.0×10^{3}	1
	Loam	27	2.3×10^{2}	4	4.6×10^{2}	4.6×10^{2}	1.2×10^{1}	1.8×10^{3}	2
	Clay	2	-	-	1.7×10^{3}	-	7.2×10^{2}	2.7×10^{3}	2
	Organic	1	-	-	6.6×10^{3}	-	-	-	1[a]
Pt	Unspecified	15	2.4×10^{1}	-	-	-	1.2×10^{1}	8.3×10^{1}	1
Pu	All soils	62	7.4×10^{2}	4	1.6×10^{3}	2.0×10^{3}	3.2×10^{1}	9.6×10^{3}	7
	Sand	11	4.0×10^{2}	4	1.0×10^{3}	2.0×10^{3}	3.3×10^{1}	6.9×10^{3}	5
	Loam	27	9.5×10^{2}	4	1.9×10^{3}	2.3×10^{3}	1.0×10^{2}	9.6×10^{3}	4
	Clay	10	1.8×10^{3}	2	2.5×10^{3}	2.2×10^{3}	4.3×10^{2}	7.6×10^{3}	4
	Organic	6	7.6×10^{2}	4	1.3×10^{3}	1.1×10^{3}	9.0×10^{1}	3.0×10^{3}	4
	Unspecified	8	2.3×10^{2}	5	6.1×10^{2}	8.5×10^{2}	3.2×10^{1}	2.1×10^{3}	2
Ra	All soils	51	2.5×10^{3}	1.3×10^{1}	3.4×10^{4}	1.3×10^{5}	1.2×10^{1}	9.5×10^{5}	8
	Sand	20	3.1×10^{3}	8	9.6×10^{3}	1.2×10^{4}	4.9×10^{1}	4.0×10^{4}	4
	Loam	19	1.1×10^{3}	1.7×10^{1}	1.5×10^{4}	3.2×10^{4}	1.2×10^{1}	1.2×10^{5}	5
	Clay	6	3.8×10^{4}	1.2×10^{1}	2.0×10^{5}	3.7×10^{4}	7.0×10^{2}	9.5×10^{5}	3
	Organic	2	-	-	1.3×10^{3}	-	2.0×10^{2}	2.4×10^{3}	2
	Unspecified	4	1.2×10^{3}	1	1.3×10^{3}	5.0×10^{2}	7.8×10^{2}	1.9×10^{3}	1
Rb	All soils	4	2.1×10^{2}	3	2.9×10^{2}	2.7×10^{2}	5.5×10^{1}	6.7×10^{2}	1[a]
	Sand	1	-	-	5.5×10^{1}	-	-	-	1
	Loam	1	-	-	1.8×10^{2}	-	-	-	1
	Clay	1	-	-	2.7×10^{2}	-	-	-	1
	Organic	1	-	-	6.7×10^{2}	-	-	-	1
Rh	Unspecified	12	4.0	-	-	-	6.0×10^{-1}	2.9×10^{1}	1
Ru	All soils	15	2.7×10^{2}	8	4.7×10^{3}	1.7×10^{4}	5.0	6.6×10^{4}	5
	Sand	3	3.6×10^{1}	6	7.7×10^{1}	9.0×10^{1}	5.0	1.7×10^{2}	3
	Loam	3	3.0×10^{2}	3	4.7×10^{2}	4.7×10^{2}	8.2×10^{1}	9.9×10^{2}	3
	Clay	4	5.0×10^{2}	2	6.0×10^{2}	3.6×10^{2}	2.0×10^{2}	9.9×10^{2}	3
	Organic	1	-	-	6.6×10^{4}	-	-	-	1[a]
	Unspecified	4	1.4×10^{2}	3	2.3×10^{2}	2.1×10^{2}	3.4×10^{1}	4.9×10^{2}	2
Sb	All soils	152	6.2×10^{1}	4	1.3×10^{2}	2.0×10^{2}	6.0×10^{-1}	2.1×10^{3}	5

	Sand	19	1.7×10^{1}	6	6.4×10^{1}	1.2×10^{2}	6.0×10^{-1}	4.7×10^{2}	4
	Loam	92	6.1×10^{1}	3	1.2×10^{2}	2.3×10^{2}	4.0	2.1×10^{3}	2
	Clay	18	1.4×10^{2}	2	2.0×10^{2}	1.7×10^{2}	3.8×10^{1}	6.1×10^{2}	2
	Organic	3	7.5×10^{1}	8	2.2×10^{2}	2.8×10^{2}	7.8	5.4×10^{2}	2
	Unspecified	20	9.9×10^{1}	4	1.7×10^{2}	1.6×10^{2}	4.0	6.1×10^{2}	2
Sc	All soils	2	-	-	2.1×10^{3}	-	6.7×10^{2}	3.5×10^{3}	1[b]
	Sand	1	-	-	6.7×10^{2}	-	-	-	1
	Unspecified	1	-	-	3.5×10^{3}	-	-	-	1
Se	All soils	172	2.0×10^{2}	3	3.6×10^{2}	3.7×10^{2}	4.0	2.1×10^{3}	10
	Sand	15	5.6×10^{1}	5	2.2×10^{2}	4.6×10^{2}	4.0	1.6×10^{3}	6
	Loam	101	2.2×10^{2}	3	3.7×10^{2}	3.5×10^{2}	1.2×10^{1}	1.6×10^{3}	4
	Clay	33	2.4×10^{2}	3	3.7×10^{2}	4.0×10^{2}	2.2×10^{1}	2.1×10^{3}	6
	Organic	2	-	-	1.0×10^{3}	-	2.3×10^{2}	1.8×10^{3}	2
	Unspecified	21	2.3×10^{2}	2	3.0×10^{2}	1.9×10^{2}	2.0×10^{1}	6.2×10^{2}	3
Si	All soils	4	1.3×10^{2}	3	1.8×10^{2}	1.6×10^{2}	3.3×10^{1}	4.0×10^{2}	1[a]
	Sand	1	-	-	3.3×10^{1}	-	-	-	1
	Loam	1	-	-	1.1×10^{2}	-	-	-	1
	Clay	1	-	-	1.8×10^{2}	-	-	-	1
	Organic	1	-	-	4.0×10^{2}	-	-	-	1
Sm	All soils	4	9.3×10^{2}	3	1.3×10^{3}	1.2×10^{3}	2.4×10^{2}	3.0×10^{3}	1[a]
	Sand	1	-	-	2.4×10^{2}	-	-	-	1
	Loam	1	-	-	8.1×10^{2}	-	-	-	1
	Clay	1	-	-	1.3×10^{3}	-	-	-	1
	Organic	1	-	-	3.0×10^{3}	-	-	-	1
Sn	All soils	12	1.6×10^{3}	6	5.7×10^{3}	9.1×10^{3}	1.3×10^{2}	3.1×10^{4}	4
	Sand	2	-	-	1.5×10^{2}	-	1.3×10^{2}	1.7×10^{2}	2[a,b]
	Loam	1	-	-	4.5×10^{2}	-	-	-	1[a]
	Clay	1	-	-	6.7×10^{2}	-	-	-	1[a]
	Organic	1	-	-	1.6×10^{3}	-	-	-	1[a]
	Unspecified	7	4.1×10^{3}	5	9.4×10^{3}	1.1×10^{4}	3.3×10^{2}	3.1×10^{4}	3
Ta	All soils	5	7.8×10^{2}	3	1.1×10^{3}	1.1×10^{3}	2.4×10^{2}	3.0×10^{3}	2
	Sand	2	-	-	3.1×10^{2}	-	2.4×10^{2}	3.8×10^{2}	2[a,b]
	Loam	1	-	-	8.1×10^{2}	-	-	-	1[a]
	Clay	1	-	-	1.3×10^{3}	-	-	-	1[a]
	Organic	1	-	-	3.0×10^{3}	-	-	-	1[a]
Tb	All soils	2	-	-	6.0×10^{3}	-	5.4×10^{3}	6.6×10^{3}	1[b]
	Sand	1	-	-	5.4×10^{3}	-	-	-	1
	Unspecified	1	-	-	6.6×10^{3}	-	-	-	1
Tc	All soils	33	2.3×10^{-1}	9	2	5	1.0×10^{-2}	1.1×10^{1}	4
	Sand	5	4.0×10^{-2}	3	5.0×10^{-2}	5.0×10^{-2}	1.0×10^{-2}	1.0×10^{-1}	2

	Loam	14	7.0×10^{-2}	3	2.0×10^{-1}	2.0×10^{-1}	1.0×10^{-2}	9.0×10^{-1}	4
	Clay	3	9.0×10^{-2}	1.0×10^{1}	4.0×10^{-1}	7.0×10^{-1}	2.0×10^{-2}	1.2	2
	Organic	11	3.1	3	6	7	9.2×10^{-1}	1.1×10^{1}	2
Te	All soils	2	-	-	4.9×10^{2}	-	1.8×10^{2}	7.9×10^{2}	1[b]
	Sand	1	-	-	1.8×10^{2}	-	-	-	1
	Unspecified	1	-	-	7.9×10^{2}	-	-	-	1
Th	All soils	46	1.9×10^{3}	1.0×10^{1}	1.6×10^{4}	4.2×10^{4}	1.8×10^{1}	2.5×10^{5}	8
	Sand	12	7.0×10^{2}	1.1×10^{1}	1.0×10^{4}	2.8×10^{4}	3.5×10^{1}	1.0×10^{5}	3
	Loam	6	1.8×10^{4}	4	5.3×10^{4}	9.7×10^{4}	5.0×10^{3}	2.5×10^{5}	2
	Clay	7	4.5×10^{3}	3	7.4×10^{3}	8.0×10^{3}	8.0×10^{2}	2.4×10^{4}	2
	Organic	5	7.3×10^{2}	4.4×10^{1}	1.9×10^{4}	3.5×10^{4}	1.8×10^{1}	8.0×10^{4}	3
	Unspecified	16	1.5×10^{3}	5	8.9×10^{3}	2.5×10^{4}	2.1×10^{2}	1.0×10^{5}	3
Tm	Sand	1	-	-	3.3×10^{2}	-	-	-	1[b]
V	All soils	2	-	-	3.0×10^{2}	-	1.8×10^{2}	4.1×10^{2}	1[b]
	Sand	1	-	-	1.8×10^{2}	-	-	-	1
	Unspecified	1	-	-	4.1×10^{2}	-	-	-	1
Y	All soils	7	4.7×10^{1}	4	6.5×10^{1}	1.0×10^{2}	1.0×10^{1}	3.8×10^{2}	1
	Sand	5	2.2×10^{1}	2	2.6×10^{1}	2.0×10^{1}	1.0×10^{1}	4.7×10^{1}	1
	Organic	2	-	-	3.2×10^{2}	-	2.6×10^{2}	3.8×10^{2}	1
Zr	All soils	11	4.1×10^{2}	2.1×10^{1}	3.0×10^{3}	3.8×10^{3}	1.5	1.0×10^{4}	4
	Sand	4	3.2×10^{1}	1.6×10^{1}	2.0×10^{2}	2.8×10^{2}	1.5	6.0×10^{2}	3
	Loam	2	-	-	5.2×10^{3}	-	2.2×10^{3}	8.1×10^{3}	2
	Clay	2	-	-	6.8×10^{3}	-	3.3×10^{3}	1.0×10^{4}	2
	Organic	2	-	-	3.7×10^{3}	-	2.3×10^{1}	7.3×10^{3}	2
	Unspecified	1	-	-	4.8×10^{2}	-	-	-	1[b]

[a] K_d values originate from the Technical Reports Series No. 364 [32].

[b] K_d values originate from experiments using neutron activation analyses [37].

2.1. A kinetic approach to examining sorption dynamics

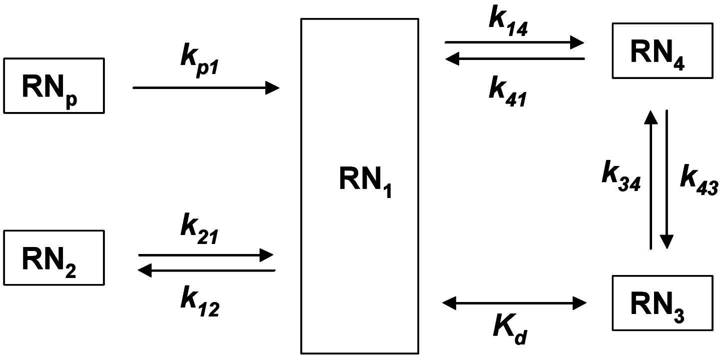
The form of radionuclides when they are first deposited on the soil is initially governed by their chemical form in the contamination source (*e.g.* fuel particles; water soluble fallout). Regarding soluble contamination sources, water soluble forms of radionuclides are sorbed onto soil particles and undergo a number of reactions in the solid soil phase. The processes underlying transformation between chemical forms of radionuclides in the soil and soil solution can be summarized as shown in Fig. 4 [38-39]. The transformation between chemical forms would eventually lead to changes (mostly, increases) in the K_d values, as a result of the transfer of the radionuclide to sites where it is no longer available to exchange with the radionuclide in solution, thus decreasing the reversibly sorbed radionuclide fraction. This process is often described as aging in the literature.

Kinetics studies have examined the changes in K_d value with time. Various models have been proposed to fit the experimental data. Although for several radionuclides their soluble complexes must be taken into account (RN_2), in most cases to postulate that the radionuclide occurs in the soil solution as a free cation is an adequate approach.

A three-box model is often used to describe the kinetics of radionuclide sorption [13, 38, 40]. This model assumes that the radionuclide in the soil solution may be sorbed either at a labile pool of sites (RN_3), thus defining an exchangeable K_d, or sorbed at non-exchangeable sites (RN_4). This process is estimated through its corresponding rate constants (see k_{14} and k_{41} in the Fig. 4), which describes a pseudo-first order reaction, assuming that forward and backward reactions are independent of the capacity of sorption sites and composition of the soil solution. Other models postulate that additional types of transformation should be considered. For instance, it has been suggested that the radionuclide reaches the non-labile pool after being sorbed in the labile pool [39]. Therefore, the corresponding rate constants (see k_{43} and k_{43} in the Fig. 4) must be quantified.

2.1.1. The case of radiostrontium

In most soil types, radiostrontium does not undergo complexation reactions in soil solution, in contrast to the transition elements. Moreover, the fixation process is almost negligible for this radionuclide [38], although slight decreases in the reversible sorbed fraction have been seen in the short-term after contamination [13, 41]. Besides the ion exchange mechanisms that control radiostrontium sorption in soils, other specific interactions (*e.g.* isomorphic substitutions in Ca-bearing minerals; formation of insoluble inorganic and organic compounds) have been postulated to explain radiostrontium aging processes. In any case, the rate constant k_{41} can be hypothesized to be much larger than k_{14}, resulting in a major part of radiostrontium being in an exchangeable form. Representative values for these rate constants in loam soils are $k_{14} = 0.01 - 0.03\ d^{-1}$ and $k_{41} = 0.13 - 0.14\ d^{-1}$ [42].



RN_1: Cationic form of a radionuclide in soil solution
RN_2: Radionuclide soluble complex compounds in the soil solution
RN_3: Radionuclide cation sorbed by ion-exchange process (labile pool)
RN_4: Non-exchangeable (strongly-bonded) form of a radionuclide (non-labile pool)
RN_p: Radionuclide in solid particles (fuel particles) as a source-term
K_d: Exchangeable (labile) distribution coefficient
k_{ij}: Rate constant for a given (kinetically controlled) transformation process

FIG. 4. Kinetic steps of radionuclide sorption in the soil solution-soil system.

2.1.2. The case of radiocaesium

As with radiostrontium, radiocaesium does not undergo complexation reactions in solution. For radiocaesium, the three-box kinetic model is the most usually applied approach to describe the kinetics of its sorption [13, 38, 40]. However, for this radionuclide it has been shown that after an initial rapid sorption at the most outer *FES* and on the organic and planar surface sites of the soil particles, there is a second kinetic step in which a moderately fast sorption on interlayer wedge sites in the *FES* takes place, where the radiocaesium remains fixed by the interlayer core. This process is often described as diffusion in the solid state, especially in the presence of expanded interlayers [43-44], which allows radiocaesium to penetrate into the crystal lattice of clays [45]. Moreover, the labile pool can be subdivided in relation to the types of site involved (regular exchange complex or specific sites at *FES*), while the sorption at the non-labile pool can be characterized as progressive fixation controlled by the solid-phase diffusion into collapsed micaceous interlayers [46]. For soils with a high organic matter content and/or extremely low clay content, this second process may be negligible. Table 17 summarizes some rate constants for ^{137}Cs fixation and release processes in various soils.

TABLE 17 ^{137}Cs RATE CONSTANTS (d^{-1}) FOR SOILS GROUPED ACCORDING TO THE TEXTURE/OM CRITERION

				k_{14}			k_{41}		
Source term	Soil group	N	GM/Value	Min	Max	GM/Value	Min	Max	# ref.
Water soluble	Sand	10	1.2×10^{-1}	5.1×10^{-2}	6.9×10^{-1}	6.7×10^{-2}	7.0×10^{-3}	1.9×10^{-1}	5[a]
Water soluble	Loam	4	2.6×10^{-1}	9.0×10^{-2}	4.3×10^{-1}	2.2×10^{-2}	7.0×10^{-3}	2.8×10^{-2}	
Water soluble	Clay	1	1.3	-	-	3.5×10^{-2}	-	-	
Water soluble	Organic	5	1.6	1.5×10^{-1}	1.2×10^{1}	2.2×10^{-1}	6.0×10^{-3}	8.7×10^{-1}	
Water soluble	Unspecified	10	1.5	6.0×10^{-1}	3.8	8.7×10^{-2}	1.9×10^{-2}	6.1×10^{-1}	
Water soluble	All soils	30	5.0×10^{-1}	5.0×10^{-2}	1.2×10^{1}	7.5×10^{-2}	6.0×10^{-3}	8.7×10^{-1}	
Leachate from particles	Unspecified	3	1.4×10^{-2}	1.2×10^{-2}	1.5×10^{-2}	1.4×10^{-3}	9.0×10^{-4}	3.7×10^{-3}	1[b]

				k_{34}			k_{43}		
Source term	Soil group	N	GM	Min	Max	GM	Min	Max	# ref.
Water soluble	Organic	5	3.0×10^{-2}	7.0×10^{-3}	2.0×10^{-1}	3.5×10^{-3}	6.0×10^{-4}	1.1×10^{-2}	1[c]

[a] [13, 40, 47-49]; [b] [47]; [c] [39].

2.2. Additional approaches to examining sorption dynamics

Other widespread experimental approaches to examining sorption dynamics are based on the construction of desorption isotherms in the same conditions as in the sorption step [50] or on the use of single and sequential extraction tests to quantify the desorption yields in controlled experimental conditions [51-54]. While the first approach may give some light on the mechanisms involved in sorption reversibility, the second represents a simpler, faster experiment to estimate the percentage of the radionuclide that may be remobilized.

Single and sequential extraction tests have been widely used when dealing with the characterization of field-contaminated soils [41, 55-56]. While single extractions focus on quantifying the reversibly sorbed fraction of a given radionuclide (f_{rev}), sequential extractions allow a better distinction between the exchangeable and non-exchangeable pools, and their changes with time. Some common patterns have been found in these studies, which agree with conclusions derived from the kinetic approach. Although it is extremely difficult to establish a range of f_{rev} values, the f_{rev} (Sr) is usually higher than f_{rev} (Cs) for mineral soils, whereas the opposite trend can be observed in organic soils. Regarding radiocaesium, the f_{rev}

(Cs) in mineral soils usually decreases with the texture sequence sand > loam > clay soils. For actinides, the f_{rev} values reported are much smaller than for radiostrontium and radiocaesium.

Although to date a successful approach to predicting desorption yields from sorption data has not been developed, the quantification of f_{rev} through the use of harmonized laboratory tests has improved radionuclide mobility predictions based solely on radionuclide distribution coefficients.

3. CONCLUSIONS

A significant amount of data has been added to the new K_d database, comparing with the former Technical Reports Series No. 364. However, there are still evident gaps of values of K_d for a substantial number of radionuclides and soil types. In some cases, values originate from a single reference. This fact restricts the possibility of proposing best estimates in many cases, and GM and single values must be considered only as approximate estimates, suitable for screening purposes, but not for specific risk assessments. For these gaps, the use of analogues (data on other elements or media, such as pure soil phases or sediments) is an option, but must be undertaken with care, with due consideration for the distinctions in chemical form and affinity for different types of binding site that may exist between the analogue and the radionuclide of interest.

The large number of experimental approaches used to quantify K_d values is partly responsible for the wide ranges of values that can be found for a given radionuclide-soil type combination. Values derived from mass transport experiments seem to lead to K_d values one-two orders of magnitude lower than those derived from batch experiments. Regarding this latter approach, the ionic composition of the contact solution may affect the K_d value by up to some orders of magnitude.

Finally, it is recommended not to use stable isotopes (at larger concentrations than radioisotopes), to quantify the K_d values of the corresponding radionuclide, since the derived values will often be significantly lower.

The K_d values derived from field-contaminated soils are usually higher than those resulting from a laboratory sorption tests, because the radionuclide quantified in the solid phase of the contaminated soil may include sorbed radionuclide not available for exchange with the soil solution due to a larger time elapsed since radionuclide incorporation, in comparison with laboratory experiments.

There is a need to have information on the reversibility of sorption and how it may change with time. The dynamics of the soil-radionuclide interaction is significant for a limited number of radionuclides, as in the cases of radiostrontium and radiocaesium. For radionuclides with very low or very high sorption, this may be a consideration of minor significance.

Soil-radionuclide interactions are governed by multiple factors that depend on the radionuclide and on various soil properties. As the quality and quantity of the mineral matter is one of the key soil properties, the definition of K_d values for soil groups based on soil texture and organic matter content is a satisfactory approach to establish the estimated values of K_d for a large number of radionuclides. However, it is recommended that consideration and use be made, as much as possible, of additional soil and radionuclide properties (cofactors) that govern soil-radionuclide interactions, for better prediction of K_d values and to decrease the variability in estimates of those values.

The main soil parameters controlling the interaction should be measured and monitored to improve the prediction of K_d, and they should also be included in models predicting the radionuclide transport in the environment. Clear examples are *RIP* and K and NH_4^+ status for radiocaesium, *CEC* and Ca and Mg concentrations for radiostrontium, and pH for heavy metal radionuclides and uranium. Therefore, modellers can choose to use the best estimates derived from the GM values of K_d for soils arranged according to texture and organic matter or, when available, according to other criteria such as specific properties, pH and speciation. Moreover, modellers and end-users can also consider using existing single and multiple correlations between soil properties and K_d to calculate best estimates from soil properties, especially in those cases where the mechanisms governing radionuclide interactions are well known.

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APPENDIX

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VERTICAL MIGRATION OF RADIONUCLIDES IN UNDISTURBED SOILS

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Abstract

Literature data on numerical values obtained for the parameters of the two most popular models for simulating the migration of radionuclides in undisturbed soils have been compiled and evaluated statistically. Parameters evaluated include D (dispersion coefficient) and v (effective convection velocity) of the Convection-Dispersion-Equation and migration rates derived from different approaches (compartment models, repeated measurements, CDE-approaches). Due to restrictions on the applicability of compartmental models, the convection–dispersion equation and its parameter values should be preferred. For temperate environments the data base is sufficiently strong, but for other environments (e.g. Tropics or Arctic) as well as for radionuclides other than caesium data on vertical migration are still scarce.

1. PHYSICAL AND CHEMICAL BEHAVIOUR OF RADIONUCLIDES IN SOILS

Soil parameters can have a distinct influence on the mobility of deposited radionuclides in soil. The most important factors governing radionuclide mobility in soil are as follows [1]:

- the composition of the soil solution (pH, concentration of inorganic ions, redox potential, concentration of organic substances),
- physical and chemical soil properties (species/characteristics and contents of clay minerals, oxides and organic matter, surface characteristics and charges of particles, soil texture/particle size distribution),
- micro-organisms and fungi (mycorrhiza) and
- temperature.

Most radionuclides in soil are present in cationic form (but iodine is an important exception). In general, therefore, low pH-values, low clay contents and a resulting low cation exchange capacity lead to increased radionuclide mobility within the soil profile [2]. Changes of soil organic matter contents can yield different effects, depending on the ability of the respective radionuclide to form organic complexes or not [3].

Furthermore, the vertical migration velocity of radionuclides is determined by the site-specific water balance. Transport of radionuclides in both soluble form and bound to soil particles (colloids) is possible, especially for less water-soluble radionuclides. For the migration of different radionuclides in the soil profile, Bunzl [4] reports typical time spans of hundreds to thousands of years for Cs; tens to hundreds of years for Sr and less than one to tens of years for Tc.

Experimental results about the fixation of radionuclides in agricultural soils are described in the earlier part of this chapter related to K_d values. A detailed description of element-specific properties of radionuclides with respect to their mobility in soils can be found in [5,6].

After deposition of Chernobyl-derived contamination it soon became evident that radionuclide behaviour in semi-natural environments differs significantly from that in agricultural areas. Especially, the radiocaesium migration in the soil profile was slower than that expected based on earlier experience with agricultural soils. Organic soil horizons, high microbial/fungal biomass in soils [7,8] and dense and superficial rooting systems all play an important role in Cs retention in semi-natural soils under highland or alpine meadows (and forests).

2. MODELING OF RADIONUCLIDE TRANSPORT IN SOIL

The basic processes controlling mobility of radionuclides (and other trace elements) in soil include convective transport by flowing water, dispersion caused by spatial variations of convection velocities, diffusive movement within the fluid, and physico-chemical interactions with the soil matrix. In addition to abiotic processes, soil fauna may contribute to the transport of radionuclides in soils [9], and their action under general conditions results in a dispersion-like translocation [10].

Since the early 1960s, scientific interest has focused on predicting the environmental fate of ^{90}Sr, ^{137}Cs and $^{238\text{-}240}$Pu, all of which have long or very long radioactive half-lives. These radionuclides strongly sorb to the soil matrix, showing K_d values mostly well above 100 L kg^{-1}. It is easily calculated that transport velocities in the unsaturated zone for these radionuclides will be about 1 cm y^{-1} or lower, indicating that a major fraction of the originally deposited activity may remain within the rooting zone for decades. Because of this slow migration velocity, models that simulate radionuclide movement in soils usually do not take into account soil moisture changes in the unsaturated zone, but (often implicitly) assume a constant mean water content.

As a second simplification, usually one-dimensional models are applied, based on the assumption that deposition rates are spatially uniform, which may be justified for weapons-testing fallout and, at least on a local scale, also for the Chernobyl fallout. With these simplifications, two approaches have become most popular for modelling the migration of radionuclides in soils–a serial compartmental approach and application of the convection-dispersion equation.

2.1. Compartment models

Compartment models are often considered as a black-box approach [11], which can be applied without detailed information about the site-specific hydrological, physico-chemical and biological processes that influence radionuclide mobility. Consequently, such models have often been used for the generic description of radionuclide migration in soils [e.g. 12]. Usually, the soil profile is split into a series of horizontal layers (compartments), which are connected by downward transport rates for the respective radionuclide. It is assumed that:

- the radionuclides are homogeneously mixed in each compartment,
- transfer rates are time-invariant,
- sorption of the radionuclide is instantaneous and characterized by a linear isotherm,
- fluxes between compartments are donor-controlled.

The radionuclide dynamics is described by a system of linear first-order differential equations with constant coefficients. If deposition of the radionuclides from the atmosphere can be approximated by pulse-like inputs into the surface compartment or continuous uniform rates of input, then analytical solutions of this system of differential equations are available [13,14]. However, numerical methods of solution that can be used with arbitrary deposition histories and initial radionuclide concentrations in the compartments have also been developed [15].

Site-specific values of transfer rates, from which migration velocities and residence times can easily be calculated, are usually estimated by tuning the model to measured concentrations in soil profile layers. Consequently, for parameter estimation, the thickness of the compartments into which the soil profile is divided is usually arbitrarily set to the dimensions of the soil layers sampled in the field.

Multi-compartmental models are simple chromatographic models that can be interpreted as discrete analogues of the dispersion-convection equation [16,17]. Hence, the compartmental model chosen should not be considered as a black-box approach, but must reflect the basic transport characteristics of the soil studied. As a consequence, the serial compartmental model is applicable only if the transport of the radionuclides is dominated by convection [18]. The number of compartments is then a discrete analogue of the dispersion characteristics of the soil under study and hence should not be chosen arbitrarily [18]. If the translocation of the radionuclide in soil is dominated by dispersion-like processes–as is indicated by concentration profiles declining exponentially with depth [e.g. 19,20]–a compartmental model with backflow should be chosen; this type of model adequately represents transport driven by vertical concentration differences [18].

2.2. Convection-dispersion models

A number of studies applying the convection-dispersion equation have been reported in the literature [e.g. 14, 21-23]. The deposition history is mostly approximated by pulse-like input functions, and effective values of the dispersion coefficient and convection velocity are obtained by fitting the analytical solution of the model equation to measured depth distributions of the radionuclides.

An extensive study has been performed by Bossew [24], who analyzed more than 1300 depth distributions taken at 477 sites in Austria. Generally, all nuclides analyzed had a limited mobility in soil with radiocaesium having smaller values of velocity and dispersion than ^{106}Ru, ^{125}Sb, ^{110m}Ag and ^{144}Ce. Bossew [24] also showed that the parameter values resulting from the fitting procedure could be grouped with regard to soil types and that parameter variability was considerably reduced within groups.

This indicates that the fitted parameter values are related to the physics of radionuclide transport in the soils studied and hence can be used for predictive modelling. In general, radionuclide mobility increased in the sequence humic alpine soils–podsols–brown earths, chernozems.

In many studies, a small fraction of the deposited radionuclide activity was found at greater depths than predicted by the convection-dispersion model [21, 22]. This effect has also been observed with other solutes [e.g. 25] and is related to stochastic spatial variations of the hydraulic properties [26].

A simple stochastic approach is the transfer function model suggested by Jury [27] and Simmons [28] and discussed by [29]. If transport is stationary and linear, all information on the system dynamics is inherent in its impulse-response function, which then can be taken as the probability density function of travel times to a fixed depth or, alternatively, of travel distances in soil during a fixed time period. Hydraulic conductivities of soils have repeatedly been found to show spatial variations that are distributed approximately log-normally [30]. In soils with such spatial properties, solute transport, which is assumed to occur by convection via individual isolated streamlines, is characterized by lognormal travel time probability density functions. This model, as proposed by Simmons [28], may be characterized as stochastic-convective transport. It shows dispersion properties that are fundamentally different from those of the familiar convective-dispersive model [31]: the variance of travel times (which is a convenient measure of a solute dispersion) grows linearly with transport distance for convective-dispersive transport, but with the square of the distance for the stochastic-convective model.

Kirchner [32] extended the stochastic-convective model to take also into account spatial variability of radionuclide sorption. After fitting both CDE and stochastic-convective models to the depth distribution of weapons fallout ^{90}Sr in a podsol soil to derive model parameter values, the latter model was found to be more adequate.

2.3. Relationships between K_d and migration parameters

If sorption of a radionuclide in soil is instantaneous, reversible and independent of its concentration (i.e. the K_d concept applies), this process is reflected in the convection-dispersion model by the following relations of the model parameters of a sorbing and a non-sorbing trace substance, respectively:

$$D_s = \frac{D}{f_{ret}} \tag{1}$$

$$v_s = \frac{v_w}{f_{ret}} \tag{2}$$

where D_s, v_s are the effective dispersion coefficient and convective velocity of the radionuclide showing sorption, D is the dispersion coefficient of a non-sorbing trace substance, v_w is the mean pore water velocity and f_{ret} is the retardation factor.

Thus, if D and v_w are known, e.g. from observations using tritium, it is possible to estimate an in-situ K_d from the depth distribution of a radionuclide (and *vice versa*). This is most easily achieved if the input of the radionuclide can be approximated by a single pulse-like function. In this case, for large t the first two moments of the depth-distribution function are asymptotically [33]

$$E[z] \cong v_s \cdot t \tag{3}$$

$$\mathrm{var}[z] \cong 2 \cdot D_s \cdot t. \tag{4}$$

Thus, v_s and D_s are estimated from the position of the peak concentration in soil, z_M, and the distance Δz between z_M and the depth, where the concentration reduces to ca. 0.6 $\left(= \frac{1}{\sqrt{e}} \right)$ of its maximum, by

$$v_s = \frac{z_M}{t} \tag{5}$$

$$D_s = \frac{(\Delta z)^2}{2t}. \tag{6}$$

Based on the values of v_s and D_s deduced in Ref. [33] from analysing more than 500 depth distributions, for ^{137}Cs this approximation can be used for t >15 y.

3. PARAMETERS FOR VERTICAL MIGRATION IN UNDISTURBED SOIL PROFILES

The following data compilation takes into account literature data on the vertical migration of radionuclides in undisturbed meadow soils (agricultural and semi-natural); all sources are given in the reference list. Most data refer to ^{137}Cs from Chernobyl fallout (deposition event in May 1986) and ^{137}Cs from weapons fallout (peak of deposition in the early 1960s). Other radionuclides are covered only in a limited number of literature sources. For convection-dispersion models the two key parameters, *v* (convection velocity) and *D* (dispersion coefficient) are reported. These values can be used in the convection-dispersion equation for a chosen time t to produce a vertical profile of the radionuclide. In some cases, authors not only reported *v* and *D* but also the migration rate, derived from the peak of the vertical distribution (or half-depth, i.e. the soil depth, above which 50% of the total activity is present) at a given time *t*.

Results from compartment model approaches are generally expressed as residence half-times or migration rates. These two figures are related by the formula [34]:

$$v = \frac{L}{\tau} \tag{7}$$

Where v represents the migration rate (cm a^{-1}); L represents the thickness of the evaluated soil layer (cm) and τ the ecological residence half-time (i.e. the time, over which the radionuclide activity decreases to 50% of the initial value, excluding physical decay).

Bonne et al. [13] have stressed the importance of separating topsoil profiles into small depth intervals to yield a good approximation of measured depth distributions. On the other hand, since multi-layer-compartment models can be interpreted as analogues of the dispersion-convection-model, layers should not be chosen arbitrarily (see above). There seems to be a depth dependence of migration rates derived from compartment models with higher values for deeper layers (Table 1).

Even if extraordinary high values, typically reported for deeper compartments of multi-layered profiles (15-20 cm depth and deeper) are excluded from the statistics, migration rates derived from compartment models (geometric mean: $0.51 \times 2.8^{\pm 1}$; n=45) are significantly higher than corresponding values derived from CDE-based models or repeated measurements (geometric mean: $0.24 \times 2.2^{\pm 1}$; n=58, Table 2).

TABLE 1. MIGRATION RATES (cm a^{-1}) FOR ^{137}Cs AND ^{134}Cs IN UNDISTURBED MEADOW SOIL PROFILES DERIVED BY USE OF COMPARTMENT MODELS[1]

Soil groups	N	GM	GSD	AM	SD	Min	Max	#ref.
Chernobyl fallout layers $\leq$ 10 cm								
All soils	45	5.1×10^{-1}	2.8	1.1	2.0	8×10^{-2}	1.0×10^{1}	12
Clay	5	1.3×10^{-1}	1.4	1.3×10^{-1}	1×10^{-2}	8×10^{-2}	2×10^{-1}	1
Loam	19	5.0×10^{-1}	2.1	6.3×10^{-1}	5×10^{-1}	1.0×10^{-1}	2.0	7
Sand	10	3.5×10^{-1}	1.3	3.5×10^{-1}	1×10^{-1}	2.0×10^{-1}	4×10^{-1}	4
Peat	3	3.0	4.2	4.9	4.1	6.0×10^{-1}	8.7	2
Unspecified	8	1.1	3.0	2.1	3.3	3.8×10^{-1}	1.0×10^{1}	3
Chernobyl fallout deep layers (>10 cm)								
All soils	15	3.2	3.1	4.9	3.9	3.2×10^{-1}	1.2×10^{1}	4
Loam	4	2.0	2.9	2.9	2.4	7.5×10^{-1}	5.0	2
Sand	1	-	-	3.2×10^{-1}	-	-	-	1
Peat	6	4.8	3.2	7.0	4.9	6.0×10^{-1}	1.2×10^{1}	2
Unspecified	4	4.7	1.3	4.8	1.6	3.90	7.2	1
Weapons fallout; layers $\leq$ 10 cm								
All soils	19	2.8×10^{-1}	2.0	3.5×10^{-1}	2×10^{-1}	9×10^{-2}	8.5×10^{-1}	6
Clay	1	-	-	2.0×10^{-1}	-	-	-	1
Loam	9	3.0×10^{-1}	1.8	3.5×10^{-1}	1×10^{-1}	9×10^{-2}	5.0×10^{-1}	5
Sand	6	3.0×10^{-1}	2.6	4.2×10^{-1}	3×10^{-1}	9×10^{-2}	8.5×10^{-1}	3
Peat	1	-	-	3.0×10^{-1}	-	-	-	1
Unspecified	2	-	-	2.5×10^{-1}	-	9×10^{-2}	4.0×10^{-1}	1
Weapons fallout; deep layers >10 cm								
All soils	8	7.1×10^{-1}	1.5	7.5×10^{-1}	2×10^{-1}	3.0×10^{-1}	1.0	2
Loam	4	8.9×10^{-1}	1.2	9.0×10^{-1}	1×10^{-1}	7.0×10^{-1}	1.0	2
Sand	3	6.9×10^{-1}	1.0	6.9×10^{-1}	-	6.9×10^{-1}	6.9×10^{-1}	1
Peat	1	-	-	3.0×10^{-1}	-	-	-	1
Artificial contamination								
All soils	23	4.1×10^{-1}	2.6	5.5×10^{-1}	4×10^{-1}	4×10^{-2}	1.80	4

[1] Values from different contamination sources and from layers deeper than 10 cm (10-15, 20-30 cm) were treated separately.

TABLE 2. MIGRATION RATES (MR, cm a^{-1}) OF ^{137}Cs IN UNDISTURBED GRASSLAND SOILS DERIVED BY CDE-APPROACHES OR REPEATED MEASUREMENT (NOT COMPARTMENT MODELS)

Soil groups	N	GM	GSD	AM	SD	Min	Max	ref.
All soils	58	2.4×10^{-1}	2.2	3.4×10^{-1}	2.9×10^{-1}	7×10^{-2}	1.16	7
Clay	2	-	-	4.2×10^{-1}	-	2.4×10^{-1}	6.0×10^{-1}	1
Loam	15	2.3×10^{-1}	2.5	3.4×10^{-1}	3.2×10^{-1}	7×10^{-2}	1.00	5
Sand	33	2.1×10^{-1}	2.0	2.7×10^{-1}	2.6×10^{-1}	8×10^{-2}	1.16	4
Peat	8	5.0×10^{-1}	1.9	5.8×10^{-1}	2.8×10^{-1}	1.4×10^{-1}	1.00	3
MR[a]	58	3.4×10^{-1}	2.6	6.0×10^{-1}	1.2	4×10^{-2}	1.0×10^{1}	29

[a] Calculated based on all available values for the layer from the soil surface down to 10 cm in depth given in Tables 1-2.

This difference may reflect the problems arising from the use of compartmental models. Moreover, this and all following comparisons are biased to some extent, mainly because the number of observations from different soil types within compared groups is never the same, and the times of evaluation (i.e. times after deposition, at which the depth profiles were studied) do not correspond.

In order to give a robust estimate, it seems justified to combine results from different modelling approaches: the selected literature represents state of the art (peer reviewed) estimations, and only if combined covers the full range of possible results for vertical migration of radionuclides in soils. The overall geometric mean for ^{137}Cs migration rates yields $0.34 * 2.6^{\pm 1}$ cm a^{-1} (see Table 2). Parameters D and v derived from undisturbed grassland soil profiles using CDE approaches are summarized in Table 3 for different sources of contamination.

4. TIME DEPENDENCE OF ^{137}Cs VERTICAL MIGRATION

Some studies have pointed to a time-dependence of migration rates. One month after the Chernobyl fallout event, Schimmack et al. [35] observed migration rates for ^{137}Cs of 0.2 cm per hour (equivalent to 1752 cm a^{-1}) in a silty loam meadow soil. In Bavaria, the Chernobyl fallout was deposited by heavy rainfall, and radioactivity was washed into the soil profile quite deeply. The fixation of radionuclides to soil components like clay minerals or humus takes some time, and as it progresses, migration rates tend to decrease with increasing time, due to irreversible fixation of part of the radionuclide content. Between 1987 and 1994 migration rates in soddy-podsolic soils of Bryansk decreased from 0.27 to 0.08 cm a^{-1} and similar decreases were observed for other soils (peaty podsol 0.24 – 0.09 /1987 – 1994). In Sweden such a decrease in migration rates was also observed [36]: one year after Chernobyl fallout migration rates in peat and gley soil profiles were determined to be 0.6 and 1.0 cm a^{-1}. In 1994 (eight years after fallout, see Fig. 1), the migration rates had decreased to 0.18 and 0.41 cm a^{-1}, respectively.

5. EFFECTS OF SOIL TEXTURE ON ^{137}Cs MIGRATION

As described above, soil characteristics, including amongst others texture and composition, have a distinct influence on the migration behaviour of radionuclides. Values for each texture group (clay, loam, sand, peat) can be found in Table 1 (migration rates) and Table 19 (CDE-parameters). For migration rates (Table 1) of Chernobyl fallout ^{137}Cs derived using compartment model approaches, a significant difference (t-test, $p < 0.05$) was found between clay and the other textures (loam, sand and peat), with lower migration rates in clay soils. For ^{137}Cs migration rates derived by other approaches, the migration rates in sand and peat soils were significantly different at the 95% confidence level (Table 2). For values of migration rates from weapons fallout (Table 1) and for parameters of the CDE-function (Table 3) only non-significant differences were found dependent on soil type.

TABLE 3. PARAMETERS OF THE CONVECTION DISPERSION MODEL (*D*, cm^{2} a^{-1} AND *v*, cm a^{-1}) FOR ^{137}Cs (AND ^{134}Cs) FROM DIFFERENT SOURCES OF CONTAMINATION (CHERNOBYL AND GLOBAL FALLOUT, ARTIFICIAL CONTAMINATION)

Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
***D*, cm^{2} a^{-1}**								
Chernobyl fallout								
All soils	31	2.2×10^{-1}	3.1	3.7×10^{-1}	4×10^{-1}	2×10^{-2}	1.9	13
Loam	4	2.0×10^{-1}	4.6	3.6×10^{-1}	3×10^{-1}	2×10^{-2}	8×10^{-1}	2
Sand	11	1.1×10^{-1}	2.3	1.6×10^{-1}	2×10^{-1}	3×10^{-2}	6×10^{-1}	5
Peat	3	9.4×10^{-1}	1.8	1.1	7×10^{-1}	6.3×10^{-1}	1.9	3
Unspecified	12	2.7×10^{-1}	2.6	3.7×10^{-1}	3×10^{-1}	4×10^{-2}	8×10^{-1}	6
Global Fallout								
All soils	12	2.2×10^{-1}	4.3	5.7×10^{-1}	9×10^{-1}	4×10^{-2}	2.9	6
Loam	2	-	-	1.6	-	3.9×10^{-1}	2.9	1
Sand	3	1.3×10^{-1}	5.9	3.6×10^{-1}	5×10^{-1}	4×10^{-2}	1.0	3
Peat	1	-	-	1.6	-	-	-	1
Unspecified	6	1.2×10^{-1}	2.2	1.6×10^{-1}	1×10^{-1}	5×10^{-2}	4×10^{-1}	1
Artificial contamination								
All soils	4	1.2	1.5	1.3	5×10^{-1}	8.0×10^{-1}	2.0	1
***v*, cm a^{-1}**								
Chernobyl fallout								
All soils	31	1.8×10^{-1}	3.3	2.7×10^{-1}	2×10^{-1}	0.00	9×10^{-1}	13
Loam	4	6×10^{-2}	1.8×10^{1}	2.4×10^{-1}	3×10^{-1}	0.00	6×10^{-1}	3
Sand	11	1.5×10^{-1}	1.7	1.7×10^{-1}	1×10^{-1}	7×10^{-2}	6×10^{-1}	5
Peat	3	6.9×10^{-1}	1.6	7.3×10^{-1}	3×10^{-1}	4.0×10^{-1}	9×10^{-1}	3
Unspecified	12	2.2×10^{-1}	1.6	2.4×10^{-1}	1×10^{-1}	9×10^{-2}	5×10^{-1}	6
Global fallout								
All soils	11	9×10^{-2}	3.3	1.6×10^{-1}	2×10^{-1}	1×10^{-2}	7×10^{-1}	6
Loam	2	-	-	2×10^{-2}	-	1×10^{-2}	1×10^{-1}	1
Sand	2	-	-	3.9×10^{-1}	-	6×10^{-2}	7×10^{-1}	3
Peat	1	-	-	1.0×10^{-1}	-	-	-	1
Unspecified	6	1.3×10^{-1}	1.4	1.3×10^{-1}	1×10^{-2}	9×10^{-2}	2×10^{-1}	1
Artificial contamination								
All soils	3	1.9×10^{-1}	1.5	1.5×10^{-1}	1×10^{-1}	1.5×10^{-1}	3×10^{-1}	1

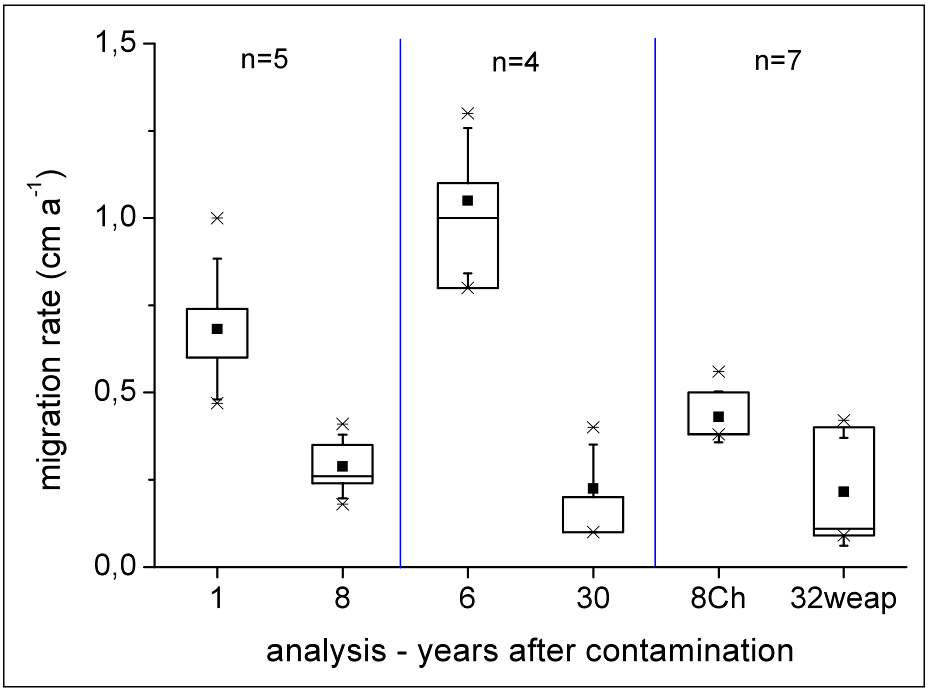

FIG. 1. Time dependence of ^{137}Cs vertical migration rates in soils. Data source: [37] (1 and 8 years after Chernobyl fallout); [38] (6 and 30 years after artificial contamination in lysimeters); [39] (comparison of ^{137}Cs from Chernobyl (Ch, 8 years) and weapons fallout (weap, 32 years).

6. ^{137}Cs MIGRATION RATES IN DIFFERENT CONTAMINATION SCENARIOS

6.1. Aerosol fallout

Geometric means of ^{137}Cs migration rates from Chernobyl fallout ($0.51 * 2.8^{\pm1}$ cm a^{-1}) are significantly higher (t-test, $p < 0.05$) in comparison with values from weapons fallout ($0.28 * 2.0^{\pm1}$ cm a^{-1}, see Table 1). Also for deeper layers this difference is significant ($3.1 * 3.1^{\pm1}$ cm a^{-1} for Chernobyl fallout compared to $0.71 * 0.5^{\pm1}$ cm a^{-1} for global weapons fallout; see Table 1). Due to the fact that weapons fallout has been present in soils for about 30 years longer than Chernobyl fallout, this finding is in agreement with the above-mentioned time dependence of migration rates.

CDE-derived migration rates from contamination experiments are intermediate between Chernobyl and global fallout data. Values derived by analysis of undisturbed grassland soil profiles (all profile depths) are given in Table 2. It seems justified to combine values from all different contamination sources to an overall estimate for ^{137}Cs migration rate of $0.34 * 2.6^{\pm1}$ cm a^{-1}.

6.2 Hot particles

During the Chernobyl NPP accident part of the reactor inventory was incinerated to gaseous aerosols and transported to the atmosphere. Part of the inventory remained in solid form and was dispersed as so-called ‘hot particles’. These are small particles of the reactor fuel consisting of original material and fission products. Radionuclides contained in hot particles are protected against downward migration with the water phase because the particles are water-insoluble. Only weathering of the particles leads to disintegration and converts the radionuclides into a more soluble form. Gastberger et al. [40] has shown that the soluble fraction of ^{90}Sr increases with decreasing presence of hot particles at the Semipalatinsk nuclear test site. Therefore, activity caused by hot particle fallout will remain at the soil surface. Only physical mixing of surface soil layers (by redistribution or bioturbation, particle transport in soil cracks during heavy rainfall events etc.) leads to vertical transport of radionuclides in the initial phase after fallout. In contrast, condensed aerosol fallout consists of radionuclides in a more mobile (cationic) form and will show a faster vertical migration into the soil profile. An illustration of this principle is given in Fig. 2, comparing soil profiles of ^{137}Cs from hot particle influenced locations and nearby sampling spots with aerosol fallout. Although differences are only significant ($p < 0.05$); for the parameter v (convection velocity) at the investigated sampling locations (median: 0.141 vs 0.215), sites with hot particles present also tend to have lower dispersion constants (median: 0.199 vs. 0.403).

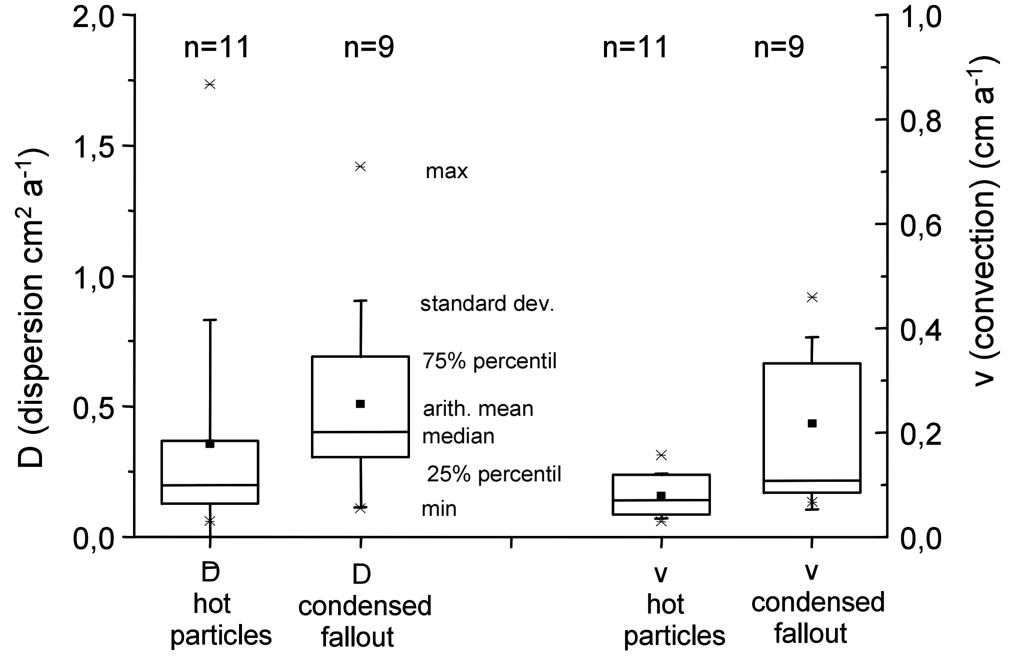

FIG. 2. Influence of hot particles on ^{137}Cs depth distribution (data source: [41]).

7. VERTICAL MIGRATION OF RADIONUCLIDES OTHERS THAN ^{137}Cs

Only a very limited number of data have been reported for the vertical migration of other radionuclides in soil. The results are listed in Table 4. Although the number of observations is far too small to yield representative values, the figures given in Table 4 agree with general descriptions of radionuclide mobility in soils (see above): migration rates decrease in the following order.: I > Sr > Ru, Ce, Co > Cs, > Pu, Am.

TABLE. 4. CDE-PARAMETERS *D* AND *v* AND MIGRATION RATES (*MR*) FOR MISCELLANEOUS RADIONUCLIDES

	Parameter	N	GM	GSD	AM	SD	Min	Max	Soil types	# ref	Notes
^{110}Ag	*D* (cm² a^{-1})	3	3.4×10^{-1}	2.0	4.0×10^{-1}	3×10^{-1}	1.8×10^{-1}	7.2×10^{-1}	Unspecified	1	
^{110}Ag	ν (cm a^{-1})	3	4.7×10^{-1}	1.8	5.3×10^{-1}	4×10^{-1}	3.2×10^{-1}	9.4×10^{-1}	Unspecified	1	
^{241}Am	*D* (cm² a^{-1})	2	-	-	6×10^{-2}	-	3×10^{-2}	8×10^{-2}	sand : 1; unspecified: 1	2	
^{241}Am	ν (cm a^{-1})	2	-	-	1.5×10^{-1}	-	1.0×10^{-1}	2.1×10^{-1}	sand : 1; unspecified: 1	2	
^{241}Am	*MR* (cm a^{-1})	5	5.0×10^{-1}	1.6	5.4×10^{-1}	3×10^{-1}	3.0×10^{-1}	1.0	Silt	1	several depth layers
^{144}Ce	*D* (cm² a^{-1})	2	-	-	1.2	-	7.6×10^{-1}	1.5	Unspecified	1	
^{144}Ce	ν (cm a^{-1})	2	-	-	5.4×10^{-1}	-	2.6×10^{-1}	8.1×10^{-1}	Unspecified	1	
^{60}Co	*D* (cm² a^{-1})	5	6.9×10^{-1}	4.8	1.2	9×10^{-1}	5×10^{-2}	2.4	loam: 4; sand: 1	2	Artificial contamination (e.g. lysimeter)
^{60}Co	ν (cm a^{-1})	4	1.9×10^{-1}	1.4	1.6×10^{-1}	1×10^{-1}	1.5×10^{-1}	3.0×10^{-1}	loam: 3; sand: 1	2	Artificial contamination (e.g. lysimeter)
$^{154/155}$Eu	*D* (cm² a^{-1})	2	-	-	9×10^{-2}	-	9×10^{-2}	$9 \text{ x} 10^{-2}$	Sand	1	
$^{154/155}$Eu	ν (cm a^{-1})	2	-	-	2.2×10^{-1}	-	2.0×10^{-1}	2.3×10^{-1}	Sand	1	
$^{129/131}$I	*MR* (cm a^{-1})	6	1.6	2.7	2.2	1.7	2.9×10^{-1}	5.0	Unspecified	1	Artificial contamination (e.g. lysimeter)
^{131}I	*MR* (cm a^{-1})	1	-	-	$2.6 \text{ x} 10^{2}$	-	-	-	Loam	1	Chernobyl fallout, 1 month after the accident
^{94g}Nb	ν (cm a^{-1})	1	-	-	1.9×10^{-1}	-	-	-	Sand	1	
^{94g}Nb	*D* (cm² a^{-1})	1	-	-	3×10^{-2}	-	-	-	Sand	1	
^{237}Np	*MR* (cm a^{-1})	3	4.3×10^{-1}	1.5	4.5×10^{-1}	1×10^{-1}	2.9×10^{-1}	5.8×10^{-1}	Silt	1	0-2, 2-5, 5-10 cm layers
^{237}Np	*MR* (cm a^{-1})	3	9.3×10^{-1}	1.4	9.7×10^{-1}	3×10^{-1}	6.3×10^{-1}	1.2	Silt	1	10-15, 15-20. 20-30 cm layer
^{238}Pu	*D* (cm² a^{-1})	2	-	-	8×10^{-2}	-	7×10^{-2}	8×10^{-2}	sand : 1; unspecified: 1	2	
^{238}Pu	ν (cm a^{-1})	2	-	-	1.5×10^{-1}	-	1.1×10^{-1}	1.9×10^{-1}	sand : 1; unspecified: 1	2	
$^{239/40}$Pu	*D* (cm² a^{-1})	2	-	-	7×10^{-2}	-	7×10^{-2}	7×10^{-2}	sand : 1; unspecified: 1	2	
$^{239/40}$Pu	ν (cm a^{-1})	2	-	-	1.4×10^{-1}	-	1.1×10^{-1}	1.7×10^{-1}	sand : 1; unspecified: 1	2	
$^{239/40}$Pu	*MR* (cm a^{-1})	18	3.5×10^{-1}	1.6	3.9×10^{-1}	2×10^{-1}	2.1×10^{-1}	9.4×10^{-1}	sand: 7; loam: 10; peat: 5	4	Global fallout, 0-2, 0-5, 5-10 cm layers
$^{239/40}$Pu	*MR* (cm a^{-1})	24	8.8×10^{-1}	2.2	1.2	9×10^{-1}	1.8×10^{-1}	3.0	loam: 14; sand: 9; peat: 1	3	Global fallout, Layers: from 10-15 to 87 cm

TABLE. 4. CDE-PARAMETERS *D* AND *v* AND MIGRATION RATES (*MR*) FOR MISCELLANEOUS RADIONUCLIDES (Cont.)

	Parameter	N	GM	GSD	AM	SD	Min	Max	Soil types	# ref	Notes
^{226}Ra	*D* (cm² a^{-1})	4	1.8	1.9	2.2	1.7	1.0	4.6	Loam	1	
^{226}Ra	v (cm a^{-1})	3	1.4×10^{-1}	1.4	1.1×10^{-1}	1×10^{-1}	1.0×10^{-1}	2.0×10^{-1}	Loam	1	
^{106}Ru	*MR* (cm a^{-1})	1	-	-	2.6×10^{3}	-	-	-	Loam	1	Chernobyl fallout, 1 month after the accident
^{106}Ru	*MR* (cm a^{-1})	2	-	-	4.4×10^{-1}	-	4.3×10^{-1}	4.5×10^{-1}	Loam	1	
^{106}Ru	*D* (cm² a^{-1})	6	5.0×10^{-1}	1.4	5.2×10^{-1}	2×10^{-1}	2.6×10^{-1}	8.2×10^{-1}	Unspecified	1	Different geological regions in Austria
^{106}Ru	v (cm a^{-1})	5	4.5×10^{-1}	1.2	4.6×10^{-1}	1×10^{-1}	3.5×10^{-1}	5.8×10^{-1}	Unspecified	1	
^{125}Sb	*D* (cm² a^{-1})	6	3.5×10^{-1}	2.3	4.4×10^{-1}	3×10^{-1}	8×10^{-2}	8.0×10^{-1}	unspecified: 5; sand: 1	1	
^{125}Sb	v (cm a^{-1})	6	3.8×10^{-1}	1.5	4.0×10^{-1}	1×10^{-1}	2.0×10^{-1}	6.3×10^{-1}	unspecified: 5; sand: 1	1	
^{90}Sr	*D* (cm² a^{-1})	10	3.8×10^{-1}	2.9	5.9×10^{-1}	6×10^{-1}	5×10^{-2}	1.7	sand: 7; loam: 2; peat: 1	2	Chernobyl fallout
^{90}Sr	v (cm a^{-1})	10	2.2×10^{-1}	2.2	3.0×10^{-1}	3×10^{-1}	6×10^{-2}	9.2×10^{-1}	sand: 7; loam: 2; peat: 1	2	Chernobyl fallout
^{90}Sr	*MR* (cm a^{-1})	16	4.8×10^{-1}	2.0	5.9×10^{-1}	4×10^{-1}	1.2×10^{-1}	1.5	sand: 11; peat: 4; loam: 1	3	Chernobyl fallout
^{90}Sr	*D* (cm² a^{-1})	1	-	-	2.4	-	-	-	Sand	1	Global fallout
^{90}Sr	v (cm a^{-1})	1	-	-	8×10^{-2}	-	-	-	Sand	1	Global fallout
^{90}Sr	*MR* (cm a^{-1})	12	8.0×10^{-1}	1.6	8.8×10^{-1}	4×10^{-1}	4.6×10^{-1}	1.4	sand: 7; loam: 3; peat: 2	2	Global fallout, 0-10 cm layer
^{90}Sr	*MR* (cm a^{-1})	24	8.9×10^{-1}	2.5	1.4	2.0	2.0×10^{-1}	9.5	unspecified: 6; loam: 6; sand: 4; clay: 2	4	Artificial contamination (e.g. lysimeter)

8. CONCLUSIONS

Results from recent studies show that mathematical constraints exist that can lead to artefacts when applying compartmental models for the description of vertical distributions of radionuclides in soil profiles. Especially, the arbitrarily chosen thickness of layers can yield unrealistic results. Therefore, literature values of migration rates derived from compartmental models should be seen as order of magnitude estimates valuable for comparison of radionuclides in different soil types, but not as the first choice for predictive purposes. This is underlined by the large variability of migration rates in the data compilation, which allows not more than a rough estimate for each radionuclide in a particular soil context.

For the modelling of vertical migration in undisturbed soils, it is highly recommended to rely on CDE-approaches or other innovative calculation methods, because they offer a more realistic representation of the observed processes. This shift of recommendations is also supported by publication history, where there is a strong preference for CDE-approaches in more recent papers.

In comparison with soil-plant transfer factors, the data base for vertical migration parameters is very limited. At the moment, no data for D and v in tropical meadow soils are available. Only a few values have been derived from arctic soils and for subtropical climate conditions. Therefore, at the present time, it is not possible to give separate values for different ecosystems.

Future studies should focus on the development of improved modelling techniques, based on stochastic or CDE approaches, to develop robust models for the description of radionuclide vertical migration in soils and improve the database of existing values for different environments and soil characteristics.

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APPENDIX

List of publication used for parameters of radionuclide vertical migration in soil evaluation

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AGRICULTURAL ECOSYSTEMS: ROOT UPTAKE

PROCESSES GOVERNING RADIONUCLIDE TRANSFER TO PLANTS

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Abstract

Processes governing radionuclide transfers to plants from soil are described. Six basic factors, including: the chemical form of depositions; the properties of radionuclide; time after depositions; properties of soil; type of crop; crop cultivation practice (application of fertilizers, irrigation, ploughing, liming, etc.); climate conditions; and the experimental conditions under which the transfer factors were obtained were identified as basic factors influencing variability of transfer factors from soil to plants.

1. INTRODUCTION

Radionuclides can enter the soil environment via several routes. Generally, most large-scale contamination is through air deposition or advection from neighbouring contaminated areas. Contamination may also occur via contaminated surface water (inundation, irrigation) or contaminated ground water (upwelling, irrigation).

Naturally occurring radionuclides (NORs) are present in many natural resources. High concentrations of these radionuclides are often found in certain geological materials, namely igneous rocks and ores. Human activities that exploit these resources may lead to enhanced concentrations of naturally occurring radionuclide materials and/or enhanced potential for exposure to naturally occurring radioactive materials (NORMs) in products, by-products, and wastes. Soil contamination routes are comparable to those for the artificial radionuclides except for large-scale air deposition. The transfer of radionuclides along food chains and through food webs has been studied extensively over the last 50 years. At the beginning of this period, the objective of the research was to evaluate the biological impact of nuclear weapons testing and releases from military sites. Studies on the impact of global fallout on man showed that some radionuclides such as ^{90}Sr, ^{131}I, and ^{137}Cs are very mobile in the environment, and that accumulation in food stuffs can be responsible for much of the effective dose to the population [1-2].

In the 1960s the development of civilian uses for nuclear energy stimulated some expansion in the list of radionuclides of interest. New experiments on the transfer of some activation products (^{54}Mn, ^{60}Co, ^{65}Zn), natural (^{238}U, ^{226}Ra, $^{228+232}$Th, ^{210}Pb and ^{210}Po), transuranic (^{237}Np, $^{239+240}$Pu, ^{241}Pu, ^{241}Am), and some other radionuclides (^{3}H and ^{14}C) were started to extend the scientific basis for predicting the contamination of agricultural products in various contamination scenarios [2].

After the Chernobyl accident, extensive, further studies of radionuclide behaviour in agricultural ecosystems were carried out, and many data on ^{137}Cs bioavailability in soil-plant systems have been obtained.

In addition, since the 1980s, many parameter values for tropical and subtropical crops have been determined. Although these data are reasonably extensive, the number of radionuclides considered is limited, geographical coverage is very variable, and a limited number of soil and crop type combinations have been studied.

The continuing growth of a body of relevant data over recent years means that there is a general need for a systematic review of radionuclide soil-to-plant transfer factors for use in dose assessments. Assessment models require well-founded default parameter values and ranges on which to base predictions and estimate uncertainties in those predictions.

Observed transfer factor values vary enormously. Six main factors (Fig. 1) determine this variability: the form in which the activity enters or is present the soil (e.g. as particles, as aerosol or in solution); the physicochemical properties of the radionuclide; time after entry into the soil; type of soil and the physicochemical characteristics of the soil environment; type of crop; crop management practices (application of fertilizers, irrigation, ploughing, liming, etc.); climate conditions; and the experimental conditions under which the transfer factors were obtained [1, 3-8, 9-11].

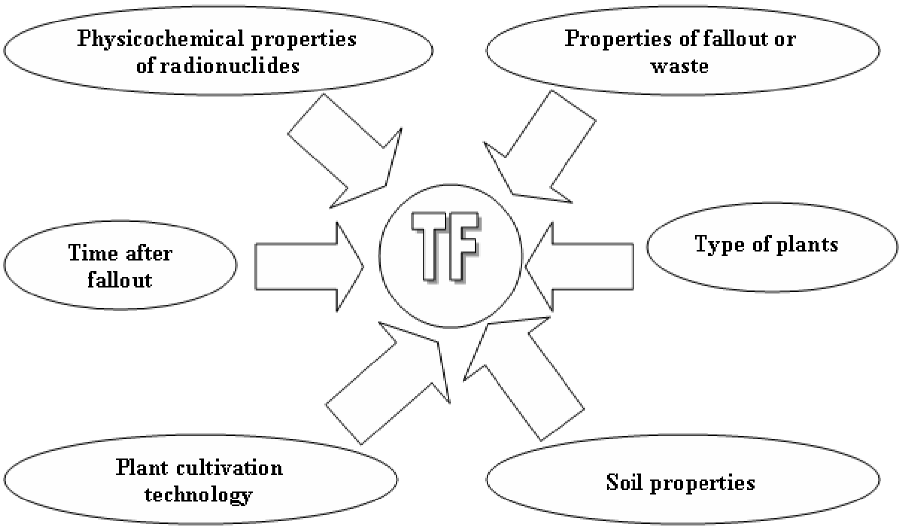

FIG. 1. Factors influencing radionuclide root uptake

2. DEPOSITION PROPERTIES

Different types of radioactive materials in routine or accidental releases from the nuclear fuel cycle can be identified according to their mobility in soil-plant systems [9, 10].

The first group exists primarily in the particulate state and has extremely low solubility (radioisotopes of the actinides and some other elements such as zirconium). For these radionuclides, the transported forms are typically either oxides (particles) or carbonates. However, in the case of process wastes, the actinides can often be in a somewhat soluble form (nitrates, organic complexes) [1, 8, 10]. Among the actinides, plutonium, thorium, and americium would be expected to have the most stable oxide/hydroxide form and very low solubility rates. Neptunium and curium have exhibited slightly higher biotic transfer rates. Uranium has the widest range in bioavailability, because it forms both cationic and anionic species [11].

The second group includes radionuclides that can exist in the cationic and/or anionic forms and are characterised by higher bioavailability to plants [1, 10]. The activation and fission products are in this group. The radionuclides of this group may be contained within refractory waste forms, e.g. fuel particles, and may or may not be present in stable chemical forms within the refractory material. However, once released to the soil environment, stable speciation of these elements will occur. For example, elements (technetium, selenium, iodine) that can readily form anions in environmental pH/Eh regimes will be more mobile, and, in many cases, more plant-available. Elements such as Ni, Cs, Sr, Ag, Sn, and Nb will likely be present in cationic forms and form stable compounds that will limit both mobility and bioavailability. The naturally occurring radionuclides $^{226/228}$Ra, ^{210}Pb and ^{210}Po may also be included in this group [10].

The physicochemical composition of deposition influences radionuclide behaviour in the soil-plant system. For example, after the Chernobyl accident, the major part of ^{137}Cs was deposited in the form of easily soluble, finely dispersed aerosols, but some of it was included in coarser material and in fuel particles. Leaching of ^{137}Cs from fuel matrices was found to be a factor that significantly affected the bioavailability of the radionuclide, which changed with time after the accident in consequence of the leaching process [9, 12].

It has been found that in the early period after contamination, ^{137}Cs F_v values for plants in the zone with the "fuel" type of deposition were significantly lower than values in the zone in which gas-aerosol deposition dominated. However, in areas with such contamination, ^{137}Cs activity concentrations in plants have increased considerably with time and, in some cases, have exceeded ^{137}Cs concentrations in plants of the ‘remote’ zone of the Chernobyl NPP [13-15]. This is due to the fixation of exchangeable and mobile ^{137}Cs in both zones, compensated by on-going leaching of ^{137}Cs from the fuel matrix in the ‘fuel’ type deposition zone.

3. TIME AFTER CONTAMINATION

A decrease in the radionuclide content in farm crops as time elapses is a typical phenomenon observed in agricultural ecosystems [13, 16]. A variety of processes are involved, including fixation to soil minerals, incorporation by microorganisms, and migration within the rooting zone. As a result, the biological availability of radionuclides for incorporation into food chains is reduced.

Fixation is partly irreversible [17]. In particular, slow adsorption of Cs occurs, especially on clay minerals [18, 19]. Usually, fixation is most important in the first years after contamination of the soil, but the degree of fixation differs among soil types [14-16, 20]. The availability of ^{137}Cs in soil, in the early period before fixation has occurred to its full extent, may be up to a factor of 10 higher than after fixation has occurred and equilibrium conditions have been achieved. The availability of ^{90}Sr also decreases with time, but the effect is much less pronounced than with ^{137}Cs [21].

At the present time, ecological half-lives have been mostly determined for ^{137}Cs. For other radionuclides, this information is almost unavailable and long-term behaviour studies are required.

4. SOIL PROPERTIES

The accumulation of radionuclides in farm crops varies considerably for different soils. The difference in transfer factors to farm crops for different soils can be up to three orders of magnitude [1, 3, 7, 8, 11, 22, 39]. Soil properties that are likely to affect bioavailability include mineralogical and granulometric composition, organic matter content, pH, and fertility. The cation exchange capacity and the nature of exchangeable bases are important. The highest availability of many radionuclides is reported for sandy soils. Soils with higher fertility generally retain radionuclides more strongly. Thus, for ^{137}Cs, the transfer factors to farm crops for high fertility soils are typically two order of magnitude lower than for low-fertility soils [1, 22-24].

5. CROP PROPERTIES

Among the key factors influencing F_v values, the biological variability inherent in plants and distinctions between different varieties and species is a likely source of the variability in transfer factors. The sources of this variability include differences in metabolic and biochemical mechanisms of radionuclide uptake by plants; crop requirements for certain nutrients (for example, analogues for radionuclides: K for Cs, Ca and Mg for Sr or Ra), detoxification and exclusion mechanisms, distribution of roots in the soil, and rhizosphere properties. Duration of the vegetative period and yield may also influence radionuclide transfer factors. Radionuclides often transfer in greater concentrations to leaves and stems and in much lower concentrations to generative parts [4, 7, 23-25]. Sr-90 is more effectively accumulated by legumes than by cereals.

6. CROP CULTIVATION PRACTICES

Cultivation of agricultural crops implies various technologies of soil management, different rates and combinations of fertilizers, irrigation in dry areas, drainage in boggy environments, etc. Crop cultivation practices may affect soil properties or lead to redistribution of radionuclides in the root zone, and, as a consequence, change radionuclide accumulation in crops. For example, under irrigation, radionuclide accumulation in plants is increased by a factor of 1.2 to 2.0 [26].

Ploughing has been found to decrease F_v values by as much as a factor of two [1]. Liming reduces plant uptake for some of radionuclides as ^{90}Sr and ^{137}Cs by a factor of 2-3 and application of fertilizers by a factor of 2-5 [1]. Thus, crop cultivation practice should be considered in the estimation of parameters for plant uptake in soil-plant systems [1, 7, 26-33].

7. RADIONUCLIDE PROPERTIES

Five groups of radionuclides can be identified according their physicochemical characteristics [1]. The first of these groups comprises light natural radionuclides (^{3}H, ^{14}C, ^{40}K). The primary uptake of ^{3}H by plants is by absorption of water and, to some degree, uptake of vapour from the atmosphere. The role of active absorbtion by roots from soil solution is considered to be insignificant. The migration of ^{14}C is determined by the characteristics of the biogeochemical carbon cycle, indicating that ^{14}C accumulates in plants mainly as a consequence of uptake of carbon dioxide through the leaves and its subsequent use in photosynthesis. The role of uptake from soil in comparison with uptake from air is insignificant. The degree of ^{40}K accumulation depends on the physiological requirements of plants [1, 22].

The second group, heavy natural radionuclides (^{232}Th, ^{238}U, ^{226}Ra, ^{210}Pb, ^{210}Po), exhibit relatively low F_v values. The F_v values are generally about 10 fold lower for Th and Po than for U, Ra and Pb. Bioavailability of ^{238}U and ^{232}Th is highly influenced by soil pH, more than by soil texture. Uptake of the bivalent ^{226}Ra is mostly ruled by factors affecting the soil exchange capacity. For ^{210}Pb and ^{210}Po F_v data are too limited to point to major soil properties ruling transfer. F_v values of heavy natural radionuclides for different species of plants differed by 10-100 times, and, for different soil types, by up to 10 times [10, 11].

Transuranic elements (Am, Cm, Pu, Np), comprising the third group, exhibit very complex soil chemistry, because of various degrees of oxidation, absence of stable carriers, and high propensities to complexation and hydrolysis. F_v values for transuranic elements varied from about 10^0 to about 10^{-6}. Concentrations of these radionuclides in fruits and grains are 10-1000 times lower than in the vegetative parts of plants. Accumulation of elements decreases in the order Np> Am> Cm> Pu. Among the higher actinides, Np appears to be the most environmentally mobile and plant available. Hydrolysis is a major factor influencing the behaviour of Am and Cm in soils. The mobility of Pu depends on its form and decreases according to Pu (V)> Pu (VI)> Pu (III)> Pu (IV) [8].

The fission products (^{89,90}Sr, ^{134,137}Cs, ^{129,131}I, ^{91}Y, ^{95}Zr, ^{95}Nb, ^{103,106}Ru, ^{141,144}Ce etc.–the fourth group) generate a rather diverse class in terms of the environmenral mobility of radionuclides. Y-91, ^{95}Zr, ^{95}Nb, ^{103,106}Ru, ^{141,144}Ce are poorly accumulated by agricultural plants because of their strong sorption in soil, a factor that is believed to dominate F_v behavior. Soil acidity and organic matter content most significantly influence the behaviour of these radionuclides. Up to 99% of the plant uptake of radionuclides is retained in the roots, and very little transfer to above-ground parts occurs. Zr-95 and ^{141,144}Ce are the least accessible of this group to plants. F_v values vary by factors of 10-30 on different soils [1, 4-6, 10, 22].

Most of the activation products (^{51}Cr, ^{54}Mn, ^{55,59}Fe, ^{60}Co, ^{65}Zn, ^{115}Cd), comprising the fifth group, are radioisotopes of biologically important micronutrients. As a rule, they have high mobility in soil-plant systems [5-6, 33]. The behaviour of these radionuclides depends strongly on the oxidation-reduction potential of the soil, the acidity of the soil solution, and the organic matter content [1]. Cadmium is known to be maximally mobile in all soil types. From 6.5 to 37% of this element is found in soil in a mobile form. It is followed by lead (5-14% in a mobile form). The next element, as far as mobility is concerned, is zinc (from 7.5% in soddy-podzolic to 1.5-2% in chernozems and chestnut soils). The content of Cu mobile forms varies between 7.5% in soddy-podzolic soils and 1% in chernozems. Chrome has the least mobility of the above elements in soddy-podzolic, grey and brown forest soils (1-4%). However in chernozems, its mobility is higher (5-5.5%) [1, 34]. More specific information for the most important radionuclides is given below.

7.1. Uranium

Uranium is widely distributed in nature. The main sources of uranium are hydrothermal veins, sedimentary rocks, and pyritic conglomerate beds. Uranium is an important constituent of about 155 minerals. It is also found in phosphate rock, lignite, and monazite sands [11, 35].

Natural uranium comprises three isotopes, ^{234}U, ^{235}U, and ^{238}U, with relative mass abundances of 0.0055, 0.72 and 99.27%, respectively [35]. The dominant isotope, ^{238}U, has a physical half-life of 4.5×10^9 yr, giving it a very low specific activity (12400 Bq g^{-1} U). In many instances, the natural abundance has been altered due either to anthropogenic uranium enrichment or to geological processes.

Uranium occurs in the valences +3, +4, +5 and +6, and, in soils, the valences +4 and +6 are the most important. U^{+4} dominates at Eh<200 mV [36], which is typical for waterlogged to wet soils. Tetravalent U has a tendency to strongly bind to organic matter and to precipitate and therefore, is immobile. The most oxidized state of U in nature is U^{+6}, and in oxidizing, aqueous environments, U is present as the uranyl ion, UO_2^{2+}. U availability is predominantly influenced by pH, soil carbonate content, amount of amorphous iron oxides and hydroxides, and organic matter content. Generally uranium is not very mobile. As it is incorporated in the mineral phase, it tends to remain *in situ*. However, uranium entering the environment due to human activities tends to be more mobile in spite of the fact that plant uptake of uranium tends to be quite limited.

Although extensive work has been done on U solubility in soils, there is comparatively little information regarding the uptake and translocation of U by plants as affected by soil properties. Generally, it appears that the U F_v value decreases from sandy through loamy to clay soils [37]. The presence of organic matter generally decreases the U F_v value [36-39].

7.2. Radium

Radium (^{226}Ra, $T_{1/2}$=1600 yr) is a natural decay product of ^{238}U. Ra-228 ($T_{1/2}$=5.75 yr) and ^{224}Ra ($T_{1/2}$= 3.82 d) are decay products from ^{232}Th and are present in the soil at comparable to slightly higher concentrations than ^{226}Ra [1, 11].

In tailings from uranium mills, phosphate mines, and phopshate processing sites, but also in various waste products from the NORM industry, the radium concentrations may be orders of magnitudes higher than in average [35, 38, 40].

Radium is the heaviest member of the alkaline earth metals, a group of metals whose lighter members (Ca and Mg) play a very important role in plant growth and nutrition [41]. As radium is highly electropositive, it reacts readily with many agents; most of these products are insoluble. It also co-precipitates with barium and strontium to form insoluble sulfate. Due to its basic characteristics, it is not easily complexed.

Compared with many other radioelements, little information exists on the uptake of radium from soil. Among the reported F_v values, large discrepancies are noticeable. Linsalata *et al.* [36] reported values ranging from 0.3×10^{-4} to 0.02. Based on data for a large series of crop and soil types, IUR [90] ninety-five percent confidence limits range from 2.9×10^{-4} to 0.12.

Radium has a high affinity for the regular exchange sites of the soil. According to Simon and Ibrahim [41], organic matter adsorbs about ten times as much radium as clay adsorbs. Increased exchangeable Ca [41], increasing pH [40], and high soil sulfate content [41] are reported to decrease the radium transfer factor.

7.3. Lead and Polonium

Uptake of ^{210}Pb and ^{210}Po in plants can occur both indirectly through the root system and by direct deposition from atmosphere [40, 42-45]. The relative importance of these two pathways depends upon the concentration of the radionuclides in the soil, the soil-plant F_v value, and the rate of deposition onto above-ground plant parts. One important implication of these considerations is that activity concentrations in crops, such as roots , tubers, cereals, and legumes, where the edible portion is protected by inedible plant parts, should not be significantly affected by direct deposition. For leafy vegetables, observed concentrations may be up to 3-5 times higher due to deposition effects. The ^{210}Pb and ^{210}Po content of soils varies with soil type. For example, Berger *et al.* [43] observed that the content was three times higher in organic soils than in mineral soils. Several sources have reported that superphosphate fertilizers contain significant concentrations of ^{210}Pb and ^{210}Po, which can provide a source of these radionuclides to plants [46-47].

7.4. Thorium

Thorium occurs naturally in the Earth's crust at an average background concentration of 8–12 mg kg^{-1}. The typical concentration range of Th in soils is 2–12 mg kg^{-1} with an average value of 6 mg kg^{-1} [11, 37, 46]. In solution, thorium shows only the +4 oxidation state Because of its high charge, Th^{4+} may be strongly adsorbed from a solution by natural ion exchangers. It has been reported that the Th content of soil is normally higher for a higher clay content of the soil. It was shown [47] that the Th ion is largely hydrolyzed at pH values above 3.2, and that hydroxyl complexes are involved in the sorption process. The adsorption of Th on clays, oxides, and organic matter increases with increasing pH and reaches a maximum at a pH of 6.5 [47]. On the other hand, it has been suggested [48] that the mobility of Th in soil may be less affected by soil pH than by soil organic matter.

7.5. Plutonium

Plutonium is a product of the nuclear fuel cycle, and exists primarily in the +4 state under normal environmental conditions. Depending on the waste release scenario, plutonium can enter the environment as a refractory oxide, with low solubility and slow conversion kinetics, or as more reactive/mobile forms, including nitrates, carbonates, and hydroxides. Plutonium behavior and transport characteristics vary with both contaminant levels and form [8, 10, 49]. Overall, F_v values for plutonium are in the range of about 10^{-3} to about 10^{-10}. This relatively low availability and low transport potential likely arises from the propensity of Pu to undergo hydrolysis and from strong sorption to soil solids [50]. With plutonium, a serious consideration is the relative role of contaminant form with respect to plant availability [53]. The relative bioavailability/uptakes of nitrate versus oxide forms were 7.5 and >10 for leaves/stems and seeds of barley, and 7.7 and >10 for pea leaves and seed, respectively [51]. The normal hydrolyzed nature of Pu(IV) was not evident for plutonium transported from root to all aerial tissues. The oxidation-reduction potential and acidity of the soil may significantly influence the mobility of plutonium [50].

7.6. Neptunium

Among the actinides, neptunium has a rather unique environmental chemistry. Although neptunium can exist in many oxidation states, NpO_2^+ appears to dominate [8, 52, 54, 55]. This chemical preference may account for the mobility and plant availability of neptunium. Not many soil/plant data are available for neptunium. In a large multi-year field lysimeter study, the relative ratio of Np/Pu in specific plants and tissues, as a measure of plant availability, was calculated [51]. The relative availability of neptunium, compared with plutonium, was 1200 and 1500 for cheatgrass and alfalfa leaves; 7500 and 43 000 for barley stem/leaves and seed; and 3500 and 370 000 for pea leaves and seed, respectively [52-53].

7.7. Americium

Americium is likely to occur in the environment as III and/or IV valence form. This would be relatively consistent with the biological behavior of the actinides, and mean F_v values for actinides can be presented as follows: Np>Cm>Am>Pu. F_v values for americium have generally been confined to laboratory studies [51]. The reported range in F_v is 10^{-6} to 10^{-1}. It is difficult to assess the reasons for the seemingly broad range, but it likely results from differences in contaminant source chemistry, amendment levels, and amendment methods. This said, there are some useful and reasonably consistent field evaluations. Livens *et al.* [50] reported F_v of 1.7×10^{-4} for mixed vegetation from an estuary impacted by a reprocessing site in the UK. On the whole, it would appear that americium transfer to plants is factor of 10 to 100-fold higher than for plutonium [8, 10].

7.8. Curium

As for americium, the environmental form of curium is likely the III and/or IV valence forms. Curium, like americium, has not been sufficiently studied to fully delineate the role of contaminant form and biological/soil factors on speciation and bioavailability. However, like americium, it appears that overall plant availability is a factor of 10 or more greater than for plutonium [8, 10].

7.9. Strontium

Natural strontium consists of 4 stable isotopes with mass numbers 84, 86, 87, 88. The average content of stable Sr in the earth's crust is 3.4×10^{-2} %. Among the fission products are two Sr radioisotopes: ^{90}Sr, ($T_{1/2} = 28.1$ years), and ^{89}Sr ($T_{1/2} = 50.5$ days). The chemical properties of strontium are typical for alkali-earth elements. Strontium is considered a highly mobile and bioavailable element [1, 5, 7, 10].

Strontium exists in the environment in the Sr(II) oxidation state. The total concentration range in soils is between 50 and 1000 mg/kg. Strontium is usually present in the surface environment as a carbonate or a sulfate mineral. As a result of nuclear-weapons testing, ^{90}Sr is distributed widely in the biosphere [56]. The chemistries of strontium and calcium are closely related [57-59]. The dominant aqueous strontium species in natural waters over a broad pH range (2 to 9) is the free divalent Sr^{2+} species.

The solubility of Sr^{2+} is not greatly affected by the presence of most inorganic anions, because Sr^{2+} forms only weak aqueous complexes with CO_3^{2-}, SO_4^{2-}, Cl^-, and NO_3^-. However, large concentrations of SO_4^{2-} or elevated alkalinity could play an important role in strontium mobility. Since strontium and calcium form humic acid complexes of similar stability [60-61], strontium should not effectively compete with calcium for humic acid ligands because calcium is typically present at much greater concentrations.

Strontium is one of the most bioavailable elements. Plants incorporate from 0.2% to 3% of the strontium in the soil [59, 62]. The transfer of ^{90}Sr from soil to plants is affected by the presence of stable strontium and its chemical analogue, calcium, in the system [7, 21, 59, 62, 63]. This factor is believed to be the main contributor to variability in F_v values.

The basic mechanism of absorption of Sr by soil is ion exchange. Strontium uptake by plants, in general, is greatest from soils of low calcium content and, in many cases, of high organic matter content. Strontium F_v values can differ by more than a factor of 100, depending on soil properties and biological features of plants. A decrease in exchangable strontium in soil does occur, but only very slowly [21]. Therefore, the availability to plants of soil ^{90}Sr decreases only slightly with time [64, 65].

7.10. Caesium

Among the numerous fission products of ^{235}U, the radioactive isotope ^{137}Cs, is of great importance, generated with a large fission yield (6.2%) and having a relatively long half-life (30.17 years) [56]. During ^{235}U fission, in addition to ^{137}Cs, 12 more isotopes of Cs are formed. However, almost all these are short-lived, except for ^{134}Cs ($T_{1/2}$=2.06 years), and very long-lived ^{135}Cs ($T_{1/2} = 2.3 \cdot 10^6$ years). The stable isotope ^{133}Cs serves a source of formation of radioactive ^{134}Cs by a (n, γ) reaction.

A peculiar feature of ^{137}Cs is its high biological mobility, caused by the fact that it is a radioisotope of an alkali element, which is a chemical analogue of a biochemically important element – K [1, 62, 66-67]. Caesium exists in the environment in the 1+ oxidation state. Stable caesium is ubiquitous in the environment with a crustal abundance of approximately 3.2 mg/kg; in soils, caesium concentrations range between 0.3 and 25 mg/kg.

The dominant aqueous species in soil and aquatic systems is thought to be free Cs^+. The Cs^+ ion forms only extremely weak aqueous complexes with SO_4^{-2}, Cl^- , and NO_3^-, and the formation of inorganic complexes is not believed to be a major influence on caesium speciation [7]. Sorption of caesium to organic colloids should follow a relationship similar to that to dissolved organic humic materials and, as a result, should not be an important sink for caesium in most soils [60]. It has been frequently demonstrated that caesium becomes associated with the clay mineral fraction of soils [68]. The association of caesium with clay minerals is characterized by a high selectivity. An important feature of the behavior of Cs isotopes is their ability for non-exchangeable sorption (fixation) by the soil solid phase [68, 69].

Accumulation of ^{137}Cs in plants depends on soil properties and decreases with increasing cation exchangeable capacity and clay fraction, with the concentrations of competing cations exerting a modifying effect [1]. Cs mobility is marked in peat and sandy soils. As noted above, caesium is an alkaline element, and its chemical analogue is potassium. With an increase of potassium concentration in soil, Cs accumulation by plants is decreased [7, 62, 66, 67]. The behaviour of caesium in soil is characterised by selective, irreversible absoption by micaceous clay minerals; as a result, over time, caesium mobility decreases considerably. On average, caesium F_v values are a factor of 2-10 lower than those of strontium [7, 65, 70, 71].

F_v values for caesium vary from about 10^{-3} up to about 1.0 [1, 62, 72-75]. For meadow grasses F_v values ranged from 0.5 to 33 in Chernobyl affected areas [70].

7.11. Iodine

Stable iodine (^{127}I) is present in soils at an average concentration of 5 mg/kg [10]. Typical terrestrial plants and food crops contain from 0.07 to 10 mg/kg of stable ^{127}I. The natural ratio of ^{129}I to ^{127}I is 10^{-14}, and it can be assumed that these two isotopes mimic each other, unless there is a contribution to ^{129}I concentrations from nuclear activities. It has long been believed that the dominant species in the aerobic soil environment are I^- and IO_3^-, and I_2, and methyl iodine for the marine system [10, 77].

Iodine F_v values vary from 0.1 to 5.0 for vegetative mass and from 0.5×10^{-3} up to 7.0×10^{-2} for generative parts. Accumulation of iodine depends on soil properties [76, 77]. High cation exchange capacity and organic matter content decrease F_v values for iodine Iodine transfers from soil-to-grain of rye and wheat are characterized by F_v values from 5×10^{-4} to 8×10^{-3}. Studies by Garland *et al.* [74] at various locations adjacent to a fuel reprocessing site showed a marked difference in the transfers of ^{127}I and ^{129}I from soil to plants. In these studies, F_v values for ^{127}I were calculated as 0.08, 1.2, 0.22, 0.2, 0.38, and 0.23 for tumble mustard, sagebrush leaves, Russian knapweed, and willow leaves, asparagus, and cottonwood leaves, respectively, while for ^{129}I, these values were as follows: 0.64, 1.0, 4.1, 0.5, 3.7, and 1.5. This would indicate that ^{127}I and ^{129}I were not showing similar behaviour, with a factor of 5-10 greater availability for ^{129}I, except for sagebrush leaves.

The potential volatile component of the iodine cycle has been investigated [10]. A series of nine soils was used; ^{129}I concentrations ranged from 0.17 to 23 pg/g. The soils were planted with soybeans, maintained in a closed growth chamber, and harvested after 21 days. Analysis of the plant tissues indicated that, although there was a factor of 135-fold difference in soil concentrations, each plant had a similar ^{129}I content, ranging from 62 to 85 pg/g tissue, and that there must be a volatile I component accounting for the loss. Analysis of volatile iodine components (both organic and inorganic) from the gas phases of ^{125}I amended soils and plants grown on contaminated soils, showed the presence of a significant organic iodine cycle. Total volatiles represented 0.028% for soils and 0.12% for plants; 75% of the volatile releases were organic in nature [10].

7.12. Ruthenium and Cerium

Natural ruthenium consists of 7 stable isotopes with mass numbers of 96, 98 to 102 and104. The content of stable Ru in the earth's crust is 5.0×10^{-7}%. Radiologically, the most important Ru isotopes are ^{103}Ru and ^{106}Ru. Ruthenium, an element of group VIII of the periodic system, has a high coordinative ability compared with other long-lived nuclear fission products [56, 78-82]. Therefore, in soil solution, Ru is practically always in the form of complex compounds, which contain oxo-groups with multiple bonds.

The number of Ru chemical forms in solution is noticeably increased due to the relatively high Ru tendency to hydrolysis [79]. In nitric acid, sulphuric acid, and hydrochloric acid solutions, the tetravalent state is typical for Ru. In the presence of deoxidants, the tri- and even the bi-valent state may be dominant.

Of crucial importance in Ru fixation on and distribution in soil constituents are the redox regime of the soil, the presence in soil of organic and inorganic ligands, and soil solution pH [79, 83]. Ru has 9 oxidation states. The fact that in the periodic table it is positioned in the group under Fe dictates a certain degree of chemical similarity with this element. In its heptavalent form, Ru resembles properties of Mn^{7+}, Te^{7+} and Re^{7+}.

The properties of trivalent Ru compounds closely resemble the behavior of Co^{3+}, Rh^{3+}, and Ir^{3+}. In soil solutions, Ru can be present in both cationic and anionic forms. Anionic forms of ^{106}Ru are more mobile, whereas cationic forms are readily sorbed by the soil solid phase. The absorbed ^{106}Ru does not enter significantly into ionic exchange reactions, as is shown by low desorption of the radionuclide by salt solutions and the absence of dependence on concentrations or charges of competing cations.

Since ^{144}Ce and ^{106}Ru are not analogues of biologically essential elements and are in soil in the sorbed state, their uptake by plants is low compared with ^{90}Sr and ^{137}Cs [81]. Up to 99% of the total content of ^{144}Ce and ^{106}Ru radionuclides taken up by plants is retained by the root system, and only a very small amount is accumulated in the above-ground mass [80-81]. Among moderate to long-lived fission products, ^{106}Ru, and especially ^{144}Ce, are two of the least available for plant uptake.

Uptake by plants of these radionuclides from soddy-podzolic sandy loam soil is much higher than from chernozem [81]. Plant accumulation of ^{144}Ce is greatly influenced by soil solution reactions. Results of greenhouse experiments with a crop grown in solution [83] demonstrate that the maximum uptake of nuclides is observed at a pH of solution close to neutral values. With pH variation from 4.5 to 8.5, ^{106}Ru accumulation in plants varies by a factor of 2–4 [78].

7.13. Niobium

Niobium is estimated to be present in surface soils at approximately 10 ppm [10]. Plant concentrations are believed to be in the range of 0.3 ppm. Nb is generally present in its +V oxidation state but it can also occur under the valence +II, +III and +IV. Tyler and Olsson [83] reported F_v of 0.38 to 1.2 and 0.005 to 0.04 for roots and shoots of *Agrostis capillaris*, respectively. Gerzabek et al. (1994) [84] reported distinctly lower F_v for a study with a Dystric Cambisol (pH 6.4) and a Calcic Chernozem (pH 8.2). Lowest F_v values were observed for bean pods (<0.0012-0.0017), then rape (0.0014-0.0025) and highest F_v values were reported for bean-shoots (0.0069-0.013). The lower F_v values were obtained for the Calcic Chernozem, attributed to a higher chelating ability of the organic matter due to the higher pH.

7.14. Cobalt

Cobalt is a chemical element of group VIII in the periodic system.The most common oxidation states of cobalt are +3 and +2 [84-86]. It occurs in different valences, is readily hydrolysed, and tends to form complexes with organic and inorganic matter. In nature, compounds of bivalent cobalt prevail, and trivalent compounds are much less often encountered [83].

The migration capacity of ^{60}Co depends on its form in solution and characteristics of the migration environment including the following: acidity, ground sorption properties, presence of agents for complex formation, and concentrations of chemical analogues [84, 87]. The migration ability of cobalt is particularly influenced by versene (a sodium salt of ethylene diamine tetraacetic acid EDTA), which forms a complex anion of bivalent cobalt, $\mathrm{Co^{II}EDTA^{2}}$, a stable compound [60-61, 78].

Co-60 concentration ratios vary widely, from 2.5×10^{-4} to 9.3×10^{-1} [85-86]. The smallest concentration ratios have been obtained for heavy textured soils (for example, chernozem) and peaty soils, and the largest have been found for sandy soils [87].

7.15. Copper and Zinc

Copper is an element of a secondary subgroup of group I in the Mendeleev Periodic System. $\mathrm{Cu^{+2}}$ is the most mobile form of Cu, but other ionic forms can also be found in soils [1, 34]. Aluminium and iron hydroxides, carbonates, and clay aluminosilicates have a high sorption power for Cu. The key reactions governing Cu behavior in most soils are chelation and complexation. Humic and fulvic acids are capable of forming stable complexes with Cu.

Adsorption of Cu ionic forms is pH dependent [87]. Solubility of different Cu forms in soil decreases with pH and approaches zero at pH 8.

The predominant and most mobile Zn form in soil is Zn^{2+}. The most crucial factors responsible for Zn mobility in soils is the content of clay minerals, hydroxides Fe and A1, and pH value; not so important is organic complexing and Zn sedimentation in the form of hydroxides, carbonates, and sulphides [1, 34]. Zn is most mobile and bioavailable in acid light mineral soils.

Zn ions are sorbed by organic matter and clay particles, with cation exchange being observed at pH <7 and precipitation at pH >7. Some 24-63% of the total Zn content in soil is bound to clay minerals and 14-38% is bound to hydroxides Fe and Al. Zn adsorption on hydroxides Fe and Mn starts from pH 5 and increases as acidity decreases. Organic matter influences Zn sorption to a much lesser extent. Iron hydroxides sorb Zn 1.5-2 times stronger than humic acids. Organic matter binds Zn into stable forms; as a result, Zn accumulation is observed in the organic horizons of soils and peat.

7.16. Nickel

The predominant form of nickel is Ni^{+2} [10. 34]. Global soil concentration for nickel averages 50 ppm. Relatively few studies directed at F_v values have been conducted, but nickel has been extensively studied in plants because it is an essential element, and it has a high absorption rate by plants. Tyler and Olsson [83] reported F_v values of 0.46 to 1.39 and 0.13 to 0.29 for, respectively, roots and shoots of *Agrostis capillaries.* Cataldo et. al. (1978) studied root transport of nickel [88]. Frissel and Bergeijk (1989) reported a value of 0.51 for clover [90]. Gerzabek et al. (1994) observed values of 0.08 and 0.29 for rape, of 0.43 and 0.56 for bean shoot and of 1.77 and 3.63 for bean pods [84].

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ROOT UPTAKE: TEMPERATE ENVIRONMENT

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Abstract

Soil-to-plant transfer factor data were compiled based on an extensive literature survey, comprising 1167 information sources, including international and national databases. In order to decrease the uncertainty associated with soil/plant factors a special classification schema was developed and soil-to-plant transfer factors were grouped according to the selected plant and soil categories. Best estimates were generally derived from geometric means calculated for major plants groups and also for some plant compartments within these groups. For the natural radionuclides 2162 data were recorded and for the artificial radionuclides - 9390. Most transfer factor data were available for ^{90}Sr, ^{137}Cs, U and Ra. Radionuclide accumulation shows high variability, which is influenced by properties of radionuclides, soil characteristics, peculiarities of plant species. In comparison to Technical Reports Series No. 364, information presented in the paper substantially increases the amount of available data on radionuclide transfer to plants. The data, providing information for specific plant and soil groups, allows more precise radiological assessments in different areas around the world.

1. INFORMATION SOURCES AND DATA EVALUATION

1.1. Sources of information

The literature review was started through Science Direct, by consideration of references in earlier reviews on F_v values for natural radionuclides and by considering the references (if available) from the IAEA coordination research projects [1, 2] and the IUR databases on radionuclide transfer to plants (Fig. 1). All useful cited references were then evaluated, as described below, and a new database was created.

With respect to the evaluation of former databases, there was a great similarity between the IAEA CRP 1998 data base and the IUR-1989 data base, and most of the data in these databases was included because they were well referenced (but only after having checked the original references in most of the cases). Use of the RADFLUX database was very limited because it was not clearly referenced.

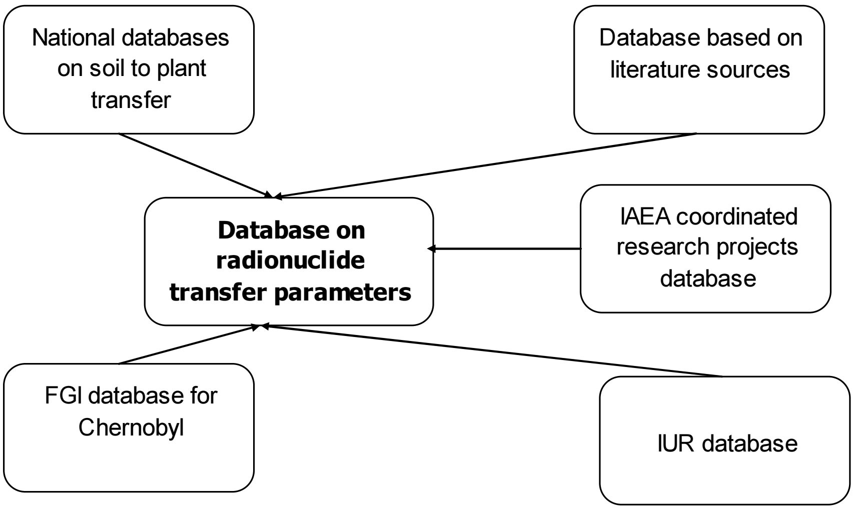

FIG. 1. The sources of information used for elaboration of the database on radionuclide transfer to plants.

Though the IUR-1992 [3] database is a well accepted database, the data were generally not considered, since it was not possible to trace back the references, giving cause for concern over double entry of data. Only for Pb were some data from the IUR-1992 database considered because of the limited data for Pb and because it was possible to verify rather easily that there were no double entries involved.

Some other databases (such as the database on transfer factor values of the French-German Initiative (FGI) for Chernobyl (Subproject 3a) [4] and the database of Russian Institute of Agricultural Radiology and Agroecology), which mainly applied to ^{90}Sr and ^{137}Cs and temperate environments, were also consulted to derive appropriate F_v values [5].

In total, 1167 information sources (books, journals, proceedings of conferences, institutional reports) (200 for natural radionuclides and 967 for artificial radionuclides), as well as international and national databases, were reviewed, and a set of databases was created to handle and process the data derived from various literature sources (Fig. 1).

1.2. Data evaluation

Each reference was critically reviewed, and data were retained/excluded based on the following criteria.

- Only individual data from matching crop-soil combinations were retained, with summary data from reviews being excluded.
- Experimental results had to be clear and concise. Clear information on sample collection and preparation was required, and if there was any doubt whether concentration in crops (or F_v) was expressed relative to fresh or dry mass, the data were not considered.
- Information on F_v was only included if there was clear indication of plant compartment sampled and analyzed.
- The minimum soil information required for associated F_v data to be entered was the concentration of radionuclides in the soil and the type of contamination. If soil concentrations were expressed on wet weight basis (and no information available on

soil water content to calculated concentrations per dry mass) or if contaminant level expressed as available fraction, F_v data were not included.

- Data from areas with high natural radioactivity were included, except when it concerned specific substrates (non-soils) (e.g. uranium tailings, red mud, phosphogypsum).
- Data from pot experiments were excluded, with the exception of experiments with natural radionuclides, where data were used independent of pot size.

For each individual reference retained, mean F_v values were calculated for specific crop/soil combinations. The F_v values were presented with SD and range, if this information was available or if the values could be calculated from information in the reference. Differences of an order of magnitude could sometimes be observed between samples collected at a same site and for the same crop. In spite of the fact that there was considerable uncertainty associated with the average F_v values recorded in the database, only the average F_v values were considered when evaluating the data and not the associated SD and range.

2. DERIVED TRANSFER FACTOR VALUES FOR NATURAL RADIONUCLIDES

Derived transfer factor values for U, Th, Ra, Pb and Po are presented in Tables 2-7 while Table 1 gives the number of entries, the number of entries retained, and entries for which a soil class was given or could be deduced for radionuclides of these elements.

TABLE 1 STATISTICAL INFORMATION ON AVAILABILITY AND USE OF DATA RELEVANT TO CALCULATING TRANSFER FACTORS FOR NORMS

Element	Entries	Entries considered*	Entries with soil group
Pb	232	208	95
Po	73	57	**
Ra	594	561	339
Th	351	335	95
U	912	781	278

* Only entries giving F_v values for specific crop groups (natural vegetation excluded, except for natural pastures); F_v values for special substrates (e.g. tailings) not included; for Pb and Po, no entry if it was clear that deposition was involved**: For Po, information on soil type was given only for few entries, and the only soils for which information was available were sandy soils.

The database for natural radionuclides included 2162 records for 5 elements. Most data were available for U followed by Ra, and minimum information was available for Po. Between 40 and 60% of the entries contained information on soil groups or sufficient information on soil texture or organic matter content to deduce a soil group. For ^{210}Po there were only a few entries containing information on soil group, all for sandy soil.

The data presented show clear differences between GMs and AMs. Overall, the value derived for AM is 3 to 5-fold higher than the value derived for GM. It should be stressed that, though information is given on arithmetic mean and SD, F_v values should generally be characterized using the GM and GSD, since the log-transformed data were generally normally distributed whereas the non-transformed data were not.

TABLE 2. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR U

Plant Group	Plant Compartment	Soil	N	GM	GSD	AM	SD	Min	Max	#ref
All			781	2.3×10^{-2}	9.1	2.1×10^{-1}	9.1×10^{-1}	3.0×10^{-5}	1.4×10^{1}	57
Cereals	Grain	All	59	6.2×10^{-3}	7.7	5.0×10^{-2}	1.4×10^{-1}	1.6×10^{-4}	8.2×10^{-1}	16
		Sand	6	8.9×10^{-3}	1.1×10^{1}	2.8×10^{-2}	2.5×10^{-2}	1.9×10^{-4}	6.2×10^{-2}	3
		Loam	20	7.7×10^{-3}	5.1	1.8×10^{-2}	1.9×10^{-2}	1.6×10^{-4}	6.2×10^{-2}	7
		Clay	11	3.8×10^{-3}	4.0	9.4×10^{-3}	1.5×10^{-2}	7.6×10^{-4}	5.0×10^{-2}	4
	Stems and shoots	All	55	2.7×10^{-2}	7.5	1.4×10^{-1}	4.8×10^{-1}	3.0×10^{-5}	3.5	13
		Sand	6	3.4×10^{-2}	6.0	7.5×10^{-2}	6.5×10^{-2}	2.1×10^{-3}	1.7×10^{-1}	3
		Loam	25	5.4×10^{-2}	6.30	2.5×10^{-1}	7.0×10^{-1}	7.4×10^{-4}	3.5	7
		Clay	8	1.0×10^{-2}	3.6	2.3×10^{-2}	3.4×10^{-2}	2.8×10^{-3}	9.8×10^{-2}	2
Maize	Grain	All	9	1.5×10^{-2}	1.2×10^{1}	1.2×10^{-1}	2.3×10^{-1}	5.0×10^{-4}	7.1×10^{-1}	5
	Stems and shoots	All	11	7.8×10^{-3}	1.4×10^{1}	1.1×10^{-1}	2.9×10^{-1}	1.6×10^{-4}	9.6×10^{-1}	6
Leafy Vegetables	Leaves	All	108	2.0×10^{-2}	7.3	2.2×10^{-1}	1.1	7.8×10^{-5}	8.8	15
		Sand	7	1.7×10^{-1}	1.5×10^{1}	1.5	3.3	1.5×10^{-3}	8.8	2
		Loam	14	4.3×10^{-2}	3.9	8.7×10^{-2}	8.8×10^{-2}	7.7×10^{-3}	2.7×10^{-1}	3
		Clay	9	3.6×10^{-3}	4.2	1.0×10^{-2}	1.6×10^{-2}	7.6×10^{-4}	4.3×10^{-2}	3
		Peat	6	1.8×10^{-1}	9.7	1.5	3.2	7.9×10^{-3}	8.0	2
Non-leafy Vegetables	Fruits, heads, berries, buds	All	38	1.5×10^{-2}	4.2	3.6×10^{-2}	5.3×10^{-2}	5.2×10^{-4}	2.0×10^{-1}	10
		Sand	7	1.9×10^{-2}	5.5	4.9×10^{-2}	6.0×10^{-2}	1.3×10^{-3}	1.6×10^{-1}	2
		Loam	4	2.3×10^{-2}	2.2	2.8×10^{-2}	1.7×10^{-2}	7.6×10^{-3}	4.7×10^{-2}	2
		Clay	7	1.8×10^{-2}	4.2	4.8×10^{-2}	7.4×10^{-2}	5.0×10^{-3}	2.0×10^{-1}	2
	Stems and shoots	All	6	5.3×10^{-2}	9.9	2.6×10^{-1}	3.5×10^{-1}	4.3×10^{-3}	7.1×10^{-1}	4
Leguminous Vegetables	Pods	All	19	2.2×10^{-3}	1.2×10^{1}	2.2×10^{-2}	4.6×10^{-2}	5.4×10^{-5}	1.5×10^{-1}	10
		Loam	4	3.0×10^{-3}	1.8×10^{1}	1.5×10^{-2}	2.2×10^{-2}	5.4×10^{-5}	4.7×10^{-2}	3
		Clay	7	5.5×10^{-4}	4.7	1.4×10^{-3}	1.9×10^{-3}	5.7×10^{-5}	5.0×10^{-3}	3

TABLE 2. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR U (Cont.)

Plant Group	Plant Compartment	Soil	N	GM	GSD	AM	SD	Min	Max	#ref
Leguminous Vegetables	Stems and shoots	All	21	6.4×10^{-2}	1.4×10^{1}	8.4×10^{-1}	2.0	7.4×10^{-4}	8.7	11
		Sand	6	2.8×10^{-1}	2.0×10^{1}	2.4	3.5	5.3×10^{-3}	8.7	4
		Loam	6	1.2×10^{-2}	6.2	3.5×10^{-2}	5.1×10^{-2}	7.4×10^{-4}	1.4×10^{-1}	5
Root Crops	Roots	All	46	8.4×10^{-3}	6.2	3.6×10^{-2}	6.5×10^{-2}	4.9×10^{-4}	2.6×10^{-1}	16
		Sand	9	7.8×10^{-3}	5.9	3.8×10^{-2}	7.6×10^{-2}	9.9×10^{-4}	2.3×10^{-1}	6
		Loam	10	2.5×10^{-2}	3.2	4.3×10^{-2}	4.3×10^{-2}	2.6×10^{-3}	1.2×10^{-1}	4
		Clay	5	6.8×10^{-3}	6.2	2.3×10^{-2}	3.9×10^{-2}	7.9×10^{-4}	9.2×10^{-2}	4
	Stems and shoots	All	37	2.8×10^{-2}	5.4	9.5×10^{-2}	1.6×10^{-1}	2.0×10^{-3}	7.0×10^{-1}	12
		Sand	9	2.5×10^{-2}	5.6	6.7×10^{-2}	8.3×10^{-2}	2.0×10^{-3}	2.4×10^{-1}	5
		Loam	11	5.0×10^{-2}	3.0	8.6×10^{-2}	9.8×10^{-2}	1.3×10^{-2}	3.2×10^{-1}	3
		Clay	5	1.1×10^{-2}	4.3	2.2×10^{-2}	2.3×10^{-2}	2.0×10^{-3}	5.8×10^{-2}	3
Tubers	Tubers	All	28	5.0×10^{-3}	6.4	1.7×10^{-2}	2.4×10^{-2}	1.8×10^{-4}	8.0×10^{-2}	11
		Sand	4	1.9×10^{-2}	3.8	3.3×10^{-2}	3.4×10^{-2}	4.3×10^{-3}	7.8×10^{-2}	3
		Loam	3	2.8×10^{-2}	3.2	4.0×10^{-2}	3.6×10^{-2}	8.2×10^{-3}	8.0×10^{-2}	3
		Clay	6	9.2×10^{-4}	3.0	1.5×10^{-3}	1.7×10^{-3}	1.9×10^{-4}	4.8×10^{-3}	3
	Stems and shoots	All	1			1.9×10^{-1}				1
Fruit	Fruits	All	11	1.2×10^{-2}	5.9	5.7×10^{-2}	1.2×10^{-1}	1.3×10^{-3}	3.7×10^{-1}	5
	Leaves	All	66	3.5×10^{-1}	3.6	6.7×10^{-1}	1.1	3.6×10^{-4}	7.5	3
Herbs		All	9	3.6×10^{-2}	4.9	1.1×10^{-1}	1.6×10^{-1}	8.6×10^{-3}	4.1×10^{-1}	4
Other Crops	Leaves (sunflower)	All	39	7.1×10^{-2}	3.9	3.1×10^{-1}	1.2	8.9×10^{-3}	7.8	3
		Sand	5	4.1×10^{-1}	5.3	1.7	3.4	1.6×10^{-1}	7.8	2
		Loam	22	7.1×10^{-2}	2.9	1.2×10^{-1}	1.4×10^{-1}	1.0×10^{-2}	6.4×10^{-1}	2
		Clay	11	2.7×10^{-2}	2.1	3.5×10^{-2}	2.8×10^{-2}	8.9×10^{-3}	1.0×10^{-1}	1
	Grain (sunflower)	All	2	1.5×10^{-2}	2.4	1.8×10^{-2}	1.5×10^{-2}	8.2×10^{-3}	2.9×10^{-2}	1
Grasses	Stems and shoots	All	147	1.7×10^{-2}	9.4	1.2×10^{-1}	4.9×10^{-1}	2.0×10^{-4}	5.5	18
		Sand	19	1.6×10^{-2}	1.7×10^{1}	2.5×10^{-1}	5.0×10^{-1}	5.5×10^{-4}	1.8	5
		Loam	34	9.8×10^{-3}	8.4	5.5×10^{-2}	1.0×10^{-1}	3.1×10^{-4}	4.6×10^{-1}	5
Pasture (Natural)	Stems and shoots	All	53	4.6×10^{-2}	5.3	4.2×10^{-1}	2.0	1.3×10^{-3}	1.4×10^{1}	9
		Sand	3	2.7×10^{-3}	1.8	3.0×10^{-3}	1.5×10^{-3}	1.3×10^{-3}	3.9×10^{-3}	1
		Loam	7	7.2×10^{-2}	3.3×10^{1}	2.7	5.2	1.8×10^{-3}	1.4×10^{1}	3
Fodder Leguminous	Stems and shoots	All	15	1.5×10^{-2}	4.2	1.2×10^{-1}	4.0×10^{-1}	2.0×10^{-3}	1.6	4
		Sand	12	1.0×10^{-2}	2..0	1.2×10^{-2}	5.9×10^{-3}	2.0×10^{-3}	2.1×10^{-2}	1

TABLE 3. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Th

Plant Group	Plant Compartment	Soil	N	GM	GSD	AM	SD	Min	Max	#ref
All		All	335	3.4×10^{-3}	1.5×10^{1}	5.8×10^{-2}	2.4×10^{-1}	1.2×10^{-6}	2.7 ×	28
Cereals	Grain	All	36	2.1×10^{-3}	3.39	3.80×10^{-3}	4.5×10^{-3}	1.6×10^{-4}	2.2×10^{-2}	10
		Sand	4	4.4×10^{-3}	1.35	4.5×10^{-3}	1.3×10^{-3}	3.0×10^{-3}	6.0×10^{-3}	1
		Loam	18	2.7×10^{-3}	3.38	4.8×10^{-3}	5.5×10^{-3}	2.1×10^{-4}	2.2×10^{-2}	7
		Clay	9	1.2×10^{-3}	1.56	1.3×10^{-3}	6.5×10^{-4}	7.0×10^{-4}	2.6×10^{-3}	3
	Stems and shoots	All	28	6.1×10^{-3}	2.43	9.0×10^{-3}	9.0×10^{-3}	1.6×10^{-3}	3.7×10^{-2}	5
		Sand	4	1.4×10^{-2}	1.26	1.4×10^{-2}	3.3×10^{-3}	1.1×10^{-2}	1.8×10^{-2}	1
		Loam	11	6.6×10^{-3}	1.94	7.8×10^{-3}	4.0×10^{-3}	2.4×10^{-3}	1.3×10^{-2}	2
		Clay	8	3.6×10^{-3}	1.57	3.9×10^{-3}	1.7×10^{-3}	2.0×10^{-3}	6.0×10^{-3}	3
		Organic	3	2.0×10^{-3}	1.49	2.1×10^{-3}	9.2×10^{-4}	1.6×10^{-3}	3.2×10^{-3}	1
Maize	Grain	All	18	6.4×10^{-5}	9.23	8.5×10^{-4}	2.6×10^{-3}	1.2×10^{-6}	1.1×10^{-2}	5
		Loam	10	2.0×10^{-4}	9.34	1.5×10^{-3}	3.4×10^{-3}	1.4×10^{-5}	1.10x 10^{-2}	3
		Clay	7	1.5×10^{-5}	3.70	2.5×10^{-5}	2.1×10^{-5}	1.2×10^{-6}	5.4×10^{-5}	2
	Stems and shoots	All	2			1.8×10^{-3}	1.7×10^{-3}	5.4×10^{-4}	3.0×10^{-3}	2
Leafy Vegetables	Leaves	All	24	1.2×10^{-3}	6.03	1.18×10^{-2}	4.3×10^{-2}	9.4×10^{-5}	2.1×10^{-1}	7
		Loam	13	8.6×10^{-4}	3.32	1.5×10^{-3}	1.6×10^{-3}	9.4×10^{-5}	5.8×10^{-3}	3
		Clay	7	4.9×10^{-4}	2.81	9.1×10^{-4}	1.4×10^{-3}	1.9×10^{-4}	4.1×10^{-3}	3
Non-leafy Vegetables	Fruits, heads, berries, buds	All	17	7.8×10^{-4}	6.76	3.4×10^{-3}	5.4×10^{-3}	6.2×10^{-5}	1.6×10^{-2}	4
		Loam	10	2.0×10^{-4}	9.34	1.5×10^{-3}	3.4×10^{-3}	1.4×10^{-5}	1.1×10^{-2}	1
		Clay	7	1.5×10^{-5}	3.70	2.5×10^{-5}	2.1×10^{-5}	1.2×10^{-6}	5.4×10^{-5}	3
	Stems and shoots	All	6	2.2×10^{-3}	5.1	6.1×10^{-3}	9.2×10^{-3}	3.3×10^{-4}	2.4×10^{-2}	4
Leguminous Vegetables	Pods	All	22	5.3×10^{-4}	9.4	2.3×10^{-2}	1.0×10^{-1}	2.5×10^{-5}	4.8×10^{-1}	8
		Loam	14	1.8×10^{-3}	3.9	4.4×10^{-3}	7.1×10^{-3}	1.7×10^{-4}	2.4×10^{-2}	4
		Clay	10	4.1×10^{-4}	2.3×10^{1}	4.9×10^{-2}	1.5×10^{-1}	2.5×10^{-5}	4.8×10^{-1}	4
		Organic	4	4.5×10^{-4}	7.6	1.4×10^{-3}	1.9×10^{-3}	8.0×10^{-5}	4.0x 10^{-3}	1
	Stems and shoots	All	7	4.3×10^{-3}	4.0	8.5×10^{-3}	9.1×10^{-3}	5.3×10^{-4}	2.4×10^{-2}	5
Root Crops	Roots	All	33	8.0×10^{-4}	1.3×10^{1}	9.3×10^{-3}	2.0×10^{-2}	8.2×10^{-6}	9.5×10^{-2}	8
		Loam	14	1.1×10^{-3}	1.6×10^{1}	9.1×10^{-3}	1.5×10^{-2}	8.2×10^{-6}	5.3×10^{-2}	4
		Clay	14	2.6×10^{-4}	5.4	2.0×10^{-3}	6.1×10^{-3}	4.5×10^{-5}	2.3×10^{-2}	4
	Stems and shoots	All	8	8.7×10^{-3}	4.4	2.1×10^{-2}	2.8×10^{-2}	2.1×10^{-3}	7.8×10^{-2}	6

TABLE 3. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Th (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Tubers	Tubers	All	24	2×10^{-4}	9.9	1.6×10^{-3}	3.7×10^{-3}	1.3×10^{-5}	1.8×10^{-2}	8
		Loam	10	2.5×10^{-4}	6.4	8.3×10^{-4}	1.2×10^{-3}	1.3×10^{-5}	3.6×10^{-3}	5
		Clay	12	9.6×10^{-5}	1.1×10^{1}	1.7×10^{-3}	5.0×10^{-3}	1.3×10^{-5}	1.8×10^{-2}	3
	Stems and shoots	All	2			1.9×10^{-2}	1.9×10^{-2}	4.8×10^{-3}	3.2×10^{-2}	2
Fruits	Fruits	All	2			6.3×10^{-3}	5.3×10^{-3}	2.5×10^{-3}	1.0×10^{-2}	3
Other Crops	Tea Leaves		1			3.4×10^{-3}				1
Grasses	Stems and shoots	All	64	4.2×10^{-2}	3.1	7.4×10^{-2}	1.0×10^{-1}	7.4×10^{-4}	6.5×10^{-1}	6
Pasture (Natural)	Stems and shoots	All	36	9.9×10^{-2}	5.5	3.7×10^{-1}	6.4×10^{-1}	2.9×10^{-3}	2.7	6
Fodder Leguminous	Stems and shoots	All	5	2.6×10^{-3}	1.6	2.8×10^{-3}	1.2×10^{-3}	1.5×10^{-3}	4×10^{-3}	2

TABLE 4. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ra

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
All		All	561	4.0×10^{-2}	8.6	6.9×10^{-1}	6.1	1.1×10^{-5}	1.3×10^{2}	47
Cereals	Grain	All	24	1.7×10^{-2}	1.20×10^{1}	1.1×10^{-1}	1.9×10^{-1}	8.0×10^{-5}	6.7×10^{-1}	8
		Loam	7	2.9×10^{-2}	9.70	1.6×10^{-1}	2.6×10^{-1}	8×10^{-4}	6.7×10^{-1}	3
		Clay	10	3.9×10^{-2}	9.88	1.3×10^{-1}	1.8×10^{-1}	2.40×10^{-4}	5.00×10^{-1}	3
	Stems and shoots	All	20	3.6×10^{-2}	4.78	9.4×10^{-2}	1.3×10^{-1}	1.60×10^{-3}	4.3×10^{-1}	4
		Loam	10	5.2×10^{-2}	4.44	1.3×10^{-1}	1.6×10^{-1}	7.20×10^{-3}	4.3×10^{-1}	2
Maize	Grain	All	28	2.4×10^{-3}	5.43	1.1×10^{-2}	2.3×10^{-2}	1.2×10^{-4}	1.1×10^{-1}	9
		Loam	4	1.7×10^{-3}	1.78	1.9×10^{-3}	1.0×10^{-3}	9.0×10^{-4}	3.0×10^{-3}	3
		Clay	16	1.4×10^{-3}	4.81	9.2×10^{-3}	2.8×10^{-2}	1.2×10^{-4}	1.1×10^{-1}	5
	Stems and shoots	All	6	1.8×10^{-2}	5.24	3.7×10^{-2}	3.3×10^{-2}	9.6×10^{-4}	8.5×10^{-2}	3
Leafy Vegetables	Leaves	All	77	9.1×10^{-2}	6.7	2.6	1.5×10^{1}	1.8×10^{-3}	1.3×10^{2}	16
		Loam	10	1.20×10^{-1}	2.5	1.6×10^{-1}	1.2×10^{-1}	1.6×10^{-2}	$4.44x\ 10^{-1}$	4
		Clay	20	4.0×10^{-2}	4.5	9.2×10^{-2}	1.1×10^{-1}	1.8×10^{-3}	4.2×10^{-1}	4
		Organic	9	4.9×10^{-2}	2.1	6.2×10^{-2}	4.60×10^{-2}	2.0×10^{-2}	1.4×10^{-1}	2
Non-leafy Vegetables	Fruits, heads, berries, buds	All	44	1.7×10^{-2}	8.4	2.6×10^{-1}	1.0	2.4×10^{-4}	6.3	12
		Sand	3	2.2×10^{-3}	2.1	2.70×10^{-3}	2.0×10^{-3}	1.10×10^{-3}	5.00×10^{-3}	4
		Loam	4	4.8×10^{-2}	5.6	1.2×10^{-1}	1.5×10^{-1}	6.9×10^{-3}	3.40×10^{-1}	2
		Clay	17	2.2×10^{-2}	2.8	4.1×10^{-2}	6.00×10^{-2}	3.9×10^{-3}	2.10×10^{-1}	1

TABLE 4. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ra (Cont.)

Plant Group	Plant Compartment	Soil	N	GM	GSD	AM	SD	Min	Max	#ref
	Stems and shoots	All	13	6.1×10^{-2}	6.4	3.1×10^{-1}	5.7×10^{-1}	6.7×10^{-3}	1.8	4
Leguminous Vegetables	Pods	All	40	1.4×10^{-2}	8.2	2.4×10^{-1}	1.0	3.20×10^{-4}	6.2	14
		Loam	12	9.8×10^{-3}	4.5	2.2×10^{-2}	2.6×10^{-2}	4.80×10^{-4}	8.7×10^{-2}	5
		Clay	15	9.3×10^{-3}	4.2	2.2×10^{-2}	3.0 × 10-2	8 × 10-4	1.10×10^{-1}	5
	Stems and shoots	All	18	2.8×10^{-2}	1.1×10^{1}	1.4×10^{-1}	3.5×10^{-1}	1.1×10^{-5}	1.50	8
		Loam	6	1.1×10^{-2}	3.2×10^{1}	4.8×10^{-2}	4.6×10^{-2}	1.1×10^{-5}	1.10×10^{-1}	3
Root Crops	Roots	All	60	7×10^{-2}	9.2	1.9	7.8	2.0×10^{-3}	5.6×10^{1}	16
		Sand	3	4.8×10^{-3}	2.3	6×10^{-3}	4.5×10^{-3}	2.0×10^{-3}	1.1×10^{-2}	2
		Loam	8	9.1×10^{-2}	1.9	1.1×10^{-1}	5.9×10^{-2}	2.9×10^{-2}	2×10^{-1}	3
		Clay	23	3.9×10^{-2}	2.9	5.1×10^{-2}	5.20×10^{-2}	3.2×10^{-3}	2.2×10^{-1}	4
Root Crops	Stems and shoots	All	22	7.1×10^{-2}	4.6	1.8×10^{-1}	2.2×10^{-1}	2.5×10^{-3}	7.1×10^{-1}	9
		Loam	6	1.5×10^{-1}	5.6	3.1×10^{-1}	2.8×10^{-1}	9.60×10^{-3}	7.1×10^{-1}	2
Tubers	Tubers	All	45	1.1×10^{-2}	6.8	1.4×10^{-1}	5.9×10^{-1}	2.40×10^{-4}	3.9	15
		Loam	8	1.20×10^{-2}	1.1×10^{1}	9.2×10^{-2}	2.1×10^{-1}	2.40×10^{-4}	6.20×10^{-1}	4
		Clay	24	5.40×10^{-3}	2.5	9.5×10^{-3}	1.6×10^{-2}	1.3×10^{-3}	8.00×10^{-2}	7
	Shoots	All	6	1.6×10^{-1}	2.2	1.9×10^{-1}	1.1×10^{-1}	4.3×10^{-2}	3.3×10^{-1}	3
Herbs	Herbs	All	20	6.9×10^{-2}	4.5	2.6×10^{-1}	7.3×10^{-1}	5.30×10^{-3}	3.3	5
Fruits	Fruits	All	12	1.2×10^{-2}	3.7	2.7×10^{-2}	4.6×10^{-2}	1.4×10^{-3}	1.7×10^{-1}	4
Other Crops	Sunflower-3/peanut-1	All	4	4.2×10^{-1}	3.0	5.8×10^{-1}	4.2×10^{-1}	8.50×10^{-2}	1.10	2
	Tea leaves	All	1			3.3×10^{-2}				1
Grasses	Stems and shoots	All	62	1.3×10^{-1}	4	2.6×10^{-1}	3.20×10^{-1}	3.6×10^{-3}	1.6	19
		Sand	24	1.4×10^{-1}	4.2	3.1×10^{-1}	4×10^{-1}	5.4×10^{-3}	1.6	7
		Loam	14	2.6×10^{-1}	2.00	3.2×10^{-1}	2.1×10^{-1}	9.6×10^{-2}	7.2×10^{-1}	2
		Clay	3	4.20×10^{-2}	1.5	4.4×10^{-2}	1.70×10^{-2}	2.70×10^{-2}	6.1×10^{-2}	1
Pasture (Natural)	Stems and shoots	All	42	7.10×10^{-2}	7.6	1.9×10^{-1}	2.8×10^{-1}	5.1×10^{-5}	1.60	16
		Sand	3	8×10^{-3}	3.8	1.2×10^{-2}	1.1×10^{-2}	1.80×10^{-3}	2.3×10^{-2}	1
		Loam	6	8.8×10^{-3}	1.9×10^{1}	3.9×10^{-2}	4×10^{-2}	5.1×10^{-5}	1.1×10^{-1}	4
Fodder Leguminous	Stems and shoots	All	16	1.7×10^{-1}	3.1	3.0×10^{-1}	3.70×10^{-1}	3.40×10^{-2}	1.5	4
		Sand	5	1.70×10^{-1}	2.5	2.4×10^{-1}	2.20×10^{-1}	8.00×10^{-2}	5.7×10^{-1}	3
		Loam	8	1.2×10^{-1}	3.9	3.1×10^{-1}	5.1×10^{-1}	3.40×10^{-2}	1.5	2

TABLE 5. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Pb

Plant Group	Plant Compartment	Soil	N	GM	GSD	AM	SD	Min	Max	#ref
All			208	2.0×10^{-2}	1.4×10^{1}	6.3×10^{-1}	2.9	1.5×10^{-4}	2.5×10^{1}	24
Cereals	Grain	All	9	1.1×10^{-2}	3.6	1.8×10^{-2}	1.6×10^{-2}	1.9×10^{-3}	4.8×10^{-2}	5
	Stems and shoots	All	4	2.3×10^{-2}	3.5	3.8×10^{-2}	4.1×10^{-2}	5.1×10^{-3}	9.6×10^{-2}	3
Maize	Grain	All	9	1.2×10^{-3}	2.3	1.7×10^{-3}	1.4×10^{-3}	5.2×10^{-4}	3.8×10^{-3}	4
	Stems and shoots	All	3	2.8×10^{-3}	6.6	8.5×10^{-3}	1.3x 10^{-2}	6.0×10^{-4}	2.3×10^{-2}	2
Leafy Vegetables	Leaves	All	31	8.0×10^{-2}	1.3×10^{1}	2.1	6.1	3.2×10^{-3}	2.5×10^{1}	9
		Sand	4	7.3×10^{-2}	1.5	7.8×10^{-2}	3.3×10^{-2}	4.9×10^{-2}	1.1×10^{-1}	1
		Loam	3	8.2×10^{-1}	1.0	8.2×10^{-1}	3.5×10^{-2}	7.9×10^{-1}	8.6×10^{-1}	1
		Clay	7	2.8×10^{-2}	4.1	5.1×10^{-2}	4.8×10^{-2}	4.1×10^{-3}	1.2×10^{-1}	3
Non-leafy Vegetables	Fruits, heads, berries, buds	All	5	1.5×10^{-2}	2.6×10^{1}	7.8×10^{-1}	1.7	1.5×10^{-3}	3.9	3
	Stems and shoots	All	2			8.8×10^{-3}	4.2×10^{-3}	5.8×10^{-3}	1.2×10^{-2}	2
Leguminous-Vegetables	Pods	All	17	5.3×10^{-3}	1.2×10^{1}	3.4×10^{-1}	1.20	4.6×10^{-4}	4.9	6
		Sand	3	2.7×10^{-3}	3.2	4.2×10^{-3}	3.4×10^{-3}	6.5×10^{-4}	8.9×10^{-3}	2
		Loam	5	1.4×10^{-3}	4.4	4.2×10^{-3}	3.4×10^{-3}	6.5×10^{-4}	8.9×10^{-3}	2
		Clay	4	8.0×10^{-4}	1.0	3.3×10^{-3}	4.7×10^{-3}	4.6×10^{-4}	1.0×10^{-2}	2
	Stems and shoots	All	1			8.0×10^{-4}				1
Root Crops	Roots	All	27	1.5×10^{-2}	1.6×10^{1}	4.1×10^{-1}	9.8×10^{-1}	2.4×10^{-4}	3.3	9
		Sand	5	6.4×10^{-2}	1.6	7.3×10^{-2}	3.4×10^{-2}	4.2×10^{-2}	1.2×10^{-1}	3
		Loam	5	2.3×10^{-3}	4.7	5.0×10^{-3}	6.8×10^{-3}	2.4×10^{-4}	1.7×10^{-2}	2
	Stems and shoots	All	12	6.3×10^{-2}	1.5×10^{1}	2.5	5.7	3×10^{-3}	1.6×10^{1}	4
Tubers	Tubers	All	30	1.5×10^{-3}	7.4	9.1×10^{-2}	4.8×10^{-1}	1.5×10^{-4}	2.6	10
		Sand	5	6.4×10^{-3}	3.5	1.20×10^{-2}	1.6×10^{-2}	1.6×10^{-3}	3.9×10^{-2}	3
		Loam	17	5.20×10^{-4}	2.4	7.3×10^{-4}	6.2×10^{-4}	1.5×10^{-4}	2.3×10^{-3}	4
Fruits	Fruits	All	5	7.7×10^{-3}	2.6	1.0×10^{-2}	6×10^{-3}	1.5×10^{-3}	1.7×10^{-2}	3
	Leaves	All	1			2.5×10^{-1}				1
Grasses	Stems and shoots	All	17	3.1×10^{-1}	1.8	3.6×10^{-1}	2.2×10^{-1}	1.1×10^{-1}	1.0	2
Pasture (Natural)	Stems and shoots	All	34	9.2×10^{-2}	4.8	2.3×10^{-1}	2.9×10^{-1}	2.2×10^{-3}	1.0	7
Fodder Leguminous	Stems and shoots	All	1			1.6×10^{-2}				1

TABLE 6. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Po

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min.	Max	#ref
All		All	57	5.6×10^{-3}	1.3×10^{1}	5.6×10^{-2}	1.6×10^{-1}	1.6×10^{-5}	1.0	6
Cereals	Grain	All	2			2.4×10^{-4}	2.6×10^{-5}	2.2×10^{-4}	2.6×10^{-4}	1
Maize	Grain	All	2			2.4×10^{-4}	3.2×10^{-4}	1.8×10^{-5}	4.7×10^{-4}	1
Leafy Vegetables	Leaves	All	12	7.4×10^{-3}	6.9	1.9×10^{-2}	1.7×10^{-2}			3
Non-leafy Vegetables	Stems and shoots	All	2			1.9×10^{-4}	2.5×10^{-4}	1.6×10^{-5}	3.7×10^{-4}	1
Leguminous Vegetables	Pods	All	4	2.7×10^{-4}	3.9	4.8×10^{-4}	4.6×10^{-4}	6.0×10^{-5}	1.0×10^{-3}	4
Root Crops	Roots	All	10	5.80×10^{-3}	4.3	1.2×10^{-2}	1.7×10^{-2}	2.4×10^{-4}	4.9×10^{-2}	4
	Stems and shoots	All	2			7.7×10^{-2}	2.7×10^{-2}	5.8×10^{-2}	9.7×10^{-2}	1
Tubers	Tubers	All	9	2.7×10^{-3}	5.8	8.0×10^{-3}	1.2×10^{-2}	1.4×10^{-4}	$3.4 \times 10^{-}2$	3
Pasture (Natural)	Stems and shoots	All	10	1.2×10^{-1}	4.2	2.6×10^{-1}	3.2×10^{-1}	2.2×10^{-2}	1.0	2
Fodder Leguminous	Stems and shoots	All	2			1.1×10^{-2}	2.0	2.6×10^{-5}	2.2×10^{-4}	1

TABLE 7. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR K

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	2			7.4×10^{-1}	3.5×10^{-3}	7.3×10^{-1}	7.4×10^{-1}	1
Leafy Vegetables	Leaves	All	2			1.3	1.2×10^{-1}	1.2	1.3	1
Pasture	Stems and shoots	All	1			7.3×10^{-1}				1
Cereals	Stems and shoots	All	2			1.1	2.0×10^{-1}	9.3×10^{-1}	1.2	1

A substantial range of variation is observed: the SD is generally more than half the value of the mean and the range between minimum and maximum values is generally more than two orders of magnitude. Table 2 gives the F_v-U estimates for selected crop groups. A generic F_v-U value of 2.15 10^{-2} (GSD 9.4) kg kg^{-1} was derived. Fodder, Pasture and Grasses, and Herbs showed the highest F_v-U (2.3-6.5 10^{-2} kg kg^{-1}), and Legumes, Cereals, and Tubers the lowest (2.2-5 10^{-3} kg kg^{-1}).

Differences in F_v-U between crop groups overall is a factor 10. Typical ranges in F_v-U within a crop group are 1 to 5 orders of magnitude. The highest F_v -U values were found for Organic soils, followed by Sand and Loam soils, and the lowest for Clay soils. Experimental and climatic conditions and contamination history did not significantly affect F_v-U.

The generic GM for F_v-Th derived is 3.4 10^{-3} (GSD 15) kg kg^{-1} (6 orders of magnitude range) (Table 3) and is, on average, 10-fold lower than the generic GM for the F_v of the other natural radionuclides in this study, except for Po.

The highest F_v-Th values were found for Pasture and Grasses (5.7 10^{-2} kg kg^{-1}), followed by Fodder and Fruits (~5.0 10^{-3} kg kg^{-1}), while the lowest F_v-Th (2.0 10^{-4} kg kg^{-1}) was observed for Tubers,. Variation within a crop group is substantial, with ranges covering 2 to 4 orders of magnitude. Very few significant differences between crop groups were observed in derived F_v-Th values. F_v-Th to Cereal Straw is a factor of two higher than to Cereal Grains. Evaluation of the overall effect of soil texture on the Th availability for plant uptake showed that Organic soils had the lowest F_v-Th, and that F_v-Th for Sand, Loam and Clay soils did not differ substantially each from other.

The derived generic F_v-Ra is 4×10^{-2} (GSD 9) kg kg^{-1} , but the range in observed values is 7 orders of magnitude (Table 4). Pasture, Grasses, Leafy Vegetables, Root Crops, Fodder, and Herbs showed the highest F_v-Ra (6×10^{-2} – 10^{-1} kg kg^{-1}), Cereals, Non-leafy Vegetables, Legumes, Tubers, and Fruits showed the lowest (9×10^{-3} – 2×10^{-2} kg kg^{-1}).

Variation within a crop group is 1 to 5 orders of magnitude, and significant differences in F_v values between crop groups were rarely observed. F_v-Ra to Cereal Straw is a factor of two lower than to Cereal Grains. Evaluation of the overall effect of soil texture on the Ra availability for plant uptake, showed that Clay and Organic soils had the lowest F_v–Ra, and Sand and Loam soils the highest, yet the difference is only 4-fold. No significant effect of experimental conditions, contamination history, or climate could be found on F_v–Ra, based on the analysis of the available data.

Table 5 shows F_v-Pb for the different crop groups. The overall GM for the F_v-Pb is 2.0×10^{-2} (GSD 14) kg kg^{-1} and the range covers 5 orders of magnitude. For most crop groups a fair number of observations were recorded, except for Non-leafy Vegetables, Fruits, and Fodder Leguminous. F_v-Pb was highest for Pasture (1.4×10^{-1} kg kg^{-1}), followed by Leafy Vegetables (8.0×10^{-2} kg kg^{-1}) and Fodder (2.5×10^{-2} kg kg^{-1}), and was lowest for Tubers (1.5×10^{-3} kg kg^{-1}).

Within a plant group variation is low (factor of 10 or less) to substantial (4 orders of magnitude). Therefore, very few F_v-Pb are significantly different between crop groups. F_v-Pb to Cereal Grain was about a factor of 2 lower than to the Cereal Straw. F_v-Pb are highest for Sand and Clay soils. Variation within a soil category is three orders of magnitude. For Organic soils, only two observations were recorded

The generic F_v value for Po was 5.6×10^{-3} (GSD 13) kg kg^{-1}, and values entered covered 5 orders of magnitude (Table 6). For several crop groups, data were very scarce (Cereals, Leguminous-Vegetables, Leguminous Fodder) or non-existant (Non-Leafy Vegetables, Fruit). Cereals and Leguminous Vegetables have generally low transfer factors (10^{-4} kg kg^{-1}) compared to the other crops. The reason can be physical–the edible parts being protected from Po deposition–or physiological, as observed for many radio-contaminants. Transfer from vegetative mass to seeds is limited. Other crops have F_v in order of 10^{-3} kg kg^{-1}. The highest F_v values were found for Pasture.

There is a clear lack of adequate F_v-Po data, and sources reporting on F_v-Po data are very few. For most crop groups, Po-TF data are scarce or even non-existing. For the other natural radionuclides, there is a very limited number of transfer data to Fruits and Leguminous Fodder. Soil-to-plant transfer data to Non-Leafy Vegetables and Tubers are also rather scarce.

Large variability exists among F_v data even at the crop group level. As a result, only very few estimates were significantly different between crop groups. Additional data acquisition may contribute to a decrease in the actually recorded variability.

A striking observation is that the majority of soil-to-plant F_v data were reported without information on soil properties. Only about 50% of the entries contained information on soil type. Information on pH, CEC, or OM was generally even less frequently recorded. Generally, F_v was highest on coarse textured soils and lowest on fine textured and organic soils, but F_v values derived per texture class were seldom significantly different. Soil characteristics and environmental conditions will affect the transfer processes, and it would be an advantage if the mechanisms of transfer could be understood and modelled. At this stage, with little information available, no mechanistic prediction of the soil-to-plant F_v based on soil properties can be made.

3. DERIVED TRANSFER FACTOR VALUES FOR ARTIFICIAL RADIONUCLIDES

The database for artificial radionuclides, derived from the database on literature sources, included 9390 records for 37 radionuclides.

Fig. 2 summarizes information on the number of records relevant to the evaluation of F_v values for the 29 radionuclides that have the most complete information in the database. For the rest of the radionuclides, only a few values are available.

The largest numbers of records relate to ^{90}Sr and ^{137}Cs, and such records account for about 50% of the total number of entries.

Figs. 3-4 show availability of the information with respect to individual soil and plant groups.

The database includes information for 13 of the plant groups. Most data were collected for Cereals, Vegetables, and Pasture (Figure 4). With respect to soil groups, most observations were found for Sand and Loam soils (Figure 3). Information for Organic soils was rather limited.

Derived transfer factor values for artificial radionuclides are given in Tables 8-40.

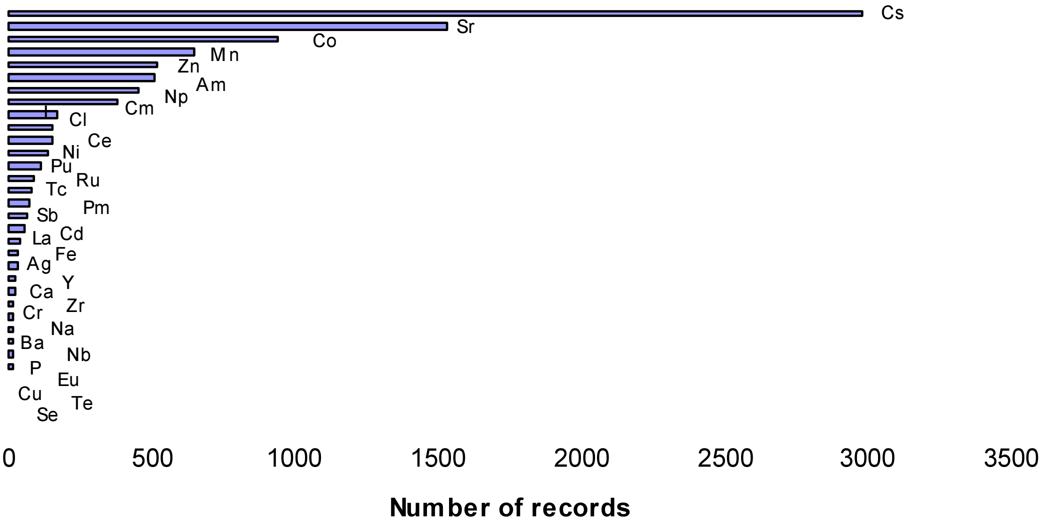

FIG. 2. Distribution of entries used for evaluation of transfer factor values for artificial radionuclides.

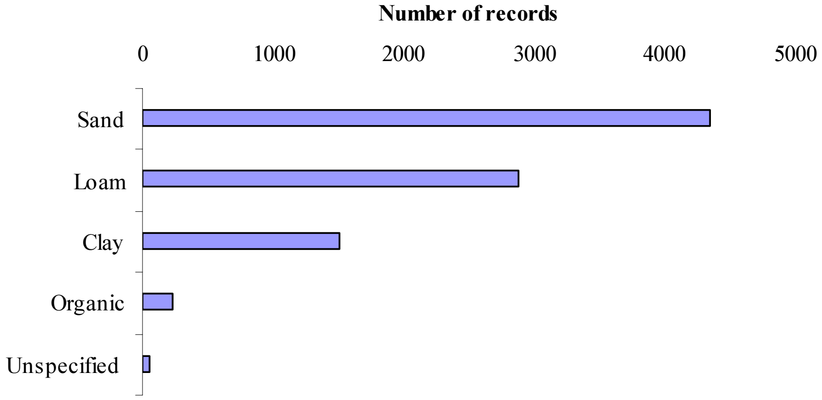

FIG. 3. Numbers of records relating to the various soil groups in the database used to define F_v values.

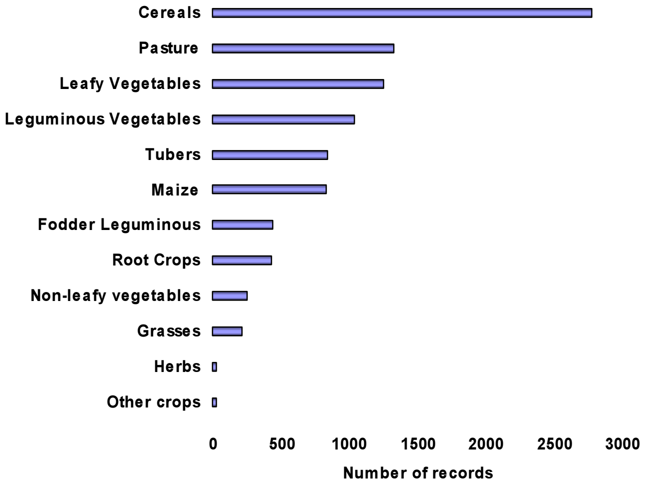

FIG 4. Numbers of records relating to the various plant groups in the database used to define F_v values.

TABLE 8. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ag

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Leafy Vegetables	Leaves	All	5	1.8×10^{-4}	3.3	3.6×10^{-4}	5.3×10^{-4}	5.9×10^{-5}	1.3×10^{-3}	3
		Sand	2			1.7×10^{-4}		9.6×10^{-5}	2.5×10^{-4}	1
		Loam	3	2.0×10^{-4}	5.0	4.9×10^{-4}	7.0×10^{-4}	5.9×10^{-5}	1.3×10^{-3}	3
Non-leafy Vegetables	Fruits	All	5	6.4×10^{-4}	2.3	8.5×10^{-4}	7.2×10^{-4}	2.5×10^{-4}	2.0×10^{-3}	3
		Sand	2			1.6×10^{-3}	6.4×10^{-4}	1.1×10^{-3}	2.0×10^{-3}	1
		Loam	3	3.7×10^{-4}	1.4	3.8×10^{-4}	1.2×10^{-4}	2.5×10^{-4}	4.7×10^{-4}	3
Root Crops	Root	All	6	1.3×10^{-3}	2.0	1.6×10^{-3}	1.2×10^{-3}	5.7×10^{-4}	3.9×10^{-3}	3
		Sand	3	1.7×10^{-3}	2.4	2.1×10^{-3}	1.6×10^{-3}	6.8×10^{-4}	3.9×10^{-3}	1
		Loam	3	1.0×10^{-3}	1.7	1.1×10^{-3}	5.7×10^{-4}	5.7×10^{-4}	1.7×10^{-3}	3

TABLE 9. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Am

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	83	2.2×10^{-5}	11.0	1.9×10^{-3}	7.1×10^{-3}	7.4×10^{-7}	3.4×10^{-2}	4
		Sand	66	2.7×10^{-5}	4.1	1.7×10^{-4}	9.8×10^{-4}	2.7×10^{-6}	8.0×10^{-3}	2
		Loam	7	4.0×10^{-4}	$2.0\ 10^{2}$	1.7×10^{-2}	1.6×10^{-2}	1.0×10^{-6}	3.4×10^{-2}	3
		Clay	9	1.6×10^{-5}	$2.5\ 10^{1}$	6.7×10^{-4}	1.4×10^{-3}	7.4×10^{-7}	4.0×10^{-3}	2
		Organic	1			1.5×10^{-7}				1
Leafy Vegetables	Leaves	All	10	2.7×10^{-4}	3.3	4.6×10^{-4}	4.8×10^{-4}	4.0×10^{-5}	1.5×10^{-3}	2
		Sand	5	5.3×10^{-4}	2.7	7.3×10^{-4}	5.6×10^{-4}	1.7×10^{-4}	1.5×10^{-3}	1
		Loam	2	1.6×10^{-4}	4.1	2.4×10^{-4}	2.5×10^{-4}	6.0×10^{-5}	4.1×10^{-4}	1
		Organic	2			1.4×10^{-4}	1.3×10^{-4}	4.0×10^{-5}	2.3×10^{-4}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	9	3.6×10^{-4}	5.0	7.9×10^{-4}	7.3×10^{-4}	2.3×10^{-5}	1.9×10^{-3}	2
		Sand	8	3.9×10^{-4}	5.5	8.6×10^{-4}	7.5×10^{-4}	2.3×10^{-5}	1.9×10^{-3}	1
Leguminous Vegetables	Seeds and pods	All	12	3.8×10^{-4}	2.6	4.8×10^{-4}	2.1×10^{-4}	2.2×10^{-5}	7.9×10^{-4}	1
		Sand	12	3.8×10^{-4}	2.6	4.8×10^{-4}	2.1×10^{-4}	2.2×10^{-5}	7.9×10^{-4}	1
Root Crops	Root	All	4	6.7×10^{-4}	2.4	8.6×10^{-4}	6.2×10^{-4}	2.0×10^{-4}	1.7×10^{-3}	2
		Sand	3	1.0×10^{-3}	1.6	1.1×10^{-3}	5.4×10^{-4}	7.3×10^{-4}	1.7×10^{-3}	1

TABLE 9. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Am (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM.	GSD	AM	SD	Min	Max	#ref
Tubers	Tubers	All	78	2.1×10^{-4}	6.0	1.4×10^{-3}	4.6×10^{-3}	1.1×10^{-5}	3.4×10^{-2}	2
		Sand	65	2.1×10^{-4}	5.5	1.5×10^{-3}	5.0×10^{-3}	1.1×10^{-5}	3.4×10^{-2}	1
		Loam	8	1.5×10^{-4}	9.0	9.1×10^{-4}	1.7×10^{-3}	1.1×10^{-5}	4.7×10^{-3}	1
		Clay	2			3.3×10^{-3}	4.5×10^{-3}	9.0×10^{-5}	6.5×10^{-3}	1
		Organic	2			8.1×10^{-4}	1.1×10^{-3}	2.1×10^{-5}	1.6×10^{-3}	1
Grasses	Stems and shoots	All	7	3.3×10^{-2}	9.0	9.3×10^{-2}	1.0×10^{-1}	4.2×10^{-4}	2.6×10^{-1}	1
		Sand	7	3.3×10^{-2}	9.0	9.3×10^{-2}	1.0×10^{-1}	4.2×10^{-4}	2.6×10^{-1}	1
Fodder Leguminous	Stems and shoots	All	20	6.5×10^{-4}	2.7	1.0×10^{-3}	9.4×10^{-4}	1.8×10^{-4}	3.1×10^{-3}	2
		Sand	12	9.9×10^{-4}	2.5	1.3×10^{-3}	8.2×10^{-4}	1.9×10^{-4}	2.9×10^{-3}	2
		Loam	1			3.1×10^{-3}				1
		Clay	7	2.5×10^{-4}	1.4	2.6×10^{-4}	1.0×10^{-4}	1.8×10^{-4}	4.8×10^{-4}	1
Pasture	Stems and shoots	All	27	1.5×10^{-3}	4.1	4.6×10^{-3}	1.1×10^{-2}	1.0×10^{-4}	4.8×10^{-2}	4
		Sand	10	5.1×10^{-3}	2.6	7.9×10^{-3}	8.7×10^{-3}	1.3×10^{-3}	2.9×10^{-2}	2
		Loam	11	1.0×10^{-3}	5.0	2.6×10^{-3}	$5.8\ 10^{-3}$	5.3×10^{-4}	2.0×10^{-2}	1
		Clay	5	1.7×10^{-4}	2.2	2.0×10^{-4}	1.0×10^{-4}	1.0×10^{-4}	3.0×10^{-4}	2
Cereals	Stems and shoots	All	5	7.9×10^{-5}	81.5	1.2×10^{-2}	2.6×10^{-2}	3.0×10^{-7}	5.8×10^{-2}	2
		Sand	1			5.8×10^{-2}				1
		Clay	4	1.5×10^{-5}	14.9	4.8×10^{-5}	4.3×10^{-5}	3.0×10^{-7}	1.0×10^{-4}	2
Maize	Stems and shoots	All	64	2.6×10^{-4}	5.5	9.6×10^{-4}	2.1×10^{-3}	1.1×10^{-5}	1.2×10^{-2}	1
		Sand	64	2.6×10^{-4}	5.5	9.6×10^{-4}	2.1×10^{-3}	1.1×10^{-5}	1.2×10^{-2}	1

TABLE 10. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ba

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			1.0×10^{-3}				1
Leafy Vegetables	Leaves	All	1			5.0×10^{-3}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			5.0×10^{-3}				1
Root Crops	Roots	All	1			5.0×10^{-3}				1
Tubers	Tuber	All	1			5.0×10^{-3}				1
Grasses	Stems and shoots	All	3	1.8	1.9	2.0	1.3	1.2	3.6	1
		Sand	1			1.3				1
		Loam	1			1.2				1
		Organic	1			3.6				1
Fodder Leguminous	Stems and shoots	All	3	0.6	3.0	9.1×10^{-1}	1.0	2.8×10^{-1}	2.1	1
		Sand	1			3.7×10^{-1}				1
		Loam	1			2.8×10^{-1}				1
		Organic	1			2.1				1

TABLE 11. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Be

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Pasture	Stems and shoots	All	1			$4.2\ 10^{-1}$				1

TABLE 12. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ca

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Leguminous Vegetables	Stems and shoots	All	6	2.0×10^{1}	3.7	3.6×10^{1}	3.3×10^{1}	5.9	7.5×10^{1}	1
		Sand	3	6.5×10^{1}	1.2	6.6×10^{1}	1.1×10^{1}	5.3×10^{1}	7.5×10^{1}	1
		Loam	3	6.2	1.0	6.2	3.0×10^{-1}	5.9	6.5	1
Cereals	Stems and shoots	All	6	8.7	3.7	1.6×10^{1}	1.6×10^{1}	2.3	3.8×10^{1}	1
		Sand	3	3.0×10^{1}	1.2	3.0×10^{1}	7.0	2.5×10^{1}	3.8×10^{1}	1
		Loam	3	2.6	1.1	2.6	3.0×10^{-1}	2.3	2.9	1

TABLE 13. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cd

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	11	8.8×10^{-1}	2.7	1.3	9.9×10^{-1}	1.4×10^{-1}	2. × 9	4
		Sand	5	1.2	2.1	1.4	8.8×10^{-1}	4.8×10^{-1}	2.5	3
		Loam	4	1.3	2.2	1.6	1.1	5.6×10^{-1}	2.9	2
		Clay	2			2.1×10^{-1}	9.9×10^{-2}	1.4×10^{-1}	2.8×10^{-1}	3
Maize	Grain	All	1			5.0×10^{-2}				1
Leguminous Vegetables	Seeds, Pod	All	2			2.7×10^{-1}	2.7×10^{-1}	8.0×10^{-2}	4.6×10^{-1}	2
		Sand	1			4.6×10^{-1}				1
		Clay	1			8.0×10^{-2}				1
Tubers	Tubers	All	1			1.5				1
Cereals	Stems and shoots	All	24	2.1	2.2	2.6	1.6	1.9×10^{-1}	5.4	4
		Sand	8	3.2	1.4	3.4	1.2	2.2	5.1	4
		Loam	12	2.2	2.1	2.7	1.7	5.8×10^{-1}	5.4	3
		Clay	4	7.1×10^{-1}	2.4	8.8×10^{-1}	4.7×10^{-1}	1.9×10^{-1}	1.3	4
Maize	Stems and shoots	All	2			1.3	1.3	3.5×10^{-1}	2.2	2
		Sand	1			2.2				1
		Clay	1			3.5×10^{-1}				1

TABLE 14. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ce

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	20	3.1×10^{-3}	3.7	5.9×10^{-3}	6.0×10^{-3}	2.4×10^{-4}	2.0×10^{-2}	5
		Sand	5	1.1×10^{-2}	2.6	1.4×10^{-2}	7.0×10^{-3}	2.0×10^{-3}	2.0×10^{-2}	3
		Loam	7	2.8×10^{-3}	3.3	4.0×10^{-3}	2.4×10^{-3}	2.4×10^{-4}	7.0×10^{-3}	3
		Clay	6	1.6×10^{-3}	4.1	2.9×10^{-3}	2.5×10^{-3}	2.8×10^{-4}	6.0×10^{-3}	3
		Organic	1			8.0×10^{-4}				1
Leafy Vegetables	Leaves	All	1			6.0×10^{-3}				1
Leguminous Vegetables	Seeds, Pod	All	2			1.3×10^{-2}	9.9×10^{-3}	6.0×10^{-3}	2.0×10^{-2}	2
		Sand	1			2.0×10^{-2}				1
		Loam	1			6.0×10^{-3}				1
Root Crops	Roots	All	1			6.0×10^{-3}				1
Tubers	Tubers	All	1			4.0×10^{-3}				1
Grasses	Stems and shoots	All	2			2.0×10^{-2}	1.4×10^{-2}	1.0×10^{-2}	3.0×10^{-2}	2
		Loam	2			2.0×10^{-2}	1.4×10^{-2}	1.0×10^{-2}	3.0×10^{-2}	2
Fodder Leguminous	Stems and shoots	All	4	8.0×10^{-3}	2.1	9.8×10^{-3}	7.3×10^{-3}	4.0×10^{-3}	2.0×10^{-2}	1
		Loam	4	8.0×10^{-3}	2.1	9.810^{-3}	7.3×10^{-3}	4.0×10^{-3}	2.0×10^{-2}	1
Pasture	Stems and shoots	All	10	3.7×10^{-1}	5.0	8.4×10^{-1}	1.0	2.0×10^{-2}	3.5	2
		Sand	1			9.6×10^{-1}				1
		Loam	4	4.0×10^{-1}	3.3	6.2×10^{-1}	5.5×10^{-1}	1.2×10^{-1}	1.2	2
		Clay	3	2.9×10^{-1}	3.3	5.0×10^{-1}	6.1×10^{-1}	1.4×10^{-1}	1.2	2
		Organic	1			3.5				1
Cereals	Stems and shoots	All	13	3.9×10^{-2}	5.5	1.3×10^{-1}	2.1×10^{-1}	3.0×10^{-3}	6.8×10^{-1}	5
		Sand	4	2.8×10^{-1}	2.5	3.6×10^{-1}	2.5×10^{-1}	8.00×10^{-2}	6.8×10^{-1}	3
		Loam	6	7.0×10^{-3}	2.4	2.8×10^{-2}	2.0×10^{-2}	7.0×10^{-3}	6.0×10^{-2}	3
		Clay	3	3.0×10^{-3}	3.0	1.3×10^{-2}	1.2×10^{-2}	3.0×10^{-3}	2.7×10^{-2}	3

TABLE 15. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cm

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	67	2.3×10^{-5}	3.3	4.1×10^{-5}	4.3×10^{-5}	1.4×10^{-6}	2.0×10^{-4}	2
		Sand	66	2.3×10^{-5}	3.3	4.2×10^{-5}	4.9×10^{-5}	1.4×10^{-6}	2.9×10^{-4}	1
Leafy Vegetables	Leaves	All	7	1.4×10^{-3}	4.5	2.8×10^{-3}	3.0×10^{-3}	2.0×10^{-4}	8.1×10^{-3}	2
		Sand	6	1.9×10^{-3}	3.7	3.3×10^{-3}	3.0×10^{-3}	3.0×10^{-4}	8.1×10^{-3}	1
Non-leafy Vegetables	Fruits	All	8	3.2×10^{-4}	4.5	6.4×10^{-4}	6.0×10^{-4}	3.6×10^{-5}	1.4×10^{-3}	2
		Sand	8	2.9×10^{-4}	4.5	6.3×10^{-4}	6.1×10^{-4}	3.6×10^{-5}	1.4×10^{-3}	1
Leguminous Vegetables	Seeds, Pod	All	17	7.5×10^{-4}	1.5	8.2×10^{-4}	3.5×10^{-4}	4.2×10^{-4}	1.6×10^{-3}	1
		Sand	17	7.5×10^{-4}	1.5	8.2×10^{-4}	3.5×10^{-4}	4.2×10^{-4}	1.6×10^{-3}	1
Root Crops	Roots	All	6	8.5×10^{-4}	3.0	1.3×10^{-3}	1.4×10^{-3}	2.0×10^{-4}	3.9×10^{-3}	2
		Sand	5	1.1×10^{-3}	2.5	1.6×10^{-3}	1.4×10^{-3}	4.1×10^{-4}	3.9×10^{-3}	1
Tubers	Tubers	All	66	1.5×10^{-4}	3.7	3.4×10^{-4}	4.2×10^{-4}	1.1×10^{-5}	2.1×10^{-3}	2
		Sand	65	1.5×10^{-4}	4.1	3.4×10^{-4}	4.2×10^{-4}	1.1×10^{-5}	2.1×10^{-3}	1
Pasture	Stems and shoots	All	17	1.0×10^{-3}	2.4	1.4×10^{-3}	1.0×10^{-3}	1.0×10^{-4}	3.6×10^{-3}	2
		Sand	6	2.1×10^{-3}	1.7	2.3×10^{-3}	1.1×10^{-3}	1.1×10^{-3}	3.6×10^{-3}	1
		Loam	8	8.3×10^{-4}	1.4	8.7×10^{-4}	2.9×10^{-4}	4.6×10^{-4}	1.4×10^{-3}	1
		Clay	2			2.5×10^{-4}	2.1×10^{-4}	1.0×10^{-4}	4.0×10^{-4}	1
Maize	Stems and shoots	All	71	2.0×10^{-4}	5.0	5.2×10^{-4}	7.8×10^{-4}	5.7×10^{-6}	4.4×10^{-3}	1
		Sand	71	2.0×10^{-4}	5.0	5.2×10^{-4}	7.8×10^{-4}	5.7×10^{-6}	4.4×10^{-3}	1

TABLE 16. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Co

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	61	8.5×10^{-3}	5.5	3.7×10^{-2}	1.0×10^{-1}	4.0×10^{-4}	7.2×10^{-1}	7
		Sand	30	1.4×10^{-2}	6.0	6.4×10^{-2}	1.4×10^{-1}	1.0×10^{-3}	7.2×10^{-1}	5
		Loam	16	4.9×10^{-3}	5.0	1.3×10^{-2}	1.8×10^{-2}	4.0×10^{-4}	6.0×10^{-2}	4
		Clay	12	5.4×10^{-3}	4.1	1.1×10^{-2}	1.1×10^{-2}	8.0×10^{-4}	3.0×10^{-2}	5
		Organic	2			3.4×10^{-3}	2.3×10^{-3}	1.7×10^{-3}	5.0×10^{-3}	1
Maize	Grain	All	40	1.0×10^{-2}	4.1	4.4×10^{-2}	1.1×10^{-1}	9.0×10^{-4}	5.6×10^{-1}	4
		Sand	26	1.5×10^{-2}	4.5	6.3×10^{-2}	1.4×10^{-1}	3.0×10^{-3}	5.6×10^{-1}	3
		Loam	10	7.2×10^{-3}	1.9	8.4×10^{-3}	4.4×10^{-3}	1.6×10^{-3}	1.6×10^{-2}	1
		Clay	4	2.0×10^{-3}	2.0	2.4×10^{-3}	1.5×10^{-3}	9.0×10^{-4}	4.0×10^{-3}	1
Leafy Vegetables	Leaves	All	185	1.7×10^{-1}	2.7	2.5×10^{-1}	2.2×10^{-1}	1.3×10^{-2}	1.0	6
		Sand	66	2.5×10^{-1}	2.4	3.4×10^{-1}	2.4×10^{-1}	1.7×10^{-2}	1.0	4
		Loam	85	1.5×10^{-1}	2.5	2.2×10^{-1}	1.9×10^{-1}	1.8×10^{-2}	8.6×10^{-1}	2
		Clay	33	9.7×10^{-2}	3.0	1.8×10^{-1}	2.1×10^{-1}	1.3×10^{-2}	6.9×10^{-1}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	7	1.4×10^{-1}	1.6	1.5×10^{-1}	6.0×10^{-2}	5.7×10^{-2}	2.3×10^{-1}	5
		Sand	2			1.4×10^{-1}	1.2×10^{-1}	5.7×10^{-2}	2.3×10^{-1}	2
		Clay	4	1.6×10^{-1}	1.2	1.6×10^{-1}	3.5×10^{-2}	1.3×10^{-1}	1.9×10^{-1}	3
Leguminous Vegetables	Seeds and pods	All	105	3.6×10^{-2}	2.3	5.5×10^{-2}	7.4×10^{-2}	5.0×10^{-3}	5.0×10^{-1}	7
		Sand	43	5.7×10^{-2}	2.1	8.3×10^{-2}	1.0×10^{-1}	2.2×10^{-2}	5.0×10^{-1}	4
		Loam	40	2.8×10^{-2}	2.3	4.0×10^{-2}	4.4×10^{-2}	5.0×10^{-3}	2.2×10^{-1}	1
		Clay	22	2.2×10^{-2}	1.9	2.6×10^{-2}	1.4×10^{-2}	6.0×10^{-3}	5.3×10^{-2}	3
Root Crops	Roots	All	14	1.1×10^{-1}	2.2	1.5×10^{-1}	1.8×10^{-1}	4.7×10^{-2}	7.2×10^{-1}	7
		Sand	7	1.4×10^{-1}	2.7	2.3×10^{-1}	2.5×10^{-1}	5.5×10^{-2}	7.2×10^{-1}	4
		Loam	4	6.5×10^{-2}	1.4	6.8×10^{-2}	2.3×10^{-2}	4.7×10^{-2}	9.9×10^{-2}	2
		Clay	2			1.0×10^{-1}	0.0	1.0×10^{-1}	1.0×10^{-1}	2
Tubers	Tubers	All	56	5.4×10^{-2}	3.0	1.1×10^{-1}	1.5×10^{-1}	1.0×10^{-2}	6.7×10^{-1}	4
		Sand	39	8.0×10^{-2}	3.0	1.4×10^{-1}	1.6×10^{-1}	1.1×10^{-2}	6.7×10^{-1}	3
		Loam	11	2.1×10^{-2}	1.7	2.4×10^{-2}	1.5×10^{-2}	1.0×10^{-2}	6.3×10^{-2}	2
		Clay	5	2.2×10^{-2}	2.0	2.8×10^{-2}	2.4×10^{-2}	1.2×10^{-2}	7.0×10^{-2}	1

TABLE 16. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Co (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Grasses	Stems and shoots	All	4	7.7×10^{-2}	2.2	9.5×10^{-2}	6.6×10^{-2}	4.0×10^{-2}	1.7×10^{-1}	3
		Sand	1			1.3×10^{-1}				1
		Loam	1			1.7×10^{-1}				1
		Clay	2			4.0×10^{-2}		4.0×10^{-2}	4.0×10^{-2}	2
Fodder Leguminous	Stems and shoots	All	38	6.6×10^{-2}	3.3	1.1×10^{-1}	1.3×10^{-1}	1.0×10^{-3}	7.2×10^{-1}	3
		Sand	15	1.4×10^{-1}	2.2	1.9×10^{-1}	1.7×10^{-1}	4.4×10^{-2}	7.2×10^{-1}	1
		Loam	10	2.9×10^{-2}	6.0	6.3×10^{-2}	5.6×10^{-2}	1.0×10^{-3}	1.8×10^{-1}	2
		Clay	9	6.9×10^{-2}	1.5	7.3×10^{-2}	2.7×10^{-2}	3.7×10^{-2}	1.2×10^{-1}	2
		Organic	4	2.8×10^{-2}	1.6	3.1×10^{-2}	1.5×10^{-2}	1.9×10^{-2}	5.3×10^{-2}	1
Pasture	Stems and shoots	All	88	4.5×10^{-2}	3.7	1.1×10^{-1}	1.7×10^{-1}	2.1×10^{-3}	8.4×10^{-1}	3
		Sand	49	8.6×10^{-2}	3.0	1.6×10^{-1}	2.1×10^{-1}	1.4×10^{-2}	8.4×10^{-1}	2
		Loam	36	1.7×10^{-2}	2.7	2.7×10^{-2}	2.9×10^{-2}	2.1×10^{-3}	1.3×10^{-1}	2
		Clay	2			1.3×10^{-1}	6.4×10^{-2}	8.0×10^{-2}	1.7×10^{-1}	1
Cereals	Stems and shoots	All	27	1.1×10^{-1}	5.0	5.3×10^{-1}	1.3	1.0×10^{-2}	4.9	5
		Sand	8	5.8×10^{-1}	4.1	1.6	2.1	1.1×10^{-1}	4.9	3
		Loam	12	6.4×10^{-2}	3.3	1.1×10^{-1}	9.7×10^{-2}	1.0×10^{-2}	2.9×10^{-1}	4
		Clay	7	3.6×10^{-2}	1.4	3.7×10^{-2}	1.1×10^{-2}	2.0×10^{-2}	5.0×10^{-2}	4
Maize	Stems and shoots	All	37	3.5×10^{-2}	2.2	4.6×10^{-2}	3.7×10^{-2}	6.0×10^{-3}	2.0×10^{-1}	2
		Sand	36	3.4×10^{-2}	2.2	4.6×10^{-2}	3.7×10^{-2}	6.0×10^{-3}	2.0×10^{-1}	1
		Loam	1			5.0×10^{-2}				1
Root Crops	Leaves	All	2			2.4×10^{-1}		2.4×10^{-1}	2.4×10^{-1}	5
		Clay	2			2.4×10^{-1}		2.4×10^{-1}	2.4×10^{-1}	2

TABLE 17. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cr

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			2.0×10^{-4}				1
Leafy Vegetables	Leaves	All	1			1.0×10^{-3}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			1.0×10^{-3}				1
Root Crops	Roots	All	1			1.0×10^{-3}				1
Tubers	Tubers	All	1			5.0×10^{-4}				1
Pasture	Stems and shoots	All	1			2.0×10^{-3}				1

TABLE 18. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cs

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	470	2.9×10^{-2}	4.1	7.6×10^{-2}	1.5×10^{-1}	2.0×10^{-4}	9.0×10^{-1}	31
		Sand	156	3.9×10^{-2}	3.3	8.0×10^{-2}	1.2×10^{-1}	2.0×10^{-3}	6.6×10^{-1}	26
		Loam	158	2.0×10^{-2}	4.1	4.3×10^{-2}	5.0×10^{-2}	8.0×10^{-4}	2.0×10^{-1}	19
		Clay	110	1.1×10^{-2}	2.7	1.8×10^{-2}	1.7×10^{-2}	2.0×10^{-4}	9.0×10^{-2}	20
		Organic	28	4.3×10^{-2}	2.7	8.3×10^{-2}	1.56×10^{-1}	1.0×10^{-2}	7.3×10^{-1}	8
Maize	Grain	All	67	3.3×10^{-2}	3.0	5.5×10^{-2}	5.7×10^{-2}	3.0×10^{-3}	2.6×10^{-1}	5
		Sand	47	4.9×10^{-2}	2.4	6.9×10^{-2}	6.2×10^{-2}	8.0×10^{-3}	2.6×10^{-1}	3
		Loam	14	1.6×10^{-2}	2.7	2.3×10^{-2}	2.0×10^{-2}	3.2×10^{-3}	7.0×10^{-2}	1
		Clay	11	1.2×10^{-2}	3.3	2.2×10^{-2}	2.4×10^{-2}	3.0×10^{-3}	7.0×10^{-2}	3
Leafy Vegetables	Leaves	All	290	6.0×10^{-2}	6.0	1.7×10^{-1}	2.1×10^{-1}	3.0×10^{-4}	9.8×10^{-1}	8
		Sand	96	1.2×10^{-1}	4.1	2.4×10^{-1}	2.4×10^{-1}	2.1×10^{-3}	9.8×10^{-1}	6
		Loam	119	7.4×10^{-2}	5.0	1.8×10^{-1}	2.0×10^{-1}	3.0×10^{-4}	7.3×10^{-1}	5
		Clay	67	1.8×10^{-2}	6.7	8.7×10^{-2}	1.7×10^{-1}	5.0×10^{-4}	7.2×10^{-1}	4
		Organic	7	2.25×10^{-2}	7.4	1.2×10^{-1}	2.0×10^{-1}	4.0×10^{-3}	4.6×10^{-1}	2
Non-leafy Vegetables	Fruits, heads, berries, buds	All	38	2.1×10^{-2}	4.1	7.0×10^{-2}	1.5×10^{-1}	7.0×10^{-4}	7.3×10^{-1}	7
		Sand	17	3.5×10^{-2}	4.1	1.1×10^{-1}	2.1×10^{-1}	1.2×10^{-2}	7.3×10^{-1}	6
		Loam	5	3.3×10^{-2}	5.5	9.4×10^{-2}	1.3×10^{-1}	6.3×10^{-3}	3.0×10^{-1}	2
		Clay	14	9.1×10^{-3}	2.2	1.1×10^{-2}	3.7×10^{-3}	7.0×10^{-4}	1.6×10^{-2}	5
Leguminous Vegetables	Seeds and pods	All	126	4.0×10^{-2}	3.7	8.5×10^{-2}	1.1×10^{-1}	1.0×10^{-3}	7.1×10^{-1}	10
		Sand	66	8.7×10^{-2}	2.5	1.3×10^{-1}	1.3×10^{-1}	3.5×10^{-3}	7.1×10^{-1}	5
		Loam	42	2.0×10^{-2}	3.3	4.3×10^{-2}	7.3×10^{-2}	1.0×10^{-3}	4.2×10^{-1}	4
		Clay	18	1.3×10^{-2}	3.0	2.2×10^{-2}	2.2×10^{-2}	2.0×10^{-3}	8.1×10^{-2}	3
Root Crops	Roots	All	81	4.2×10^{-2}	3.0	7.5×10^{-2}	1.1×10^{-1}	1.0×10^{-3}	8.8×10^{-1}	10
		Sand	37	6.2×10^{-2}	2.5	9.1×10^{-2}	8.8×10^{-2}	8.0×10^{-3}	4.0×10^{-1}	10
		Loam	21	3.0×10^{-2}	3.7	5.3×10^{-2}	4.7×10^{-2}	1.0×10^{-3}	1.6×10^{-1}	5
		Clay	17	2.4×10^{-2}	2.2	3.0×10^{-2}	2.0×10^{-2}	5.0×10^{-3}	6.0×10^{-2}	6
		Organic	5	5.9×10^{-2}	5.0	2.0×10^{-1}	3.8×10^{-1}	1.6×10^{-2}	8.8×10^{-1}	3

TABLE 18. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cs (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Tubers	Tubers	All	138	5.6×10^{-2}	3.0	1.0×10^{-1}	1.2×10^{-1}	4.0×10^{-3}	6.0×10^{-1}	9
		Sand	69	9.3×10^{-2}	3.0	1.5×10^{-1}	1.4×10^{-1}	4.0×10^{-3}	6.0×10^{-1}	9
		Loam	40	3.5×10^{-2}	2.3	4.6×10^{-2}	3.3×10^{-2}	4.8×10^{-3}	1.4×10^{-1}	5
		Clay	21	2.5×10^{-2}	2.2	3.2×10^{-2}	2.3×10^{-2}	5.0×10^{-3}	9.0×10^{-2}	6
		Organic	7	5.8×10^{-2}	3.7	1.3×10^{-1}	1.9×10^{-1}	1.610^{-2}	5.4×10^{-1}	4
Grasses	Stems and shoots	All	64	6.3×10^{-2}	36.6	1.3×10^{-1}	1.8×10^{-1}	4.8×10^{-3}	9.9×10^{-1}	8
		Sand	41	8.4×10^{-2}	3.3	1.6×10^{-1}	2.1×10^{-1}	1.0×10^{-2}	9.9×10^{-1}	6
		Loam	10	4.8×10^{-2}	2.3	6.5×10^{-2}	5.8×10^{-2}	1.2×10^{-2}	2.1×10^{-1}	4
		Clay	9	1.2×10^{-2}	2.1	1.6×10^{-2}	1.3×10^{-2}	4.8×10^{-3}	4.3×10^{-2}	3
		Organic	4	2.8×10^{-1}	1.2	2.9×10^{-1}	5.8×10^{-2}	2.1×10^{-1}	3.4×10^{-1}	1
Fodder Leguminous	Stems and shoots	All	85	1.6×10^{-1}	3.3	3.1×10^{-1}	3.7×10^{-1}	1.0×10^{-2}	1.8	8
		Sand	29	2.4×10^{-1}	3.7	4.6×10^{-1}	4.9×10^{-1}	1.8×10^{-2}	1.8	7
		Loam	51	1.5×10^{-1}	3.0	2.5×10^{-1}	2.6×10^{-1}	1.0×10^{-2}	1.2	5
		Clay	4	4.6×10^{-2}	4.1	9.5×10^{-2}	1.4×10^{-1}	1.3×10^{-2}	3.0×10^{-1}	3
Pasture	Stems and shoots	All	401	2.5×10^{-1}	4.1	5.5×10^{-1}	8.2×10^{-1}	1.0×10^{-2}	5.0	16
		Sand	169	2.9×10^{-1}	4.1	6.4×10^{-1}	8.6×10^{-1}	1.0×10^{-2}	4.8	14
		Loam	124	1.9×10^{-1}	4.1	4.3×10^{-1}	5.7×10^{-1}	1.0×10^{-2}	2.6	12
		Clay	75	1.8×10^{-1}	3.7	3.4×10^{-1}	3.3×10^{-1}	1.0×10^{-2}	1.2	11
		Organic	31	7.6×10^{-1}	2.2	1.1	1.2	3.0×10^{-1}	5.0	6
Herbs	Stems and shoots	All	4	6.6×10^{-2}	14.9	7.2×10^{-1}	1.4	4.8×10^{-3}	2.8	3
Other Crops		All	9	3.1×10^{-1}	4.5	6.8×10^{-1}	8.0×10^{-1}	3.6×10^{-2}	2.2	4
Cereals	Stems and shoots	All	130	1.5×10^{-1}	5.0	3.6×10^{-1}	7.2×10^{-1}	4.3×10^{-3}	3.7	22
		Sand	35	2.1×10^{-1}	3.3	4.1×10^{-1}	5.0×10^{-1}	4.1×10^{-2}	1.9	8
		Loam	36	1.1×10^{-1}	4.5	2.7×10^{-1}	3.9×10^{-1}	6.5×10^{-3}	1.5	9
		Clay	37	5.6×10^{-2}	3.7	1.1×10^{-1}	1.2×10^{-1}	4.3×10^{-3}	5.3×10^{-1}	9
Maize	Stems and shoots	All	101	7.3×10^{-2}	3.0	1.2×10^{-1}	1.1×10^{-1}	3.0×10^{-3}	4.9×10^{-1}	9
		Sand	77	1.0×10^{-1}	2.3	1.4×10^{-1}	1.2×10^{-1}	1.4×10^{-2}	4.9×10^{-1}	7
		Loam	10	1.5×10^{-2}	2.5	2.1×10^{-2}	1.8×10^{-2}	3.0×10^{-3}	5.2×10^{-2}	4
		Clay	11	2.2×10^{-2}	2.1	2.9×10^{-2}	2.0×10^{-2}	7.8×10^{-3}	6.0×10^{-2}	5
		Organic	3	1.4×10^{-1}	1.3	1.4×10^{-1}	3.2×10^{-2}	1.0×10^{-1}	1.6×10^{-1}	2

TABLE 18. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES Cs (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Root crops	Leaves	All	12	3.5×10^{-2}	3.0	6.8×10^{-2}	1.2×10^{-1}	6.0×10^{-3}	4.5×10^{-1}	2
		Sand	3	1.1×10^{-1}	3.3	1.9×10^{-1}	2.1×10^{-3}	5.1×10^{-2}	4.5×10^{-1}	1
		Loam	2			2.6×10^{-2}	2.4×10^{-2}	9.0×10^{-3}	4.3×10^{-2}	2
		Clay	7	2.6×10^{-2}	2.1	3.0×10^{-2}	1.4×10^{-2}	6.0×10^{-3}	4.7×10^{-2}	2

TABLE 19. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Fe

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			2.0×10^{-4}				1
Leafy Vegetables	Leaves	All	1			1.0×10^{-3}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			1.0×10^{-3}				1
Leguminous Vegetables	Stems and shoots	All	3	3.7×10^{-1}	1.3	3.8×10^{-1}	9.3×10^{-2}	3.0×10^{-1}	4.8×10^{-1}	2
Root Crops	Roots	All	1			1.0×10^{-3}				1
Tubers	Tubers	All	1			5.0×10^{-4}				1
Pasture	Stems and shoots	All	1			2.0×10^{-3}				1
Cereals	Stems and shoots	All	3	2.9×10^{-1}	1.3	2.9×10^{-1}	6.6×10^{-2}	2.2×10^{-1}	3.5×10^{-1}	2

TABLE 20. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR I

Plant Group	Plant Compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	13	6.3×10^{-4}	2.3	1.4×10^{-3}	2.8×10^{-3}	1.0×10^{-4}	1.1×10^{-2}	5
		Sand	2			5.8×10^{-3}	6.7×10^{-3}	1.0×10^{-3}	1.1×10^{-2}	1
		Loam	5	3.6×10^{-4}	2.5	5.0×10^{-4}	4.3×10^{-4}	1.0×10^{-4}	1.2×10^{-3}	3
		Clay	6	5.7×10^{-4}	2.3	7.5×10^{-4}	5.8×10^{-4}	2.0×10^{-4}	1.6×10^{-3}	4
Leafy Vegetables	Leaves	All	12	6.5×10^{-3}	3.7	1.6×10^{-2}	2.9×10^{-2}	1.1×10^{-3}	1.0×10^{-1}	4
		Sand	1			4.0×10^{-2}				1
		Loam	8	4.1×10^{-3}	1.9	4.7×10^{-3}	2.3×10^{-3}	1.1×10^{-3}	8.0×10^{-3}	3
		Clay	2	4.6×10^{-3}	4.5	7.3×10^{-3}	8.1×10^{-3}	1.6×10^{-3}	1.3×10^{-2}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			1.0×10^{-1}				1
Leguminous Vegetables	Seeds and pods	All	23	8.5×10^{-3}	7.4	3.0×10^{-2}	3.9×10^{-2}	2.0×10^{-4}	1.4×10^{-1}	3
		Sand	2			3.5×10^{-3}	2.8×10^{-4}	3.3×10^{-3}	3.7×10^{-3}	1
		Loam	3	4.4×10^{-4}	1.5	4.7×10^{-4}	2.1×10^{-4}	3.0×10^{-4}	7.0×10^{-4}	2
		Clay	2			2.5×10^{-4}	7.1×10^{-5}	2.0×10^{-4}	3.0×10^{-4}	1
Root Crops	Roots	All	28	7.7×10^{-3}	3.0	1.3×10^{-2}	1.2×10^{-2}	1.4×10^{-3}	4.7×10^{-2}	4
		Sand	9	2.3×10^{-2}	1.5	2.5×10^{-2}	1.1×10^{-2}	1.2×10^{-2}	4.7×10^{-2}	2
		Loam	12	4.7×10^{-3}	2.1	6.0×10^{-3}	4.7×10^{-3}	1.5×10^{-3}	1.6×10^{-2}	3
		Clay	7	4.5×10^{-3}	3.0	7.6×10^{-3}	9.5×10^{-3}	1.4×10^{-3}	2.8×10^{-2}	3
Tubers	Tubers	All	1			1.0×10^{-1}				1
Pasture	Stems and shoots	All	12	3.7×10^{-3}	6.0	4.5×10^{-2}	1.4×10^{-1}	9.0×10^{-4}	5.0×10^{-1}	3
		Sand	9	1.8×10^{-3}	2.1	2.4×10^{-3}	2.5×10^{-3}	9.0×10^{-4}	8.5×10^{-3}	3
		Clay	2			8.7×10^{-3}	4.2×10^{-4}	8.4×10^{-3}	9.0×10^{-3}	1
Cereals	Stems and shoots	All	16	5.2×10^{-2}	3.3	1.1×10^{-1}	1.8×10^{-1}	7.0×10^{-3}	7.5×10^{-1}	3
		Sand	2			4.3×10^{-1}	4.6×10^{-1}	1.1×10^{-1}	7.5×10^{-1}	1
		Loam	7	3.6×10^{-2}	3.3	5.8×10^{-2}	6.5×10^{-2}	7.0×10^{-3}	2.0×10^{-1}	2
		Clay	7	4.5×10^{-2}	2.5	6.3×10^{-2}	6.1×10^{-2}	1.0×10^{-2}	1.9×10^{-1}	2

TABLE 21. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR La

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			2.0×10^{-5}				1
Leafy Vegetables	Leaves	All	7	5.7×10^{-3}	2.7	8.2×10^{-3}	6.0×10^{-3}	1.1×10^{-3}	1.5×10^{-2}	2
Non-leafy Vegetables	Fruits	All	2			6.0×10^{-3}	7.1×10^{-5}	5.9×10^{-3}	6.0×10^{-3}	2
Leguminous Vegetables	Seeds and pods	All	4	4.2×10^{-4}	3.0	6.7×10^{-4}	7.7×10^{-4}	1.6×10^{-4}	1.8×10^{-3}	2
Root crops	Roots	All	9	1.6×10^{-3}	2.7	2.4×10^{-3}	2.1×10^{-3}	4.5×10^{-4}	6.0×10^{-3}	2
Tubers	Tubesr	All	8	3.9×10^{-4}	3.7	8.6×10^{-4}	1.3×10^{-3}	7.0×10^{-5}	4.0×10^{-3}	2
Grasses	Stems and shoots	All	4	1.8×10^{-5}	2.3	2.3×10^{-5}	1.7×10^{-5}	6.0×10^{-6}	4.7×10^{-5}	2
Pastures	Stems and shoots	All	1			2.0×10^{-2}				1
Maize	Stems and shoots	All	2			8.8×10^{-5}	1.6×10^{-5}	7.6×10^{-5}	9.9×10^{-5}	2

TABLE 22. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Mn

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	78	2.8×10^{-1}	3.3	4.9×10^{-1}	5.3×10^{-1}	1.4×10^{-2}	2.7	11
		Sand	33	3.4×10^{-1}	3.3	6.1×10^{-1}	6.5×10^{-1}	1.4×10^{-2}	2.7	7
		Loam	22	2.0×10^{-1}	2.6	3.1×10^{-1}	3.2×10^{-1}	5.6×10^{-2}	1.1	5
		Clay	15	2.2×10^{-1}	4.1	4.0×10^{-1}	3.2×10^{-1}	2.4×10^{-2}	1.0	5
		Organic	6	6.5×10^{-1}	2.1	8.2×10^{-1}	6.1×10^{-1}	2.7×10^{-1}	1.7	1
Maize	Grain	All	19	7.5×10^{-2}	2.1	9.7×10^{-2}	7.7×10^{-2}	1.8×10^{-2}	3.0×10^{-1}	2
		Sand	7	1.4×10^{-1}	1.8	1.6×10^{-1}	9.5×10^{-2}	6.4×10^{-2}	3.0×10^{-1}	2
		Loam	9	5.6×10^{-2}	1.6	6.2×10^{-2}	3.0×10^{-2}	3.1×10^{-2}	1.1×10^{-1}	2
		Clay	3	4.5×10^{-2}	2.4	5.6×10^{-2}	4.1×10^{-2}	1.8×10^{-2}	9.9×10^{-2}	2
Leafy Vegetables	Leaves	All	103	4.1×10^{-1}	2.4	5.8×10^{-1}	5.3×10^{-1}	5.2×10^{-2}	3.0	6
		Sand	35	8.5×10^{-1}	1.8	1.0	6.4×10^{-1}	2.5×10^{-1}	3.0	5
		Loam	49	3.4×10^{-1}	1.9	4.2×10^{-1}	2.6×10^{-1}	7.4×10^{-2}	1.0	5
		Clay	18	1.7×10^{-1}	2.3	2.3×10^{-1}	2.1×10^{-1}	5.2×10^{-2}	7.7×10^{-1}	5
Non-leafy Vegetables	Fruits	All	3	3.1×10^{-1}	4.1	6.0×10^{-1}	7.7×10^{-1}	1.0×10^{-1}	1.5	3
Leguminous Vegetables	Seeds and pod	All	92	2.2×10^{-1}	2.5	3.4×10^{-1}	3.9×10^{-1}	2.2×10^{-2}	2.8	11
		Sand	37	5.0×10^{-1}	1.9	6.2×10^{-1}	4.9×10^{-1}	1.6×10^{-1}	2.8	7
		Loam	43	1.4×10^{-1}	1.6	1.6×10^{-1}	7.2×10^{-2}	5.0×10^{-2}	3.4×10^{-1}	8
		Clay	12	7.8×10^{-2}	2.3	1.2×10^{-1}	1.6×10^{-1}	2.2×10^{-2}	6.0×10^{-1}	5
Root crops	Roots	All	13	4.2×10^{-1}	5.5	1.1	1.4	1.5×10^{-2}	3.9	5
		Sand	8	1.3	2.4	1.7	1.4	3.9×10^{-1}	3.9	5
		Loam	4	6.7×10^{-2}	3.7	1.2×10^{-1}	1.4×10^{-1}	1.5×10^{-2}	3.1×10^{-1}	3
Tubers	Tubers	All	23	4.7×10^{-2}	2.2	6.6×10^{-2}	6.7×10^{-2}	1.2×10^{-2}	3.0×10^{-1}	5
		Sand	9	8.1×10^{-2}	2.2	1.1×10^{-1}	9.1×10^{-2}	2.6×10^{-2}	3.0×10^{-1}	5
		Loam	9	3.6×10^{-2}	1.6	4.0×10^{-2}	2.0×10^{-2}	1.6×10^{-2}	7.6×10^{-2}	5
		Clay	4	2.4×10^{-2}	2.2	3.1×10^{-2}	2.8×10^{-2}	1.2×10^{-2}	7.3×10^{-2}	2

TABLE 22. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Mn (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Fodder Leguminous	Stems and shoots	All	32	1.5	3.3	3.0	3.5	2.4×10^{-1}	1.2×10^{1}	3
		Sand	15	2.7	3.0	4.4	4.1	3.4×10^{-1}	1.2×10^{1}	2
		Loam	6	1.4	4.1	3.0	3.7	3.9×10^{-1}	8.4	2
		Clay	7	6.3×10^{-1}	2.2	7.8×10^{-1}	4.9×10^{-1}	2.4×10^{-1}	1.3	2
		Organic	4	9.2×10^{-1}	2.1	1.156	9.8×10^{-1}	4.9×10^{-1}	2.6	2
Pastures	Stems and shoots	All	83	6.4×10^{-1}	1.9	7.7×10^{-1}	4.8×10^{-1}	1.1×10^{-1}	2.7	3
		Sand	42	9.7×10^{-1}	1.5	1.1	4.6×10^{-1}	4.0×10^{-1}	2.7	2
		Loam	40	4.3×10^{-1}	1.7	4.8×10^{-1}	2.7×10^{-1}	1.1×10^{-1}	1.8	2
Cereals	Stems and shoots	All	30	2.2	4.1	4.7	6.0	2.0×10^{-1}	2.7×10^{1}	4
		Sand	9	9.0	1.9	1.1×10^{1}	7.4	4.8	2.7×10^{1}	4
		Loam	16	1.2	3.0	2.1	2.1	2.0×10^{-1}	6.2	4
		Clay	5	9.8×10^{-1}	4.1	2.2	3.4	2.0×10^{-1}	8.3	4

TABLE 23. SOIL-TO PLANT TRANSFER FACTORS (F_v) FOR Mo

Plant Group	Plant Compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			8.0×10^{-1}				1
Leafy Vegetables	Leaves	All	1			5.1×10^{-1}		2.1×10^{-1}	8.0×10^{-1}	1
Root Crops	Root	All	3			3.2×10^{-1}		2.3×10^{-2}	4.2×10^{-1}	2
Maize	Stems and shoots	All	3			7.3×10^{-1}		1.0×10^{-1}	3.8×10^{1}	2
Fodder Leguminous	Stems and shoots	All	1			5.4×10^{0}				1

TABLE 24. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Na

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			1.0×10^{-2}				1
Leafy Vegetables	Leaves	All	1			3.0×10^{-2}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			3.0×10^{-2}				1
Root Crops	Roots	All	1			3.0×10^{-2}				1
Tubers	Tubers	All	1			3.0×10^{-2}				1
Pastures	Stems and shoots	All	1			1.0×10^{-1}				1

TABLE 25. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Nb

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	2			1.4×10^{-2}		2.0×10^{-3}	2.5×10^{-2}	2
Leafy Vegetables	Leaves	All	2			1.7×10^{-2}		8.0×10^{-3}	2.5×10^{-2}	2
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			8.0×10^{-3}				1
Root Crops	Roots	All	2			1.7×10^{-2}		8.0×10^{-3}	2.5×10^{-2}	2
Tubers	Tubers	All	1			4.0×10^{-3}				1
Pastures	Stems and shoots	All	1			2.0×10^{-2}				1

TABLE 26. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ni

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	44	2.7×10^{-2}	2.7	4.2×10^{-2}	3.9×10^{-2}	3.1×10^{-3}	1.7×10^{-1}	3
		Sand	26	3.7×10^{-2}	2.4	5.2×10^{-2}	4.3×10^{-2}	8.2×10^{-3}	1.7×10^{-1}	3
		Loam	4	7.6×10^{-3}	1.7	8.5×10^{-3}	5.1×10^{-3}	4.9×10^{-3}	1.6×10^{-2}	2
		Clay	9	3.2×10^{-2}	2.4	4.3×10^{-2}	3.0×10^{-2}	6.3×10^{-3}	9.3×10^{-2}	2
		Organic	4	6.1×10^{-3}	1.6	6.6×10^{-3}	2.8×10^{-3}	3.1×10^{-3}	1.0×10^{-2}	1
Grasses	Stems and shoots	All	38	1.7×10^{-1}	2.6	2.3×10^{-1}	1.5×10^{-1}	1.8×10^{-2}	5.8×10^{-1}	4
		Sand	18	2.6×10^{-1}	1.6	2.8×10^{-1}	1.1×10^{-1}	8.2×10^{-2}	5.1×10^{-1}	3
		Loam	5	1.1×10^{-1}	1.6	1.2×10^{-1}	4.7×10^{-2}	5.6×10^{-2}	1.7×10^{-1}	1
		Clay	10	2.5×10^{-1}	1.8	2.9×10^{-1}	1.7×10^{-1}	1.1×10^{-1}	5.8×10^{-1}	1
		Organic	5	2.4×10^{-2}	1.5	2.6×10^{-2}	1.4×10^{-2}	1.8×10^{-2}	5.0×10^{-2}	1
Fodder Leguminous	Stems and shoots	All	27	4.0×10^{-1}	2.5	5.8×10^{-1}	5.2×10^{-1}	7.3×10^{-2}	2.6	1
		Sand	14	6.5×10^{-1}	1.8	7.8×10^{-1}	6.0×10^{-1}	2.8×10^{-1}	2.6	1
		Loam	3	2.5×10^{-1}	2.6	3.4×10^{-1}	3.4×10^{-1}	1.2×10^{-1}	7.4×10^{-1}	1
		Clay	6	3.2×10^{-1}	2.4	4.2×10^{-1}	2.9×10^{-1}	1.1×10^{-1}	7.3×10^{-1}	1
		Organic	4	1.5×10^{-1}	3.3	2.9×10^{-1}	4.1×10^{-1}	7.3×10^{-2}	9.1×10^{-1}	1

TABLE 27. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Np

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	85	2.9×10^{-3}	5.0	7.5×10^{-3}	1.2×10^{-2}	2.3×10^{-5}	7.1×10^{-2}	3
		Sand	79	3.5×10^{-3}	4.1	8.0×10^{-3}	1.2×10^{-2}	2.5×10^{-4}	7.1×10^{-2}	2
		Loam	2			8.5×10^{-4}	7.9×10^{-4}	2.9×10^{-4}	1.4×10^{-3}	2
		Clay	2			3.9×10^{-5}	2.2×10^{-5}	2.3×10^{-5}	5.4×10^{-5}	2
		Organic	1			9.7×10^{-5}				1
Maize	Grain	All	2			4.8×10^{-3}	6.6×10^{-3}	1.0×10^{-4}	9.4×10^{-3}	1
Leafy Vegetables	Leaves	All	5	2.7×10^{-2}	3.0	4.0×10^{-2}	3.310^{-2}	5.0×10^{-3}	8.0×10^{-2}	2
Non-leafy vegetables	Fruits	All	9	1.8×10^{-2}	2.4	2.3×10^{-2}	1.6×10^{-2}	4.0×10^{-3}	5.7×10^{-2}	1
Leguminous Vegetables	Seeds and pods	All	17	1.7×10^{-2}	1.8	1.9×10^{-2}	8.9×10^{-3}	4.0×10^{-3}	3.8×10^{-2}	1
		Sand	17	1.7×10^{-2}	1.8	1.9×10^{-2}	8.9×10^{-3}	4.0×10^{-3}	3.8×10^{-2}	1
Root Crops	Roots	All	7	2.2×10^{-2}	2.0	2.6×10^{-2}	1.1×10^{-2}	5.0×10^{-3}	3.6×10^{-2}	3
		Sand	6	2.9×10^{-2}	1.2	2.9×10^{-2}	6.2×10^{-3}	2.1×10^{-2}	3.6×10^{-2}	2
Tubers	Tubers	All	57	5.7×10^{-3}	2.5	8.2×10^{-3}	6.9×10^{-3}	7.1×10^{-4}	2.7×10^{-2}	3
		Sand	56	5.8×10^{-3}	2.5	8.3×10^{-3}	7.0×10^{-3}	7.1×10^{-4}	2.7×10^{-2}	2
Grasses	Stems and shoots	All	3	3.1×10^{-2}	3.7	4.8×10^{-2}	3.9×10^{-2}	7.2×10^{-3}	8.6×10^{-2}	2
Fodder Leguminous	Stems and shoots	All	34	2.5×10^{-2}	3.3	4.2×10^{-2}	3.4×10^{-2}	2.0×10^{-3}	1.2×10^{-1}	2
		Sand	23	4.6×10^{-2}	1.8	5.4×10^{-2}	3.1×10^{-2}	1.6×10^{-2}	1.2×10^{-1}	1
		Loam	2			6.1×10^{-2}	2.8×10^{-3}	5.9×10^{-2}	6.3×10^{-2}	1
		Clay	9	4.1×10^{-3}	1.5	4.4×10^{-3}	1.7×10^{-3}	2.0×10^{-3}	8.1×10^{-3}	1
Pasture	Stems and shoots	All	16	6.1×10^{-2}	2.7	1.0×10^{-1}	1.3×10^{-1}	1.3×10^{-2}	4.7×10^{-1}	2
		Sand	5	2.1×10^{-1}	2.0	2.5×10^{-1}	1.6×10^{-1}	8.9×10^{-2}	4.7×10^{-1}	2
		Loam	10	3.4×10^{-2}	1.7	3.7×10^{-2}	1.5×10^{-2}	1.3×10^{-2}	5.7×10^{-2}	2
Maize	Stems and shoots	All	58	1.9×10^{-2}	3.3	3.4×10^{-2}	3.1×10^{-2}	1.4×10^{-3}	1.1×10^{-1}	1
		Sand	58	1.9×10^{-2}	3.3	3.4×10^{-2}	3.1×10^{-2}	1.4×10^{-3}	1.1×10^{-1}	1

TABLE 28. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR P

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			2.0×10^{-1}				1
Leafy Vegetables	Leaves	All	1			1.0				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			1.0				1
Root Crops	Roots	All	1			1.0				1
Tubers	Tubers	All	1			5.0×10^{-1}				1
Pasture	Stems and shoots	All	1			2.0				1

TABLE 29. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Pm

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	17	1.4×10^{-2}	6.0	4.7×10^{-2}	6.8×10^{-2}	1.7×10^{-3}	2.4×10^{-1}	3
		Sand	10	6.6×10^{-3}	6.0	4.1×10^{-2}	8.2×10^{-2}	1.7×10^{-3}	2.4×10^{-1}	3
		Loam	6	4.6×10^{-2}	2.7	6.3×10^{-2}	4.6×10^{-2}	1.0×10^{-2}	1.3×10^{-1}	2
		Clay	1			2.0×10^{-2}				2
Leguminous-Vegetables	Seeds and pod	All	4	1.7×10^{-1}	7.4	4.8×10^{-1}		2.0×10^{-2}	1.2	1
		Sand	1			5.0×10^{-2}				1
		Loam	1			2.0×10^{-2}				1
Root Crops	Root	All	5	4.2×10^{-2}	1.2	4.3×10^{-2}	9.6×10^{-3}	3.6×10^{-2}	6.0×10^{-2}	1
Tubers	Tuber	All	3	1.0×10^{-2}	1.3	1.0×10^{-2}	2.3×10^{-3}	7.5×10^{-3}	1.2×10^{-2}	1
Cereals	Stems and shoots	All	19	2.3×10^{-1}	4.1	4.4×10^{-1}	4.2×10^{-1}	2.2×10^{-2}	1.4	3
		Sand	10	1.2×10^{-1}	5.0	3.2×10^{-1}	4.5×10^{-1}	2.2×10^{-2}	1.4	3
		Loam	7	5.8×10^{-1}	1.8	6.6×10^{-1}	3.6×10^{-1}	2.9×10^{-1}	1.2	2
		Clay	2			2.8×10^{-1}	2.0×10^{-1}	1.3×10^{-1}	4.2×10^{-1}	2
Root Crops	Leaves	All	5	1.9×10^{-1}	1.2	2.0×10^{-1}	3.9×10^{-2}	1.6×10^{-1}	2.5×10^{-1}	1

TABLE 30. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Pu

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	105	9.5×10^{-6}	6.7	5.2×10^{-5}	1.3×10^{-4}	2.0×10^{-7}	1.1×10^{-3}	4
		Sand	76	1.0×10^{-5}	4.5	3.3×10^{-5}	6.6×10^{-5}	5.0×10^{-7}	3.6×10^{-4}	4
		Loam	10	4.9×10^{-6}	11.0	4.7×10^{-5}	9.8×10^{-5}	3.5×10^{-7}	3.1×10^{-4}	4
		Clay	16	7.4×10^{-6}	14.9	8.4×10^{-5}	1.5×10^{-4}	2.0×10^{-7}	5.1×10^{-4}	4
		Organic	2			5.4×10^{-4}	7.6×10^{-4}	2.3×10^{-6}	1.1×10^{-3}	1
Maize	Grain	All	1			3.0×10^{-6}				1
Leafy Vegetables	Leaves	All	13	8.3×10^{-5}	2.7	1.2×10^{-4}	9.3×10^{-5}	1.0×10^{-5}	2.9×10^{-4}	2
		Sand	4	1.1×10^{-4}	2.7	1.5×10^{-4}	1.1×10^{-4}	2.9×10^{-5}	2.9×10^{-4}	1
		Loam	1			2.8×10^{-4}				1
		Organic	1			2.7×10^{-5}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	9	6.5×10^{-5}	2.7	8.7×10^{-5}	5.4×10^{-5}	6.0×10^{-6}	2.0×10^{-4}	2
		Loam	8	6.2×10^{-5}	2.7	8.5×10^{-5}	5.8×10^{-5}	6.0×10^{-6}	2.0×10^{-4}	1
Leguminous Vegetables	Seeds and pods	All	18	6.3×10^{-5}	1.4	6.7×10^{-5}	2.8×10^{-5}	3.7×10^{-5}	1.5×10^{-4}	1
		Sand	18	6.3×10^{-5}	1.4	6.7×10^{-5}	2.8×10^{-5}	3.7×10^{-5}	1.5×10^{-4}	1
Root Crops		All	5	3.9×10^{-4}	10.0	1.7×10^{-3}	2.5×10^{-3}	7.0×10^{-5}	5.8×10^{-3}	1
		Sand	4	5.5×10^{-4}	10.0	2.2×10^{-3}	2.7×10^{-3}	7.0×10^{-5}	5.8×10^{-3}	1
Tubers	Tubers	All	87	1.1×10^{-4}	5.5	3.8×10^{-4}	6.7×10^{-4}	3.8×10^{-6}	5.0×10^{-3}	3
		Sand	72	1.0×10^{-4}	5.0	3.0×10^{-4}	4.1×10^{-4}	3.8×10^{-6}	2.0×10^{-3}	3
		Loam	9	1.5×10^{-4}	11.0	9.9×10^{-4}	1.7×10^{-3}	6.2×10^{-6}	5.0×10^{-3}	3
		Clay	3	3.6×10^{-4}	3.7	5.4×10^{-4}	4.3×10^{-4}	8.0×10^{-5}	9.4×10^{-4}	2
		Organic	2			4.1×10^{-4}	$5. 6 \times 10^{-4}$	1.3×10^{-5}	8.0×10^{-4}	1
Grasses	Stems and shoots	All	2			1.6×10^{-4}	1.6×10^{-4}	5.0×10^{-5}	2.7×10^{-4}	2
Fodder Leguminous	Stems and shoots	All	74	4.9×10^{-4}	2.2	6.6×10^{-4}	5.3×10^{-4}	1.1×10^{-4}	2.9×10^{-3}	1
		Sand	33	4.8×10^{-4}	2.2	6.3×10^{-4}	5.0×10^{-4}	1.1×10^{-4}	2.0×10^{-3}	1
		Loam	25	5.8×10^{-4}	2.4	8.0×10^{-4}	6.6×10^{-4}	1.1×10^{-4}	2.9×10^{-3}	1
		Clay	16	4.1×10^{-4}	1.9	4.9×10^{-4}	2.9×10^{-4}	1.2×10^{-4}	1.1×10^{-3}	1
Pasture	Stems and shoots	All	22	5.5×10^{-4}	3.0	9.5×10^{-4}	1.0×10^{-3}	6.3×10^{-5}	3.9×10^{-3}	2
		Sand	5	4.6×10^{-4}	1.8	5.2×10^{-4}	2.8×10^{-4}	2.1×10^{-4}	9.4×10^{-4}	1
		Loam	10	3.0×10^{-4}	3.0	5.9×10^{-4}	9.7×10^{-4}	6.3×10^{-5}	3.3×10^{-3}	1
		Clay	5	2.0×10^{-3}	1.5	2.2×10^{-3}	1.0×10^{-3}	1.2×10^{-3}	3.9×10^{-3}	1
		Organic	1			1.1×10^{-3}				1

TABLE 30. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Pu (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Stems and shoots	All	10	4.4×10^{-5}	16.4	2.7×10^{-4}	3.3×10^{-4}	4.4×10^{-7}	9.0×10^{-4}	2
		Sand	1			4.0×10^{-5}				1
		Loam	5	4.5×10^{-4}	2.0	5.3×10^{-4}	2.9×10^{-4}	1.5×10^{-4}	9.0×10^{-4}	2
		Clay	4	2.4×10^{-6}	5.5	6.4×10^{-6}	9.2×10^{-6}	4.4×10^{-7}	2.0×10^{-5}	2
Maize	Stems and shoots	All	58	5.2×10^{-5}	2.7	8.0×10^{-5}	7.8×10^{-5}	$2. \times 10^{-6}$	3.2×10^{-4}	1
		Sand	58	5.2×10^{-5}	2.7	8.0×10^{-5}	7.8×10^{-5}	$2. \times 10^{-6}$	3.2×10^{-4}	1
Root Crops	Leaves	All	10	1.2×10^{-3}	2.5	1.6×10^{-3}	1.4×10^{-3}	2.5×10^{-4}	4.9×10^{-3}	2
		Sand	4	7.7×10^{-4}	1.9	8.8×10^{-4}	5.2×10^{-4}	3.4×10^{-4}	1.6×10^{-3}	2
		Loam	5	2.2×10^{-3}	1.8	2.5×10^{-3}	1.5×10^{-3}	1.1×10^{-3}	4.9×10^{-3}	2
		Organic	1			2.5×10^{-4}				1

TABLE 31`. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Rb

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			9.0×10^{-1}				1
Leafy Vegetables	Leaves	All	1			6.2×10^{-1}		3.4×10^{-1}	$9.0x\ 10^{-1}$	2
Root Crops	Roots	All	1			9.0×10^{-1}				1

TABLE 32. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Ru

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	12	3.0×10^{-3}	2.6	4.2×10^{-3}	3.0×10^{-3}	6.0×10^{-4}	1.0×10^{-2}	4
		Sand	2			6.5×10^{-3}	4.9×10^{-3}	3.0×10^{-3}	1.0×10^{-2}	2
		Loam	6	3.4×10^{-3}	2.2	4.3×10^{-3}	2.8×10^{-3}	1.0×10^{-3}	8.0×10^{-3}	2
		Clay	3	1.3×10^{-3}	3.3	2.1×10^{-3}	2.5×10^{-3}	6.0×10^{-4}	5.0×10^{-3}	3
Leafy Vegetables	Leaves	All	3	9.0×10^{-2}	3.7	1.410^{-1}	1.1×10^{-1}	2.0×10^{-2}	2.3×10^{-1}	1
Non-leafy Vegetables	Fruits	All	1			2.0×10^{-2}				1
Leguminous Vegetables	Seeds and pods	All	2			1.5×10^{-2}	7.1×10^{-3}	1.0×10^{-2}	2.0×10^{-2}	2
Root Crops	Roots	All	1			1.0×10^{-2}				1
Tubers	Tubers	All	1			5.0×10^{-3}				1
Cereals	Stems and shoots	All	19	1.6×10^{-1}	2.7	2.6×10^{-1}	2.7×10^{-1}	3.0×10^{-2}	1.0	3
		Sand	3	5.9×10^{-1}	1.9	6.6×10^{-1}	3.5×10^{-1}	3.0×10^{-1}	1.0	2
		Loam	10	2.0×10^{-1}	2.2	2.5×10^{-1}	1.9×10^{-1}	5.0×10^{-2}	6.5×10^{-1}	3
		Clay	5	6.2×10^{-2}	1.7	6.8×10^{-2}	3.0×10^{-2}	3.0×10^{-2}	9.5×10^{-2}	3

TABLE 33. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Sb

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	24	1.8×10^{-3}	2.7	2.8×10^{-3}	2.7×10^{-3}	3.0×10^{-4}	9.0×10^{-3}	3
		Sand	4	1.2×10^{-3}	3.7	2.5×10^{-3}	3.6×10^{-3}	4.5×10^{-4}	7.8×10^{-3}	2
		Loam	19	2.0×10^{-3}	2.7	3.0×10^{-3}	2.6×10^{-3}	3.0×10^{-4}	9.0×10^{-3}	3
Leafy Vegetables	Leaves	All	5	9.4×10^{-5}	2.6	1.3×10^{-4}	8.7×10^{-5}	2.2×10^{-5}	2.3×10^{-4}	2
		Sand	2			2.2×10^{-4}	1.6×10^{-5}	2.0×10^{-4}	2.3×10^{-4}	1
		Loam	3	5.5×10^{-5}	2.2	6.6×10^{-5}	4.1×10^{-5}	2.2×10^{-5}	1.0×10^{-4}	1
Non-leafy Vegetables	Fruits	All	5	1.3×10^{-4}	6.7	4.2×10^{-4}	6.7×10^{-4}	1.5×10^{-5}	1.6×10^{-3}	2
Leguminous Vegetables	Seeds and pods	All	1			7.0×10^{-3}				1
Root Crops	Roots	All	5	6.2×10^{-4}	1.5	6.6×10^{-4}	2.7×10^{-4}	4.0×10^{-4}	1.1×10^{-3}	2
Tubers	Tubers	All	1			2.0×10^{-3}				1
Cereals	Stems and shoots	All	7	2.5×10^{-2}	1.6	2.7×10^{-2}	1.3×10^{-2}	1.2×10^{-2}	5.3×10^{-2}	2

TABLE 34. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Sr

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	282	1.1×10^{-1}	2.7	1.8×10^{-1}	1.9×10^{-1}	3.6×10^{-3}	1.0	18
		Sand	123	1.4×10^{-1}	3.0	2.3×10^{-1}	2.3×10^{-1}	3.6×10^{-3}	1.0	16
		Loam	71	1.1×10^{-1}	2.4	1.6×10^{-1}	1.5×10^{-1}	1.6×10^{-2}	7.2×10^{-1}	10
		Clay	72	7.8×10^{-2}	2.4	1.2×10^{-1}	1.3×10^{-1}	5.3×10^{-3}	7.1×10^{-1}	12
		Organic	10	9.7×10^{-2}	4.1	1.8×10^{-1}	1.4×10^{-1}	1.2×10^{-2}	3.6×10^{-1}	3
Maize	Grain	All	39	3.2×10^{-1}	4.1	5.9×10^{-1}	6.1×10^{-1}	2.0×10^{-3}	2.6	3
		Sand	19	5.2×10^{-1}	3.3	8.8×10^{-1}	7.5×10^{-1}	4.0×10^{-2}	2.6	1
		Loam	13	3.6×10^{-1}	1.6	4.0×10^{-1}	1.9×10^{-1}	1.5×10^{-1}	8.6×10^{-1}	1
		Clay	7	6.9×10^{-2}	6.7	1.6×10^{-1}	1.5×10^{-1}	2.0×10^{-3}	3.9×10^{-1}	3
Leafy vegetables	Leaves	All	217	7.6×10^{-1}	6.0	1.9	1.8	3.9×10^{-3}	7.8	5
		Sand	72	1.7	4.1	2.9	2.1	6.4×10^{-2}	7.8	4
		Loam	84	1.2	4.1	2.0	1.2	4.1×10^{-2}	5.0	2
		Clay	54	1.5×10^{-1}	6.0	5.9×10^{-1}	8.1×10^{-1}	3.9×10^{-3}	2.2	4
		Organic	6	2.1×10^{-1}	1.4	2.2×10^{-1}	6.8×10^{-2}	1.5×10^{-1}	3.0×10^{-1}	2
Non-leafy Vegetables	Fruits	All	19	3.6×10^{-1}	5.5	9.8×10^{-1}	1.8	7.1×10^{-3}	7.9	5
		Sand	5	8.7×10^{-1}	4.1	2.1	3.3	2.0×10^{-1}	7.9	1
		Loam	3	1.4	1.6	1.5	7.1×10^{-1}	9.0×10^{-1}	2.3	1
		Clay	8	1.3×10^{-1}	6.0	3.2×10^{-1}	3.1×10^{-1}	7.1×10^{-3}	8.6×10^{-1}	3
		Organic	2			2.2×10^{-1}	4.2×10^{-2}	1.9×10^{-1}	2.5×10^{-1}	3
Leguminous Vegetables	Seeds and pod	All	148	1.4	2.3	1.8	1.3	1.3×10^{-1}	6.0	4
		Sand	55	2.2	2.1	2.7	1.5	3.0×10^{-1}	6.0	4
		Loam	68	1.3	1.9	1.5	8.5×10^{-1}	1.7×10^{-1}	4.6	4
		Clay	25	6.2×10^{-1}	2.2	8.1×10^{-1}	6.2×10^{-1}	1.3×10^{-1}	2.6	4
Root Crops	Roots	All	56	7.2×10^{-1}	4.1	1.5	1.4	3.0×10^{-2}	4.8	5
		Sand	26	1.1	3.7	1.8	1.4	3.0×10^{-2}	4.8	5
		Loam	16	6.1×10^{-1}	4.5	1.3	1.5	4.4×10^{-2}	4.5	4
		Clay	13	4.1×10^{-1}	4.5	1.0	1.3	5.2×10^{-2}	3.9	4

TABLE 34. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Sr (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Tubers	Tubers	All	106	1.6×10^{-1}	3.0	2.4×10^{-1}	2.2×10^{-1}	7.4×10^{-3}	1.6	8
		Sand	39	2.2×10^{-1}	2.6	3.3×10^{-1}	3.0×10^{-1}	2.6×10^{-2}	1.6	6
		Loam	41	1.3×10^{-1}	3.0	2.0×10^{-1}	1.4×10^{-1}	7.4×10^{-3}	4.5×10^{-1}	5
		Clay	21	1.3×10^{-1}	2.3	1.7×10^{-1}	1.5×10^{-1}	2.6×10^{-2}	6.7×10^{-1}	5
		Organic	4	5.8×10^{-2}	4.5	1.0×10^{-1}	9.8×10^{-2}	8.0×10^{-3}	2.3×10^{-1}	2
Grasses	Stems and shoots	All	50	9.1×10^{-1}	1.9	1.1	6.2×10^{-1}	2.5×10^{-1}	2.8	4
		Sand	34	1.1	1.7	1.3	5.9×10^{-1}	2.6×10^{-1}	2.8	4
		Loam	6	6.0×10^{-1}	2.5	8.9×10^{-1}	8.6×10^{-1}	2.9×10^{-1}	2.0	3
		Clay	7	7.9×10^{-1}	1.3	8.1×10^{-1}	1.7×10^{-1}	4.8×10^{-1}	9.7×10^{-1}	3
		Organic	3	2.6×10^{-1}	1.1	2.6×10^{-1}	1.5×10^{-2}	2.5×10^{-1}	2.8×10^{-1}	2
Fodder Leguminous	Stems and shoots	All	35	3.7	1.9	4.6	3.6	1.3	1.8×10^{1}	6
		Sand	14	4.9	2.0	6.2	4.9	1.3	1.8×10^{1}	4
		Loam	11	3.3	1.8	3.8	2.4	1.4	9.8	3
		Clay	10	2.8	1.7	3.2	1.6	1.3	5.8	3
		Organic	1			3.9×10^{1}				1
Pasture	Stems and shoots	All	172	1.3	2.2	1.7	1.2	5.6×10^{-2}	7.3	8
		Sand	87	1.7	5.5	2.2	1.4	9.8×10^{-2}	7.3	7
		Loam	58	1.1	1.6	1.2	5.6×10^{-1}	3.7×10^{-1}	2.6	6
		Clay	22	8.0×10^{-1}	2.2	1.0	7.6×10^{-1}	9.0×10^{-2}	2.8	5
		Organic	4	3.5×10^{-1}	3.7	5.5×10^{-1}	4.7×10^{-1}	5.6×10^{-2}	1.2	2
Herbs	Stems and shoots	All	1			4.5				1
Other Crops		All	9	8.8×10^{-1}	6.0	2.4	3.1	2.0×10^{-2}	8.2	5
Cereals	Stems and shoots	All	37	1.1	2.5	1.7	1.9	1.5×10^{-1}	9.8	9
		Sand	11	2.1	2.3	3.0	2.9	9.3×10^{-1}	9.8	7
		Loam	3	1.8	2.3	2.2	1.5	7.2×10^{-1}	3.6	3
		Clay	20	7.5×10^{-1}	2.4	1.0	7.8×10^{-1}	1.5×10^{-1}	2.8	5
Maize	Stems and shoots	All	36	7.3×10^{-1}	6.0	9.9×10^{-1}	7.6×10^{-1}	1.2×10^{-1}	3.0	5
		Sand	23	8.2×10^{-1}	2.6	1.2	8.7×10^{-1}	1.2×10^{-1}	3.0	4
		Loam	7	7.0×10^{-1}	1.7	7.8×10^{-1}	3.7×10^{-1}	2.8×10^{-1}	1.4	2
		Clay	6	5.0×10^{-1}	1.9	5.9×10^{-1}	3.4×10^{-1}	1.8×10^{-1}	1.1	2

TABLE 35. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Te

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			1.0×10^{-1}				1
Leafy Vegetables	Leaves	All	1			3.0×10^{-1}				1
Non-leafy Vegetables	Fruits	All	1			3.0×10^{-1}				1
Root Crops	Roots	All	1			3.0×10^{-1}				1
Tubers	Tubers	All	1			2.0×10^{-1}				1
Pasture	Stems and shoots	All	1			1.0				1

TABLE 36. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Y

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			5.0×10^{-4}				1
Leafy Vegetables	Leaves	All	1			2.0×10^{-3}				1
Non-leafy Vegetables	Fruits	All	1			2.0×10^{-3}				1
Root crops	Roots	All	1			2.0×10^{-3}				1
Tubers	Tubers	All	1			1.0×10^{-3}				1
Pasture	Stems and shoots	All	1			5.0×10^{-3}				1

TABLE 37. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Zn

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	86	1.8	2.7	2.8	2.7	2.0×10^{-2}	1.4×10^{1}	7
		Sand	42	2.0	2.5	2.9	2.9	3.9×10^{-1}	1.4×10^{1}	5
		Loam	21	1.5	2.5	2.3	2.2	5.2×10^{-1}	7.0	3
		Clay	17	1.4	1.6	1.8	1.6	6.6×10^{-1}	3.6	6
		Organic	4	8.6×10^{-1}	1.7	9.5×10^{-1}	4.9×10^{-1}	4.7×10^{-1}	1.6	1
Maize	Grain	All	17	5.8×10^{-1}	1.4	6.1×10^{-1}	1.8×10^{-1}	2.8×10^{-1}	9.1×10^{-1}	3
		Sand	7	5.6×10^{-1}	1.5	6.0×10^{-1}	2.1×10^{-1}	2.8×10^{-1}	8.8×10^{-1}	3
		Loam	7	5.8×10^{-1}	1.3	6.0×10^{-1}	1.6×10^{-1}	3.4×10^{-1}	8.0×10^{-1}	3
		Clay	3	6.6×10^{-1}	1.4	6.9×10^{-1}	2.2×10^{-1}	4.8×10^{-1}	9.1×10^{-1}	2
Leafy Vegetables	Leaves	All	112	2.4	2.4	3.4	2.8	1.0×10^{-1}	1.7×10^{1}	5
		Sand	39	4.2	2.0	5.1	3.4	7.4×10^{-1}	1.7×10^{1}	4
		Loam	53	1.8	2.1	2.4	1.8	3.4×10^{-1}	9.3	4
		Clay	19	2.1	2.5	2.8	2.1	3.2×10^{-1}	8.6	2
Non-leafy Vegetables	Fruits	All	3	4.2×10^{-1}	3.7	6.1×10^{-1}	4.5×10^{-1}	1.0×10^{-1}	9.5×10^{-1}	3
		Organic	2		1.0	8.6×10^{-1}	1.2×10^{-1}	7.8×10^{-1}	9.5×10^{-1}	1
Leguminous Vegetables	Seeds and pods	All	86	9.1×10^{-1}	2.4	1.5	2.5	2.5×10^{-1}	1.3×10^{1}	6
		Sand	31	9.7×10^{-1}	1.8	1.2	9.9×10^{-1}	2.7×10^{-1}	5.9	5
		Loam	14	3.9×10^{-1}	1.2	4.0×10^{-1}	6.7×10^{-2}	2.5×10^{-1}	4.7×10^{-1}	3
		Clay	13	1.6	2.5	2.4	2.7	2.5×10^{-1}	8.8	4
Root Crops	Roots	All	3	1.9	13.5	6.4	7.5	1.0×10^{-1}	1.5×10^{1}	2

TABLE 37. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Zn (Cont)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Tubers	Tubers	All	20	3.0×10^{-1}	1.8	3.4×10^{-1}	1.5×10^{-1}	5.0×10^{-2}	6.3×10^{-1}	4
		Sand	6	3.5×10^{-1}	1.5	3.7×10^{-1}	1.5×10^{-1}	1.9×10^{-1}	6.3×10^{-1}	3
		Loam	10	3.0×10^{-1}	1.6	3.3×10^{-1}	1.5×10^{-1}	1.5×10^{-1}	5.8×10^{-1}	3
		Clay	3	3.7×10^{-1}	1.4	3.8×10^{-1}	1.2×10^{-1}	2.4×10^{-1}	4.6×10^{-1}	3
Pasture	Stems and shoots	All	73	1.0	1.9	1.2	6.0×10^{-1}	5.4×10^{-2}	3.2	2
		Sand	38	1.3	1.4	1.4	4.8×10^{-1}	4.8×10^{-1}	2.5	2
		Loam	34	7.8×10^{-1}	2.1	9.7×10^{-1}	6.3×10^{-1}	5.4×10^{-2}	3.2	2
Cereals	Stems and shoots	All	28	5.3	1.7	6.1	3.4	2.0	1.5×10^{1}	6
		Sand	6	8.2	1.5	8.8	3.3	4.2	1.2×10^{1}	5
		Loam	14	4.4	1.5	4.7	1.9	2.5	9.5	5
		Clay	8	3.8	1.6	4.2	1.9	2.0	7.2	5
Maize	Stems and shoots	All	2			5.8	1.8	4.5	7.0	

TABLE 38. SOIL-TO-PLANT TRANSFER FACTOR (F_v) VALUES FOR Zr

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	1			1.0×10^{-3}				1
Leafy Vegetables	Leaves	All	1			4.0×10^{-3}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	1			4.0×10^{-3}				1
Root Crops	Roots	All	1			4.0×10^{-3}				1
Tubers	Tubers	All	1			2.0×10^{-3}				1
Pasture	Stems and shoots	All	1			1.0×10^{-2}				1

By definition, soil-plant transfer factors concept implies equilibrium or quasi equilibrium conditions in the soil-plant system. This precondition is valid with some accuracy, if flows of radionuclides from available pool of radionuclides to plants (or unavailable pool) are negligible. Special care was taken in data selection for the current document to avoid data obtained in non-equilibrium conditions because of interference by soil sorption/adsorption processes and it has been assumed that radionuclide flows from soil to plants are low compared to the available pool of radionuclide in soil.

This is not the case for very mobile radionuclides (such as chlorine and technetium), i.e. for radionuclides with soil-plant transfer factors values which are around 100 or even higher. It should be mentioned that these values have been also determined properly, as the ratio of the activity concentrations in plants and soil at harvest; but most likely this was done for the conditions with no equilibrium in the soil-plant system.

Another point is that chlorine and technetium are very mobile in soil and may be subject to a considerable migration to deeper soil layers; i.e. the soil activities at the end of the vegetation are much lower than at the beginning. Such observations are made in the study by Kashparov et al. [5.24-5.25], where the chlorine activity dropped by a factor of 10-100 due to heavy rainfall during the growing period. However, the activity in plant is due to uptake from soil during the whole vegetation period. Transfer factors from the radionuclide activity concentrations in soil and plant are usually being determined at the end of the vegetation period. Applying those values to activity concentration determined at the start of the vegetation period may cause serious overestimations.

Therefore, for the current document average values for Cl concentration in soil were selected. Besides, for some of these radionuclides, e.g. ^{3}H, ^{14}C, and ^{36}Cl, transfer parameters and models are normally formulated in terms of specific activity concepts. Therefore, data for these particular radionuclides are mainly treated separately and given in a specific paper of this TECDOC.

As previously reported for natural radionuclides, the data given in Tables 8-40 show clear differences between the geometric and arithmetic mean values. The mean ratio of the arithmetic mean to the geometric mean, calculated across all the datasets used for estimation of radionuclide transfer factors from all soils to all plants, was 1.8 with the standard deviation around that mean of 1.14. However, the majority of such ratios calculated for individual databases is less than factor 3 (Fig. 5).

The F_v geometric mean values for transuranic elements range from 4.3×10^{-7} to 5.1×10^{-1}; for fission products from 5.1×10^{-6} to 5.3×10^{2} (Fig. 7); for neutron activation products from 4.5×10^{-4} to 5.2×10^{2}.

Radionuclide accumulation shows high variability, which is influenced by the physical and chemical properties of radionuclides, soil properties, and peculiarities of plant species. Figs. 6-8 illustrate effects of radionuclide properties, representing F_v values to Cereals planted on sandy soils.

At the same time, the general point, relevant to all crop and soil groups, is that extremely large variability in transfer factors was not observed. For any combination of soil/crop, the uncertainity factor, expressed as a GSD, is less than a factor of five for the majority of datasets. The exceptions were the data for Am (Cereals, all plant compartments, Clay and Loam soils), Cs (Herbs: Stems and shoots), Pu (Cereals and Tubers, 6 datasets), Tc (Maize and Leafy Vegetables) and the data for Zn transfer to Root Crops.

TABLE 39. SOIL-TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cl

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	7	3.6×10^1	1.6	4.0×10^1	2.3×10^1	2.0×10^1	8.6×10^1	3
		Sand	2			2.5×10^1	6.4	2.0×10^1	2.9×10^1	2
		Loam	3	4.7×10^1	1.8	5.3×10^1	3.0×10^1	2.6×10^1	8.6×10^1	3
		Clay	2			3.7×10^1	1.2×10^1	2.8×10^1	4.5×10^1	2
Leafy Vegetables	Leaves	All	6	2.6×10^1	1.7	2.9×10^1	1.5×10^1	1.4×10^1	4.8×10^1	1
		Sand	1			1.6×10^1				2
		Loam	4	2.5×10^1	1.7	2.8×10^1	1.5×10^1	1.4×10^1	4.8×10^1	1
		Clay	1			4.5×10^1				2
Leguminous Vegetables	Seeds, Pod	All	7	1.1×10^1	1.3	1.1×10^1	2.5	7.0	1.5×10^1	1
		Sand	2			1.3×10^1	2.8	1.1×10^1	1.5×10^1	2
		Clay	2			9.0	2.8	7.0	1.1×10^1	2
Root Crops	Roots	All	14	1.2×10^1	1.8	1.4×10^1	9.8	4.8	3.6×10^1	3
		Sand	4	1.2×10^1	1.4	1.2×10^1	3.8	8.6	1.7×10^1	2
		Loam	6	1.1×10^1	2.0	1.4×10^1	1.1×10^1	4.8	3.6×10^1	3
		Clay	4	1.4×10^1	2.0	1.7×10^1	1.3×10^1	8.1	3.6×10^1	2
Cereals	Stems and shoots	All	7	3.4×10^2	1.5	3.6×10^2	1.5×10^2	2.1×10^2	6.2×10^2	3
		Sand	2			3.0×10^2	1.3×10^2	2.1×10^2	3.9×10^2	2
		Loam	3	3.4×10^2	1.7	3.8×10^2	2.2×10^2	2.2×10^2	6.2×10^2	3
		Clay	2			4.0×10^2	7.9×10^1	3.4×10^2	4.5×10^2	2

TABLE 40. SOIL-TO-PLANT TRANSFER FACTORS (F_v) FOR Tc

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereal	Grain	All	2			1.3	1.6	1.8×10^{-1}	2.4	2
Maize	Grain	All	8	3.8	8.2	1.7×10^{1}	2.3×10^{1}	5.0×10^{-1}	5.2×10^{1}	3
Leafy Vegetables	Leaves	All	10	1.8×10^{2}	13.5	1.1×10^{3}	1.4×10^{3}	4.5	3.4×10^{3}	2
		Sand	4	1.1×10^{2}	33.1	1.2×10^{3}	1.4×10^{3}	4.5	2.9×10^{3}	1
		Loam	6	2.5×10^{2}	8.2	1.1×10^{3}	1.6×10^{3}	2.5×10^{1}	3.4×10^{3}	1
Leguminous Vegetables	Seeds and pods	All	5	4.3	5.2	1.1×10^{1}	1.4×10^{1}	1.1	3.0×10^{1}	2
		Sand	3	1.3	1.1	1.3	1.6×10^{-1}	1.1	1.4	2
		Loam	2			2.6×10^{1}	4.7	2.3×10^{1}	3.0×10^{1}	2
Root Crops	Roots	All	2			4.6×10^{1}	4.6×10^{1}	1.4×10^{1}	7.9×10^{1}	2
Tubers	Tubers	All	8	2.3×10^{-1}	3.7	3.5×10^{-1}	2.2×10^{-1}	1.3×10^{-2}	6.5×10^{-1}	2
		Sand	6	3.9×10^{-1}	1.6	4.3×10^{-1}	1.8×10^{-1}	1.8×10^{-1}	6.5×10^{-1}	2
		Loam	2			9.4×10^{-2}	1.2×10^{-1}	1.3×10^{-2}	1.8×10^{-1}	2
Grasses	Stems and shoots	All	1							1
Pasture	Stems and shoots	All	18	7.6×10^{1}	3.0	1.2×10^{2}	1.3×10^{2}	7.9	4.7×10^{2}	2
Maize	Stems and shoots	All	20	6.4	3.3	1.1×10^{1}	1.1×10^{1}	8.4×10^{-1}	3.7×10^{1}	2

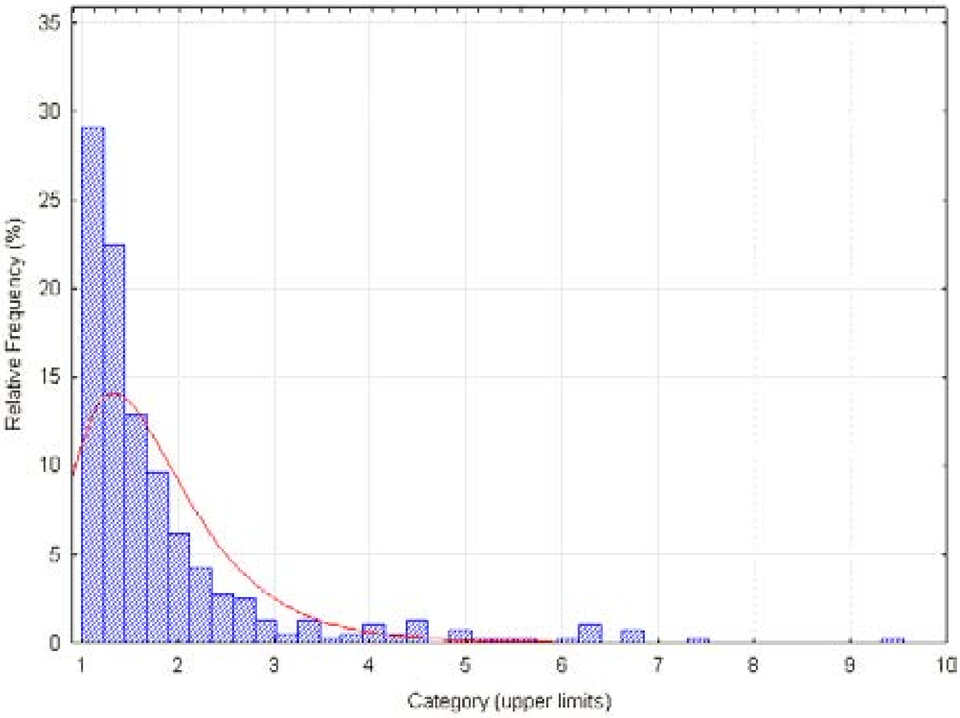

FIG. 5. Relative frequencies distribution plot of ratios of the arithmetic means of the transfer factors to those of the geometric means.

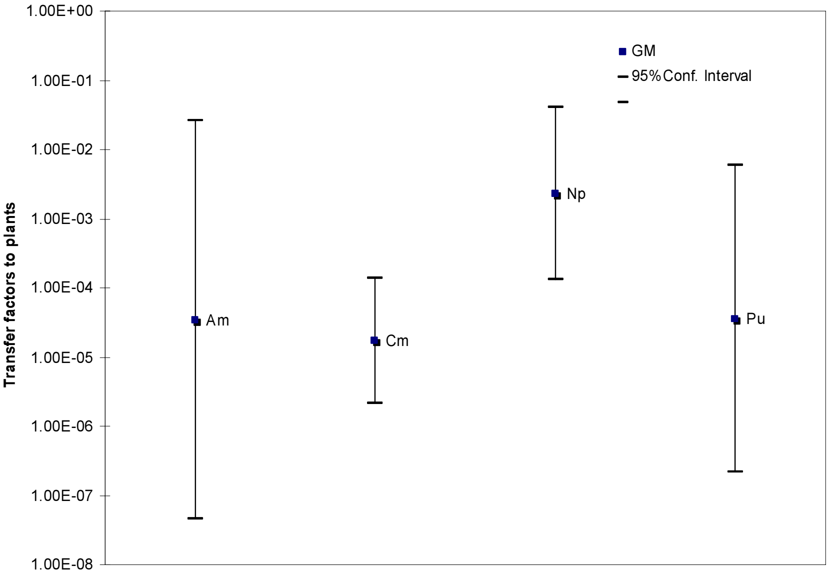

FIG. 6. Chart representation of distribution of F_v values to cerals for transuranic elements (sandy soils).

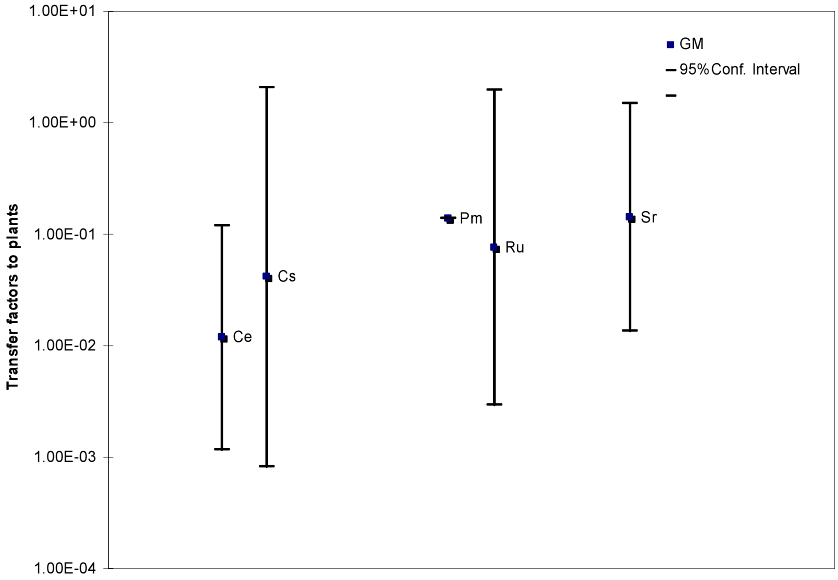

FIG. 7. Chart representation of distribution of F_v values to cereals for fission products (sandy soils).

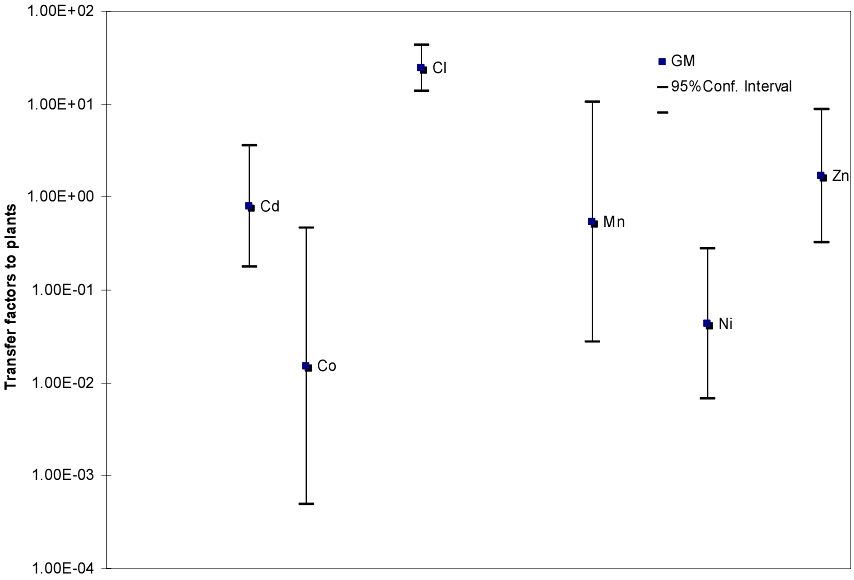

FIG. 8. Chart representation of distribution of F_v values to cereals for neutron activation radionuclides and Cl (sandy soils).

The F_v geometric mean values, across all the radionuclides considered, vary from 3.0×10^{-6} to 4.0×10^{2}. The most biologically available radionuclides are Ca, Cl, Na, P, Pm, Sr, Tc, Cu, and Mo. Some clear differences exist between the transfers of radionuclides to different crops and between soils. The highest transfers for the majority of radionuclides are observed on organic soils, and, on mineral soils, the rate of uptake largely decreases with increasing clay content. In terms of plant groups, the highest radionuclide transfer is observed for Pasture, Grasses and Fodder Leguminous followed by Cereals, Maize and Root Crops.

4. COMPARISON OF THE AVAILABLE DATA WITH TECHNICAL REPORTS SERIES NO. 364

The present database contains information about eight elements, in addition to those previously included in Technical Reports Series No. 364, and in particular, Ca, Cd, Cl, K, Pm, and P. Data for these are given in the current document (Figure 9). For some radionuclides, e.g. ^{3}H, ^{14}C, and ^{36}Cl, transfer parameters and models are normally formulated in terms of specific activity concepts. Therefore, data for these particular radionuclides are treated separately and presented in the separate paper[1] presented in this document below.

For Cu, Nd, Pr, Rh and W, no additional information was obtained; as a result, the present database contains F_v values only from Technical Reports Series No. 364 [6]. It should be recognized that the expected values for these five elements presented in Technical Reports Series No. 364 are mainly a product of expert judgement and based only on one reference, namely [7]. Therefore, results of radiological assessments with these values involved should be interpreted with great caution.

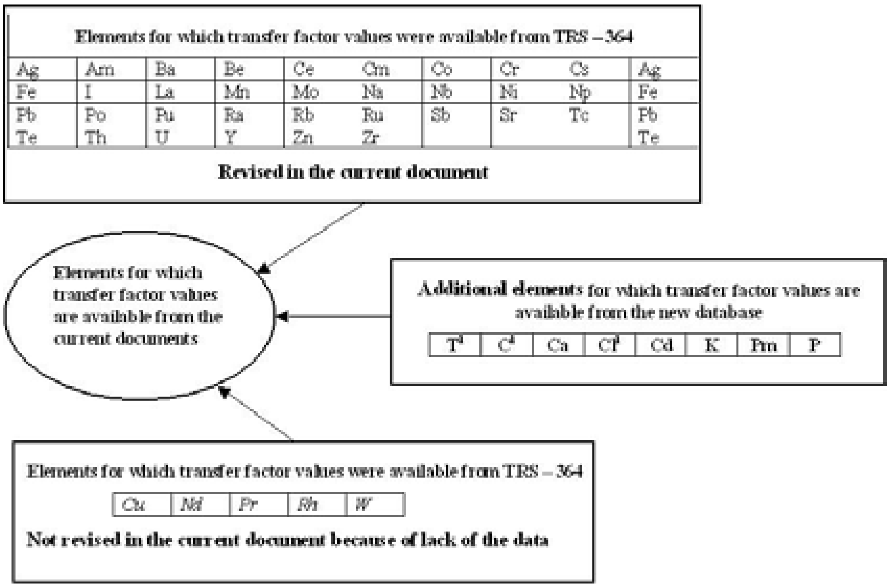

FIG. 9. Comparison of data available from Technical Reports Series No. 364 and from the current document.

TABLE 41 SOIL-TO PLANT TRANSFER FACTORS FOR RADIONUCLIDES PRESENTED ONLY IN Technical Reports Series No. 364

Element	Plant Group	Plant compartment	Soil Group	N	Expected Value	Uncertainity Factor	#ref
Cu	Not specified		All	1	0.8	-	1
Nd	Not specified		All	1	2.0×10^{-2}	-	1
Pr	Cereal	Grain	All	1	2.0×10^{-2}	-	1
	Leafy Vegetables	Leaves	All		2.0×10^{-2}	-	
	Root Crops	Roots	All		2.0×10^{-2}	-	
Rh	Not specified		All	1	9.0×10^{-1}	-	1
W	Not specified		All	1	1.0×10^{-1}	-	1

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[1] See Davis et al. ‘Specific activity models and parameter values for tritium, ^{14}C and ^{36}Cl’ in this documment

In comparison to Technical Reports Series No. 364, the current document contains many more data for radionuclide transfer to different crop groups and for different soils and allows, at least in part, for better differentiation between specific contamination scenarios.

5. CONCLUSIONS

Assessment of F_v values based on the literature sources is always associated with many shortcomings, and very often considerable judgment must be exercised in evaluating the available data. First, such data are normally based on studies that were not originally intended for transfer factors assessments. Second, the experimental design may deviate greatly from the transfer factor definition. For example, vertical distribution of radionuclides in soil profiles can depart from the uniform distribution assumed by the transfer factor definition, etc. Radionuclide transfer to plants depends on numerous factors including physical and chemical forms of the radionuclide, soil properties, plant species, plant compartment, farming practices, etc. Such factors result in high variability, and the individual F_v values themselves can vary over three orders of magnitude [2, 3, 8].

In order to decrease the uncertainty associated with soil/plant factors, several classifications were developed and soil-to-plant transfer factors for temperate environments were reviewed and grouped according to the selected plant and soil categories. New information presented in this paper substantially increases the amount of available data on radionuclide transfer to plants. The data, providing information for specific plant and soil groups, allows more precise radiological assessments in different areas around the world. However, there are still clear gaps in transfer factor values for a substantial number of radionuclides, plants, and soil groups.

Acceptable numbers of data entries for radionuclide transfer to plants are available only for a few elements, such as Cs, Sr, and partially for U, Ra, Mn, and Co. From 100 to 500 estimates are available for some other elements such as Pu, Ni, Ce, Cl, I, Cm, Np, Am, Zn, Th and Pb while for 19 elements (Cu, Eu, P, Nb, Ba, Na, Cr, Zr, Ca, Y, Ag, Fe, La, Cd, Sb, Pm, Tc, Ru and Po), only from 10 to 100 entries are available. For some elements such as Te, Mo and Rb, only a few values are available, and for elements such as Cu, Nd, Pr, Rh, and W, single values available from the literature are provided, based on expert judgement, or estimated, based on behaviour of stable analogues. Thus, for many radionuclides, there are no experimental data to support radiological assessments, and an analogue approach should be used to derive the necessary values.

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APPENDIX

List of publication used for evaluation of radionuclide transfer factors from soil to plants: temperate environment

Natural Radionuclides

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ROOT UPTAKE: TROPICAL AND SUB-TROPICAL ENVIRONMENTS

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Abstract

In this paper we analyse the radionuclide soil-to-plant transfer factors obtained for tropical and subtropical environments. In these regions several soil types occur in which radionuclide uptake by crops consistently deviates from the values characteristic of temperate environments. F_v values from 6 countries from tropical regions and 10 countries belonging subtropical environments have been gathered in the corresponding databases. In total, more than 2700 individual transfer factor values were considered mainly for radioisotopes of Cs, Ra, U, Sr, Co, Th, Pb, K, Mn and Zn, but also for radioisotopes of I, Ag, Tc, Pu, and Am. Four broad soil groups and 13 plant groups were separately investigated. Additionally, different plant compartments were distinguished. The wide variability and uncertainty observed in transfer factors is considerably reduced when data are independently grouped into clusters containing the same radionuclide/soil type/plant group/plant part combinations. For each cluster, standard statistical quantities were determined.

1. INTRODUCTION

Previously, existing reviews of environmental radionuclide transfers were principally limited to temperate climatic zones, as data from other climatic zones were typically unavailable [1-7]. However, recently new data on tropical environments have become available, and from these new data it appears that, although the direct influence of climatic conditions on radioecological transfer parameters seems to be minimal, its indirect effects can be significant [2, 6, 8, 9-11].

Climate and parent rock material determine, to a large extent, the characteristics of soil development. In tropical areas, several soil types occur in which radionuclide uptake by crops consistently deviates from the values characteristic of temperate environments. In typical tropical environments, almost all of the organic material that reaches the soil surface decomposes rapidly, and surface accumulation of soil organic matter is minimal. Consequently, there is rapid recycling of nutrients and contaminants into the vegetation. In temperate zones, the decomposition of organic debris is slower, and the accumulation of soil organic matter is usually greater than the rate of decomposition, resulting in highly organic surface soil [12-15].

2. MAIN FACTORS GOVERNING RADIONUCLIDE BEHAVIOUR IN TROPICAL AND SUBTROPICAL ENVIRONMENTS

Tropical climatic region is the geographic area of the Earth centred on the equator and limited in latitude, respectively, by the tropics of Cancer (northern hemisphere) and Capricorn (southern hemisphere). In this region, ecosystems can consist of rainforests, dry deciduous forest, spiny forest, desert, and other types of environments. In tropical environments, almost all organic material that reaches the soil surface decomposes rapidly, and the surface accumulation of soil organic matter is therefore minimal. Consequently, there is rapid recycling of nutrients and contaminants into the vegetation. In the tropics, due to the long age of soils and the high mineral weathering rates, clays of low exchange activity such as kaolinite are more common than in the temperate zone. This lead to soils that, in spite of having a high clay content, can have a low exchange capacity. Subtropical region refers to the zones of the Earth immediately north and south of tropic zone including range of latitudes between 23.5 and approximately 40 degrees (parts of arid and semiarid regions). These areas typically have hot summer, and, throughout the annual period, the air temperature usually does not go below freezing (0 °C). Subtropical areas do not usually have markedly wet or dry seasons, and the rain distribution is fairly regular throughout the year [17]. It should be recognised that there are several quite different classifications of the geographical zones according to the climatic conditions. For the purpose of the current report, the simplest is used as a first experience in the application of an geographical approach for the classification of agricultural systems for radiological assessments.

3. INFORMATION SOURCES AND DATA ANALYSIS

Values of soil-to-plant transfer factors in tropical and subtropical environments determined before 1998 were mostly obtained from the IAEA CRP [18], and IUR databases [6, 12]. In total, 180 references were reviewed, of which only 55 were retained and used for transfer factor value evaluation. These are identified in the reference list given in Appendix 3. The database on tropical soil-to-plant F_v values contains 1269 entries, principally for radioisotopes of Cs, Ra, U, Sr and Zn, but also for radioisotopes of Co, Pb, Th, K, Pu and Am. Fig. 1 shows the number of entries for each radionuclide.

The database on subtropical soil-to-plant F_v values contains 940 entries. Most of them are for radioisotopes of Cs and Sr, but also included are radioisotopes of I, K, Co, Mn, Ag, Zn, Tc, and Pu. The number of entries for each radionuclide in this database are given in Fig. 2. Figures 3 and 4 show the number of records for different plant groups accumulated in the tropical and subtropical databases, respectively. Soils were grouped based on classification given in previous paper, considering primarily the texture classification. Figures 5 and 6 show the number of entries for each soil type represented in the two databases. In terms of number of records associated with different plant groups, the largest contribution to the tropical database is for Leafy and Leguminous Vegetables, followed by Grasses and Non-Leafy Vegetables, and finally Tubers, Rice, Maize, Fruits and Cereals. Rather a large percentage (≈20%) is related to plants outside the classification scheme. This emphasises the need for alternative guidance as to the grouping of tropical plants. For both tropical and subtropical ecosystems, the major contribution to the database is for Leafy Vegetables (around 30%), followed by Grasses, Root/Tuber Crops, Leguminous Vegetables.

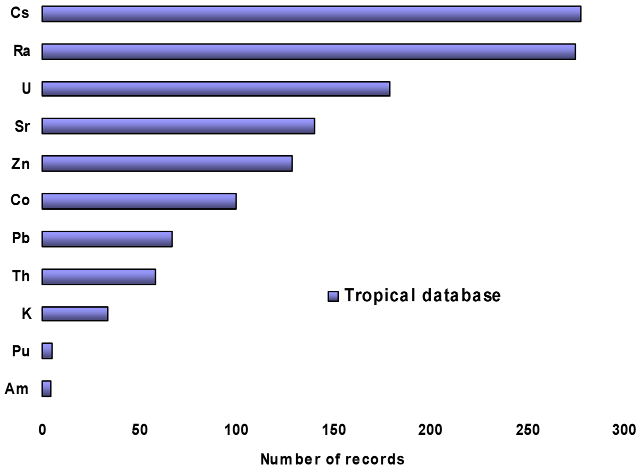

FIG. 1. Number of entries for every radionuclide in the tropical database.

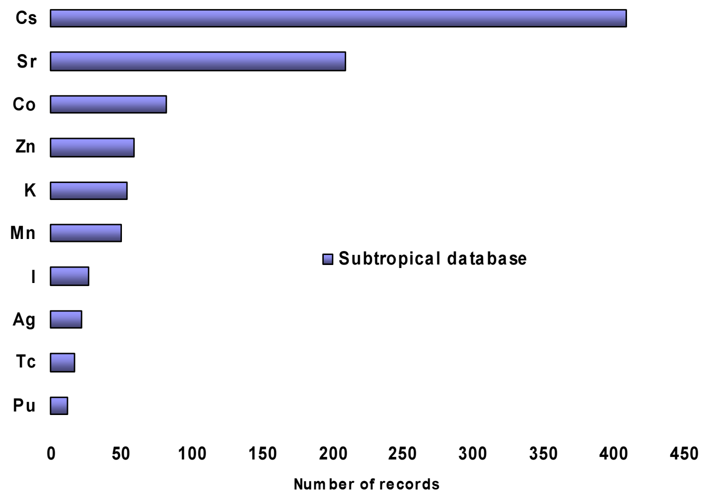

FIG. 2. Number of entries for every radionuclide in the subtropical database.

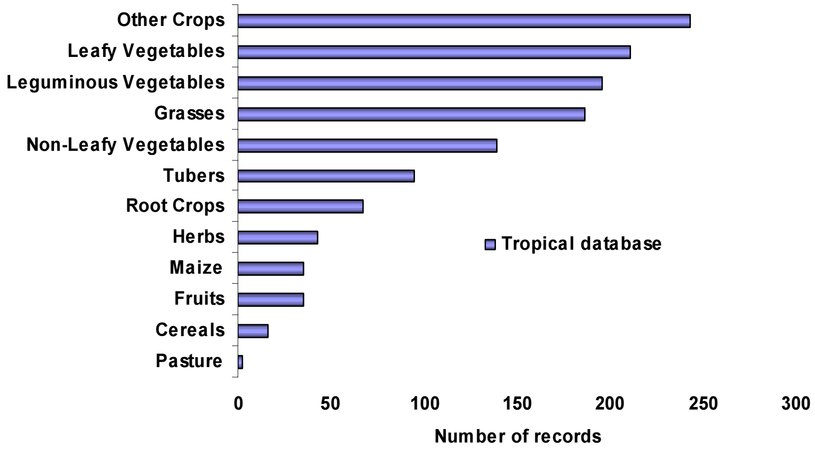

FIG. 3. Number of entries for each individual plant group in the tropical database.

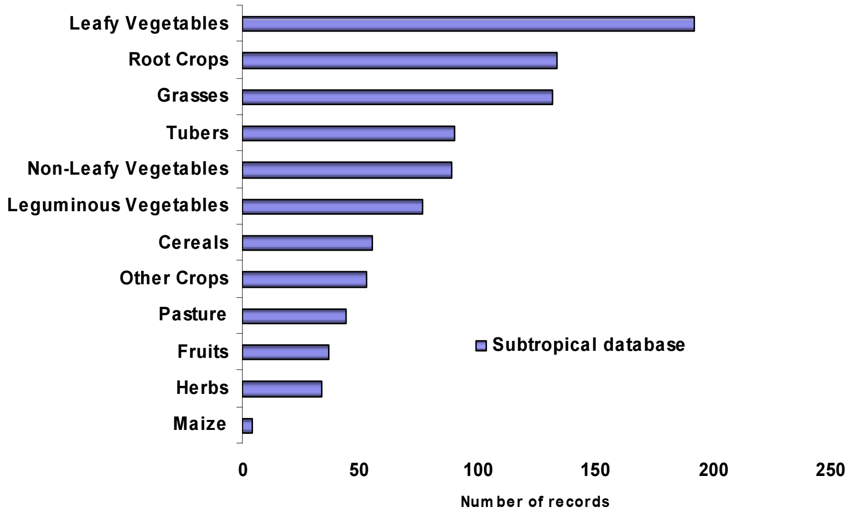

FIG. 4. Number of entries for each individual plant group in the subtropical database.

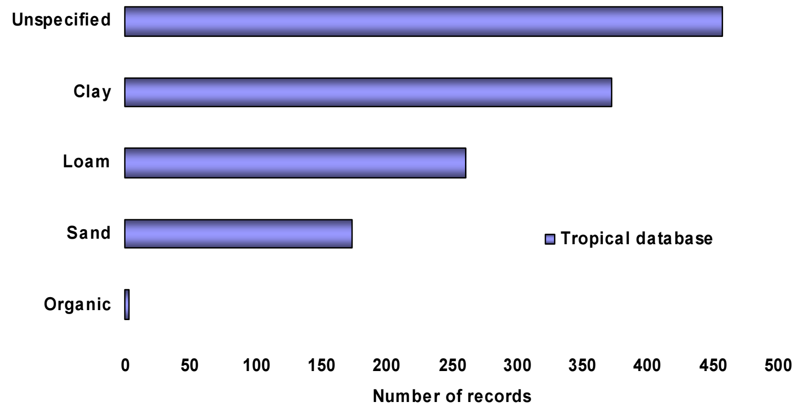

FIG. 5. Number of entries for each soil group in the tropical database.

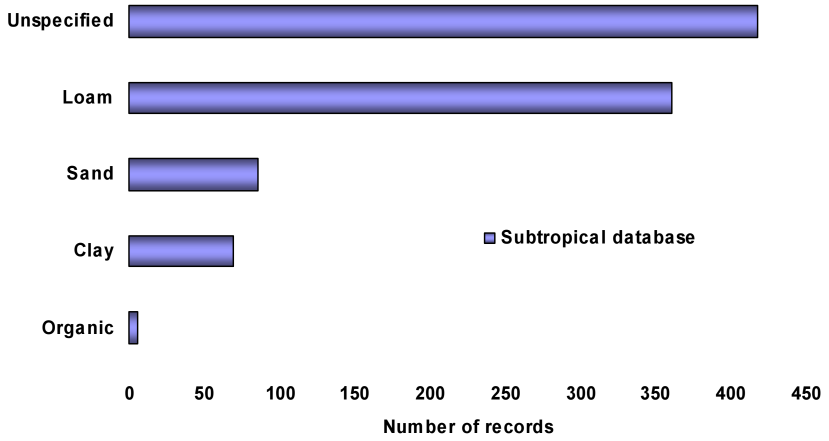

FIG. 6. Number of entries for each soil group in the subtropical database.

It should be also noted that between 30-50% of available publications do not provide sufficient information on soil properties to define their class according to the approach given in previous paper of this IAEA-TECDOC. For subtropical environments, in terms of the data availability, soil groups can be ranked as follows: Loam>Sand>Clay>Others. For tropical environments, the majority of the data available are for clay soils (around 30%), followed by loam>sand and finally organic and others.

4. DERIVED TRANSFER FACTOR VALUES

The available information for both tropical and subtropical environments is markedly heterogeneous in terms of locations around the world. For each respective environment, Table 1 shows the number of entries in the database associated with each country. Tables 2-22 give F_v values for each element included in the data collected in the tropical and subtropical databases, respectively.

TABLE 1. NUMBER OF ENTRIES FOR EACH COUNTRY IN EACH DATABASE

Environment	Country/Part of country[4]	Number of Entries
Tropical	Australia	345
	Brazil	229
	India	224
	Indonesia	40
	Marshall Islands	51
	Vietnam	380
	Total	1269
Subtropical	Bangladesh	101
	Cuba	30
	China	43
	Iraq	15
	Japan	263
	Republic of Korea	24
	Syrian Arab Republic	200
	Taiwan	226
	Turkey	38
	Total	940
	Total	2209

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[4] From some countries located partly in temperate and partly in subtropical environments data from the areas with subtropical conditions were selected.

TABLE 2. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Am

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Fruits	Fruits	Others[1]	2			3.7×10^{-5}	1.6×10^{-5}	2.6×10^{-5}	4.8×10^{-5}	1
	Coconut Milk[2]	Others[1]	1			3.7×10^{-5}				1
Non-leafy Vegetables	Fruits	Others[1]	1			1.1×10^{-5}				1

[1] Others refer to soils which are out of the classification schema used by the current document such as Marsharll Island soils, classified by authors as coral sand soil. [2] Name is beyond the classification used. Authors report data on radionuclides in plant compartments for the same plants. [2] The plant compartment is beyond of the classification used by the current document.

TABLE 3. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Co

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Leguminous Vegetables	Seeds and pods	All	19	6.6×10^{-1}	2.3	9.4×10^{-1}	$9.74x\ 10^{-1}$	2×10^{-3}	3.6	1
		Clay	18	6.5×10^{-1}	2.3	9.4×10^{-1}	1.0	2×10^{-1}	3.6	1
Leafy Vegetables	Leaves	All	41	9.2×10^{-2}	1.9	1.1×10^{-1}	7.2×10^{-2}	3.2×10^{-2}	2.8×10^{-1}	2
		Clay	39	9.1×10^{-2}	2	1.1×10^{-1}	7.4×10^{-2}	3.2×10^{-2}	2.8×10^{-1}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	28	3.1×10^{-1}	1.7	3.5×10^{-1}	1.7×10^{-1}	1.4×10^{-1}	6.9×10^{-1}	2
		Clay	26	3.1×10^{-1}	1.7	3.5×10^{-1}	1.70×10^{-1}	1.4×10^{-1}	6.9×10^{-1}	1
Root Crops	Roots	All	7	1.2×10^{-1}	1.7	1.3×10^{-1}	6.4×10^{-2}	6.3×10^{-2}	2.1×10^{-1}	2
		Clay	5	1.2×10^{-1}	1.7	1.3×10^{-1}	6.9×10^{-2}	6.3×10^{-2}	2.1×10^{-1}	1
Tubers	Tubers	All	4	3.7×10^{-1}	1.0	3.7×10^{-1}	1.4×10^{-2}	3.6×10^{-1}	3.9×10^{-1}	2
		Clay	3	3.7×10^{-1}	1.2	3.7×10^{-1}	1.7×10^{-2}	3.6×10^{-1}	3.9×10^{-1}	1

TABLE 4. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR K

Plant Group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Fruits	Fruits	Organic	2			5.5×10^{1}	2.1	4.0×10^{1}	7.0×10^{1}	1
	Leaves[1]	All	2			3.2×10^{1}	4.3×10^{1}	9.0×10^{-1}	6.2×10^{1}	2
		Loam	1			9.0×10^{-1}				1
		Organic	1			6.2×10^{1}				1
Grasses	Stems and shoots	Loam	1			8.7×10^{-1}				1
Other Crops	Leaves	Loam	13	1.4	2.0	1.7	1.5	4.9×10^{-1}	5.6	1
	Stems and shoots	Loam	5	1.7	1.3	1.7	4.8×10^{-1}	1.2	2.3	1
Tubers	Tubers	Loam	1			2.7				1

TABLE 5 TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cs

Plant Group	Plant Compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	4	2.3×10^{-1}	3.4	3.8×10^{-1}	2.1×10^{-1}	6.0×10^{-2}	1.0	3
		Sand	1			1.3×10^{-1}				1
		Loam	1			3.3×10^{-1}				1
Fruits	Fruits	All	13	4.8×10^{-1}	5.7	1.7	2.9	5.0×10^{-2}	8.7	3
		Sand	1			3.6×10^{-1}				1
		Unspecified	5	2.6	3.6	4.1	3.5	3.6×10^{-1}	8.7	2
	Coconut Milk[1]	Unspecified	2			4.3	5.6	3.2×10^{-1}	8.2	1
	Leaves	Unspecified	1			5.8				1
Grasses	Grain	All	34	1.4×10^{-2}	2.9×10^{1}	1.6	3.5	1.5×10^{-4}	1.3×10^{1}	4
		Sand	24	6.6×10^{-3}	2.8×10^{1}	1.2	2.8	1.5×10^{-4}	8.6	1
		Unspecified	2			1.2×10^{1}	2.1	1.0×10^{1}	1.3×10^{1}	1
	Stems and shoots	Sand	24	1.1×10^{-2}	2.5×10^{1}	1.5	3.4	4.2×10^{-4}	9.60	1
Herbs	Leaves	All	3	5.3×10^{-1}	1.4×10^{1}	1.7	1.6	$2. \times 10^{-2}$	3.20	2
		Others	1			1.9				1

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[1] Name is beyond the classification skhema used. Authors report data on radionuclides in plant compartments for the same plants.

TABLE 5. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cs (Cont.)

Plant Group	Plant Compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Leguminous Vegetables	Grain	All	7	1.1	4.45	2.2	1.9	1.9×10^{-1}	4.1	3
		Sand	6	1.4	4.07	2.5	1.8	2.2×10^{-1}	4.1	3
		Loam	1			1.9×10^{-1}				1
	Stems and shoots	Sand	4	7.3	1.30	7.5	1.9	5.4	9.6	1
	Leaves	All	61	1.1×10^{-1}	3.94	2.9×10^{-1}	5.4×10^{-1}	1.0×10^{-2}	3.9	3
		Sand	1			4.4×10^{-1}				1
		Clay	53	1.0×10^{-1}	3.67	2.2×10^{-1}	2.6×10^{-1}	1.0×10^{-2}	7.7×10^{-1}	2
		Unspecified	1			3.9				1
Other Crops	Leaves	All	19	5.9×10^{-1}	7.19	3	5.4	4.0×10^{-2}	2.1×10^{1}	3
		Unspecified	6	5.6	2.65	7.9	6.9	1.1	2.1×10^{1}	2
	Fruits	Unspecified	2			6.6	3.4	4.2	9.0	1
	Stems and shoots	Loam	4	7.0×10^{-2}	1.1	7.0×10^{-2}	8.2×10^{-3}	6.0×10^{-2}	8.0×10^{-2}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	38	7.0×10^{-1}	3.3	1.2	1.8	5.0×10^{-2}	1.1×10^{1}	3
		Clay	26	9.3×10^{-1}	1.7	1.1	5.6×10^{-1}	3.6×10^{-1}	2.3	1
		Sand	4	5.6×10^{-2}	1.1	5.7×10^{-2}	6.0×10^{-3}	5.0×10^{-2}	6.3×10^{-2}	1
		Unspecified	2			7.3	5.2	3.6	1.1×10^{1}	1
Root Crops	Roots	All	9	4.3×10^{-1}	2	5.1×10^{-1}	2.3×10^{-1}	1.3×10^{-1}	8.1×10^{-1}	2
		Clay	5	6×10^{-1}	1.3	6.1×10^{-1}	1.4×10^{-1}	4.5×10^{-1}	8.1×10^{-1}	1
Tubers	Tubers	All	8	4.3×10^{-1}	3.4	7.8×10^{-1}	9.5×10^{-1}	6.0×10^{-2}	3.0	4
		Sand	1			2.0×10^{-1}				1
		Clay	4	3.7×10^{-1}	3.5	5.5×10^{-1}	3.9×10^{-1}	6.0×10^{-2}	10×10^{-1}	2
		Unspecified	1			3.00				1
Leafy Vegetables	Leaves	All	49	9.8×10^{-1}	2.3	1.2	6.3×10^{-1}	1.1×10^{-1}	2.9	3
		Clay	39	1.3	1.5	1.4	5.0×10^{-1}	3.6×10^{-1}	2.9	1
		Unspecified	1			4.1×10^{-1}				1
Maize	Grain	Unspecified	2			2.0	7.8×10^{-1}	1.4	2.5	1

TABLE 6. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Pu

Plant Group	Plant Compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Fruits	Fruits	Unspecified	2			2.3×10^{-5}	9.2×10^{-6}	1.6×10^{-5}	2.9×10^{-5}	1
	Coconut milk	Unspecified	1			3.2×10^{-5}				1
Other Crops		Unspecified	1			6.7×10^{-5}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	Unspecified	1			1.7×10^{-5}				1

TABLE 7. TROPICAL DATABASE: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Pb

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	Sand	1			2.5×10^{-3}				1
Grasses	Leaves	Sand	9	2.1×10^{-1}	2.2	2.9×10^{-1}	2.8×10^{-1}	5.9×10^{-2}	1.00	1
Herbs	Leaves	Sand	2			3.7×10^{-1}	4.9×10^{-1}	2.0×10^{-2}	7.1×10^{-1}	1
Leguminous Vegetables	Grain	All	9	3.3×10^{-3}	2.4	4.2×10^{-3}	2.7×10^{-3}	6.5×10^{-4}	8.9×10^{-3}	1
		Sand	3	3.4×10^{-3}	1.3	3.4×10^{-3}	8.5×10^{-4}	2.8×10^{-3}	4.4×10^{-3}	1
		Loam	6	3.2×10^{-3}	3.0	4.6×10^{-3}	3.3×10^{-3}	6.5×10^{-4}	8.9×10^{-3}	1
Other crops		Sand	18	2.3×10^{-1}	2.7	3.2×10^{-1}	2.4×10^{-1}	1.4×10^{-2}	1.0	1
Non-leafy Vegetables		All	2			7.0×10^{-3}	0.0	7.0×10^{-3}	7.0×10^{-3}	2
Pasture	Stems and shoots	Unspecified	1			3.0×10^{-1}				1
Root Crops	Roots	Loam	3	2.4×10^{-3}	1.5	2.6×10^{-3}	1.2×10^{-3}	1.8×10^{-3}	4.0×10^{-3}	1
Tubers	Tubers	All	16	5.7×10^{-4}	2.4	8.0×10^{-4}	6.6×10^{-4}	1.5×10^{-4}	2.3×10^{-3}	1
		Sand	1			1.6×10^{-3}				1
		Loam	15	5.3×10^{-4}	2.4	7.4×10^{-4}	6.4×10^{-4}	1.5×10^{-4}	2.3×10^{-3}	1
Maize	Grain	All	6	8.5×10^{-4}	2.1	1.2×10^{-3}	1.3×10^{-3}	5.2×10^{-4}	3.8×10^{-3}	1
		Sand	1			5.2×10^{-4}				1
		Loam	5	9.3×10^{-4}	2.2	1.3×10^{-3}	1.4×10^{-3}	5.9×10^{-4}	3.8×10^{-3}	1

TABLE 8. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ra

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals		All	3	3.5×10^{-3}	2.5	4.7×10^{-3}	4.6×10^{-3}	1.7×10^{-3}	1.0×10^{-2}	2
		Sand	2			2.1×10^{-3}	5.7×10^{-4}	1.7×10^{-3}	$2. \times 10^{-3}$	1
Fruits	Leaves	Loam	1			1.0×10^{-1}				1
Grasses	Stems and shoots	All	33	1.7	4.3	4.4	1.0×10^{1}	1.8×10^{-2}	5.80×10^{1}	2
		Loam	1			1.9×10^{-1}				1
Herbs	Stems and shoots	Unspecified	4	7.5×10^{-2}	1.4×10^{1}	7.8×10^{-1}	1.5	1.0×10^{-2}	3.0	1
	Leaves	Unspecified	11	1.1×10^{-1}	4.8	2.9×10^{-1}	3.6×10^{-1}	1.1×10^{-2}	1.0	1
Leguminous Vegetables	Grain	All	31	2.1×10^{-2}	4.3	5.1×10^{-2}	7.1×10^{-2}	7.6×10^{-4}	2.7×10^{-1}	1
		Sand	5	3.6×10^{-2}	3.0	4.9×10^{-2}	3.2×10^{-2}	5.7×10^{-3}	8.7×10^{-2}	1
		Loam	26	1.9×10^{-2}	4.5	5.1×10^{-2}	7.7×10^{-2}	7.6×10^{-4}	2.7×10^{-1}	1
Other Crops	Leaves	All	12	1.1×10^{-1}	2.1	1.4×10^{-1}	1.1×10^{-1}	3.7×10^{-2}	3.7×10^{-1}	2
		Loam	11	1.2×10^{-1}	2,0	1.51×10^{-1}	1.2×10^{-1}	4.0×10^{-2}	3.7×10^{-1}	1
	Roots	All	3	9.8×10^{-2}	5.8	1.8×10^{-1}	1.5×10^{-1}	1.3×10^{-2}	2.7×10^{-1}	1
	Stems and shoots	All	6	1.2×10^{-1}	2.5	1.5×10^{-1}	8.3×10^{-2}	2.0×10^{-2}	2.7×10^{-1}	2
		Loam	5	1.6×10^{-1}	1.4	1.7×10^{-1}	6.1×10^{-2}	1.0×10^{-1}	2.7×10^{-1}	1
		All	57	2.6	5.5	9.4	1.9×10^{1}	5.7×10^{-2}	1.3×10^{2}	1
		Loam	1			8.0×10^{-2}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	9	3.2×10^{-3}	5.6	1.2×10^{-2}	2.3×10^{-2}	5.2×10^{-4}	7.0×10^{-2}	4
		Loam	6	1.4×10^{-3}	3.4	3.2×10^{-3}	5.3×10^{-3}	5.2×10^{-4}	1.4×10^{-2}	2
Pasture	Stems and shoots	Unspecified	1			7.0×10^{-2}				1
Root Crops	Roots	Loam	22	1.1×10^{-2}	4.9	3.4×10^{-2}	5.5×10^{-2}	1.2×10^{-3}	2.2×10^{-1}	1
Tubers	Tubers	All	42	1.9×10^{-3}	3.8	8.1×10^{-3}	2.9×10^{-2}	2.6×10^{-4}	1.9×10^{-1}	2
		Sand	1			1.6×10^{-3}				1
		Loam	41	2.0×10^{-3}	3.9	8.3×10^{-3}	3.0×10^{-2}	2.6×10^{-4}	1.9×10^{-1}	2
Leafy Vegetables		Loam	22	2.7×10^{-2}	4.5	7.1×10^{-2}	1.0×10^{-1}	3.0×10^{-3}	4.3×10^{-1}	1
Maize	Grain	All	18	1.1×10^{-3}	2.4	1.7×10^{-3}	2.0×10^{-3}	1.9×10^{-4}	8.3×10^{-3}	1
		Sand	3	3.8×10^{-3}	2.0	4.6×10^{-3}	3.3×10^{-3}	2.0×10^{-3}	8.3×10^{-3}	1
		Loam	15	8.7×10^{-4}	2.0	1.1×10^{-3}	8.7×10^{-4}	1.9×10^{-4}	3.8×10^{-3}	1

TABLE 9. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Sr

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	2			6.0×10^{-1}	2.3×10^{-1}	4.4×10^{-1}	7.6×10^{-1}	1
		Sand	1			4.4×10^{-1}				1
		Loam	1			7.6×10^{-1}				1
Fruits	Fruits	Others	3	1.7×10^{-2}	6.2	4.8×10^{-2}	7.1×10^{-2}	3.7×10^{-3}	1.3×10^{-1}	1
Grasses	Grain	Sand	24	1.9×10^{-1}	7.9	1.2	2.2	1.4×10^{-2}	6.8	1
	Leaves	Sand	24	3.0×10^{-1}	7.6	1.9	3.5	2.8×10^{-2}	9.7	1
Herbs	Leaves	All	1			3.6				1
Leguminous Vegetables	Grain	All	6	3.7	1.9	4.4	2.9	1.8	8.2	2
		Sand	5	3.9	2.0	4.7	3.1	1.8	8.2	2
		Loam	1			2.7				1
	Leaves	All	2			5.0×10^{1}	1.8×10^{1}	3.7×10^{1}	6.3×10^{1}	1
		Sand	1			3.7×10^{1}				1
		Loam	1			6.3×10^{1}				1
	Stems and shoots	Sand	4	7.5	1.3	7.6	1.8	6.0	9.5	1
		Clay	17	1.2	6.2	2.9	2.1	7.9×10^{-2}	5.9	3
Other Crops		Others	1			4.8×10^{-1}				1
Non-leafy Vegetables	Fruits, heads, berries, buds	All	16	1.2	2.6	1.7	1.3	2.6×10^{-1}	4.2	3
		Clay	13	1.2	2.6	1.8	1.4	3.9×10^{-1}	4.2	1
		Others	1			2.6×10^{-1}				1
Root Crops	Roots	All	4	1.8	1.6	2.0	9.3×10^{-1}	1.2	2.8	2
		Clay	2			2.0	1.1	1.2	2.8	1
Tubers	Tubers	All	2			6.8×10^{-1}	3.0×10^{-2}	6.6×10^{-1}	7.0×10^{-1}	2
		Clay	1			6.6×10^{-1}				1
Leafy Vegetables	Leaves	All	34	3.6	1.9	4.4	2.8	1.4	1.2×10^{1}	2
		Clay	32	3.6	1.9	4.4	2.9	1.4	1.2×10^{1}	1

TABLE 10. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Th

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Herbs	Leaves	Unspecified	3	5.8×10^{-2}	1.1×10^{1}	3.2×10^{-1}	5.2×10^{-1}	1.2×10^{-2}	9.2×10^{-1}	1
	Stems and shoots	Unspecified	9	1.8×10^{-1}	5.5	4.5×10^{-1}	4.7×10^{-1}	1.8×10^{-2}	1.2	1
Leguminous Vegetables	Grain	Loam	4	6.3×10^{-5}	2.5	8.7×10^{-5}	8.7×10^{-5}	2.6×10^{-5}	2.1×10^{-4}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	Loam	2			5.3×10^{-6}	2.8×10^{-6}	3.3×10^{-6}	7.3×10^{-6}	1
Other Crops	Leaves	Unspecified	3	5.8×10^{-2}	2.3	7.5×10^{-2}	6.9×10^{-2}	3.5×10^{-2}	1.5×10^{-1}	1
	Stems and shoots	Unspecified	3	2.6×10^{-1}	1.1×10^{1}	1.3	2.2	5.3×10^{-2}	3.9	1
		Unspecified	4	8.2×10^{-3}	1.6	8.9×10^{-3}	4.4×10^{-3}	5.0×10^{-3}	1.5×10^{-2}	1
Root Crops	Roots	Loam	5	1.9×10^{-5}	1.7	2.1×10^{-5}	1.1×10^{-5}	9.0×10^{-6}	3.9×10^{-5}	1
Tubers	Tubers	Loam	13	8.9×10^{-6}	2.6	1.3×10^{-5}	1.0×10^{-5}	2.9×10^{-6}	3.5×10^{-5}	1
Leafy Vegetables	Leaves	Loam	6	3.4×10^{-5}	1.9	4.1×10^{-5}	2.8×10^{-5}	1.8×10^{-5}	7.6×10^{-5}	1
Maize	Grain	Loam	6	1.2×10^{-5}	3.5	2.1×10^{-5}	2.1×10^{-5}	1.9×10^{-6}	5.0×10^{-5}	1

TABLE 11. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR U

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	3	1.8×10^{-2}	3.8×10^{1}	2.8×10^{-1}	4.7×10^{-1}	6.0×10^{-4}	8.2×10^{-1}	3
		Sand	1			6.0×10^{-4}				1
	All plant	Unspecified	1			4.6×10^{-1}				1
Fruits	Fruits	Unspecified	3	4.4×10^{-2}	3.9	7.3×10^{-2}	7.7×10^{-2}	1.1×10^{-2}	1.6×10^{-1}	2
	All plant	Unspecified	1			6.2×10^{-1}				1
Grasses	Stems and shoots	Unspecified	10	6.4×10^{-1}	1.5	6.9×10^{-1}	2.3×10^{-1}	2.5×10^{-1}	8.8×10^{-1}	1
Herbs	Leaves	Unspecified	5	7.8×10^{-3}	1.4	8.2×10^{-3}	2.8×10^{-3}	5.0×10^{-3}	1.2×10^{-2}	1
	Fruits	Unspecified	1			3.7×10^{-1}				1
		Unspecified	3	4.9×10^{-2}	1.9	5.6×10^{-2}	3.7×10^{-2}	2.8×10^{-2}	9.8×10^{-2}	1
Leguminous Vegetables	Grain	All	7	3.8×10^{-2}	1.1×10^{1}	2.0×10^{-1}	3.3×10^{-1}	2.3×10^{-3}	9.2×10^{-1}	2
		Sand	2			3.4×10^{-3}	1.5×10^{-3}	2.3×10^{-3}	4.4×10^{-3}	1
		Loam	1			3.2×10^{-3}				1
	All plant	Unspecified	2			8.5×10^{-1}	2.8×10^{-3}	8.5×10^{-1}	8.6×10^{-1}	1
Other Crops	Leaves	Unspecified	8	4.9×10^{-3}	1.5	5.2×10^{-3}	2.0×10^{-3}	3.3×10^{-3}	8.4×10^{-3}	1
	Roots	Unspecified	10	2.5×10^{-2}	1.9	2.9×10^{-2}	1.7×10^{-2}	1.1×10^{-2}	5.5×10^{-2}	1
	Tubers	Unspecified	7	8.9×10^{-3}	2.0	1.1×10^{-2}	7.6×10^{-3}	2.9×10^{-3}	2.6×10^{-2}	1
	All plant	Unspecified	27	2.2×10^{-1}	6.1	4.3×10^{-1}	2.7×10^{-1}	8.0×10^{-4}	9.4×10^{-1}	1
Non-leafy Vegetables	Fruits	Unspecified	14	2.6×10^{-2}	2.8	4.3×10^{-2}	5.4×10^{-2}	4.3×10^{-3}	1.8×10^{-1}	2
	All plant	Unspecified	2			7.0×10^{-1}	7.1×10^{-4}	7.0×10^{-1}	7.1×10^{-1}	1
Roots	Leaves	Unspecified	1			2.5×10^{-1}				1
	Roots	Unspecified	6	4.7×10^{-2}	5.1	1.1×10^{-1}	1.2×10^{-1}	8.3×10^{-3}	2.6×10^{-1}	2
	Stems and shoots	Unspecified	1			1.7×10^{-1}				1
Tubers	Tubers	Unspecified	4	2.0×10^{-2}	2.3	2.5×10^{-2}	1.7×10^{-2}	7.3×10^{-3}	4.3×10^{-2}	1
Leafy Vegetables	Leaves	Unspecified	19	4.8×10^{-2}	3.2	8.8×10^{-2}	1.1×10^{-1}	4.4×10^{-3}	4.1×10^{-1}	2
	All plant	Unspecified	1			6.8×10^{-1}				1

TABLE 11 TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR U (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Maize	Grain	All	2			8.7×10^{-2}	1.2×10^{-1}	1.5×10^{-3}	1.7×10^{-1}	2
		Sand	1			1.5×10^{-3}				1
	All plant	Unspecified	1			9.6×10^{-1}				1

TABLE 12. TROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Zn

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	2			2.2×10^{1}	$5.7 \times$	1.8×10^{1}	2.6×10^{1}	1
		Sand	1							1
		Loam	1			2.6×10^{1}				1
Grasses	Grain	Sand	12	2.2×10^{-1}	1.8	2.6×10^{-1}	1.8×10^{-1}	1.1×10^{-1}	7.0×10^{-1}	1
	Stems and shoots	Sand	12	2.0×10^{-1}	1.9	2.5×10^{-1}	1.7×10^{-1}	9.5×10^{-2}	5.7×10^{-1}	1
Leguminous Vegetables	Grain	All	2			1.8×10^{1}	3.5	1.5×10^{1}	2.0×10^{1}	1
		Sand	1			1.5×10^{1}				1
		Loam	1			2.0×10^{1}				1
	Leaves	All	2			1.9×10^{1}	5.0	1.5×10^{1}	2.2×10^{1}	1
		Sand	1			1.5×10^{1}				1
		Loam	1			2.2×10^{1}				1
		All	19	1.5	1.4	1.6	5.6×10^{-1}	9.3×10^{-1}	2.5	1
		Clay	18	1.5	1.4	1.6	5.8×10^{-1}	9.3×10^{-1}	2.5	1
Non-leafy Vegetables	Fruits	All	28	1.7	1.6	1.9	7.8×10^{-1}	5.8×10^{-1}	3.4	2
		Clay	26	1.7	1.6	1.8	7.8×10^{-1}	5.8×10^{-1}	3.4	1
Root Crops	Roots	All	7	1.2	1.8	1.4	6.8×10^{-1}	5.6×10^{-1}	2.2	2
		Clay	5	1.3	1.8	1.4	7.1×10^{-1}	5.6×10^{-1}	2.2	1
Tubers	Tubers	All	4	1.1	1.2	1.1	2.5×10^{-1}	9.2×10^{-1}	1.5	2
		Clay	3	1.1	1.3	1.1	3.1×10^{-1}	9.2×10^{-1}	1.5	1
Leafy Vegetables	Leaves	All	41	1.7	1.4	1.8	6.7×10^{-1}	7.3×10^{-1}	4.8	2
		Clay	39	1.7	1.4	1.8	6.8×10^{-1}	7.3×10^{-1}	4.8	1

TABLE 13. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Ag

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Herbs	Grain	Sand	2			1.1×10^{-2}		1.1×10^{-2}	1.1×10^{-2}	1
Leguminous Vegetables	Grain	Sand	2			8.0×10^{-3}		8.0×10^{-3}	8.0×10^{-3}	2
Non-leafy Vegetables	Stems and shoots	Sand	1			3.0×10^{-3}				1
	Fruits	Sand	1			1.0×10^{-2}				1
	Tubers	Sand	2			1.2×10^{-2}	4.24×10^{-3}	9.0×10^{-3}	1.5×10^{-2}	1
		Sand	4	8.0×10^{-3}	2.0	9.3×10^{-3}	4.92×10^{-3}	3.0×10^{-3}	1.5×10^{-2}	1
Root Crops	Roots	Sand	2			2.3×10^{-2}	0.0	2.3×10^{-2}	2.3×10^{-2}	2
Leafy Vegetables	Leaves	Sand	8	2.1×10^{-2}	5.8	5.3×10^{-2}	5.13×10^{-2}	2.0×10^{-3}	1.2×10^{-1}	2

TABLE 14. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Co

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Grasses		Sand	19	2.6×10^{-1}	2.5	3.6×10^{-1}	2.6×10^{-1}	4.0×10^{-2}	9.2×10^{-1}	2
Leguminous Vegetables	Grain	Loam	3	1.1×10^{-1}	1.5	1.2×10^{-1}	5.3×10^{-2}	8.0×10^{-2}	1.8×10^{-1}	1
		Unspecified	1			5.5×10^{-2}				1
Other Crops		Loam	3	6.7×10^{-1}	1.2	6.7×10^{-1}	9.6×10^{-2}	6.0×10^{-1}	7.8×10^{-1}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	Loam	3	7.9×10^{-1}	1.1	7.9×10^{-1}	7.9×10^{-2}	7.3×10^{-1}	8.8×10^{-1}	1
Pasture	Stems and shoots	Loam	2			2.8×10^{-1}	7.1×10^{-3}	2.7×10^{-1}	2.8×10^{-1}	1
Root Crops	Leaves	Unspecified	10	1.3×10^{-3}	3.8	2.5×10^{-3}	2.5×10^{-3}	1.9×10^{-4}	8.4×10^{-3}	1
	Roots	Unspecified	11	1.3×10^{-3}	4.8	4.4×10^{-3}	9.6×10^{-3}	1.7×10^{-4}	3.3×10^{-2}	1

TABLE 14. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Co (Cont.)

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Leafy-Vegetables	Leaves	All	19	1.1×10^{-1}	5.8	3.4×10^{-1}	4.6×10^{-1}	4.8×10^{-3}	1.5 ×	3
		Loam	7	5.1×10^{-1}	2.0	6.2×10^{-1}	4.0×10^{-1}	2.0×10^{-1}	1.2	1
	Roots	Loam	2			4.7×10^{-2}	5.1×10^{-2}	1.1×10^{-2}	8.3×10^{-2}	1
	Stems and shoots	Loam	2			1.1×10^{-2}	1.4×10^{-3}	1.0×10^{-2}	1.2×10^{-2}	1

TABLE 15. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR K

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Grasses	Stems, shoots		33	1.5	1.5	1.6	5.62×10^{-1}	6.7×10^{-1}	2.84	1
Other Crops		Others	3	2.8	2.3	3.5	3.18	1.6	7.20	1
Tubers	Tubers	All	18	2.4×10^{-1}	1.5	2.5×10^{-1}	8.11×10^{-2}	1.0×10^{-1}	4.10×10^{-1}	1
		Sand	6	1.8×10^{-1}	1.5	12.0×10^{-1}	7.50×10^{-2}	1.0×10^{-1}	2.70×10^{-1}	1
		Loam	8	2.9×10^{-1}	1.3	3.0×10^{-1}	6.90×10^{-2}	1.8×10^{-1}	4.10×10^{-1}	1
		Clay	4	2.4×10^{-1}	1.4	2.5×10^{-1}	6.85×10^{-2}	1.5×10^{-1}	3.00×10^{-1}	1

TABLE 16. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cs

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	All	23	3.1×10^{-3}	2.4	4.8×10^{-3}	5.8×10^{-3}	1.0×10^{-3}	2.6×10^{-2}	3
		Loam	15	2.5×10^{-3}	2.4	4.2×10^{-3}	6.4×10^{-3}	1.0×10^{-3}	2.6×10^{-2}	2
	Stems and shoots	Loam	11	1.0×10^{-2}	1.9	1.2×10^{-2}	6.0×10^{-3}	3.4×10^{-3}	2.0×10^{-2}	1
Fruits	Fruits	All	20	2.0×10^{-2}	4.3	6.8×10^{-2}	1.5×10^{-1}	2.8×10^{-3}	6.6×10^{-1}	5
		Loam	12	2.1×10^{-2}	6.3	1.0×10^{-1}	1.9×10^{-1}	2.8×10^{-3}	6.6×10^{-1}	3
		Clay	8	1.7×10^{-2}	2.0	2.1×10^{-2}	1.2×10^{-2}	6.0×10^{-3}	3.5×10^{-2}	2
	Skin	Clay	1			9.0×10^{-3}				1
Grasses	Grain	Loam	9	2.7×10^{-3}	1.3	2.8×10^{-3}	6.8×10^{-4}	1.9×10^{-3}	4.0×10^{-3}	1
	Stems and shoots	All	51	2.5×10^{-1}	6.3	8.3×10^{-1}	1.1	6.0×10^{-3}	3.7	4
		Loam	21	2.7×10^{-1}	1.6×10^{1}	1.6	1.5	6.0×10^{-3}	3.7	2
Herbs	Leaves	All	18	1.1×10^{-1}	3.9	2.1×10^{-1}	2.5×10^{-1}	5.4×10^{-3}	8.9×10^{-1}	2
		Loam	8	9.6×10^{-2}	1.7	1.1×10^{-1}	5.8×10^{-2}	4.6×10^{-2}	1.90×10^{-1}	1
		Clay	8	2.4×10^{-1}	2.9	3.7×10^{-1}	3.1×10^{-1}	5.1×10^{-2}	8.9×10^{-1}	1
	Fruit	Clay	1			7.0×10^{-4}				1
	Stems, shoots	Unspecified	1			6.8×10^{-3}				1
Leguminous Vegetables	Grain	All	31	1.6×10^{-2}	4.1	4.5×10^{-2}	7.7×10^{-2}	2.0×10^{-3}	3.1×10^{-1}	3
		Loam	28	1.5×10^{-2}	4.1	4.4×10^{-2}	8.0×10^{-2}	2.0×10^{-3}	3.1×10^{-1}	3
		Clay	2			5.9×10^{-2}	7.2×10^{-2}	8.0×10^{-3}	1.1×10^{-1}	1
Other Crops	Grain	Clay	1			3.0×10^{-4}				1
	Leaves	All	10	1.3×10^{-1}	1.1×10^{1}	5.7×10^{-1}	6.5×10^{-1}	3.7×10^{-3}	1.9	2
		Sand	4	8.9×10^{-3}	2.4	1.2×10^{-2}	1.2×10^{-2}	3.7×10^{-3}	2.9×10^{-2}	1
	Roots	All	2			6.2×10^{-2}	3.0×10^{-2}	4.0×10^{-2}	8.3×10^{-2}	2
		Sand	1			8.3×10^{-2}				1
		Loam	1			4.0×10^{-2}				1
	Stems and shoots	Unspecified	9	5.5×10^{-1}	2.4	7.5×10^{-1}	5.7×10^{-1}	2.1×10^{-1}	1.5	1
		All	18	6.9×10^{-1}	9.6	3.0	4.1	3.0×10^{-3}	1.0×10^{1}	9
		Loam	6	2.4×10^{-1}	4.6	4.5×10^{-1}	3.7×10^{-1}	2.6×10^{-2}	8.9×10^{-1}	3
		Clay	2			5.7×10^{-2}	7.6×10^{-2}	3.0×10^{-3}	1.1×10^{-1}	1
		Others[2]	3	7.3	1.8	8.0	3.4	3.8	1.0×10^{1}	1

[1] Name is beyond of the classification schema used . [2] Others refer to sawdust as a substrate where this plants group was grown.

TABLE 16. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Cs (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Non-Leafy Vegetables	Fruits, heads, berries, buds	All	13	1.9×10^{-2}	6.5	7.9×10^{-2}	1.2×10^{-1}	2.3×10^{-3}	3.0×10^{-1}	7
		Loam	10	2.5×10^{-2}	7.8	1.0×10^{-1}	1.3×10^{-1}	2.3×10^{-3}	3.0×10^{-1}	3
		Clay	3	7.3×10^{-3}	1.8	8.1×10^{-3}	4.0×10^{-3}	4.0×10^{-3}	1.2×10^{-2}	2
	Leaves	Loam	6	2.6×10^{-2}	2.1	3.7×10^{-2}	3.1×10^{-2}	1.0×10^{-2}	8.0×10^{-2}	1
	Roots	Loam	6	5.3×10^{-3}	3.3	8.7×10^{-3}	8.3×10^{-3}	1.0×10^{-3}	2.3×10^{-2}	1
		All	6	3.2×10^{-2}	4.1	5.1×10^{-2}	3.5×10^{-2}	2.0×10^{-3}	1.1×10^{-1}	4
		Loam	2			5.0×10^{-2}	0.0	5.0×10^{-2}	5.0×10^{-2}	1
		Clay	2			1.7×10^{-2}	2.1×10^{-2}	2.0×10^{-3}	3.1×10^{-2}	2
Pasture	Stems, shoots	All	34	1.9×10^{-1}	2.5	2.6×10^{-1}	1.8×10^{-1}	2.0×10^{-2}	6.3×10^{-1}	4
		Loam	6	7.5×10^{-2}	2.8	1.1×10^{-1}	8.6×10^{-2}	2.0×10^{-2}	2.2×10^{-1}	3
Root crops	Leaves	Unspecified	10	3.5×10^{-2}	4.5	8.7×10^{-2}	1.2×10^{-1}	3.9×10^{-3}	3.5×10^{-1}	1
	Roots	All	15	1.5×10^{-2}	4.4	4.1×10^{-2}	6.4×10^{-2}	1.4×10^{-3}	2.3×10^{-1}	3
		Loam	2			2.6×10^{-2}	6.4×10^{-3}	2.1×10^{-2}	3.0×10^{-2}	1
Tubers	Tubers	All	34	6.5×10^{-2}	2.4	9.8×10^{-2}	1.1×10^{-1}	9.0×10^{-3}	4.1×10^{-1}	4
		Sand	8	1.5×10^{-1}	2.3	2.0×10^{-1}	1.5×10^{-1}	4.8×10^{-2}	4.1×10^{-1}	2
		Loam	8	4.2×10^{-2}	2.0	4.9×10^{-2}	2.3×10^{-2}	9.0×10^{-3}	8.0×10^{-2}	2
		Clay	4	4.7×10^{-2}	1.6	5.1×10^{-2}	2.4×10^{-2}	3.0×10^{-2}	8.0×10^{-2}	1
Leafy Vegetables	Leaves	All	35	3.8×10^{-2}	6.2	1.7×10^{-1}	3.1×10^{-1}	1.1×10^{-3}	1.4	7
		Sand	6	1.04×10^{-2}	5.49	2.71×10^{-2}	3.42×10^{-2}	1.1×10^{-3}	8.90×10^{-2}	1
		Loam	22	4.1×10^{-2}	6.0	1.7×10^{-1}	2.6×10^{-1}	6.0×10^{-3}	8.9×10^{-1}	3
		Clay	1			8.0×10^{-3}				1
Maize	Grain	Clay	2			5.0×10^{-3}	4.2×10^{-3}	2.0×10^{-3}	8.0×10^{-3}	1

TABLE 17. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR I

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	Unspecified	1			1.5×10^{-4}				1
	Leaves	Unspecified	1			2.0×10^{-2}				1
	Stems, shoots	Unspecified	3	1.0×10^{-2}	4.5	2.2×10^{-2}	3.1×10^{-2}	4.1×10^{-3}	5.8×10^{-2}	1
Herbs	Leaves	Unspecified	1			2.4×10^{-1}				1
	Stems, shoots	Unspecified	1							1
Leguminous Vegetables	Grain	Unspecified	1			3.0×10^{-3}				1
Non-leafy vegetables	Fruits, heads, berries, buds	Unspecified	3	1.2×10^{-3}	2.1	1.4×10^{-3}	1.1×10^{-3}	6.5×10^{-4}	2.7×10^{-3}	2
	Leaves	Unspecified	1			4.5×10^{-2}				1
	Roots	Unspecified	1			1.1×10^{-2}				1
Root Crops	Leaves	Unspecified	1			1.2×10^{-1}				1
	Roots	Unspecified	2			5.6×10^{-2}	7.1×10^{-3}	5.1×10^{-2}	6.1×10^{-2}	1
Leafy Vegetables	Leaves	All	8	3.0×10^{-2}	2.4	3.9×10^{-2}	2.7×10^{-2}	6.7×10^{-3}	8.0×10^{-2}	1
		Sand	5	3.5×10^{-2}	2.0	4.1×10^{-2}	2.1×10^{-2}	1.2×10^{-2}	6.3×10^{-2}	3
	Roots	Unspecified	1			1.3×10^{-1}				1
	Stems and shoots	Unspecified	1			3.0×10^{-3}				1

TABLE 18. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Mn

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Grasses	Stems, shoots	Unspecified	6	1.0	2.4	1.4	1.2	4.0×10^{-1}	3.30	2
Leguminous Vegetables	Grain	Unspecified	1			1.0×10^{-1}				1
Root crops	Leaves	Unspecified	10	3.7×10^{-2}	4.8	8.5×10^{-2}	9.7×10^{-2}	2.9×10^{-3}	2.9×10^{-1}	1
	Roots	Unspecified	11	6.0×10^{-3}	9.7	1.5×10^{-1}	4.6×10^{-1}	4.7×10^{-4}	1.5	2
Leafy Vegetables	Leaves	Unspecified	4	1.0	6.4	3.6	6.0	2.3×10^{-1}	1.3×10^{1}	2
	Roots	Unspecified	2			3.9×10^{-1}	3.3×10^{-1}	1.5×10^{-1}	6.2×10^{-1}	1
	Stems, shoots	Unspecified	2			1.6×10^{-1}	9.2×10^{-2}	9.0×10^{-2}	2.2×10^{-1}	1

TABLE 19 SUBTROPICAL DATABASE: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Pu

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Non-Leafy Vegetables	Fruits, heads, berries, buds	Unspecified	2			8.2×10^{-4}	5.4×10^{-4}	4.3×10^{-4}	1.2×10^{-3}	1
Root crops	Roots	Unspecified	2			4.6×10^{-3}	5.7×10^{-3}	5.3×10^{-4}	8.6×10^{-3}	1
Tubers	Tubers	Unspecified	6	1.46×10^{-3}	2.4	2.0×10^{-3}	1.8×10^{-3}	6.2×10^{-4}	4.8×10^{-3}	1
Leafy Vegetables	Leaves	Unspecified	2			1.1×10^{-3}	1.3×10^{-3}	1.9×10^{-4}	2.0×10^{-3}	1

TABLE 20. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Sr

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	Loam	8	5.1×10^{-2}	1.3	5.3×10^{-2}	1.3×10^{-2}	3.6×10^{-2}	6.5×10^{-2}	1
	Stems, shoots	Loam	7	1.5×10^{-1}	2.5	2.0×10^{-1}	1.5×10^{-1}	4.2×10^{-2}	4.2×10^{-1}	1
Fruits	Fruits	All	16	1.0×10^{-1}	3.7	2.2×10^{-1}	3.0×10^{-1}	1.1×10^{-2}	8.8×10^{-1}	4
		Loam	10	1.5×10^{-1}	3.7	3.1×10^{-1}	3.6×10^{-1}	2.9×10^{-2}	8.8×10^{-1}	3
		Clay	6	5.2×10^{-2}	3.0	8.1×10^{-2}	7.8×10^{-2}	1.1×10^{-2}	1.9×10^{-1}	1
Grasses	Grain	Loam	9	3.4×10^{-2}	2.3	5.2×10^{-2}	7.5×10^{-2}	1.7×10^{-2}	2.5×10^{-1}	1
	Stems, shoots	Loam	9	5.2×10^{-1}	1.4	5.5×10^{-1}	1.6×10^{-1}	2.9×10^{-1}	8.0×10^{-1}	1
Herbs	Fruits	Clay	2			2.0×10^{-2}	1.8×10^{-2}	7.1×10^{-3}	3.3×10^{-2}	1
Leguminous Vegetables	Grain	Loam	26	2.8×10^{-1}	3.0	4.9×10^{-1}	5.7×10^{-1}	2.0×10^{-2}	2.5	3
Other Crops		Loam	4	2.1×10^{-1}	1.2	2.2×10^{-1}	3.0×10^{-2}	1.8×10^{-1}	2.4×10^{-1}	2
Non-leafy Vegetables	Fruits	All	15	1.1×10^{-1}	3.7	2.2×10^{-1}	2.3×10^{-1}	1.9×10^{-2}	6.5×10^{-1}	4
		Loam	10	2.1×10^{-1}	2.8	3.1×10^{-1}	2.4×10^{-1}	4.8×10^{-2}	6.5×10^{-1}	3
		Clay	5	3.2×10^{-2}	2.2	4.4×10^{-2}	4.8×10^{-2}	1.9×10^{-2}	1.3×10^{-1}	1
	Grain	Loam	3	1.1×10^{-1}	1.4	1.1×10^{-1}	3.4×10^{-2}	8.2×10^{-2}	1.5×10^{-1}	1
	Leaves	Loam	3	1.7	1.5	1.7	6.8×10^{-1}	1.2	2.5	1
	Roots	Loam	3	9.7×10^{-1}	1.1	9.8×10^{-1}	1.3×10^{-1}	8.5×10^{-2}	1.1	1

TABLE 20. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Sr (Cont.)

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Pasture	Stems, shoots	All	5	7.8×10^{-1}	1.1	7.9×10^{-1}	8.8×10^{-2}	6.9×10^{-1}	9.1×10^{-1}	3
		Sand	1			7.2×10^{-1}				1
		Loam	4	8.0×10^{-1}	1.1	8.1×10^{-1}	9.2×10^{-2}	6.9×10^{-1}	9.1×10^{-1}	2
Root Crops	Leaves	Unspecified	10	1.4×10^{-1}	5.4	3.5×10^{-1}	3.7×10^{-1}	1.3×10^{-2}	9.4×10^{-1}	1
	Roots	All	12	4.1×10^{-2}	5.4	1.3×10^{-1}	2.4×10^{-1}	3.2×10^{-3}	8.7×10^{-1}	3
		Clay	1			5.4×10^{-2}				1
Tubers	Tubers	All	29	4.5×10^{-1}	3.0	7.6×10^{-1}	8.0×10^{-1}	5.3×10^{-2}	3.6	3
		Sand	6	8.2×10^{-1}	1.9	9.6×10^{-1}	5.8×10^{-1}	3.5×10^{-1}	1.9	1
		Loam	8	3.6×10^{-1}	2.5	4.9×10^{-1}	3.5×10^{-1}	9.0×10^{-2}	1.1	1
		Clay	7	4.0×10^{-1}	5.7	1.1	1.4	5.3×10^{-2}	3.6	2
Leafy Vegetables	Leaves	All	36	9.8×10^{-1}	3.5	1.7	1.4	5.2×10^{-2}	5.0	6
		Loam	22	1.8	1.9	2.2	1.3	6.6×10^{-1}	5.0	3
		Clay	2			8.6×10^{-2}	4.8×10^{-2}	5.2×10^{-2}	1.2×10^{-1}	1
Maize	Grain	Clay	2			3.4×10^{-2}	1.6×10^{-2}	2.2×10^{-2}	4.5×10^{-2}	1

TABLE 21 SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Tc

Plant Group	Plant Compartment	Soil Group	N	GM	GSD	AM	SD	Min	Max	#ref
Cereals	Grain	Unspecified	1			3.0×10^{-2}				1
Leguminous Vegetables	Grain	Unspecified	2			5.0×10^{-1}	4.2×10^{-1}	2.0×10^{-1}	8.0×10^{-1}	1
Non-leafy Vegetables	Fruits, heads, berries, buds	Unspecified	2			3.0×10^{-1}		3.0×10^{-1}	3.0×10^{-1}	1
	All plant	Unspecified	2			3.0×10^{-1}	2.8×10^{-1}	1.0×10^{-1}	5.0×10^{-1}	1
Root Crops	Roots	Unspecified	1			1.9				1
Tubers	Tubers	Unspecified	3	5.0×10^{-1}	7.15	1.5	2.2	8.0×10^{-2}	4.0	1
Leafy Vegetables	Leaves	Unspecified	6	7.2×10^{-1}	2.08	8.4×10^{-1}	3.8×10^{-1}	1.7×10^{-1}	1.3	1

TABLE 22. SUBTROPICAL ENVIRONMENT: SOIL TO PLANT TRANSFER FACTOR (F_v) VALUES FOR Zn

Plant group	Plant compartment	Soil group	N	GM	GSD	AM	SD	Min	Max	#ref
Leguminous Vegetables	Grain, seeds and pods	All	4	6.3×10^{-1}	2.0	7.2×10^{-1}	3.5×10^{-1}	2.2×10^{-1}	1.0	2
		Loam	3	8.9×10^{-1}	1.1	8.9×10^{-1}	1.2×10^{-1}	8.0×10^{-1}	1.0	1
Other crops		Loam	3	1.1	1.1	1.1	1.4×10^{-1}	9.8×10^{-1}	1.3	1
Non-Leafy Vegetables	Fruits, heads, beries, buds	Loam	3	1.3	1.1	1.3	1.6×10^{-1}	1.2	1.5	1
Pasture	Stems, shoots	Loam	3	1.8	1.1	1.8	1.2×10^{-1}	1.7	1.9	1
Root Crops	Leaves	Unspecified	10	1.1×10^{-1}	3.5	2.2×10^{-1}	2.7×10^{-1}	2.2×10^{-2}	8.7×10^{-1}	1
	Roots	Unspecified	11	1.1×10^{-1}	3.2	2.3×10^{-1}	4.0×10^{-1}	2.4×10^{-2}	1.4	2
Leafy Vegetables	Leaves	All	18	8.9×10^{-1}	3.8	2.0	3.3	1.1×10^{-1}	1.4×10^{1}	3
		Loam	6	1.5	2.1	1.9	1.2	7.1×10^{-1}	3.1	1
	Roots	Unspecified	2			3.3×10^{-1}	1.1×10^{-1}	2.5×10^{-1}	4.1×10^{-1}	1
	Stems, shoots	Unspecified	2			2.2×10^{-1}	9.9×10^{-2}	1.5×10^{-1}	2.9×10^{-1}	1

4. EVALUATION OF THE DATA ON RADIONUCLIDE TRANSFERS TO PLANTS IN TROPICAL AND SUBTROPICAL ENVIRONMENTS

Altogether, experiments performed in the tropical and sub-tropical environments provided information on 15 radionuclides. Of the available data, 29% were related to ^{137}Cs, 17% to ^{90}Sr, 16% to neutron activation products (Co, Zn, Mn), around 21% to naturally occurring radionuclides, and around 17% to other radionuclides (Ag, Am, Pu, Tc etc.).

The statistical parameters of radionuclide transfers from all studied soils to all studied plants are given in Table 23. For most cases, the orders of magnitude of difference between maximum and minimum F_v values were larger than a factor of three, reaching a factor of five for radiocaesium transfers in tropical environments.

The coefficient of variation (ratio of the standard deviation to the mean–*CV*) was, in all these cases, larger than one. *CV*s for Cs were 2.6 and 3.2, respectively, for tropical and subtropical environments. The maximum variability was found for Ra transfers to plants in tropical environments ($CV = 4$).

For Cs, arithmetic and geometric mean values calculated for the tropical environments were one order of magnitude higher than those calculated for subtropical environments. Similar behavior was observed for Sr. However, this difference is chiefly attributable to the different plants considered for each climatic region.

For both tropical and subtropical environments, the highest F_v values were observed for Zn and Sr (geometric means for the entire set of data for these radionuclides were 1.2 and 0.96 for tropical environments, and 0.41 and 0.24 for subtropical environments). The lowest F_v value were those for Th (geometric mean of 4.0×10^{-4}). Geometric means for Cs for these data sets were 0.21 for tropical and 0.05 for subtropical environments.

TABLE 23 MAIN RADIONUCLIDES IN TROPICAL AND SUBTROPICAL DATABASES: STATISTICAL VALUES

Elem.	Tropical					Subtropical				
	N	AM	GM	CV	DIF	N	AM	GM	CV	DIF
Co	100	4.5×10^{-1}	1.0×10^{-1}	1.5	4	82	2.5×10^{-1}	4.5×10^{-2}	1.3	4
Cs	278	1.6	2.1×10^{-1}	2.6	5	409	3.6×10^{-1}	4.9×10^{-2}	3.2	4
I	-	-	-	-	-	27	4.5×10^{-2}	1.6×10^{-2}	1.3	3
Pb	67	1.4×10^{-1}	1.3×10^{-2}	1.6	4	-	-	-	-	-
Ra	275	2.5	6.2×10^{-2}	4.0	5	-	-	-	-	-
Sr	140	3.4	9.6×10^{-1}	1.9	4	209	6.3×10^{-1}	2.4×10^{-1}	1.4	3
U	179	2.3×10^{-1}	6.1×10^{-2}	1.3	3	-	-	-	-	-
Zn	129	2.3	1.2	1.8	3	59	1.0	4.1×10^{-1}	1.9	2

* Orders of magnituded of difference between maximum and minimum values.

Statistical distributions of soil-to-plant transfer factors for some plant groups were examined when a reasonable quantity of F_v values were available. Transfer factor distributions were analysed for various combinations of environment/plant group/radionuclide (Table 24). In every case, it was found that the uncertainity in assessment of transfer factor values was significantly less than that for the case when only environment/radionuclide combinations were considered (Table 23). In most cases, when the Shapiro-Wilk normality test [11, 19] was applied to the data set, the data were acceptably fitted by a lognormal distribution. The last column of Table 24 gives information on the coefficient of variation for the lognormal distribution:

$$CVlog=exp(\mu)[exp(\sigma^2)-1]^{1/2}$$

where μ and σ the mean and the standard deviation of $\ln(F_v)$.

Exploring the influence of soil group on radionuclide transfer factors was difficult due to insufficient data clusters involving the same combinations of environment/plant group/radionuclide/ for different soil types.

In the case of the Cs transfer to Tubers, a substantial difference was observed. Sand soil showed a significantly larger transfer than clay or Loam soils. Transfer of Cs to Fruits shows similar behavior on Clay and Loam soils, though with greater variability in the later case, and a relatively larger AM value. The transfer of Sr to Tubers is greatest in Sand soils. In the case of K and transfers from soil to Tuber plants, Loam soil shows the greatest transfer, whereas for Sand soil, the transfer is less.

CONCLUSIONS

As it is was found in the current and in the previous section, F_v values present a wide variability which constitutes a strong limitation for their application. The concentration of a radionuclide in a soil is not the only factor influencing its uptake by plants. Mean values reported in this investigation would be used only as a tendency for the radionuclide transfer from soil to plants. In this paper more than 2200 F_v values from tropical and subtropical environments were statistically analyzed with the purpose of exploring the influence of crop types and soil properties on the radionuclide uptake by plants. Various radionuclides have been examined, however more information was found for radioisotopes of Cs, Ra, Sr, U, Zn and Co.

The foremost results obtained are the following.

Wide F_v variability was found for all radionuclides when data were unified for all plants and soils. CV ranged from 1.3 to 4.0 (Table 23). The difference in the orders of magnitude between maximum and minimum Fv values ranged from 2 (Zn) to 5 (Cs).

When different soil group/plant type combinations are separately considered Fv variability was markedly lower. In most combinations, CV is less than 2. Additionally, it was found that for many combinations Fv reasonably fit a log-normal probability distribution (see Table 24).

For most plant groups, Zn and Sr present the highest F_v. Th and U present the lowest F_v values. A relatively high transfer is found for Ra accumulation in grasses. Transfer factors for Cs present an intermediate value.

Although more detailed further analysis is still necessary, the derived data allows the preliminarily conclusion that F_v values assessed for the 'all soils' soil group based on the tropical database tend to be higher that those defined based on subtropical database.

TABLE 24 TRANSFER FACTOR PROBABILITY DISTRIBUTIONS AND COEFFICIENTS OF VARIATION FOR DIFFERENT ENVIRONMENT (PLANT GROUP) RADIONUCLIDE COMBINATIONS

Environment	Plant Group	Rad.	Probability distribution	*CV*	CV_{log}
Tropical	Grasses	Cs	-	2.2	-
		Pb	Lognormal	1.0	0.9
		Ra	Lognormal	2.3	2.7
		Sr	-	1.9	-
		U	-	0.3	-
		Zn	Lognormal	0.7	0.7
	Leguminous Vegetables	Co	Lognormal	1.0	1.0
		Cs	-	2.0	-
		Pb	Lognormal	0.6	1.1
		Ra	Lognormal	1.4	2.7
		Sr	-	1.8	-
		Th	Lognormal	1.0	1.2
		U	Lognormal	1.2	2.4
		Zn	Lognormal	0.4	0.4
	Leafy Vegetables	Co	-	0.6	-
		Cs	-	0.5	-
		Ra	Lognormal	1.5	3.0
		Sr	Lognormal	0.6	0.7
		Th	Lognormal	0.7	0.7
		U	Lognormal	1.4	2.0
		Zn	Lognormal	0.4	0.3
	Non-Leafy Vegetables	Co	Lognormal	0.5	0.5
		Cs	-	1.5	-
		Ra	Lognormal	1.8	4.3
		Sr	Lognormal	0.8	1.2
		U	Lognormal	1.8	2.8
		Zn	Lognormal	0.4	0.5

TABLE 24 TRANSFER FACTOR PROBABILITY DISTRIBUTIONS AND COEFFICIENTS OF VARIATION FOR DIFFERENT ENVIRONMENT (PLANT GROUP) RADIONUCLIDE COMBINATIONS (Cont.)

Environment	Plant Group	Elem..	Probability distribution	CV	CV_{log}
	Root Crops	Co	Lognormal	0.5	0.5
		Cs	Lognormal	0.5	0.7
		Ra	Lognormal	1.6	3.3
		Sr	Lognormal	0.5	0.5
		Pb	Lognormal	0.5	0.4
		Th	Lognormal	0.5	0.6
		U	Lognormal	1.0	3.9
		Zn	Lognormal	0.5	0.6
Subtropical	Leguminous vegetables	Co	Lognormal	0.5	0.5
		Cs	Lognormal	1.7	2.5
		Sr	Lognormal	1.2	1.5
		Zn	-	0.4	-
	Leafy vegetables	Co	Lognormal	1.5	4.8
		Cs	Lognormal	1.5	5.1
		I	Lognormal	0.9	1.8
		Sr	Lognormal	0.9	1.9
		Zn	Lognormal	1.8	2.1
	Non-leafy vegetables	Co	Lognormal	0.1	0.1
		Cs	Lognormal	1.6	2.7
		I	Lognormal	1.5	3.4
		Sr	Lognormal	1.2	2.9
		Zn	Lognormal	0.1	0.1
	Root crops	Co	Lognormal	2.2	3.2
		Cs	Lognormal	1.5	3.0
		I	Lognormal	0.5	0.5
		Sr	Lognormal	1.4	4.5
		Zn	Lognormal	1.7	1.7
	Pasture	Cs	Lognormal	0.7	1.1
		Sr	Lognormal	0.1	0.1
		Zn	Lognormal	0.1	0.1

The numbers of entries available for tropical and subtropical ecosystems are much less than those used for the data evaluation from the temperate environments. In addition, some changes in plant classification adopted for this document should be made to cover diversity of exposure pathways typical for the tropical environments. Additional uncertainty in the application of the data provided by the IAEA-TECDOC can be assigned with the use of different climate classification schemes. The simplest approach has been used in this document. However, further analysis is needed in order to estimate sensitivity of the derived transfer factor values for various soil/plant groups to the approach used for climate classification. Perhaps a better grouping of the available data can be done on this basis.

Overall, these facts show a need for further research aimed at the improving information on radionuclide transfer to tropical and subtropical plants including experimental studies and advanced data analysis.

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APPENDIX

List of publication used for evaluation of radionuclide transfer factors from soil to plants: tropical and subtropical environments

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TRANSFER TO RICE

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Abstract

One of the critical foods for the intake of radionuclides by humans is rice (*Oryza sativa L.*), which is the dominant staple food crop in humid tropical and sub-tropical countries across the globe. Cultivation methods have important effects on plant uptake of radionuclides from soil. Most rice is produced under flooded conditions, i.e. in fields with a water layer of 5-15 cm deep on soil. This is unlike the situation in unsaturated fields where the soil conditions are significantly different, meaning that the soil-to-rice transfer factors (F_vs) need to be discussed separately from F_v values of other cereals grown under unsaturated field conditions. In this section, rice refers to the crop grown on wet paddy fields; since water management is the dominant method for rice cultivation this is the most common type. Rice *Fv* values were collected from papers in international journals, books and proceedings with peer reviewing processes and F_v values based on dry weight rice grain were summarized here. F_v data for fallout isotopes (e.g. ^{90}Sr and ^{137}Cs), neutron activation products (e.g. ^{54}Mn, ^{60}Co, *etc.*) or naturally existing radionluclides (e.g. ^{210}Pb, ^{226}Ra, ^{238}U, *etc.*) and those for naturally existing stable elements under agricultural field conditions were listed.

1. MAIN FEATURES INFLUENCING RADIONUCLIDE UPTAKE IN RICE

One of the critical foods for determining the intake of radionuclides by humans is rice (*Oryza sativa L.*), which is the dominant staple food crop in humid tropical countries across the globe. Although the total planted area for rice is smaller than that for wheat, more than half of the world's population depends on rice as their main source of calories (i.e. rice supplies a third of their total caloric intake). Almost 90% of the world's rice is produced and consumed in Asia. Since the population there is growing, the domestic demand for rice is increasing.

Rice varieties are classified according to their area of cultivation: paddy or wet rice, which is suited to paddy fields; upland or dry rice, which is suited to dry fields; and floating rice, which is grown in flood-prone areas. Cultivation methods have important effects on plant uptake of radionuclides from soil. Most rice is produced under flooded conditions, e.g. in fields with a water layer of 5-15 cm deep.[6] In this paper, rice refers to the crop grown on wet paddy fields; since water management is the dominant method for rice cultivation, this paddy

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[6] Rice paddy fields are generally unflooded during the non-cultivated period, however, during the cultivation period, they are flooded and readily reach reducing conditions in most cases.

fields with a water layer of 5-15 cm deep.[1] In this chapter, rice refers to the crop grown on wet paddy fields; since water management is the dominant method for rice cultivation, this paddy rice is the most common type. In addition, no data are available for upland rice and floating rice; further studies are needed for these farming technologies.

For paddy rice, no crop rotation is necessary, and rice has been planted in the same places for hundreds of years. Under flooded conditions, oxygen is depleted relatively quickly by the respiration of soil microorganisms and plant roots. After the disappearance of molecular oxygen, various degrees of anaerobiosis occur and the chemical reduction of mineral nutrients takes place. Subsequently, phosphate and iron species sorbed on the soil particles are gradually changed to soluble forms, which are easily taken up by rice plants. In addition, under these conditions, nitrogen is present as the positive ammonium ion, which is efficiently used by rice plants.

This is unlike the situation in unsaturated fields[2] where the soil conditions are significantly different. Because of this difference, the soil-to-rice transfer factor (F_v) needs to be discussed separately from F_v values of other cereals grown under unsaturated field conditions. For instance, in Fig.1, F_v data for stable elements for brown rice samples are compared with those for cereal samples (wheat and barley 'WB': 7 samples) collected in Japan: the comparison is made by plotting the ratios of F_v-GMs for brown rice divided by F_v-GMs for WB. Most of the elements, with the exception of K, Fe, As, Sr, and Mo, do not show significant differences (t-test: $p<0.05$) with the F_v-GMs of WB.

The F_vs of rare earth elements, Th and U, are amongst the lowest in brown rice and in WB, and the F_vs tend to be lower in brown rice than in WB. For Sr, the value is significantly higher for WB; indeed, all alkaline earth metals show higher F_v-GMs for WB than for brown rice. It should be noted that Sr and other alkaline earth metals, i.e. Ca and Ba, have almost the same concentrations in paddy fields as in upland field soils [1].

The results indicate it is likely that Ca, Sr and Ba transfers to WB are higher than those to brown rice. On the other hand, As accumulation is significantly higher in brown rice than in WB, although As is not an essential element for brown rice. The mechanisms should be clarified because As can be a toxic element if too much is ingested. Mo and Fe, essential elements to plants, are redox sensitive so that the waterlogged condition might affect their mobility in paddy fields. Both F_v similarities and differences exist between wheat and barley (WB) and brown rice samples.

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[1] Rice paddy fields are generally unflooded during the non-cultivated period, however, during the cultivation period, they are flooded and readily reach reducing conditions in most cases.

[2] Here unsaturated field conditions extend to soils at field capacity. It is recognized that seasonally saturated soils are used for arable and pastoral purposes, but this is distinct from the special flooded conditions under which rice is grown. It is also recognized that some varieties of rice, i.e. upland rice, may be cultivated under unsaturated conditions like other cereal crops.

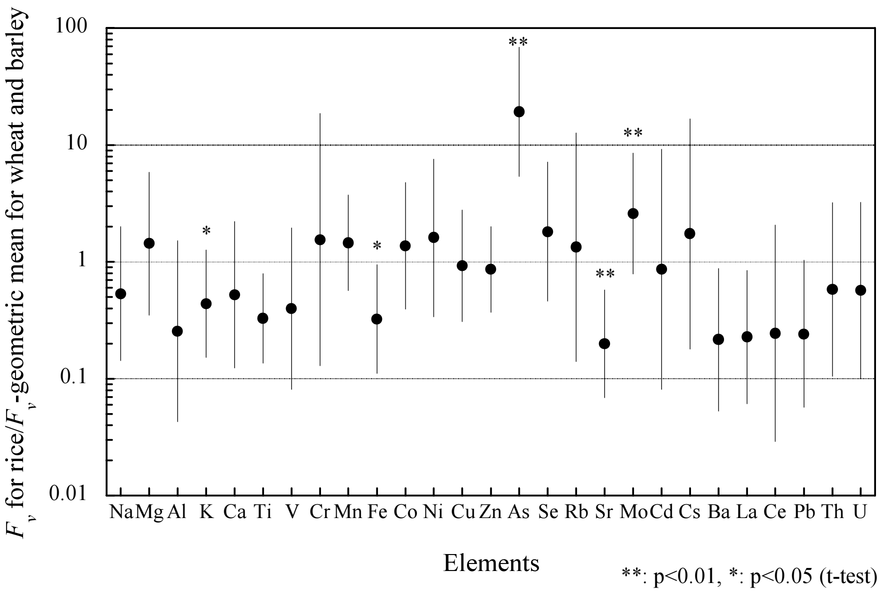


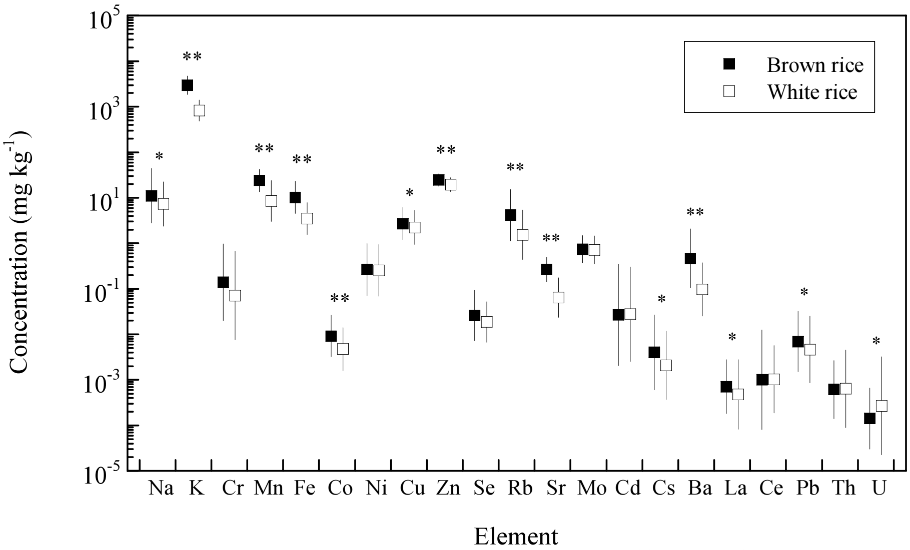
FIG. 1 Comparison of geometric means of F_vs for brown rice (for all the samples and three soil groups) and for wheat and barley. Bars show upper and lower 95% confidence limits [modified after 1].

A whole grain of rice has *several* layers; the F_v may refer to unhulled rice, brown rice (hulled rice), or white rice. People do not consume the outer layer of rice grain, known as the hull, and almost no data exist for unhulled rice. Brown rice is produced by removing the external hull, so brown rice is sometimes referred to as hulled rice. White, or polished, rice is obtained when brown rice is milled to remove the inner hull and polished to remove the bran, the germ layer, and the aleurone layer (a layer filled with health-supportive and essential fats). White rice is simply a refined starch; consequently, the chemical compounds and elements in brown rice and white rice can be different. Figure 2 shows geometric mean values of concentrations of different elements for white and brown rice [1].

The concentrations are different for many elements, indicating that similar differences in terms of F_v values for many elements can be expected. This information shows that specifying in every case which type of rice has been analyzed is extremely important; however, many reports do not give this information.

2. CRITERIA FOR DATA SELECTING

The IAEA INIS database, as well as the most widespread publishers' databases, such as *Science Direct,* were used for reviewing the literature information sources. Rice F_v data were collected from papers in international journals, books and proceedings with a peer reviewing processes. To avoid double counting of a single datum, summary data from reviews were excluded from the database.



**: $p<0.01$, *: $p<0.05$ (t-test)

FIG. 2. Comparison of concentrations of different chemical elements in white and brown rice (modified after [1]).

Since dry weight basis values are compiled in this IAEA-TECDOC, data that were not clearly indicated as dry or wet mass basis F_v (or concentrations in grains and soils) were excluded. If the grain treatment process (white rice or brown rice) was not clearly indicated, such data were considered as brown rice.

As already pointed out in this IAEA-TECDOC, types of soil and cultivation information are important factors because behaviours of radionuclides depend on soil properties and sampling conditions. However, F_v data without soil information were included when the sampling conditions was clearly indicated. It should be noted that such data were not classified by F_v for soil categories such as Sand, Loam, Clay, and Organic, but were included to the 'all' data category.

3. DERIVED TRANSFER FACTOR VALUES

Available F_v values are given in Table 1. The data for elements followed by an asterisk (*) were obtained for fallout isotopes (e.g. ^{90}Sr and ^{137}Cs) or neutron activation products (e.g, ^{54}Mn, ^{60}Co, *etc.*). Absence of an asterisk indicates F_v data for naturally existing elements under agricultural field conditions. Because there are no available F_v values for some important radionuclides used in radiation dose assessment, information on relevant stable elements is also given in this Table. F_v values of some elements, i.e. Sr, Cs, Pb, Ra, Th, and U, for brown rice and white rice are given in Fig. 3.

TABLE 1. TRANSFER FACTORS OF RADIONUCLIDES FROM SOIL TO RICE GRAIN

Element	Rice Type	Soil Group	N	GM	GSD	AM	SD	Min	Max	# Ref
Ba	White rice	All	37	3.9×10^{-4}	2.0	5.0×10^{-4}	4.3×10^{-4}	8.5×10^{-5}	1.9×10^{-3}	1
Ba	Brown rice	All	50	1.8×10^{-3}	2.0	2.2×10^{-3}	1.6×10^{-3}	3.4×10^{-4}	7.8×10^{-3}	1
Ca	White rice	All	37	4.1×10^{-3}	2.0	5.2×10^{-3}	3.8×10^{-3}	1.3×10^{-3}	1.7×10^{-2}	1
Ca	Brown rice	All	50	8.8×10^{-3}	2.1	1.2×10^{-2}	9.7×10^{-3}	2.2×10^{-3}	4.6×10^{-2}	1
Cd	White rice	All	37	9.1×10^{-2}	3.1	1.7×10^{-1}	2.1×10^{-1}	1.3×10^{-2}	8.5×10^{-1}	1
Cd	Brown rice	All	50	9.4×10^{-2}	3.3	1.9×10^{-1}	2.5×10^{-1}	9.0×10^{-3}	1.1×10^{0}	1
Ce	White rice	All	27	3.3×10^{-5}	2.6	6.3×10^{-5}	1.2×10^{-4}	8.5×10^{-6}	5.7×10^{-4}	1
Ce	Brown rice	All	33	3.3×10^{-5}	2.9	5.4×10^{-5}	6.1×10^{-5}	1.9×10^{-6}	3.4×10^{-4}	1
Co	White rice	All	37	4.6×10^{-4}	1.9	5.8×10^{-4}	5.0×10^{-4}	1.3×10^{-4}	2.7×10^{-3}	1
Co	Brown rice	All	49	9.3×10^{-4}	1.9	1.1×10^{-3}	9.4×10^{-4}	2.4×10^{-4}	6.4×10^{-3}	1
Co*	Brown rice	All	5	5.1×10^{-3}	1.7	5.7×10^{-3}	2.8×10^{-3}	2.2×10^{-3}	1.0×10^{-2}	2
Cr	White rice	All	37	1.5×10^{-3}	3.5	2.9×10^{-3}	3.5×10^{-3}	1.1×10^{-4}	1.5×10^{-2}	1
Cr	Brown rice	All	50	2.0×10^{-3}	3.5	4.0×10^{-3}	4.7×10^{-3}	2.3×10^{-4}	1.9×10^{-2}	1
Cs	White rice	All	59	6.0×10^{-3}	2.3	8.8×10^{-4}	1.1×10^{-3}	1.1×10^{-4}	6.4×10^{-3}	3
Cs	White rice	Loam	10	7.7×10^{-4}	3.5	1.5×10^{-3}	1.9×10^{-3}	1.1×10^{-4}	6.4×10^{-3}	1
Cs	White rice	Clay	15	5.4×10^{-4}	1.7	6.3×10^{-4}	3.6×10^{-4}	2.4×10^{-4}	1.4×10^{-3}	2
Cs	Brown rice	All	37	9.4×10^{-4}	3.1	1.9×10^{-3}	3.3×10^{-3}	1.3×10^{-4}	1.6×10^{-2}	1
Cs	Brown rice	Loam	16	1.2×10^{-3}	4.0	3.0×10^{-3}	4.8×10^{-3}	1.3×10^{-4}	1.6×10^{-2}	1
Cs	Brown rice	Clay	21	7.8×10^{-4}	2.4	1.1×10^{-3}	9.1×10^{-4}	1.4×10^{-4}	3.3×10^{-3}	1
Cs*	White rice	All	227	4.7×10^{-3}	6.6	2.6×10^{-2}	4.7×10^{-2}	1.3×10^{-4}	1.9×10^{-1}	4
Cs*	White rice	Sand	3	8.8×10^{-2}	1.9	1.0×10^{-1}	6.0×10^{-2}	4.9×10^{-2}	1.7×10^{-1}	1
Cs*	White rice	Clay	7	4.0×10^{-2}	5.2	6.7×10^{-2}	3.7×10^{-2}	1.1×10^{-3}	1.0×10^{-1}	2
Cs*	Brown rice	All	239	1.2×10^{-2}	5.3	4.9×10^{-2}	9.4×10^{-2}	1.1×10^{-3}	6.1×10^{-1}	12
Cs*	Brown rice	Sand	4	4.3×10^{-2}	4.9	8.8×10^{-2}	8.8×10^{-2}	7.1×10^{-3}	1.7×10^{-1}	2
Cs*	Brown rice	Loam	24	7.5×10^{-3}	4.1	2.5×10^{-2}	5.9×10^{-2}	1.1×10^{-3}	2.8×10^{-1}	5
Cs*	Brown rice	Clay	16	1.7×10^{-2}	5.8	4.9×10^{-2}	5.4×10^{-2}	1.4×10^{-3}	1.5×10^{-1}	5

TABLE 1. TRANSFER FACTORS OF RADIONUCLIDES FROM SOIL TO RICE GRAIN (Cont.)

Element	Rice Type	Soil Group	N	GM	GSD	AM	SD	Min	Max	# Ref
Fe	White rice	All	37	9.8×10^{-5}	1.8	1.2×10^{-4}	1.1×10^{-4}	3.8×10^{-5}	6.5×10^{-4}	1
Fe	Brown rice	All	50	2.9×10^{-4}	1.7	3.3×10^{-4}	2.0×10^{-4}	7.5×10^{-5}	1.3×10^{-3}	1
I	White rice	All	28	2.0×10^{-3}	3.2	3.1×10^{-3}	2.3×10^{-3}	1.3×10^{-4}	7.8×10^{-3}	1
I	Brown rice	All	14	5.0×10^{-3}	2.6	6.9×10^{-3}	5.1×10^{-3}	1.5×10^{-4}	2.0×10^{-2}	1
I *	Brown rice	All	8	3.8×10^{-3}	2.1	4.6×10^{-3}	2.5×10^{-3}	1.1×10^{-3}	7.6×10^{-3}	2
K	White rice	All	37	6.2×10^{-2}	1.8	7.2×10^{-2}	4.3×10^{-2}	1.8×10^{-2}	2.4×10^{-1}	1
K	Brown rice	All	50	2.2×10^{-1}	1.7	2.5×10^{-1}	1.6×10^{-1}	8.7×10^{-2}	7.8×10^{-1}	1
La	White rice	All	34	3.5×10^{-5}	2.5	7.8×10^{-5}	2.4×10^{-4}	4.6×10^{-6}	1.4×10^{-3}	1
La	Brown rice	All	45	4.9×10^{-5}	1.9	6.3×10^{-5}	6.3×10^{-5}	1.8×10^{-5}	4.2×10^{-4}	1
Mn	White rice	All	37	1.5×10^{-2}	1.8	1.8×10^{-2}	9.9×10^{-3}	5.4×10^{-3}	4.9×10^{-2}	1
Mn	Brown rice	All	50	4.6×10^{-2}	1.6	5.1×10^{-2}	2.5×10^{-2}	1.7×10^{-2}	1.2×10^{-1}	1
Mn*	Brown rice	All	5	2.6×10^{-1}	1.7	2.9×10^{-1}	1.5×10^{-1}	1.2×10^{-1}	5.2×10^{-1}	3
Mn*	Brown rice	Sand	1			2.3×10^{-1}				1
Mn*	Brown rice	Loam	4	2.6×10^{-1}	1.8	3.0×10^{-1}	1.7×10^{-1}	1.2×10^{-1}	5.2×10^{-1}	3
Na	White rice	All	37	6.9×10^{-4}	1.9	8.7×10^{-4}	8.1×10^{-4}	2.0×10^{-4}	4.7×10^{-3}	1
Na	Brown rice	All	50	9.8×10^{-4}	1.9	1.3×10^{-3}	1.2×10^{-3}	3.2×10^{-4}	6.9×10^{-3}	1
Ni	White rice	All	37	1.4×10^{-2}	2.2	1.9×10^{-2}	1.6×10^{-2}	3.0×10^{-3}	6.8×10^{-2}	1
Ni	Brown rice	All	50	1.3×10^{-2}	2.2	1.8×10^{-2}	1.6×10^{-2}	3.4×10^{-3}	8.7×10^{-2}	1
P	Brown rice	All	50	2.4×10^{0}	1.8	2.8×10^{0}	1.6×10^{0}	3.7×10^{-1}	9.4×10^{0}	1
Pb*	Brown rice	All	2	7.5×10^{-3}	1.9	8.4×10^{-3}	5.2×10^{-3}	4.7×10^{-3}	1.2×10^{-2}	2
Pb	White rice	All	25	2.5×10^{-4}	2.9	4.1×10^{-4}	4.6×10^{-4}	3.6×10^{-5}	1.8×10^{-3}	2
Pb	White rice	Loam	10	2.0×10^{-4}	2.5	3.0×10^{-4}	3.3×10^{-4}	7.2×10^{-5}	1.1×10^{-3}	1
Pb	White rice	Clay	14	2.5×10^{-4}	3.0	4.2×10^{-4}	4.7×10^{-4}	3.6×10^{-5}	1.8×10^{-3}	1
Pb	Brown rice	All	38	3.3×10^{-4}	2.4	5.7×10^{-4}	1.0×10^{-3}	8.8×10^{-5}	5.9×10^{-3}	2
Pb	Brown rice	Loam	16	2.7×10^{-4}	1.8	3.2×10^{-4}	2.0×10^{-4}	1.2×10^{-4}	9.0×10^{-4}	1
Pb	Brown rice	Clay	21	3.4×10^{-4}	2.3	5.1×10^{-4}	7.2×10^{-4}	8.8×10^{-5}	3.5×10^{-3}	1

TABLE 1. TRANSFER FACTORS OF RADIONUCLIDES FROM SOIL TO RICE GRAIN (Cont.)

Element	Rice Type	Soil Group	N	GM	GSD	AM	SD	Min	Max	# Ref
Po*	Brown rice	All	2	1.3×10^{-2}	1.5	1.3×10^{-2}	5.2×10^{-3}	9.4×10^{-3}	1.7×10^{-2}	2
Ra*	White rice	All	3	4.7×10^{-4}	2.8	6.8×10^{-4}	7.1×10^{-4}	2.2×10^{-4}	1.5×10^{-3}	2
Ra*	Brown rice	All	37	9.1×10^{-4}	3.1	2.3×10^{-3}	5.1×10^{-3}	1.9×10^{-4}	2.8×10^{-2}	5
Ra*	Brown rice	Loam	14	7.8×10^{-4}	2.4	1.3×10^{-3}	2.2×10^{-3}	2.7×10^{-4}	8.8×10^{-3}	2
Ra*	Brown rice	Clay	18	5.7×10^{-4}	1.7	6.8×10^{-4}	5.9×10^{-4}	2.5×10^{-4}	2.9×10^{-3}	2
Rb	White rice	All	37	5.6×10^{-2}	2.7	1.0×10^{-1}	1.5×10^{-1}	7.3×10^{-3}	6.6×10^{-1}	1
Rb	Brown rice	All	50	1.2×10^{-1}	3.1	2.4×10^{-1}	4.1×10^{-1}	8.9×10^{-3}	2.2×10^{0}	1
Se	White rice	All	26	5.4×10^{-2}	1.9	6.3×10^{-2}	3.3×10^{-2}	9.0×10^{-3}	1.4×10^{-1}	1
Se	Brown rice	All	41	6.7×10^{-2}	2.0	8.2×10^{-2}	5.8×10^{-2}	9.5×10^{-2}	2.9×10^{-1}	1
Sr	White rice	All	41	1.0×10^{-3}	1.8	1.2×10^{-3}	7.7×10^{-4}	3.8×10^{-4}	3.3×10^{-3}	3
Sr	White rice	Loam	10	8.4×10^{-4}	1.7	9.4×10^{-4}	4.7×10^{-4}	4.1×10^{-4}	1.7×10^{-3}	1
Sr	White rice	Clay	15	1.1×10^{-3}	1.9	1.3×10^{-3}	8.3×10^{-4}	3.8×10^{-4}	3.3×10^{-3}	2
Sr	Brown rice	All	35	3.0×10^{-3}	1.7	3.4×10^{-3}	1.9×10^{-3}	1.3×10^{-3}	8.2×10^{-3}	1
Sr	Brown rice	Loam	16	2.5×10^{-3}	1.6	2.8×10^{-3}	1.5×10^{-3}	1.3×10^{-3}	6.4×10^{-3}	1
Sr	Brown rice	Clay	19	3.5×10^{-3}	1.7	4.0×10^{-3}	2.0×10^{-3}	1.6×10^{-3}	8.2×10^{-3}	1
Sr *	White rice	All	55	$9.4 x 10^{-3}$	3.0	1.8×10^{-2}	2.4×10^{-2}	2.1×10^{-3}	8.8×10^{-2}	4
Sr *	White rice	Sand	3	3.7×10^{-2}	2.6	4.7×10^{-2}	3.1×10^{-2}	1.2×10^{-2}	7.2×10^{-2}	1
Sr *	White rice	Clay	7	1.7×10^{-2}	3.4	2.8×10^{-2}	2.5×10^{-2}	2.1×10^{-3}	7.5×10^{-2}	2
Sr *	Brown rice	All	40	4.6×10^{-2}	4.4	2.3×10^{-1}	9.4×10^{-1}	5.5×10^{-3}	6.0×10^{0}	10
Sr *	Brown rice	Sand	3	9.8×10^{-2}	2.3	1.2×10^{-1}	9.2×10^{-2}	4.1×10^{-2}	2.2×10^{-1}	1
Sr *	Brown rice	Loam	4	9.5×10^{-2}	8.1	2.8×10^{-1}	3.7×10^{-1}	5.5×10^{-3}	8.3×10^{-1}	3
Sr *	Brown rice	Clay	7	6.0×10^{-2}	1.6	6.6×10^{-2}	2.8×10^{-2}	2.8×10^{-2}	1.1×10^{-1}	2
Tc *	Brown rice	All	2			$<2 \times 10^{-4}$				1
Th*	White rice	All	26	1.7×10^{-4}	2.0	2.2×10^{-4}	1.6×10^{-4}	4.3×10^{-5}	6.9×10^{-4}	3
Th*	White rice	Loam	10	1.7×10^{-4}	1.9	2.1×10^{-4}	1.4×10^{-4}	6.7×10^{-5}	4.6×10^{-4}	1
Th*	White rice	Clay	14	1.9×10^{-4}	2.1	2.4×10^{-4}	1.8×10^{-4}	5.1×10^{-5}	6.9×10^{-4}	1
Th*	Brown rice	All	31	1.4×10^{-4}	4.5	1.3×10^{-3}	5.4×10^{-3}	2.2×10^{-5}	3.0×10^{-2}	4
Th*	Brown rice	Loam	12	1.3×10^{-4}	4.1	4.6×10^{-4}	1.1×10^{-3}	2.2×10^{-5}	4.0×10^{-3}	3
Th*	Brown rice	Clay	17	1.1×10^{-4}	2.7	1.9×10^{-4}	2.2×10^{-4}	2.6×10^{-5}	8.3×10^{-4}	1
Th*	Brown rice	Organic	1			3.0×10^{-2}				1

TABLE 1. TRANSFER FACTORS OF RADIONUCLIDES FROM SOIL TO RICE GRAIN (Cont.)

Element	Rice Type	Soil Group	N	GM	GSD	AM	SD	Min	Max	# Ref
U*	White rice	All	24	1.9×10^{-4}	2.0	2.3×10^{-4}	1.3×10^{-4}	4.5×10^{-5}	6.3×10^{-4}	4
U*	White rice	Loam	8	1.7×10^{-4}	2.3	2.3×10^{-4}	1.9×10^{-4}	4.7×10^{-5}	6.3×10^{-4}	1
U*	White rice	Clay	12	2.5×10^{-4}	1.4	2.6×10^{-4}	8.8×10^{-5}	1.1×10^{-4}	4.5×10^{-4}	1
U*	Brown rice	All	41	2.8×10^{-4}	8.9	3.8×10^{-3}	1.4×10^{-4}	8.6×10^{-6}	9.0×10^{-2}	8
U*	Brown rice	Sand	3	5.4×10^{-3}	2.6	7.0×10^{-3}	5.3×10^{-3}	1.9×10^{-3}	1.3×10^{-2}	2
U*	Brown rice	Loam	15	2.3×10^{-4}	10	2.3×10^{-3}	6.1×10^{-3}	8.6×10^{-6}	2.4×10^{-3}	4
U*	Brown rice	Clay	17	1.4×10^{-4}	5	4.1×10^{-4}	5.5×10^{-4}	2.3×10^{-5}	1.8×10^{-3}	3
U*	Brown rice	Organic	1			9.0×10^{-2}				1
Zn	White rice	All	37	2.0×10^{-1}	1.6	2.2×10^{-1}	8.4×10^{-2}	6.1×10^{-2}	4.0×10^{-1}	1
Zn	Brown rice	All	50	2.4×10^{-1}	1.5	2.6×10^{-1}	1.1×10^{-1}	7.3×10^{-2}	6.8×10^{-1}	1
Zn *	Brown rice	All	5	1.5×10^{0}	2.0	1.7×10^{0}	9.3×10^{-1}	5.8×10^{-1}	2.7×10^{0}	3
Zn *	Brown rice	Sand	1			2.3×10^{0}				1
Zn *	Brown rice	Loam	3	1.5×10^{0}	2.3	1.8×10^{0}	1.1×10^{0}	5.8×10^{-1}	2.7×10^{0}	1

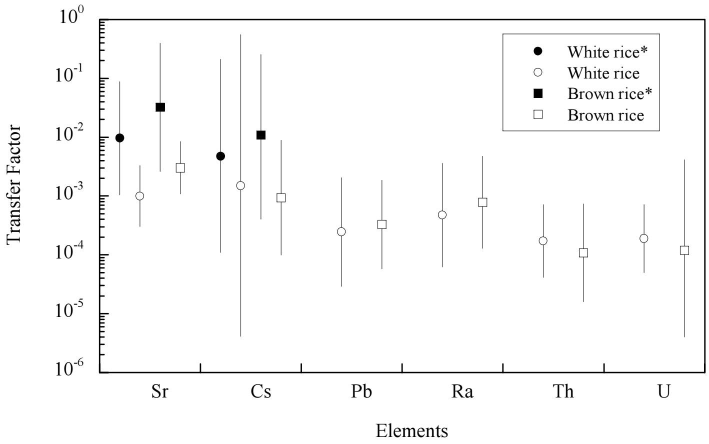


FIG. 3. Geometric means of transfer *factors of Sr, Cs, Pb, Ra, Th and U for brown rice and white rice (on a dry weight basis). F_v results for White rice* (●) and Brown rice*(■) are F_v values obtained using fallout isotopes and radiotracers. The results for White rice (○) and Brown rice (□) without asterisk were obtained using stable elements. Bars show upper and lower 95% confidence levels for F_v values.*

Under equilibrium conditions, i.e. under long-time contact with the environment, radionuclides behave similarly to their naturally existing isotopes so that naturally existing elements can be suitable analogues. However, under freshly contaminated conditions, F_v values will often be higher than for naturally existing elements due to different mobility in soil-to-plant systems (ageing effect) (Table 2).

Recently compiled data in this IAEA-TECDOC for cereals and F_v values for brown rice (from Table 1) are plotted in Fig. 4. F_v values for brown rice obtained using radionuclides are almost the same as those for cereals. Analyses of these results for some radionuclides are given below.

3.1. Strontium and Caesium

A number of reports are available on F_v values for Sr and Cs isotopes. These elements, as well as many other elements, do not distribute uniformly in the rice plant. For Sr, only 0.7 and 1.1% of the total ^{90}Sr in the whole rice plant were found in polished rice and brown rice, respectively, whereas, 5.3% and 7.4% of the total ^{137}Cs were observed in polished rice and brown rice, respectively [2]. Tsukada *et al.* [3] reported that 7% and 10% of the total stable Cs were distributed in polished rice and bran, respectively.

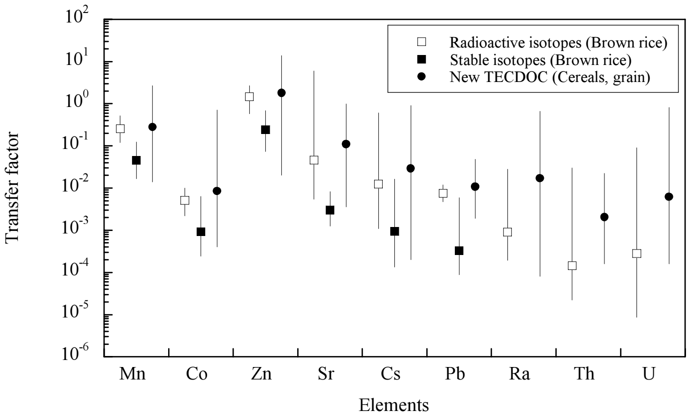


FIG. 4 Geometric means of TFs for brown rice and TFs for cereals compiled in this IAEA-TECDOC. Bar ends shows minimum and maximum TF values.

The values obtained from pot experiments are usually higher than values obtained from field observations, due to the aging effect that occurs with the water management used in paddy fields (Table 2).

The reported F_v values for strontium in brown rice from pot experiments are usually one order of magnitude higher than those from field observations (fallout ^{90}Sr) and range from 1.4×10^{-2} to 8.6×10^{-2}. The data from field observations are generally lower than values recommended

TABLE 2. SOIL-TO-RICE TRANSFER FACTORS FOR Sr AND Cs

Country/State, organization	Experimental conditions	Elements	
		Strontium	Caesium
Bangladesh	Rice, not specified (Pot experiment)	$(7.8 \times 10^{-1}–8.9 \times 10^{-1})$	$(2.5 \times 10^{-1}–3.3 \times 10^{-1})$
China	Brown rice (Pot experiment)	6.4×10^{-3} [e] $(2.1 \times 10^{-3}– 2.0 \times 10^{-2})$ $(1.8\pm0.4) \times 10^{-1}$ [c,e] $(6.7\pm0.9) \times 10^{-2}$ [c,e]	4.0×10^{-3} [e] $(2.8 \times 10^{-4} – 5.7 \times 10^{-2})$ $(5.7\pm2.8) \times 10^{-3}$ [c,e]
Republic of Korea	Brown rice (F[a], Field observation)	-	4.5×10^{-3} $(1.2 \times 10^{-3} – 1.1 \times 10^{-2})$
	Brown rice (Pot experiment)	1.2×10^{-1}	3.1×10^{-2} $(1.0 \times 10^{-2}–1.0 \times 10^{-1})$
Japan	Brown rice (F, Field observation)	1.4×10^{-2} [f]	4.1×10^{-3} [f]
	White rice (F, Field observation)	4.0×10^{-3} $(3.8\pm2.6) \times 10^{-3}$	1.6×10^{-3} $(2.6\pm2.8) \times 10^{-3}$
	White rice (S[b], Field observation)	$(7.4\pm2.7) \times 10^{-4}$	$(7.8\pm6.9) \times 10^{-4}$
Japan	White rice (F, Field observation)	$(2.1\pm0.7) \times 10^{-3}$	1.6×10^{-3} $(2.1 \times 10^{-4} – 1.2 \times 10^{-2})$
	White rice (S, Field observation)		5.6×10^{-4} $(1.1 \times 10^{-4} – 2.8 \times 10^{-3})$
Japan	Brown rice (F, Field observation)		3.3×10^{-3} $(1.1 \times 10^{-3} – 2.4 \times 10^{-2})$
	Brown rice (S, Field observation)	3.1×10^{-3} $(1.1 \times 10^{-3} – 9.1 \times 10^{-3})$	9.5×10^{-4} $(9.9 \times 10^{-5} – 9.1 \times 10^{-3})$
	White rice (S, Field observation)	8.6×10^{-4} $(2.2 \times 10^{-4} – 3.4 \times 10^{-3})$	5.9×10^{-4} $(8.9 \times 10^{-5} – 3.9 \times 10^{-3})$
Taiwan, China	White rice (F, Field observation)	3.7×10^{-2} $(5 \times 10^{-3}–8.8\ 10^{-2})$	9.7×10^{-2} $(3 \times 10^{-2} – 1.8 \times 10^{-1})$
	Brown rice (F, Field observation)	8.6×10^{-2} $(2.7 \times 10^{-2} – 2.2 \times 10^{-1})$	1.4×10^{-1} $(7 \times 10^{-2} – 2.7 \times 10^{-1})$
Taiwan, China	Rice, not specified (F, Field observation)	3.1×10^{-2}	2.2×10^{-3}
Vietnam	Rice, not specified (Pot experiment)	$(3.05\pm2.83) \times 10^{-1}$	$(1.67\pm3.41) \times 10^{-1}$
Vietnam	Rice, not specified (Pot experiment)	1.13×10^{-1} $(5.1 \times 10^{-2} – 3.3 \times 10^{-1})$	5.05×10^{-2} $(9 \times 10^{-4} – 1.4)$
Technical Reports Series No. 364 [d]	Cereals grain	1.2×10^{-1} (clay, loam pH=6) 2.1×10^{-1}(sand, pH=5) 2.0×10^{-2} (peat, pH=4)	1.0×10^{-2} (clay, loam pH=6) 2.6×10^{-2} (sand, pH=5) 8.3×10^{-2} (peat, pH=4)
Technical Reports Series No. 364 [d]	Rice (soil-to-plant)		5.0×10^{-3}

[a]F: Fallout radionuclides, [b]S: stable elements, [c] arithmetic mean, [d] expected values, [e] values were converted from fresh weight basis F_v to dry weight basis F_v by applying dry/wet ratio of 0.85, and [f] calculated from ref. [5].

by the IAEA (Technical Reports Series No. 364) [4] previously for Cereals (1.2×10^{-1} for Clay Loam soils). Also, the geometric mean for all ^{90}Sr F_v values for Cereals compiled in this IAEA-TECDOC, 1.1×10^{-1}, is higher than that obtained in field observations.

For Cs in brown rice, an F_v range of 5.7×10^{-3}–3.3×10^{-1} is observed for pot experiments, whereas field observation F_v values are usually on the order of 10^{-3}. For most cases, again, the F_v values observed under field conditions are lower than those listed in Technical Reports Series No. 364 for cereals. Among recent data reported in this IAEA-TECDOC, the geometric mean of Cs F_v values for cereals, 2.9×10^{-2}, is also higher than the field observation results for stable Cs and fallout ^{137}Cs, as shown in the Table 2.

3.2. Manganese, Cobalt and Zinc

The F_v results for ^{54}Mn, ^{60}Co, and ^{65}Zn [6, 7] are given in Table 3. For these radionuclides, the only data available were from radiotracer experiments, and they are one order of magnitude higher than F_v values of naturally existing stable elements as shown in Table 3. The best-estimate F_v values listed in Technical Reports Series No. 364 for Cereals or Wheat were 0.3 for Mn, 3.7×10^{-3} for Co, and 1.6 for Zn. GMs of recently compiled data in this IAEA-TECDOC are 2.8×10^{-1} for Mn, 8.5×10^{-3} for Co, and 1.8 for Zn, which are almost the same as those compiled previously. The GM of F_v values for Co in brown rice is higher than that in Technical Reports Series No. 364, whereas the geometric mean values for Mn and Zn are on the same order of magnitude. It would be highly advisable to obtain more data for Co in order to evaluate whether the difference is significant or not.

TABLE 3. SOIL-TO-RICE TRANSFER FACTORS FOR Mn, Co, Zn, Tc and I OBTAINED IN POT EXPERIMENTS

Nuclides	Type of rice	N	GM	Min	Max	# Ref
^{60}Co	Rice, not specified	3	1.0×10^{-2}	9×10^{-3}	1.1×10^{-2}	1
	Brown rice	4	4.3×10^{-3}	2.2×10^{-3}	6×10^{-3}	1
^{125}I	Brown rice	4	2.5×10^{-3}	1.1×10^{-3}	7.3×10^{-3}	1
	Brown rice	5	5.9×10^{-3}	4.4×10^{-3}	7.6×10^{-3}	1
^{54}Mn	Brown rice	5	2.6×10^{-1}	1.2×10^{-1}	5.2×10^{-1}	1
^{95m}Tc	Brown rice	2	$< 2 \times 10^{-4}$ [a]			1
^{65}Zn	Rice, not specified	3	8.9×10^{-1}	7.9×10^{-1}	9.6×10^{-1}	1
	Brown rice	4	1.7	5.8×10^{-1}	2.7	1

[a] Arithmetic mean.

3.3. Technetium

F_v values for Tc in rice and wheat plants have been obtained from laboratory radiotracer experiments [8], and the data for rice grain are listed in Table 3. The mean F_v for rice plants is less than 0.2×10^{-3} for the hulled rice and 5.1 for the lower leaf blade. For wheat plants, higher F_v values are found: 2.7×10^{-3} for the hulled wheat and 230 for the lower leaf blade. In Technical Reports Series No. 364, the best estimate value of Tc for cereals was 7.3×10^{-1} with a 95% confidence level of 7.3×10^{-2}-3.7 [4], and in recently compiled data in this IAEA-TECDOC, the arithmetic mean of F_v values for cereals was 1.3 (range: 0.18-2.4).

The Tc concentrations in the grains of both rice and wheat are much lower than those in their leaves, and F_v values of Tc for rice grain are lower than those for wheat grain. The level of Tc in the soil solution collected from the flooded soil of rice pots decreased rapidly with time, whereas the Tc concentration in the soil solution of wheat pots decreased slowly. When a Tc tracer was added as TcO_4^- to a flooded paddy soil sample, Tc was easily transformed to immobile forms due to the reducing conditions generated by microbial activity. Thus, low F_v values observed for the rice plants could be explained by fixation of Tc in the soil.

3.4. Iodine

When rice plants were grown on Andosol and Gray lowland soil, the F_v values obtained for radioiodine for brown rice were 2.5×10^{-3} and 5.9×10^{-3}, respectively [9, 10] (Table 3). F_v values for polished rice were 0.7×10^{-3} for the Andosol and 0.2×10^{-2} for the Gray lowland soil. Stable iodine concentrations in paddy soil and rice samples collected in Japan have been analyzed to estimate the F_v values for brown rice and polished rice [11]. The F_v values for polished rice range from 0.1×10^{-3} to 0.8×10^{-2} and the geometric mean is 0.2×10^{-2}, whereas those for brown rice range from 0.1×10^{-2}-0.2×10^{-1} and the geometric mean is 0.5×10^{-2}.

Iodine sorbed on the soil solid phase is readily dissolved into the soil solution under flooded conditions, as it is more soluble in anoxic than oxic conditions; however, the F_v values of iodine for brown rice and polished rice are extremely small. One of reasons is that iodine is mainly distributed in the leaves, and hardly any iodine translocates into the grains.

3.5. Radium, Thorium, and Uranium

As shown above, fewer data for other radionuclides are available. As a result, naturally existing elements have been studied as analogues of these radionuclides to add information. Recently, rice and associated soil samples have been collected from 50 sampling sites throughout Japan to obtain F_v values of stable isotopes, Th, and U, for white rice and brown rice grains [1]. The results are presented in Table 1, along with previous results: F_v-Th and F_v-U, are listed, and no substantial differences were observed between the F_v values obtained in recent and previous studies [12-15]. The F_v data are not separated by soil types since the sample data presented in the previous report [1] showed a log-normal distribution of F_v values. Usually, F_v values are higher in brown rice than in white rice. Some elements, such as Fe, Mn, Co, Sr, Ba, and U, show significantly different F_v values between white rice and brown rice. White rice is simply a refined starch, and chemical element components in brown rice and polished rice are different as shown in Fig. 3.

Ra occurs naturally in the environment; however, its concentration is extremely low so that F_v data obtained under natural conditions are limited. The F_v values for ^{226}Ra from the paddy soil to brown rice range from 1.9×10^{-4} to 2.8×10^{-2} whereas those for white rice range from 2.2×10^{-4} to 1.5×10^{-3}. The F_v values of brown rice and white rice are rather similar except for a few elements. Technical Reports Series No. 364 has no Ra data for rice or cereals, but the values reported here for white and brown rice are close to the best-estimate value for maize (1.2×10^{-3}). The Ra data for cereals have been summarized here, and the geometric mean value is 1.7×10^{-2} with 95% confidence level of $9.0 \times 10^{-4} – 3.2 \times 10^{-1}$. Cereals show slightly higher Ra F_v values than brown rice (Fig. 2).

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ROOT UPTAKE FOLLOWING ACUTE SOIL DEPOSITION DURING PLANT GROWTH

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Abstract

Root uptake of radionuclides is generally evaluated using transfer factor (F_v) values expressed as the ratios of the plant concentrations to the soil concentrations. F_v values are measured in the soils mixed with radionuclides to a certain depth before planting. In contrast, radionuclides acutely deposited during plant growth will usually be localised with respect to the soil surface till harvest. This makes the use of F_v inappropriate and makes it necessary to use aggregated transfer factors specified for the time periods from deposition until harvest (T^a_{ag}, m^2 kg^{-1}). In this paper, time-dependent T^a_{ag} values for Mn, Co, Sr, Cs and/or tritium for several food crops are summarized. The values vary considerably with the radionuclides, plant species, soil types and times of deposition. They can be used for predicting the root uptake following an acute soil deposition during the growth of crop plants. To date, however, a limited number of T^a_{ag} values are available. In order to reduce the uncertainty in the prediction, it will be necessary to perform a substantial number of further experiments.

1. DEFINITION AND PROCESSES

The previous papers largely describes radionuclide transfers from soil to plants for equilibrium conditions, i.e. in some time period after the deposition. However, a certain amount of radionuclides can also be transferred to plants from soil in the year of deposition. Such a situation can be of importance, in particular, for the case of acute (short term) deposition. Many models allow the contamination of plants to be calculated for this scenario with some uncertainty. In addition, some direct experimental data are now available, giving the opportunity for testing and enhancing models of radionuclide transfer from soil to plants after acute deposition.

As mentioned earlier, root uptake of radionuclides is generally evaluated using transfer factor (F_v) values expressed as the ratios of the plant concentrations to those in soil. F_v values are measured in the soils mixed with radionuclides to a certain depth before planting because ploughing is a common practice done before planting.

In contrast, radionuclides acutely deposited during plant growth will usually be localised with respect to the soil surface till harvest because farmlands are not ploughed within the growing period. This makes the use of F_v inappropriate and makes it necessary to use aggregated transfer factors [1-3] specified for the time periods from deposition until harvest (T^a_{ag}):

$$T^a_{ag} = \frac{Concentration\ of\ radionuclide\ in\ plant\ at\ harvest\ (Bq\ kg^{-1})}{Activity\ acutely\ deposited\ onto\ soil\ per\ unit\ area\ of\ farmland\ (Bq\ m^{-2})}$$

Due to the lack of such data, most of the dynamic food-chain models use F_v values even for a growing-time deposition [4-6] in spite of its inappropriateness in this context [1-3]. Since the physiological activity and organ development of plants, and the bioavailability of deposited radionuclides change with time, the T^a_{ag} may greatly depend on the growth stage when a deposition occurs.

2. DERIVED T_{ag}^{a} VALUES

2.1. T_{ag}^{a} values for sandy soil

Available data on radionuclide transfers from soil to plants obtained in greenhouse experiments for acid sandy soils [7-10] are given in Tables 1-3. The T_{ag}^{a} values vary considerably with the radionuclide, plant species, and time of deposition. Data in Table 1 show that, in terms of T_{ag}^{a} values, radionuclides used in that study can generally be ranked as follows: Sr > Mn > Co > Cs. Comparatively high uptake of Mn by rice can likely be explained by a rapid reduction of Mn in the water-logged soil [11]. In most cases, the depositions at the middle or late growth stages produced higher T_{ag}^{a} values than those at the early growth stage. This can largely be explained by the comparatively weak absorbing-powers of the young roots and increasing soil fixation of the deposited radionuclides. Particularly for rice, such an enhanced uptake of Cs, following its deposition at the middle growth stage, may be associated with uptake via plant bases. The bases of rice plants are normally submerged in standing water. This water contains considerable amounts of radionuclides for some time after an accidental deposition, possibly leading to significant plant-base uptake. Some experiments have shown that plant-base uptake in rice was very efficient for Cs [12-13]. In Table 2, the T_{ag}^{a} values of tritiated water (HTO) to organically bound tritium (OBT) in rice straw and brown rice were highest when HTO was deposited at the most active growth stage of each compartment of plant [9].

Rice straw is supplied to cattle for feeding in some Asian countries including Republic of Korea. The T_{ag}^{a} values for the grain OBT are several times higher than that for the straw OBT when HTO is deposited at the active grain-developing stage, whereas the opposite is generally true following deposition at the other growth stages.

The T_{ag}^{a} values of HTO for both TFWT (tissue free water tritium) and, to a lesser degree, OBT in the Chinese cabbage increase sharply as the deposition time becomes closer to harvest (Table 3) [10]. This can be explained by the dilution of tissue tritium being less for a shorter duration. Such a dilution is caused by evapotranspiration, gas exchange, biomass increase, and respiration.

2.2. ^{90}Sr and ^{137}Cs T_{ag}^{a} values to rice from various soils

For rice, further experiments were performed to measure the T_{ag}^{a} values for ^{90}Sr and ^{137}Cs in various paddy soils [2, 3, 8]. The experimental soils, collected undisturbed into small lysimeters, were acid loams and sands. The results of these experiments are summarised in Table 4. It can be seen from the table that in terms of radionuclides transfer to plants there was no big difference recorded between these two soil types. Generally, ^{90}Sr T_{ag}^{a} values were 30-40 fold higher in the rice straw than in the rice grain, whereas those of ^{137}Cs are only about two times higher in the rice straws than in the rice grain. This fact suggests a higher mobility of ^{137}Cs within a plant's body than of ^{90}Sr. The values of ^{137}Cs deposited at around the starting time of the ear emergence (83 d after transplanting) are several times higher than those from the early-tillering-stage deposition. This difference between the depositions at two different times might be caused mainly by the difference in the plant-base uptake.

TABLE 1 Co, Mn, Sr AND Cs T_{ag}^{a} VALUES TO VARIOUS PLANTS

Plant Compartment	Time of Deposition [a]		T_{ag}^{a} ($m^2\,kg^{-1}$-dry plant) [b]			
	(DAP)	(DTH)	Mn	Co	Sr	Cs
Brown rice,	13	125	2.9×10^{-3}	3.6×10^{-5}	3.9×10^{-4}	1.4×10^{-4}
Rice [14]	40	98	2.6×10^{-3}	4.9×10^{-5}	4.7×10^{-4}	2.0×10^{-4}
	67	71	2.0×10^{-3}	3.0×10^{-5}	5.5×10^{-4}	7.0×10^{-4}
	89	49	5.0×10^{-3}	4.6×10^{-5}	7.6×10^{-4}	5.4×10^{-4}
	112	26	1.2×10^{-3}	2.6×10^{-4}	1.1×10^{-4}	5.2×10^{-5}
Leaves,	13	58	1.3×10^{-2}	9.6×10^{-4}	8.6×10^{-2}	6.6×10^{-4}
Chinese cabbage	27	44	1.3×10^{-2}	6.4×10^{-4}	7.2×10^{-2}	8.2×10^{-4}
[1]	42	29	1.3×10^{-2}	9.4×10^{-4}	5.4×10^{-2}	7.0×10^{-4}
	56	15	2.2×10^{-2}	4.8×10^{-3}	4.4×10^{-2}	1.2×10^{-3}
Roots,	13	62	1.4×10^{-3}	1.7×10^{-4}	9.0×10^{-3}	1.1×10^{-4}
Radish [1]	27	48	2.4×10^{-3}	2.9×10^{-4}	1.3×10^{-2}	2.1×10^{-4}
	42	33	2.1×10^{-3}	6.8×10^{-4}	1.1×10^{-2}	6.5×10^{-4}
	58	17	2.7×10^{-3}	8.0×10^{-4}	6.2×10^{-3}	3.2×10^{-4}
Tubers,	4	86	6.0×10^{-4}	2.6×10^{-4}	5.5×10^{-4}	3.5×10^{-4}
Potato [7]	31	59	9.0×10^{-4}	4.5×10^{-4}	1.6×10^{-3}	3.5×10^{-4}
	63	27	2.5×10^{-4}	5.5×10^{-4}	2.0×10^{-4}	5.5×10^{-5}
Fruits,	13	53/63	8.8×10^{-3}	1.0×10^{-3}	2.8×10^{-2}	7.3×10^{-4}
Cucumber [15]	31	35/45	1.2×10^{-2}	1.6×10^{-3}	1.6×10^{-2}	5.5×10^{-4}
	50	16/26	1.3×10^{-2}	2.1×10^{-3}	2.0×10^{-2}	7.5×10^{-4}
	61	5/15	9.3×10^{-3}	1.2×10^{-3}	1.2×10^{-2}	2.5×10^{-4}

[a] DAP: days after planting, DTH : days to harvest; [b] The values for Chinese cabbage, radish, potato and cucumber are based on dry-to-fresh weight ratios of 5%, 6.7%, 20% and 4%, respectively. The values for potato are about the tubers after peeled and those for cucumber are averages for the harvests made at two different times.

TABLE 2. TRANSFER FACTORS (T_{ag}^{a}) OF HTO TO RICE OBT

Transfer of Interest	Time of Deposition [a]		T_{ag}^{a} ($m^2\,kg^{-1}$- dry weight)	
	(DAP)	(DTH)	Brown rice	Straw
HTO-to-rice OBT [9]	5	132	1.3×10^{-5}	4.1×10^{-5}
	57	80	3.3×10^{-5}	2.0×10^{-4}
	79	58	1.6×10^{-4}	1.5×10^{-4}
	98	39	3.2×10^{-4}	8.4×10^{-5}
	113	24	2.5×10^{-5}	8.8×10^{-5}

[a] DAP : days after planting, DTH : days to harvest.

TABLE 3. TRANSFER FACTORS (T_{ag}^{a}) OF HTO TO CHINESE CABBAGE

Plant Compartment	Time of Deposition [a]		T_{ag}^{a} ($m^2\,kg^{-1}$-plant) [b]	
	(DAP)	(DTH)	TFWT	OBT
Leaves, Chinese cabbage [10]	8	67	1.6×10^{-5}	9.0×10^{-5}
	26	49	6.0×10^{-5}	1.7×10^{-4}
	44	31	4.4×10^{-4}	4.6×10^{-4}
	63	12	4.9×10^{-3}	8.6×10^{-4}

[a] DAP : days after planting, DTH : days to harvest. [b] on the basis of the fresh weight for TFWT and of the dry weight for OBT.

TABLE 4. ^{137}Cs AND ^{90}Sr T_{ag}^{a} VALUES TO RICE FOLLOWING CONTAMINATION AT THE STAGES OF EARLY TILLERING AND EARLY EAR EMERGENCE

Plant Compartment		(DAP^{a})	Soil Group	N	AM	SD	GM	GSD	Min	Max
^{137}Cs	Brown rice									
		12	Total	3	2.8×10^{-4}	1.6×10^{-4}	2.5×10^{-4}	1.87	1.3×10^{-4}	4.5×10^{-4}
			Loam	2	2.9×10^{-4}	2.3×10^{-4}	-	-	1.3×10^{-4}	4.5×10^{-4}
			Sand	1	2.7×10^{-4}	-	-	-	-	-
		83	Total	8	2.5×10^{-3}	1.4×10^{-3}	2.0×10^{-3}	2.26	3.9×10^{-4}	4.2×10^{-3}
			Loam	5	3.0×10^{-3}	1.3×10^{-3}	2.7×10^{-3}	1.81	1.0×10^{-3}	4.2×10^{-3}
			Sand	3	1.6×10^{-3}	1.2×10^{-3}	1.2×10^{-3}	2.75	3.9×10^{-4}	2.7×10^{-3}
	Straw	12	Total	3	6.0×10^{-4}	3.8×10^{-4}	5.2×10^{-4}	2.05	2.4×10^{-4}	1.0×10^{-3}
			Loam	2	6.2×10^{-4}	5.4×10^{-4}	-	-	2.4×10^{-4}	1.0×10^{-3}
			Sand	1	5.7×10^{-4}	-	-	-	-	-
		83	Total	8	3.8×10^{-3}	2.4×10^{-3}	2.9×10^{-3}	2.63	3.8×10^{-4}	6.9×10^{-3}
			Loam	5	4.5×10^{-3}	2.1×10^{-3}	4.0×10^{-3}	1.73	1.8×10^{-3}	6.9×10^{-3}
			Sand	3	2.7×10^{-3}	2.9×10^{-3}	1.6×10^{-3}	3.97	3.8×10^{-4}	9×10^{-3}
^{90}Sr	Brown rice									
		83	Total	5	6.4×10^{-4}	3.3×10^{-4}	7×10^{-4}	1.78	2.5×10^{-4}	1.1×10^{-3}
			Loam	3	6.2×10^{-4}	1.9×10^{-4}	6.0×10^{-4}	1.40	4.1×10^{-4}	7.8×10^{-4}
			Sand	2	6.8×10^{-4}	6.0×10^{-4}	-	-	2.5×10^{-4}	1.1×10^{-3}
	Straw	83	Total	5	2.1×10^{-2}	7.5×10^{-3}	2.0×10^{-2}	1.40	1.4×10^{-2}	3.3×10^{-2}
			Loam	3	1.8×10^{-2}	4.7×10^{-3}	1.7×10^{-2}	1.29	1.4×10^{-2}	2.3×10^{-2}
			Sand	2	2.6×10^{-2}	9.9×10^{-3}	-	-	1.9×10^{-2}	3.3×10^{-2}

[a] DAP : days after planting. Rice plants were harvested 145 d after transplanting.

Fig. 1 shows the T_{ag}^{a} values of ^{137}Cs for brown rice and straw in three different soils contaminated at two different times during rice growth. The patterns of the deposition time-dependent variations of the values are generally similar in different soils. A similar pattern of the variation can be found also for the ^{137}Cs T_{ag}^{a} values to rice (Table 4).

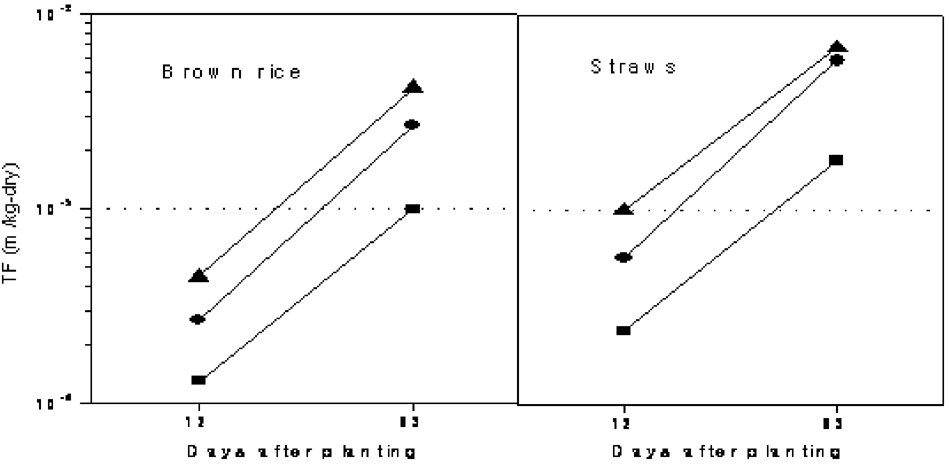

FIG. 1. T_{ag}^{a} values of ^{137}Cs for rice in different soils contaminated at two different times during the growing period. Different symbols denote different soils.

3. CONCLUSIONS

T_{ag}^{a} values can be used for evaluating a root uptake following an acute soil deposition during the growth of crop plants. For this purpose, it is important to use the appropriate values for the time of deposition. If the deposition time deviates from that in the experiments, some interpolation or extrapolation needs to be implemented.

When a dynamic food chain model applies an F_v value to an acute growing-time deposition, it is assumed that the root uptake is proportional to the biomass increase or uptake duration. By this assumption, an earlier deposition will generally result in a higher uptake at harvest [1]. This kind of time-dependence rarely happened in the above-mentioned experiments. A more realistic approach may be, therefore, to directly use such empirical data as presented here.

To date a limited number of T_{ag}^{a} values are available. Many more data are needed to sufficiently cover various accidental and agricultural conditions. The values given in the section were mainly obtained from single experiments. Therefore, the data obtained are characterised by a high uncertainty and should be interpreted with some caution. In order to reduce the uncertainty in the prediction, it will be necessary to perform a substantial number of further experiments.

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EVOLUTION OF PLANT CONTAMINATION WITH TIME

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1. BASIC DEFINITIONS

Evaluation of radionuclide transfer in the environment implies consideration of the decrease of radionuclide activity concentrations in plants in the course of time after single release of radionuclides into the environment. This arises because radionuclides transferred to the environment are gradually fixed by natural sorbents (soils, bottom sediments in water ecosystems, etc.), are lixiviated to lower soil layers and become less biologically available for inclusion into food chains.

As mentioned earlier[1], the long-term, time-dependent behaviour of radionuclides is often quantified by reference to the ecological half-life, $T_{1/2}^{eco}$, which is an integral parameter that relates to the reduction of activity or activity concentration in a specific medium. According to the definition (see section 2.1), $T_{1/2}^{eco}$ is equal to the period over which the concentration of a radionuclide, in some definite component of a trophic chain, is decreased by a factor of two, but excluding the effects of radioactive decay. Based on this approach, changes in the radionuclide activity concentrations in plants normalised at the radionuclide activity concentrations in soil F^*_v can be estimated as follows.

$$F^*{}_v(t) = F^*{}_v(0) \cdot e^{-\frac{0.693 \cdot t}{T_{1/2}}}$$

Although field data on variations over time in radionuclide transfer factors after clearly defined depositions are rather scarce, there are three prime sources of the information on the radionuclides half-lives in plants: global fallout and the Kyshtym and Chernobyl accidents.

Global fallout represented a variable source term of radionuclides for the environment, combining foliar and root uptakes, with a maximum depositions observed in 1962-1964. However, long term monitoring data following the above time period provided an opportunity for long term half-life assessments for ^{90}Sr and ^{137}Cs.

A unique situation for the assessment of environmental half-lives, at least for ^{137}Cs, developed after the Chernobyl NPP accident: the single release with well-known characteristics, high contamination levels, and various conditions of the environment (soil, climate etc.). As a result, extensive data on ^{137}Cs bioavailability in soils and soil-plant systems have been obtained, and the results are discussed below.

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[1] See paper by Fesenko et al. «Radioecological definitions, soil, plant classifications and reference ecological data for radiological assessments » in this publication.

2. DYNAMICS OF ^{137}CS AND ^{90}SR TRANSFER FACTORS TO PLANTS FOLLOWING MILITARY TEST EXPLOSIONS

In order to determine year-to-year variations in F_v values for ^{90}Sr and ^{137}Cs observed after the period of intensive global fallout, F_v values in brown and white rice samples, and associated soil samples collected throughout Japan in 1959-2000 by the National Institute for Agro-Environmental Sciences, Japan [1] were reviewed. Fig. 1 gives geometric means (GMs) for ^{90}Sr and ^{137}Cs F_v values for brown and white rice. The results allow the conclusion that F_v values from soil to rice were decreasing with effective half-lives ranging from 12-18 years.

Different estimates are given in some other studies following the nuclear weapons testing, provided by UNSCEAR, which suggested a half-life for ^{137}Cs from eight to fifteen years [2].

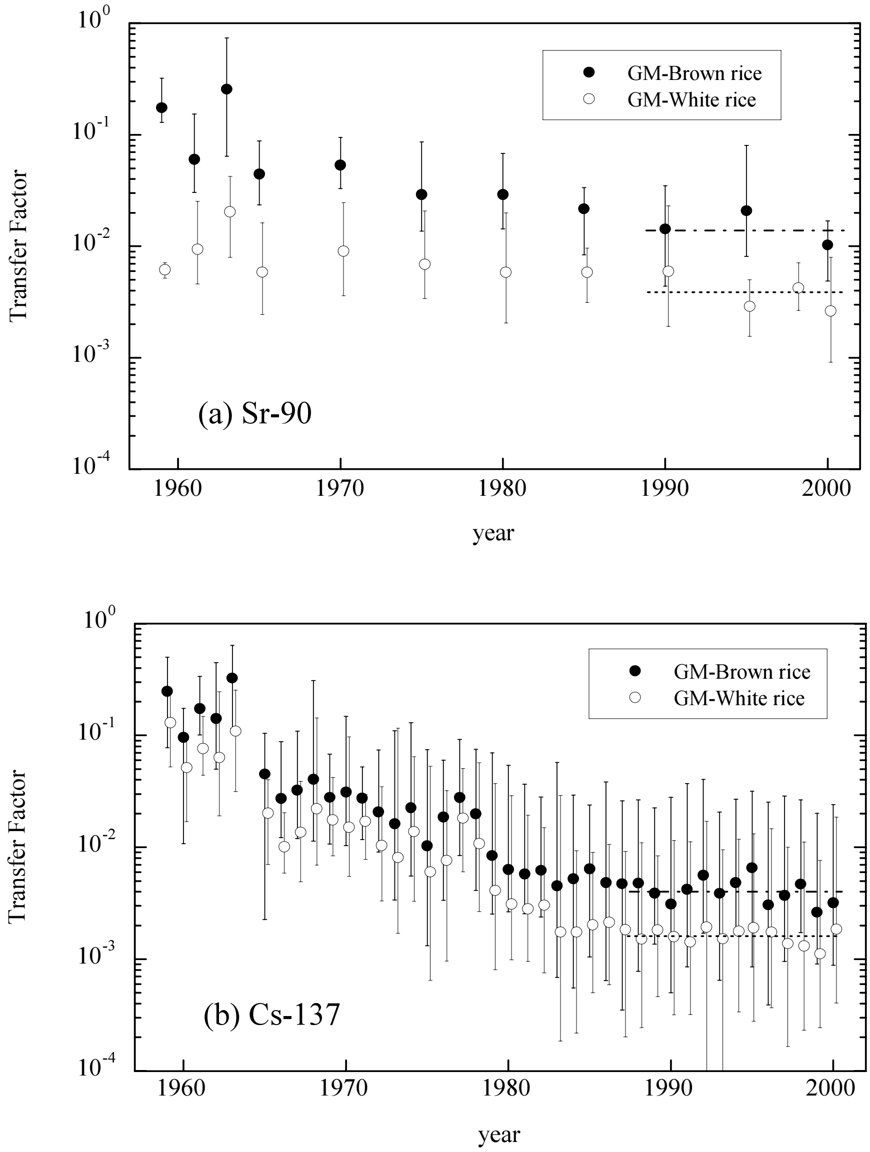

FIG. 1. Variations of (a) ^{90}Sr-transfer factor values and (b) ^{137}Cs-transfer factor values for brown rice and white rice collected in Japan as a function of time (modified from [1]). Bars show maximum and minimum values.

3. DYNAMICS OF ^{137}Cs AND ^{90}Sr TRANSFER FACTORS TO PLANTS FOLLOWING RADIATION ACCIDENTS

The analysis of the dynamics of ^{137}Cs F_v values has shown that, for a 13-year period after the Chernobyl accident, F_v reduced 3 to 10 times. This decrease was found to be dependent on crop type, cultivation practices, and soil properties. The F_v ecological half-lives for two periods were calculated in a comparative analysis of radionuclide availability for root uptake. The first period included the years 1987 to 1989, when the fixation of radionuclides (primarily ^{137}Cs) was most intense; the second covered 1989-1999, when fixation slowed down [3]. Consequently, the dynamics of F_v decrease were described as a sum of two exponentials, and effective half-lives have been calculated for the time periods 1987 to 1989 and 1989 to 1999 (Table 1) while the average half-lives from 1987-1999 according to [4, 5] are given in Table 2.

TABLE 1 ^{137}Cs HALF-LIVES IN PLANTS, YEARS [3]

Soil Group	The first period, 1987-1989			The second period, 1989-1999		
	$T_{1/2}^{eco}$	$F^*_v(0)$	R^2	$T_{1/2}^{eco}$	$F^*_v(0)$	R^2
Barley						
Sand	1.3	4.3×10^{-1}	9.9×10^{-1}	6.2	1.4×10^{-1}	6.5×10^{-1}
Loam	1.9	2.1×10^{-1}	9.8×10^{-1}	6.7	9×10^{-2}	7.1×10^{-1}
Clay	1.4	1.7×10^{-1}	9.8×10^{-1}	3.8	7×10^{-2}	8.7×10^{-1}
Potato						
Sand	1.2	5.7×10^{-1}	8.8×10^{-1}	7.5	1.2×10^{-1}	9.9×10^{-1}
Loam	2.4	1.4×10^{-1}	9.8×10^{-1}	8.5	1×10^{-1}	(2.0×10^{-1})*
Clay	2.9	7×10^{-2}	9.5×10^{-1}	5.0	4×10^{-2}	6.4×10^{-1}
Beetroot						
Sand	2.9	3.2×10^{-1}	9.2×10^{-1}	5.2	2.9×10^{-1}	8.3×10^{-1}
Loam	2.6	2.5×10^{-1}	8.2×10^{-1}	5.9	1.2×10^{-1}	9.8×10^{-1}
Clay	2.9	1.8×10^{-1}	7.2×10^{-1}	7.2	1.5×10^{-1}	9.6×10^{-1}
Natural Grasses						
Sand	1.6	2.9×10^{1}	9.9×10^{-1}	1.5×10^{1}	1.8×10^{1}	8.7×10^{-1}
Loam	1.3	7.0	9.9×10^{-1}	4.7	2.3	(3.7×10^{-1})*
Clay	1.3	3.6	9.9×10^{-1}	4.9	1.4	5.6×10^{-1}
Organic	1.8	8.3×10^{1}	8.4×10^{-1}	1.1×10^{1}	3.0×10^{1}	7.2×10^{-1}
Perennial Grasses						
Sand	2.3	1.2×10^{1}	9.9×10^{-1}	4.8	3.5	9.1×10^{-1}
Loam	2.5	1.2×10^{1}	9.5×10^{-1}	4.6	6×10^{-1}	6.0×10^{-1}
Clay	2.5	1.9	9.5×10^{-1}	1.0×10^{1}	6.5×10^{-1}	6.6×10^{-1}
Organic	2.6	2.3×10^{1}	9.9×10^{-1}	2.1×10^{1}	9.9	(2.1×10^{-1})*

*–insufficient data for adequate estimation.

TABLE 2 AVERAGE ^{137}Cs F_v HALF-LIVES (1987-1999) [4, 5]

Crop	Soil Group	Half-lives, years
Cereal	Sand	7-11.8
	Loam	5.1-22.0
	Clay	14.4-40.4
	Peat	9.2-14.4
Potato, Beetroot	Sand	7.0-9.2
	Loam	10
	Clay	13.3
	Peat	10
Perennial Grasses	Sand	6.4
	Loam	4.0
	Clay	9.8
	Peat	17.3
Natural Grasses	Sand	5.8
	Loam	18.0
	Clay	13.3
	Peat	17.3

The analysis of the results presented suggests the following:

- The initial ecological half-lives for F_v are rather similar for different soil groups and farm crops and vary between 1.3 and 2.9 years.
- The longer-term ecological half-lives are in the range of 4.6 to 17.3 years, with the highest F_v values being reported for sand, loam, and organic soils.

It should be taken into account in the analysis that radionuclide transfers to plants depend on many factors that are responsible for their considerable variations observed under field conditions. The variability in soil characteristics within each of the identified groups caused a certain degree of variability in F_v values. In addition, because of the specific weather conditions of any year, F_v values can vary up to a factor of 3.

There is no information on the systematic observations of variations with time in ^{90}Sr concentrations in plants following the Kyshtym accident, while some data on environmental half-lives in cow milk are available. Peremyslova et al [6] have calculated effective half-life for ^{90}Sr in milk of around 20 years for the period 1964-1999.

Although these data can not be directly used for the evaluation of plant contamination with time, the decline of radionuclide concentrations in animal products can be considered as an integral parameter that reflects the decrease of environmental mobility of radionuclides and, in particular, ecological half-lives of radionuclides in plants.

4. CONCLUSIONS

During the years after depositions, the rate of radionuclide transfer to plant declines markedly, resulting in large changes in contamination of vegetation. Evolution of plant contamination with time is a complex task requiring analysis of big datasets with time-dependent information on radionuclide transfer to plants. To date, the most information is available for

^{137}Cs. The dynamics of the decrease of ^{137}Cs availability in soil-plant system is considerably influenced by soil properties, and the rates of decreasing ^{137}Cs uptake by plants can differ by the factor of 3-5, being dependent on soil characteristics. The rate of decrease of radionuclide uptake by plants is irregular by its nature, and several time periods should be considered in applying a half-life approach for data evaluation.

In the first years after deposition, bioavailability of some radionuclides in soil reaches its maximum, resulting in maximum radionuclide transfer rate to plants. The data allows the conclusion that ecological half-lives for ^{137}Cs in plants are in range of 1-2 years in first years after the deposition, declining up to 12-20 years in the long term after the deposition. The half-lives of ^{90}Sr tend to be slightly longer and can be estimated as 20-30 years.

Unfortunately, existing literature data are rather scarce even for ^{90}Sr and ^{137}Cs, and any such estimates should be interpreted with a great caution. No data are available for radionuclides other than ^{90}Sr and ^{137}Cs, and long-term research is needed to estimate variation over time in changes of radionuclide transfer of other radionuclides to plants.

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AGRICULTURAL ECOSYSTEMS: TRANSFER TO ANIMALS

TRANSFER TO ANIMALS

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Abstract

Data have been compiled to derive animal product transfer coefficients for radionuclides to update the values given in Technical Reports Series No. 364. Significant new data inputs have been incorporated from an extensive review of Russian language information and inclusion of data published since the early 1990s. The resultant database has been used to provide reference transfer coefficient values for a range of radionuclides to (i) cow, sheep and goat milk, (ii) meat (muscle) of cattle, sheep, goats, pigs and poultry and (iii) eggs. The approaches and procedures used to identify and collate data, and assumptions used are given. For most animal products, transfer coefficient values for elements additional to those in Technical Reports Series No. 364 are provided, although some elements were considered in the earlier evaluation which were not included in this review. Differences between the Technical Reports Series No. 364 'expected' values and the reference values from this document, which will be incorporated into the revised transfer parameter handbook, are discussed. An alternative approach to quantifying transfer by using concentration ratios is evaluated and CR values which could be applied across animal species have been provided for milk and meat. Information on fractional gastrointestinal absorption in adult ruminants has been compiled and reference values presented. Despite these improvements many data gaps remain.

1. INTRODUCTION

Contamination of food products derived from both farmed and wild animals can represent major sources of radiation exposure to humans from both routine and accidental releases of radioactivity to the environment.

The primary health effect arising from the Chernobyl accident was the development of thyroid cancers, especially in young children through exposure to radioiodine in contaminated milk [1]. Animal products were the only agricultural food type requiring intervention outside of the former Soviet Union after the Chernobyl accident, and in some countries radiocaesium countermeasures were still being used twenty years after the accident [2].

An understanding of the factors affecting the transfer of radionuclides to, and their behaviour in, animals is therefore essential if we are to be able to interpret monitoring results, accurately predict activity concentrations in animal-derived food products and develop effective and appropriate countermeasures.

Radionuclide activity concentrations in animal food products depend primarily on the rate of intake, gastrointestinal absorption and turnover in tissues. The extent of available data on gastrointestinal absorption, transfer to tissues and milk varies widely for different radionuclides.

2. DATA SOURCES AND ANALYSIS

Information sources such as books, journals, conference proceedings, institutional reports, and international and national databases were reviewed, and a database created to handle and process the data derived from five different sources (Figure 1). The database of transfer parameter values for domestic animals was based on an adaptation of the transfer coefficient database for meat and milk of Green and Woodman [3], which originally contained data from 292 references. The entries in this database were individually reviewed and revised where necessary. They were then supplemented, where appropriate, by source data used in the previous IAEA transfer parameter handbook TRS364 [4], by going back to original sources, rather than using review values where used in the tables in the handbook.

In addition to transfer coefficients from these two sources, we also collated additional transfer coefficient and gastrointestinal absorption values. This was achieved by additional searches of English language literature, as well as a major review of relevant information published in the Russian language documents which is being published in a series of peer-reviewed papers [5, 6]. Most of the latter data have not previously been taken into account in either Technical Reports Series No. 364 [4] or other national or international reviews, and these data constitute an important new source of information.

Where there are sufficient data we have provided the following values in the transfer tables: number of data, geometric mean, and geometric standard deviation, arithmetic mean, standard deviation, and minimum and maximum values. For less well documented radionuclides with a single value identified we give only the value itself in the arithmetic mean column.

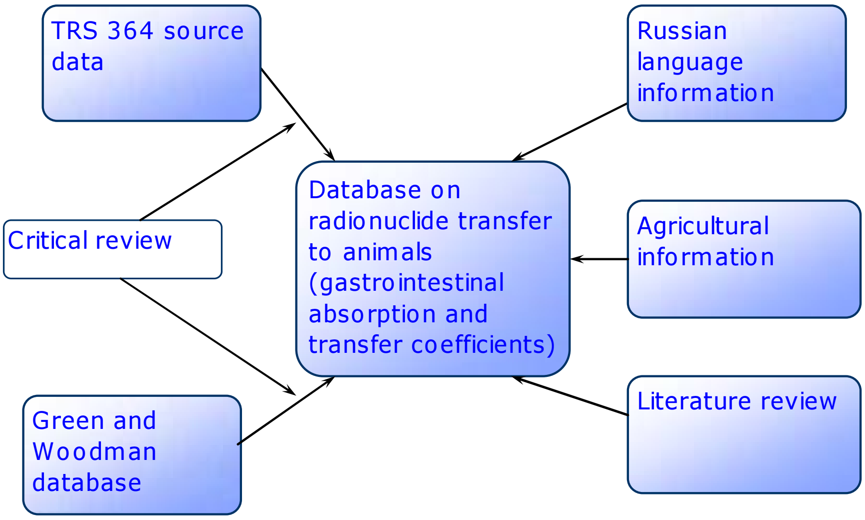

FIG. 6.1. The sources of information used to construct the database of radionuclide transfer parameters for domestic animals.

3. CONTAMINATION ROUTES

Animals can be contaminated by three different routes: through the skin, by inhalation, and, most importantly, by ingestion. Uptake through the skin is not usually an important route of contamination, although skin lesions can provide a direct entry for radionuclides into an animal’s circulation system, and this route is not considered here.

Inhalation is potentially more important than skin absorption since the lung surfaces, the site of gaseous exchange, are more permeable to a wider range of elements.

Radionuclides may be inhaled in different forms, including gaseous compounds, aerosols and particles. The ability of radionuclides to pass through the pulmonary membranes varies considerably; despite low transfer rates for actinides, such as plutonium, they are often more readily absorbed via the lungs than via the gastrointestinal tract. Due to their limited solubility, noble gases may be neglected as a source of contamination of animals.

In contrast, gaseous iodine is readily absorbed. Inhalation, however, is generally not a major contamination route of agricultural animals and is not considered here. Transfer factors for ingestion can be modified for inhalation by multiplying by the ratio of a model-estimated fractional systemic uptake from the respiratory system to the fractional uptake from the gastrointestinal tract.

The most important transfer pathway to animals is the ingestion of contaminated feed, soil and drinking water. Intake via drinking water is generally a small contributor to the total radionuclide intake and normally restricted to soon after an accident. Radionuclide intake via soil can be significant, but the availability for absorption of soil-associated radionuclides is often low (see section on transfer). Hence, it is the ingestion of contaminated feed and processes influencing absorption and retention that usually determines the radionuclide content of animals.

4. GASTROINTESTINAL ABSORPTION

The degree of absorption from the gastrointestinal tract is generally the most important factor in determining the degree of radionuclide contamination of animal tissues and milk.

Ruminants have a four compartment stomach: in the first two compartments (rumen and reticulum) fermentation of chewed vegetation occurs as the gastrointestinal microflora digest the vegetation and some absorption of breakdown products occurs. The anaerobic, reducing environment of the rumen can lead to changes in radionuclide speciation and therefore bioavailability. In the case of radiocobalt, rumen micro-flora synthesise vitamin B_{12} from inorganic cobalt and hence radiocobalt ingested in an inorganic form may be absorbed as vitamin B_{12} (which affects the transfer and residence times in tissues) [7].

Subsequently, the fermented rumen fluids pass into the omasum and abomasum and are subject to enzymatic digestion, which is similar to that occurring in the stomach of monogastric animals. Some fermentation does occur in non-ruminants (e.g. in the caecum in mammals and the crop of some birds). For some radionuclides, absorption is homeostatically controlled and the animal's mineral status with regard to stable isotopes of the radionuclide (e.g. stable Fe and ^{55}Fe) or analogous elements (e.g. calcium and radiostrontium) determines the degree of transfer from the diet to the circulatory system.

Absorption has often been reported as the difference in dietary intake and faecal output, expressed as a proportion of dietary intake (the *apparent absorption coefficient,* A_a). Whilst for some radionuclides this approach will provide an accurate estimate of absorption, it is too insensitive to measure absorption from sources with a low availability and negative values of absorption can be derived.

The calculation also does not take into account endogenous secretion of absorbed radioactivity from the body to faeces which may be important for some radionuclides (e.g. liver excretion via bile) For example, endogenous excretion of radiocaesium in ruminants is typically 20-25 % of that absorbed. The absorption of some radionuclides has been measured in ruminants as the *true absorption coefficient* (A_t), which takes endogenous secretion of radionuclides into the gastrointestinal tract into account [8].

A discussion of the relative benefits of the different methods of estimating A_t and suitable protocols by which such studies can be conducted is given in [8, 9]. Previously reported fractional absorption values have been estimated by a number of methods [10-13] some of which measure apparent absorption (A_a) rather than true absorption since they do not take account of endogenous excretion. Recently, substantial new information has been made available from Russian language publications [5] using methodologies that give true absorption values.

Absorption of most essential elements is controlled by dietary supply and the animal's requirement (absorption tending to decrease with increasing dietary concentrations when requirement is met) and, in some instances, other essential elements (e.g. interactions between Ca and P, Fe and Zn etc.). A frequent limitation of the radioecological data reported here is that there is no information on dietary intakes of relevant stable elements and information is often lacking on the animal's nutritional requirements or status.

4.1. Absorption in Ruminants

Here, we report fractional absorption values for ruminants (Table 1) derived from either the database or from authoritative agricultural reviews for many of the radionuclides which are isotopes of essential nutrient elements (e.g. Ca, Cl, P, Na, Zn). Data on ^{90}Sr ^{137}Cs and ^{131}I comprise more than 50 % of the total entries. Detailed discussion of the derivation of the values and literature used can be found in Howard et al. submitted [14].

Where we have used agricultural review values we have occasionally extended the reported range given using values from the database. The values presented largely represent those that may be expected from ingestion of contaminated feeds.

We have not used data where there may have been effects on absorption of high stable element intakes (e.g. Cd). We also omitted a few values that previous reviews have considered because we had concerns about the validity, or relevance, of the data. Values in Table 1 were derived from ruminants aged over 100 days since there is evidence for enhanced absorption in young animals [5].

Some previous publications have derived values of absorption from tissue retention only, after single oral administrations (in some instances after considerable time gaps) or daily ingestion of radionuclides. We have not used such data here because losses via excreta and milk are not accounted for.

For radiocaesium, we excluded some values derived from sources of known low bioavailability (e.g. contaminated soils/sediments), but there were insufficient data to make such exclusions for other elements.

The fractional absorption values for the different radionuclides in ruminants have been grouped into orders of magnitude in Table 2.

The absorption of the three major dose determining radionuclides varies; it is complete for radioiodine, and higher for radiocaesium than radiostrontium. The absorption of essential elements is, not surprisingly, relatively high. In contrast, elements with high atomic weights, which are not essential elements, or analogues of essential elements, are poorly absorbed from the gastrointestinal tract. For example, the absorption of transuranic elements, such as Pu, is low compared with that of many elements. Some factors influencing absorption in ruminants are discussed below.

TABLE 1. GASTROINTESTINAL FRACTIONAL ABSORPTION VALUES[1] FOR ADULT RUMINANTS TAKEN FROM THE DATABASE AND ANIMAL NUTRITION REVIEWS[2]

Radionuclide	N	Fractional absorption						Source[3]
		GM[2]	GSD	AM	SD	Min	Max	
Ag	1	5.6×10^{-2}						A_t : database [15]
Am	2			1.4×10^{-4}		1.4×10^{-4}	1.4×10^{-4}	A_t : database [16]
Ba	2			5.5×10^{-2}		5.0×10^{-2}	6.0×10^{-2}	A_t : database
Ca		3.0×10^{-1}				8.0×10^{-2}	4.2×10^{-1}	NRC advised value for forage with reported range [17]
Cd	1	1.2×10^{-3}						A_t : database [18]
Ce	5	6.1×10^{-4}	2.9	9.9×10^{-4}	1.2×10^{-3}	1.9×10^{-4}	3.0×10^{-3}	A_t and A_a : database
Cl		9.0×10^{-1}				7.1×10^{-1}	1.0	NRC advised value for all feedstuffs with reported range [17]
Co	9	4.7×10^{-2}	2.9	5.4×10^{-2}	2.8×10^{-2}	1.5×10^{-2}	1.1×10^{-1}	A_t and A_a : database
Cs	14	8.0×10^{-1}	1.1	8.0×10^{-1}	8.2×10^{-2}	6.7×10^{-1}	9.3×10^{-1}	A_t : See text, excludes data where bioavailability is low due to chemical form and values from [5]
Fe		1.0×10^{-1}				2.0×10^{-2}	2.0×10^{-1}	NRC advised value for forage [17]
I	13	9.8×10^{-1}	1.4	9.9×10^{-1}	1.1×10^{-1}	7.0×10^{-1}	1.1	Database , A_t only
Mn		7.5×10^{-3}				5.0×10^{-3}	4.0×10^{-2}	NRC advised value and maximum [17], minimum from database
Na		9.0×10^{-1}				7.4×10^{-1}	1.0	NRC advised value with reported range [17]
Nb	1	$>1.4 \times 10^{-3}$						A_a : database [19]
P		6.7×10^{-1}				5.8×10^{-1}	1.0	NRC advised value for forage/concentrates with range [17]
Pb	9	4.0×10^{-2}	2.2	5.0×10^{-2}	3.1×10^{-2}	1.0×10^{-2}	1.1×10^{-1}	Database, all A_a
Pm	1	$>5.2 \times 10^{-4}$						A_a ; database [19]

TABLE 1. GASTROINTESTINAL FRACTIONAL ABSORPTION VALUES[1] FOR ADULT RUMINANTS TAKEN FROM THE DATABASE AND ANIMAL NUTRITION REVIEWS[2] (Cont.)

Radionuclide	N	Fractional absorption						Source[3]
		GM[2]	GSD	AM	SD	Min	Max	
Pu	3	8.5×10^{-5}	1.4	8.8×10^{-5}	2.9×10^{-5}	6.5×10^{-5}	1.2×10^{-4}	A_t : based on papers cited in review of Beresford et al. [9]
Ru	6	5.8×10^{-3}	4.9	1.7×10^{-2}	2.8×10^{-2}	1.4×10^{-3}	7.1×10^{-2}	A_t : database
Se		5.2×10^{-1}				4.0×10^{-1}	6.5×10^{-1}	NRC reported range for feed for ruminants [17], mean estimated from reported range
Sr	21	1.1×10^{-1}	2.0	1.2×10^{-1}	6.2×10^{-2}	5.5×10^{-2}	2.7×10^{-1}	A_t : database. See text excludes values for low Ca intakes
Te	1	$>1.6 \times 10^{-1}$						A_a : database [20]
Y	2			1.2×10^{-3}		5.0×10^{-4}	1.9×10^{-3}	A_t : database [19, 21]
U	2			1.1×10^{-2}		1.0×10^{-2}	1.2×10^{-2}	A_t : database [22; 23]
Zn		1.5×10^{-1}				5.3×10^{-2}	3.1×10^{-1}	NRC advised value [17], minimum from database and maximum from ARC [24]
Zr	1	6.8×10^{-3}						A_t: database [25]

[1] Some values are apparent absorption – these are identified in the source column; [2] GM is given when n≥3, otherwise only AM is given; [3] Source is specified as agricultural review data (NRC 2001–National Research Council [17] and ARC 1980–Agricultural Research Council [25]) or the references used from data compilation, unless n >5 when the references are given in the text. Note summarise values represent means as reported in the literature and do not reflect ranges observed between individual animals

TABLE 2 GROUPING OF FRACTIONAL ABSORPTION VALUES FOR DIFFERENT ELEMENTS IN RUMINANTS

Fractional absorption magnitude	Radionuclide
10^{-1} – 1	I, Cl, Na, Cs, P, Se, Ca, Te, Zn, Sr, Fe
10^{-2}–10^{-1}	Ag, Ba, Co, Pb, U
10^{-3}–10^{-2}	Mn, Ru , Cd, Y
10^{-4}–10^{-3}	Zr, Ce , Pm, Am, Nb
10^{-5}–10^{-4}	Pu

Most forms of iodine are rapidly reduced to iodide within the digestive tract [26]. Therefore, it is unlikely that the source of radioiodine ingested by lactating ruminants will be a major factor in determining its activity concentration in milk. For radioiodine, absorption has previously been reported to be complete regardless of the source of radioiodine [9] or dietary stable iodine intake [27]. The fractional absorption value for I of 1.0 in Table 1 is consistent with these previous recommendations.

In the case of radiocaesium, the source ingested is a major factor determining subsequent concentrations in tissues and milk with the true absorption coefficient ranging from <0.10 to >0.80 when all data are considered. Absorption of particle or soil-associated radiocaesium is considerably lower than that within plants. Sources with a low bioavailability were excluded from Table 1.

The variation in A_t for radiostrontium, with values ranging in the entire database from 0.05–0.72, does not appear to be related to source. The behaviour of strontium is strongly influenced by that of its analogue, calcium, which is a homeostatically controlled essential element [28].

The extent of calcium absorption in the gastrointestinal tract is governed by the animal's calcium requirement which depends on factors such as age, growth rate and milk yield [16, 25, 29]. At a given calcium requirement, there is an inverse relationship between the absorption of calcium and the amount of calcium in the diet [25]. Beresford et al. [9] have suggested an inverse relationship between the ratio between the calcium intake and requirement and the true absorption of radiostrontium in ruminants (Figure 2). Under normal levels of calcium intake, the source of radiostrontium ingested is unlikely to influence either the extent of absorption or the concentrations in tissues and milk to any great extent [9].

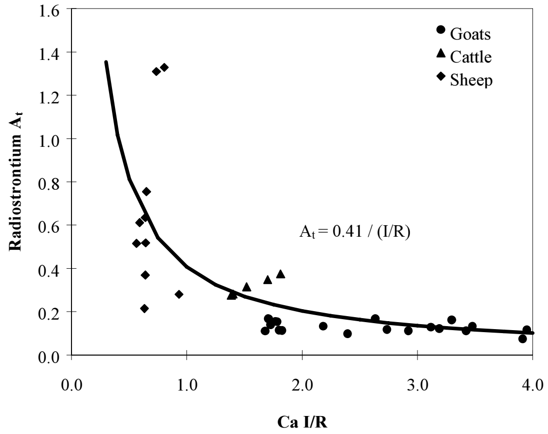

FIG. 2. Relationship between true absorption of radiostrontium and the ratio of calcium intake to requirement (Ca I/R).

The geometric mean fractional absorption value derived for Sr from the database is 0.11. The data from the Russian literature tends to be lower than that from elsewhere, possibly because the animals were slaughtered only one day after radioisotope administration and absorption estimated from body retention. This value is somewhat lower than that previously reported by Beresford et al. [9] which is consistent with previous recommendations such as those of Coughtrey and Thorne [10] although the latter include Russian data with low Ca intakes.

There are a considerable number of data in the literature documenting changes in gastrointestinal absorption with age. The decrease of fractional absorption with age may be due to the lower permeability of the membranes of the gastrointestinal wall of mature animals compared with young animals which have a greater need to absorb a wide range of nutrients and essential elements.

Some of the best evidence for the decline in fractional absorption values with age has been reported for ^{90}Sr. Long-term, large-scale experiments measuring variation in ^{90}Sr fractional absorption values with age were carried out by Burov et al. [31, 32] and by Panchenko et al. [33]. As Figure 3 demonstrates, there are considerable reductions with age in fractional absorption for all species, which is likely to be the result of high calcium requirements for skeletal development as well as the decreasing efficiency of calcium absorption with increasing age [16]. Further evidence of changes in fractional absorption values with age have also been collated for ruminants by Fesenko et al. [5] for ^{137}Cs, ^{65}Zn, ^{59}Fe and ^{60}Co and ^{238}U.

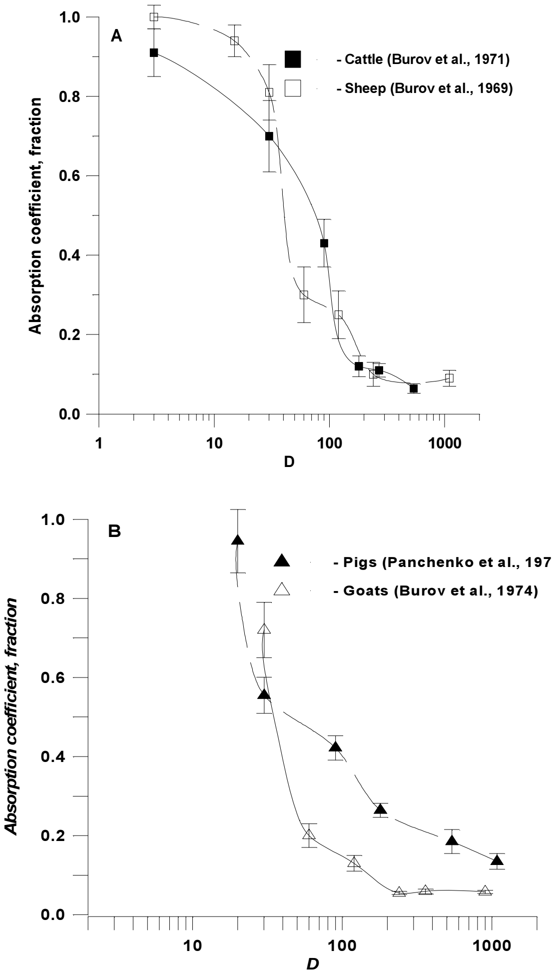

FIG. 3. Change with age in the true absorption coefficients of ^{90}Sr for: cattle and sheep (a) and pigs and goats (b) [5, 31-34].

4.2. Absorption in Monogastric animals

Gastrointestinal absorption values have been recently reported by the ICRP [35] and this provides the most relevant values for monogastric animals (Table 3). Other values reported in reviews such as those of Coughtrey and colleagues [10, 36-39] recommending values for modelling purposes tend to be conservative and also often refer to 'domestic animals' which implies both ruminants and monogastrics. For most radionuclides, the values recommended by ICRP are similar to those in Table 1 for ruminants. However, direct comparisons with Table 1 are difficult because (i) the Reference ICRP values are sometimes based on data for both ruminants and non-ruminants and (ii) the procedure for deriving the values differs. In this document we have used expert judgement to exclude data for a number of defined reasons and then derived values mathematically. For some radionuclides, this process has considerably reduced the number of values used to derive the value in Table 1 compared with similar values derived elsewhere. ICRP has described relevant values and used expert judgement to select the recommended value.

TABLE 3 FRACTIONAL ABSORPTION VALUES FOR ADULT HUMANS, ICRP [35]

Radionuclide	Fractional absorption
H, C, Cs, S, Mo, I	1.0
Se	0.8
Zn, Tc, Po	0.5
Te, Sr, Ca	0.3
Ba, Ra, Pb	0.2
Co, Fe, Sb	0.1
Ru, Ni, Ag	0.05
U	0.02
Zr, Nb	0.01
Ce, Th, Np, Pu, Am, Cm	0.0005

Recently, data for the absorption of a few radionuclides in pigs and hens has become available from Russian language publications [5]. These are generally in agreement with the ICRP values with the exception of a Sr fractional absorption value in laying hens of 0.6 (probably as the consequence of a high requirement for Ca for egg production).

5. DISTRIBUTION IN ANIMALS

Once absorbed, radionuclides enter the circulatory system and are distributed into various tissues of the body. The form of some radionuclides may be changed after absorption, for instance, certain radionuclides (e.g. radioisotopes of silver and cadmium) are bound to proteins (metallothioneins).

In some cases, radionuclides are bio-transformed within tissues and subsequently are present within the animal in more than one form. For instance, cobalt may exist as both Co^{2+} and incorporated into vitamin B_{12}, iodine as both iodide and incorporated within a number of thyroid hormones, and ^3H as tissue water or incorporated into the protein and fat of tissues. The relative proportions of these different forms can be important when considering doses to human consumers as the form ingested can influence the degree of absorption and retention of the radionuclide. Other radionuclides (e.g. radiocaesium) remain in an ionic form after absorption.

Different radionuclides are accumulated in different tissues (Table 4). For some radionuclides, the site of deposition is determined by the biological role of the corresponding stable element or analogue. The major iodine storage organ in the body is the thyroid and the element is also actively taken up by the mammary gland and transferred into milk. The stable iodine status of an animal will directly affect the behaviour of radioiodine: animals with a low stable iodine status will accumulate proportionally more in the thyroid whereas those with an excess of iodine may secrete proportionally more to milk. Radiostrontium behaves as a Ca analogue and is therefore accumulated in bone and also transferred into milk. Radiocaesium is an analogue of potassium and is, therefore, found in all soft tissues particularly muscle.

The influence of an animal's potassium status on radiocaesium behaviour is unclear although it appears to be considerably less than that of calcium on the behaviour of radiostrontium. Strontium, radium, plutonium and rare earth elements are all accumulated in bone. Liver and kidneys are common storage tissues for many pollutants including some radionuclides (e.g. actinide elements). Carbon-14, ^{3}H, ^{35}Cl and ^{35}S, all of which are released by the nuclear industry or present in waste products, are rather special cases since they have stable isotopes that are basic constituents of animals' tissues.

TABLE 4. TARGET ORGANS FOR SELECTED RADIONUCLIDES

Radionuclide	Target organ	Reference
Ag	Liver	[40]
Am	Bone and liver	[39]
Ce	Bone and liver	[14]
Co[1]	Liver	[7]
Cs	All soft tissue and milk	[41]
I	Thyroid and milk	[42]
Pu	Bone and liver	[38]
Ru	Kidney	[14]
Sr	Bone and milk	[41]
Tc	Thyroid, liver and stomach wall	[37]

[1] Co-vitamin B_{12}

6. TRANSFER TO ANIMAL PRODUCTS

The transfer coefficient was first proposed (in the open literature) as a measure of the transfer of radionuclides to animal-derived food products by Ward et al. [43] to describe the transfer of radiocaesium from the diet to the milk of dairy cattle. They defined the transfer coefficient as the ratio between the radiocaesium activity concentration in milk and the daily dietary radionuclide intake. Ward et al. [43] reported that this parameter exhibited less variability between individual animals within the experimental herd than expressing transfer as the total amount of Cs excreted in milk expressed as a percentage of intake. The same authors also defined the meat transfer coefficient as the ratio of the ^{137}Cs activity concentration in boneless meat to the dietary daily ^{137}Cs intake [44].

Other early papers by the same group define transfer coefficient as a percentage and not a fraction [45-46]. In further discussion of the use of the transfer coefficient, Ward and Johnson

[47] commented that the wider use of the diet to milk transfer coefficient, F_m, appeared to be justified and that factors such as stable element intake, soil intake, milk production rate, metabolic rate and inhalation could be ignored for most situations. However, they had previously acknowledged that assessment of transfer coefficients was based primarily on Cs data for cow milk and noted the lack of critical evaluations for other species and radionuclides [48].

To estimate transfer coefficients the dietary composition of the animal must be quantified. For agricultural animals this varies according to feeding strategies (indoors or grazing), maintenance requirements, agricultural practices and diet composition and characteristics such as dry matter digestibility. The relative proportion of grass, grain and other dietary constituents is important in determining radionuclide intake by agricultural animals, since grassy vegetation tends to be more highly contaminated. Typical dietary constituents for agricultural animals vary between and within countries, and with season.

Following the publications of Ward and co-workers in the 1960's, the transfer coefficient was widely adopted as the basis for quantifying transfer to both milk (F_m, d L^{-1} or d kg^{-1}) and meat (F_f, d kg^{-1}) for all radionuclides (the definition in later publications generally being the *equilibrium* ratio of the activity concentration in milk/meat to the daily dietary radionuclide intake). By the late 1970's–early 1980's, transfer coefficient values were being recommended for most radionuclide-animal product combinations [49-52]; such recommended values are incorporated into many food chain models [53-56].

6.1. Factors influencing estimation of transfer coefficients

In contrast to the statement by Ward and Johnson [47], a number of authors have reported variations in transfer coefficients for some radionuclides.

6.1.1. Intake estimation

Confidence in estimates of the amount of feed intake by experimental animals is clearly greater for most experimental studies under controlled conditions than it is for field studies where intake is often not measured. For the latter, different approaches are used for estimating mass intake, some based on agricultural production criteria but others using 'expert' judgement and this can lead to variability in reported F_m values [3].

6.1.2. Assumption of equilibrium

By definition, for a transfer coefficient to be valid the radionuclide activity concentration in tissues or milk needs to be at equilibrium with the dietary intake of the radionuclide. There can be considerable temporal variation in an animal's intake of radionuclides and hence tissue concentrations may be constantly changing. In the case of milk (the product for which Ward et al. [46] originally suggest transfer coefficients) an approximate equilibrium is reached rapidly for many radionuclides. However, experimental observations, from which transfer coefficients are derived, are often not conducted for long enough for equilibrium to have been reached in tissues or milk. The requirement of equilibrium conditions is often not met for radionuclides with short physical half lives or for those radionuclides with long radioactive and biological half lives in tissues (Pu) so that activity concentrations in tissues will not have equilibrated with the diet by the time of slaughter. For this reason, dynamic models describing the behaviour of radionuclides within animal tissues have been developed. These models can be used to predict radionuclides activity concentrations in different tissues following continuous, single or varying intakes [14, 57-59].

6.1.3. Physicochemical form

Physicochemical form affects transfer to tissue, mostly through changing the extent of gastrointestinal absorption. The best documented example of an effect of form is for radiocaesium, where transfer coefficients vary with a number of factors. Differences of over two orders of magnitude in transfer coefficients to the milk and meat of ruminants have been reported for different environmental dietary sources of radiocaesium, including soils, sediments, milk (for suckling lambs) and fungi [60-63].

Furthermore, Voigt et al. [63] noted differences in radiocaesium transfer coefficients to hen eggs and meat between contaminated wheat and grass pellet diets. Further variation in transfer coefficients has been explained by metabolic factors including dietary intake rates [64], lactation [65] and exercise [66].

After the Chernobyl accident transfer coefficients for recently deposited radiocaesium were reported to be lower than for plant-incorporated radiocaesium [47, 60, 61, 67]. The proportion of soil-associated radionuclides is an important factor affecting overall transfer to animals. The ingestion of soil adhered to vegetation can be the major source of radionuclide intake, especially for those radionuclides that have low soil-plant transfer [68]. For radiocaesium, although soil adhered to ingested vegetation often forms an important component of the total radiocaesium intake [69, 70], it is generally not an important contribution to radiocaesium in animal-derived products [71]. This is because the transfer of radiocaesium associated with mineral soil to animal products is much lower than the transfer of radiocaesium from vegetation [62].

However, for certain scenarios the ingestion of radiocaesium via soil/sedimentary adhesion can be important. For example, animals grazing tidally inundated pastures near the Sellafield reprocessing plant, can ingest over 90 % of their radiocaesium intake in the form of contaminated silt particles deposited by the tide on vegetation surfaces, and roughly 60 % of radiocaesium in the animal tissues can be estimated to be derived from the ingestion of contaminated silt [72]. However, because of the low bioavailability of the sediment associated radiocaesium, contamination of animals is lower than would be expected for plant-incorporated sources [73].

6.1.4. Effect of age/body weight

Transfer coefficients of radionuclides are generally higher to animals with a lower body mass [74]. Thus, for instance, transfer coefficients to lambs will generally be larger than those to ewes. For food production purposes, the liveweight at slaughter will be the most relevant basis for estimation.

6.1.5. Stable element

The clearest example of the effect of stable element analogue status on radionuclide transfer coefficients is that for radiostrontium transfer with respect to Ca intake. A relationship (Fig. 4) between the transfer coefficient for radiostrontium to milk and calcium intake has been proposed [75, 76]:

$$F_m Sr = \frac{OR_{milk-diet} \times [Ca]_{milk}}{I_{Ca}} \qquad (1)$$

where F_mSr is the transfer coefficient of radiostrontium to milk; $[Ca]_{milk}$ the concentration of calcium in milk; I_{Ca} the daily intake of calcium and $OR_{milk\text{-}diet}$ the observed ratio between the diet and milk. The observed ratio is a measure of the discrimination in the transfer of strontium from the diet to milk compared with that of calcium [77], i.e.:

$$OR_{milk-diet} = \frac{[Sr]_{milk} \,/\, [Ca]_{milk}}{[Sr]_{diet} \,/\, [Ca]_{diet}} \qquad (2)$$

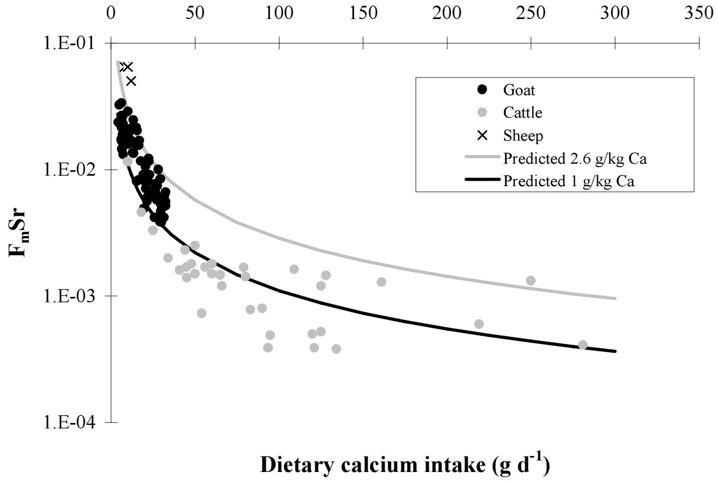

FIG. 4. Comparison between calcium intake and F_m for strontium in ruminants based on Beresford et al. [76-78]. The lines represent predicted values from equation ~~6.~~2 based upon calcium contents in milk of 1 g kg^{-1} for cows and goats and 2.6 g kg^{-1} and sheep.

6.2. Transfer to milk

Whilst compiling the database we did not use review values, in most cases the original data were consulted. Source information for the database included F_m values derived from:

i. experiments with daily administration of radionuclides to animals, or stable elements in the feed, for which the estimated plateau concentration could be divided by the daily intake;
ii. experiments (of an adequate length) where following a single oral administration of an isotope, the time-integrated activity concentration in milk could be estimated, for instance, by measuring the fraction of the total activity recovered in milk and dividing by the daily rate of milk secretion;
iii. for elements where we did not have F_m values we used the approach adopted previously by Ng et al.[50] of comparing stable element concentrations in milk with those in feeds reported by agricultural science review publications and assuming a daily herbage intake (see detailed explanation below).

For some literature, critical information was missing from the document. Where we felt it was justified to do so, we derived values using the following guidelines:

i. If daily herbage intake or daily milk output were not given we assumed values from similar literature in the same decade (this was often taken from papers by the same authors);

ii. Excluded low bioavailability sources for Cs F_m values (see text below);
iii. Excluded experimental data for animals where the diet was deficient in the element considered (e.g. < 40 g Ca d^{-1} for cattle, where the information was provided, when determining Sr F_m values);
iv. Excluded experimentally increased stable element intake data and only used data for control animals. For some elements there is good evidence that increasing stable element intakes does not lead to a proportional increase in the milk concentrations (e.g. Cd, Zn). In situations where animals are ingesting higher than normal rates of the elements of interest (e.g. heavy metals), then the transfer parameters we have derived in the tables below will overestimate concentrations in milk.

Reports by Ng et al. [49-52] constituted a major source of information for Technical Reports Series No. 364 [4]. Unfortunately, for many of the data reported for different elements it is not possible to trace the reference sources used as only a bibliographic reference number is given in the Ng et al.references. Therefore, where it was not possible to identify source material we have largely confined our use of the Ng et al.information to comparisons with other sources. The agricultural literature contains a wealth of data on many stable elements in milk and herbage which can be used to derive transfer parameters.

We have used a number of key reviews [16, 25, 79-85] to identify typical concentrations of elements within milk and herbage from which we have derived concentration ratios for cow, sheep, goat and horse milk. By assuming dry matter intake rates of 16 kg d^{-1} DM for lactating cows and 1.5 kg d^{-1} DM for sheep and goats, we have also estimated transfer coefficient values. This may potentially overestimate transfer, as for some nutrient elements a considerable proportion of the diet may be supplied in feed supplements within developed farming systems (i.e. we may have underestimated the nutrient intake rate). There is obviously variation in dry matter intake (DMI) but this is unlikely to influence the derived F_m numbers by more than a factor of 2-3 (DMI is not required for the estimation of CR).

Many of the stable elements considered will be under homeostatic control and transfer will therefore not be linear with intake rate. However, given the large databases we have based these values on, they are likely to be representative of 'typical values' taking into account the provisos above. Where this method has been used to select the recommended value of F_m it is identified on the tables below.

6.2.1. Transfer coefficients

Tables 5, 6 and 7 give the F_m values for cow, sheep and goat milk. All data for F_m values below are in units of d L^{-1}. The type of data source is specified for each element in the table. For cow milk there were adequate data in the database to derive values for most of the elements whereas for sheep and goat milk stable element compilation was used more often. Where recommended values are derived from the database, summary statistics are provided in the tables; if derived from stable element review data only the best estimate is presented. This work has resulted in F_m values being recommended for a number of additional elements not included in Technical Reports Series No. 364 [4]:

Cow – Be, Cd, Cl, Cu, P, Pb, Se, W and Zn

Goat – Ca, Cd, Ce, Co, Cr, Fe, Mn, Na, Ni, P, Pb, Pm, Po, S, Se, U, Y and Zn

Sheep – Ba, Ca, Cd, Co, Cr, Fe, Mn, Na, Ni, P, Pb, S, Te and Zn.

The review considered any isotope (including the stable element) for most radionuclides and the recommended F_m values are based upon an analyses of values derived for all of these.

In some instances, F_m values derived from the database are based on stable element data only. These have been used in preference to values derived from the stable element review as they are based upon paired milk-feed samples. However, in most case values derived from both stable element approaches are in close agreement with each other.

The only exception to the approaches used to derive F_m values in Tables 5-7 was the Pu F_m value for cow milk which was taken from the critical review of Howard et al. [86] (see below).

In many instances, the F_m values presented in Tables 5-7 are similar to those in Technical Reports Series No. 364 [4] although often based on more data. Where there are substantial variations (near or over an order of magnitude) between the geometric mean values presented here and the recommended values in Technical Reports Series No. 364 [4] these differences are discussed below.

Some elements in Technical Reports Series No. 364 milk tables [4] were not included in Tables 5-7, in particular:

> *^{3}H* – It is recommended that F_m values are not used for this radionuclide, an alternative approach is given below[1].
>
> *Mg & Cu* – Not selected for consideration in this work
>
> *Tc* – The Technical Reports Series No. recommended isotope specific values for cow and goat milk. Studies of the transfer of Tc to animals have tended to use the short-lived gamma-emitting radioisotopes ^{99m}Tc and ^{95m}Tc. These studies have shown considerable isotope-specific variation in transfer. Ennis et al. [87] report the transfer of ^{99m}Tc to the milk of goats to be 1.5×10^{-4} d l^{-1}, compared with values of 8.5×10^{-4} d l^{-1} and 1.1×10^{-2} d l^{-1} for ^{95m}Tc and ^{99}Tc respectively. Other reviewers have suggested that ^{99m}Tc and ^{95m}Tc data should not be used to model ^{99}Tc [88]. The F_m values of Ennis et al. were inversely proportional to the specific activity of the three Tc isotopes. The authors suggested that this is due to the differential rates of reduction of pertechnate between the three isotopes in the rumen. However, neither Ennis et al. nor the accompanying paper of Johnson et al. [89] give details of the amount of Tc administered and it therefore makes it difficult to comment on the relevance of their results for determining transfer at likely environmental levels. Given this uncertainty we have not included recommended values in the tables presented here. Data for the transfer of Tc isotopes for cattle and goats can be found in Ennis et al. [87], Johnson et al. [89], Voigt et al. [90], Von Wiechen et al. [91], Jones et al. [92] and Bondietti and Garten [93].

Cow Milk

In the case of Cr, the value derived here is more than an order of magnitude greater than in Technical Reports Series No. 364 [4]. The latter was based upon a single reported value (which is included in the database compiled here) whereas the revised value is based upon three values. Additionally, the dietary source used in the studies quoted in Technical Reports Series No. 364 [4] may have had comparatively low availability (the Cr source was chromate).

The F_m value for Ni in Table 5 is over an order of magnitude lower than the Technical Reports Series No. 364 [4] value. However, the Technical Reports Series No. 364 [4] value

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[1] See Davis et al. 'Specific activity models and parameter values for tritium, ^{14}C and ^{36}Cl' in this publication.

was based on a model prediction and did not consider any actual data; the value recommended here derived from the database is similar to that which can be derived from stable element reviews of 5×10^{-3} d kg^{-1}.

The Ca value reported here is almost an order of magnitude higher than that in Technical Reports Series No. 364 [4]. The value recommended in Technical Reports Series No. 364 [4] cannot be checked because it is wrongly referenced. We are confident that the value recommended here is reasonable as it is approximately one order of magnitude higher than that for Sr which is to be expected given that the observed ratio between the transfer of Ca and Sr from the diet to milk is generally accepted to be *circa* 0.1 [76]. The F_m values recommended in Technical Reports Series No. 364 [4] for Sr and Ca were approximately equal and therefore unlikely to be reliable.

The value recommended here for Pu is nearly an order of magnitude higher than that in Technical Reports Series No. 364 [4]. The value in Table 5 is based upon the finding of recent experimental work and a recent critical evaluation [86].

The value for U transfer to cow milk of 1.8×10^{-3} d L^{-1} in Table 5, which is based on three Russian language data sources, seems rather high taking into account the gastrointestinal absorption value. The value can be compared with the value in Ng et al. [50] of 7.3×10^{-5} d L^{-1} based on unpublished data obtained at a weapons test site and 6.1×10^{-4} d L^{-1} based on a single unreferenced balance study of linked milk and vegetation assuming 10 kg d^{-1}(DW) herbage intake. Thorne [94], in a recent review of the Ng et al. [50, 51] values and those of Prister [22] (which contains the same information as one of the original data used in Table 5), derived an F_m value of 2.0×10^{-4} d L^{-1} with a range of 6.0×10^{-5}–6.0×10^{-4} d L^{-1}. The F_m value of 4.0×10^{-4} d L^{-1} recommended in Technical Reports Series No. 364 has an incorrect data source attributed to it.

Table 5 does not give a value for Np, the Technical Reports Series No. 364 value for Np is taken from Ng et al. [49], who refers to recommended review values from [55]. We have not used review values in our compilation.

Goat milk

There were no substantial differences between the geometric mean values in Table 6 and the recommended values given in Technical Reports Series No. 364 [4]. An F_m value for Pu given for goat milk in Technical Reports Series No. 364 is based on a model derivation by Coughtrey [10], but there are no supporting relevant data so it is not included in Table 6.

Sheep milk

The Pu F_m value given here (Table 7) is more than an order of magnitude higher than that in Technical Reports Series No. 364 [4]. It is based upon the study of Beresford et al. [96] and for sheep grazing a UK salt marsh contaminated by marine discharges from the Sellafield reprocessing plant. Such data would not be used for Cs as the bioavailability is known to be comparatively low. However, evidence is not available to suggest a low bioavailability of Pu to animals for this area [9]. Furthermore, the recommended value is reasonable compared to that in Table 5 for cow milk.

An F_m value for Am for sheep milk was recommended in Technical Reports Series No. 364 this was based on a model derivation by Coughtrey [10]. However, there are no supporting data and hence it is not included in Table 7. The Technical Reports Series No. 364 value for Ag was derived from milk activity concentrations reported as the limit of detection; such values are not included in this review.

TABLE 5. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO COW MILK d L^{-1}

Element	N	GM	GSD	AM	SD	Min	Max	Source
Am	1			4.2×10^{-7}				Database
Ba	15	1.6×10^{-4}	2.7	2.5×10^{-4}	2.4×10^{-4}	3.8×10^{-5}	7.3×10^{-4}	Database–includes stable data
Be	1			8.3×10^{-7}				Database
Ca	15	1.0×10^{-2}	1.7	1.2×10^{-2}	5.9×10^{-3}	4.0×10^{-3}	2.5×10^{-2}	Database–includes stable data
Cd	8	1.9×10^{-4}	15	1.9×10^{-3}	3.4×10^{-3}	1.8×10^{-6}	8.4×10^{-3}	Database–includes stable data
Ce	6	2.0×10^{-5}	5.8	4.7×10^{-5}	4.9×10^{-5}	2.0×10^{-6}	1.3×10^{-4}	Database
Co	4	1.1×10^{-4}	2.0	1.3×10^{-4}	1.1×10^{-4}	6.0×10^{-5}	3.0×10^{-4}	Database
Cr	3	4.3×10^{-4}	26	2.0×10^{-3}	2.1×10^{-3}	1.0×10^{-5}	4.3×10^{-3}	Database–includes stable data
Cs	288	4.6×10^{-3}	2.0	6.1×10^{-3}	6.3×10^{-3}	6.0×10^{-4}	6.8×10^{-2}	Database–includes stable data
Fe	7	3.5×10^{-5}	2.0	4.2×10^{-5}	2.7×10^{-5}	1.0×10^{-5}	9.7×10^{-5}	Database–includes stable data
I	104	5.4×10^{-3}	2.4	9.1×10^{-3}	7.0×10^{-3}	4.0×10^{-4}	2.5×10^{-2}	Database–includes stable data
Mn	4	4.1×10^{-5}	4.9	1.0×10^{-4}	1.5×10^{-4}	7.0×10^{-6}	3.3×10^{-4}	Database
Mo	7	1.1×10^{-3}	2.3	1.5×10^{-3}	1.7×10^{-3}	4.3×10^{-4}	5.2×10^{-3}	Database–includes stable data
Na	7	1.3×10^{-2}	2.0	1.6×10^{-2}	1.5×10^{-2}	5.0×10^{-3}	5.0×10^{-2}	Database–includes stable data
Nb	1			4.1×10^{-7}				Database
Ni	2			9.5×10^{-4}		6.5×10^{-4}	1.3×10^{-3}	Database–stable data only
P				2.0×10^{-2}				From stable element review data
Pb	15	1.9×10^{-4}	1.0	3.3×10^{-4}	3.5×10^{-4}	7.3×10^{-6}	1.2×10^{-3}	Database–includes stable data
Po	4	2.1×10^{-4}	1.8	2.3×10^{-4}	9.7×10^{-5}	8.9×10^{-5}	3.0×10^{-4}	Database
Pu				1.0×10^{-5}				Recent review paper
Ra	11	3.8×10^{-4}	2.3	5.1×10^{-4}	3.8×10^{-4}	9.0×10^{-5}	1.4×10^{-3}	Database
Ru	6	9.4×10^{-6}	8.5	3.6×10^{-5}	5.3×10^{-5}	6.7×10^{-7}	1.4×10^{-4}	Database
S	1			7.9×10^{-3}				Database
Sb	3	3.8×10^{-5}	2.5	5.2×10^{-5}	5.1×10^{-5}	2.0×10^{-5}	1.1×10^{-4}	Database
Se	12	4.0×10^{-3}	2.1	5.2×10^{-3}	4.5×10^{-3}	1.5×10^{-3}	1.6×10^{-2}	Database–includes stable data
Sr	154	1.3×10^{-3}	1.7	1.5×10^{-3}	8.1×10^{-4}	3.4×10^{-4}	4.3×10^{-3}	Database–includes stable data
Te	11	3.4×10^{-4}	2.4	4.5×10^{-4}	2.9×10^{-4}	7.8×10^{-5}	1.0×10^{-3}	Database
U	3	1.8×10^{-3}	3.5	2.9×10^{-3}	2.9×10^{-3}	5.0×10^{-4}	6.1×10^{-3}	Database
W	7	1.9×10^{-4}	3.1	2.9×10^{-4}	2.4×10^{-4}	3.4×10^{-5}	6.8×10^{-4}	Database
Zn	8	2.7×10^{-3}	3.9	4.3×10^{-3}	3.0×10^{-3}	1.3×10^{-4}	9.0×10^{-3}	Database–includes stable data
Zr	6	3.6×10^{-6}	4.3	7.1×10^{-6}	6.9×10^{-6}	5.5×10^{-7}	1.7×10^{-5}	Database

TABLE 6. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO GOAT MILK d L^{-1}

Element	N	GM*	GSD	AM	SD	Min	Max	Source
Am	2			6.9×10^{-6}		3.7×10^{-6}	1.0×10^{-5}	Database
Ba	3	1.1×10^{-2}	9.9	5.4×10^{-2}	8.7×10^{-2}	2.1×10^{-3}	1.5×10^{-1}	Database–includes stable data
Ca	12	7.3×10^{-2}	1.9	8.3×10^{-2}	3.4×10^{-2}	1.2×10^{-2}	1.4×10^{-1}	Database–includes stable data
Cd	1			1.6×10^{-2}				Database–stable data only
Ce	1			4.0×10^{-5}				Database
Co	1			5.0×10^{-3}				Database–stable data only
Cr	2			1.5×10^{-2}		2.9×10^{-3}	2.8×10^{-2}	Database–stable data only
Cs	28	1.1×10^{-1}	2.2	1.3×10^{-1}	8.0×10^{-2}	7.0×10^{-3}	3.3×10^{-1}	Database–stable data only
Fe		5.2×10^{-2}						From stable element review data
I	24	2.2×10^{-1}	2.9	3.3×10^{-1}	2.3×10^{-1}	2.7×10^{-2}	7.7×10^{-1}	Database
Mn		1.0×10^{-3}						From stable element review data
Mo	4	8.2×10^{-3}	1.4	8.5×10^{-3}	2.5×10^{-3}	5.0×10^{-3}	1.1×10^{-2}	Database–includes stable data
Na		1.2×10^{-1}						From stable element review data
Nb	1			6.4×10^{-6}				Database
Ni	2			8.3×10^{-2}		3.2×10^{-3}	1.6×10^{-1}	Database–stable data only
Np	1			5.3×10^{-5}				Database
P		2.9×10^{-1}						From stable element review data
Pb	1			6.0×10^{-3}				Database–stable data only
Pm	1			2.7×10^{-5}				Database
Po	2			2.3×10^{-3}		1.8×10^{-3}	2.7×10^{-3}	Database
S	12	3.8×10^{-2}	1.7	4.3×10^{-2}	1.9×10^{-2}	1.6×10^{-2}	6.8×10^{-2}	Database–includes stable data
Se	2			6.9×10^{-2}		5.9×10^{-2}	7.9×10^{-2}	Database–stable data only
Sr	21	1.6×10^{-2}	2.0	2.1×10^{-2}	2.0×10^{-2}	5.8×10^{-3}	8.1×10^{-2}	Database–includes stable data
Te	1			4.4×10^{-3}				Database
U	1			1.4×10^{-3}				Database
Y	1			2.0×10^{-5}				Database
Zn		6.4×10^{-2}						From stable element review data
Zr	1			5.5×10^{-6}				Database

* Where the value is derived from a stable element review it is not a GM

TABLE 7. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO SHEEP MILK d L^{-1}

Element	N	GM*	GSD	AM	SD	Min	Max	Source
Ba	1			4.1×10^{-2}				Database–stable data only
Ca		2.3×10^{-1}						From stable element review data
Cd	1			4.9×10^{-2}				Database–stable data only
Co	2			2.7×10^{-3}		1.2×10^{-3}	4.1×10^{-3}	Database–includes stable data
Cr	1			2.0×10^{-2}				Database–stable data only
Cs	28	5.8×10^{-2}	2.3	7.7×10^{-2}	6.1×10^{-2}	6.0×10^{-3}	3.2×10^{-1}	Database
Fe		7.9×10^{-2}						From stable element review data
I	7	2.3×10^{-1}	3.3	3.5×10^{-1}	3.0×10^{-1}	3.0×10^{-2}	9.4×10^{-1}	Database
Mn	1			2.4×10^{-3}				Database–stable data only
Na	1			1.0×10^{-1}				Database
Ni	1			2.8×10^{-1}				Database–stable data only
P		3.1×10^{-1}						From stable element review data
Pb		3.5×10^{-2}						From stable element review data
Pu	1			1.0×10^{-4}				Database
S		1.5×10^{-1}						From stable element review data
Sr	4	2.7×10^{-2}	1.2	3.0×10^{-2}	1.2×10^{-2}	1.3×10^{-2}	4.0×10^{-2}	Database
Te	1			2.9×10^{-3}				Database
Zn		8.1×10^{-2}						From stable element review data

* Where the value is derived from a stable element review it is not a GM

6.2.2. Concentration ratios – an alternate method of quantifying transfer to animal products

Over the last 40 years following the introduction of the transfer coefficient concept many studies have been conducted to determine values for a range of radionuclide – animal products. On the basis of these studies, it is generally accepted that transfer coefficients for smaller animals are higher than those for larger animals, and that those for adults are lower than those for (smaller) young livestock. Beresford et al. [97] suggest that much of this difference could be because transfer coefficients incorporate dry matter intake which increases with animal size.

Smith et al. [98] used the recommended milk and meat radiocaesium transfer coefficient values for different farm animals from Technical Reports Series No. 364 [4] together with recommended dry matter intake rates to estimate CR values (Table 8). Whilst the transfer coefficient values for meat varied over three orders of magnitude from 10 d kg^{-1} (chicken) to 7.9×10^{-3} d kg^{-1} (beef) the range in estimated CR values was only two-fold. Similarly, estimated CR values for milk varied approximately two-fold compared to the roughly order of magnitude range in F_m values. Concentration ratio values for meat and milk, estimated from the data given in Technical Reports Series No. 364 [4] for most other radionuclides would also have a considerably lower variability between species than that seen for recommended transfer coefficients. The similarity in CR values between species should perhaps not be surprising given that the concentrations of many elements in meat or milk are similar across species [99-100]; Table 8 demonstrates this for potassium (a chemical analogue of caesium). The use of concentration ratios for radioisotopes of the macro-elements S, H and C rather than transfer coefficients has been suggested by Howard et al. [101] and Galeriu et al. [102] as the elemental contents of meat and milk do not vary significantly with factors, such as milk yield and live-weight, which influence dry matter intake rates and hence estimated transfer coefficients.

Differences in dry matter intake may also contribute to the higher transfer coefficients observed for younger livestock compared with adults. For instance, Beresford et al. [96] presents *CR* and F_f values for two groups of sheep, adult ewes and lambs, fed vegetation contaminated with ^{60}Co, ^{95}Nb, ^{106}Ru, ^{134}Cs, ^{137}Cs, ^{238}Pu, ^{239,240}Pu and ^{241}Am. Transfer coefficients suggested significantly higher transfer of radionuclides from the vegetation to the tissues of lambs compared with ewes more frequently than did transfer expressed as *CR*; the daily dry matter intake rate of the lambs was 0.6 kg d^{-1} compared with 0.9 kg d^{-1} for the ewes.

TABLE 8. RECOMMENDED TRANSFER COEFFICIENTS FOR RADIOCAESIUM AND DRY MATTER FEED INTAKE RATES [4] AND ESTIMATED CONCENTRATION RATIOS, TYPICAL POTASSIUM CONCENTRATIONS ARE ALSO SHOWN (ADAPTED FROM SMITH ET AL. [98])

Animal	Daily dry matter intake (kg d^{-1})	F_f (d kg^{-1}) or F_m, (d l^{-1})	*CR* [dimensionless]	Typical K concentration (mg kg^{-1} FW)
		Milk		
Cow	16.1	7.9×10^{-3}	0.13	1430
Goat	1.3	1.0×10^{-1}	0.13	1930
Sheep	1.3	5.8×10^{-2}	0.075	1370
		Meat		
Beef	7.2	5.0×10^{-2}	0.36	3040
Lamb	1.1	4.9×10^{-1}	0.54	3060
Pork	2.4	2.4×10^{-1}	0.58	3765
Chicken	0.07	1.0×10^{1}	0.7	2570

Previously when considering wild animals we suggested that, for some radionuclides, whole-body *CR*s could be considered as a constant across animal species for radioisotopes of Co, Cs, Ra, Sb, Sr, U, Zn and Zr [103]. This was based upon an algebraic derivation of *CR* from allometric (body mass dependent) relationships for radionuclide biological half-life [104-105] and dry matter intake [106]. A constant *CR* could also be similarly hypothesised from the allometric relationships presented for F_f for radiocaesium and radioiodine by MacDonald [29] [see 103, 107]. However, constant *CR* values across species could not be suggested for the actinides and lanthanides on the basis of current information.

The *CR* also has the advantage in field studies that dietary dry matter intake does not need calculating or, as is more often the case, a value is not assumed. However, in many circumstances when the diet is comprised of a number of foodstuffs, the relative proportions of all dietary components will be required to apply *CR* values in assessments.

On the basis of the above discussion we have estimated *CR* values for the milk of cows, sheep and goats from the database compiled to derive F_m values. In some instances, this has been possible as the originating authors reported the concentrations in the milk and complete diet, whereas in others we have estimated *CR* as the product of the F_m and dry matter intake. Unfortunately, many authors do not report the information required to estimate *CR* values; few data entered from the Russian language literature could be used to derive *CR*s. If data were not available in the database, then the stable element review values have been used (see discussion above). From the stable element review it was also possible to estimate *CR* values for horse milk. Table 9 compares the *CR* values for the four animal species.

There is no consistent ranking between species in *CR* values as accepted for F_m values. For most elements, the *CR* values available differ little between the species considered (see the ratio of minimum to maximum values in Table 9). The only elements for which there is a greater than one-order of magnitude variation between the species are Ba, Fe and Pb. However, variation in *CR* values is still much less than that between F_m values (compare Table 9 to Tables 5-7) even though *CR* values are available for an extra species in the case of Ba and Fe.

A particular advantage of being able to assume that *CR* for many radionuclides varies little between species is that generic values can be derived for food-producing animals for which no data are currently available. The mean value presented in Table 9 is recommended for this purpose. Pragmatically, however, transfer coefficients will continue to be used for some time, not least because they are more numerous in the literature than *CR* values.

Whilst we propose that *CR* is a more robust and generic parameter than transfer coefficients for milk, it is still subject to variation due to a number of the parameters which have been shown to influence transfer coefficients (e.g. bioavailability of the dietary source, concentration of stable/analogue elements in the diet).

6.3. Transfer to meat

6.3.1. Transfer coefficients

Approaches to deriving F_f values for meat were as described above for the compilation of F_m values. Exceptions were that: (i) no additional stable element review of animal nutrition literature was conducted although the database does include some stable element values; (ii) single dose studies were not used unless sufficient time series data were available. Considerable Russian language data has recently been made available for use in this review by Fesenko et al. [108].

TABLE 9. CONCENTRATION RATIOS FOR THE MILK OF DIFFERENT ANIMALS[1]

Element	Cow					Goat					Sheep					Horse	Mean	Ratio
	CR	SD	Min	Max	N	*CR*	SD	Min	Max	N	*CR*	SD	Min	Max	N	*CR*	All species	Min/ Max
Ba	1.3×10^{-2}	1.6×10^{-3}	1.2×10^{-2}	1.5×10^{-2}	3	1.2×10^{-1}		1.4×10^{-2}	2.3×10^{-1}	2	6.1×10^{-2}				1	3.5×10^{-3}	5.0×10^{-2}	2.9×10^{-2}
Ca	2.5×10^{-1}					2.0×10^{-1}	8.3×10^{-2}	1.3×10^{-1}	2.9×10^{-1}	4	3.4×10^{-1}					1.5×10^{-1}	2.4×10^{-1}	4.4×10^{-1}
Cd	4.3×10^{-2}	7.4×10^{-2}	2.7×10^{-5}	1.3×10^{-1}	3	2.4×10^{-2}				1	7.4×10^{-2}				1		4.7×10^{-2}	3.3×10^{-1}
Cl	6.9×10^{-2}				1												6.9×10^{-2}	
Ce	3.2×10^{-3}				1												3.2×10^{-3}	
Co	2.5×10^{-3}				1	7.6×10^{-3}				1	6.2×10^{-3}				1		5.4×10^{-3}	3.3×10^{-1}
Cr	4.0×10^{-2}		3.7×10^{-2}	4.3×10^{-2}	2	4.1×10^{-2}				1	3.0×10^{-2}				1		3.7×10^{-2}	7.2×10^{-1}
Cs	1.1×10^{-1}	1.2×10^{-1}	3.6×10^{-3}	6.9×10^{-1}	119	1.8×10^{-1}	6.5×10^{-2}	6.3×10^{-2}	3.0×10^{-1}	12	1.7×10^{-1}	1.3×10^{-1}	2.0×10^{-2}	5.8×10^{-1}	17		1.5×10^{-1}	6.4×10^{-1}
Fe	1.2×10^{-3}	2.4×10^{-4}	1.0×10^{-3}	1.5×10^{-3}	3	3.4×10^{-2}				1	5.2×10^{-2}				1	9.3×10^{-3}	2.4×10^{-2}	2.4×10^{-2}
I	3.0×10^{-1}	2.8×10^{-1}	3.0×10^{-3}	7.9×10^{-1}	44	5.0×10^{-1}	5.8×10^{-1}	8.4×10^{-2}	1.2	3	5.8×10^{-1}	2.5×10^{-1}	2.5×10^{-1}	8.8×10^{-1}	5		4.6×10^{-1}	5.2×10^{-1}
Mn	4.5×10^{-3}		8.6×10^{-4}	8.2×10^{-3}	2	1.5×10^{-3}				1	3.6×10^{-3}				1	1.6×10^{-3}	2.8×10^{-3}	3.2×10^{-1}
Mo	2.8×10^{-2}	1.3×10^{-2}	1.9×10^{-2}	4.3×10^{-2}	3	2.7×10^{-2}				1						2.1×10^{-2}	2.5×10^{-2}	7.5×10^{-1}
Na	3.7×10^{-1}		2.3×10^{-1}	5.0×10^{-1}	2	1.8×10^{-1}					1.6×10^{-1}					6.0×10^{-2}	1.9×10^{-1}	1.6×10^{-1}
Nb	1.0×10^{-5}				1	1.9×10^{-5}				1							1.5×10^{-5}	5.3×10^{-1}
Ni	8.2×10^{-2}					2.5×10^{-1}				1	4.2×10^{-1}				1		2.5×10^{-1}	1.9×10^{-1}
P	3.1×10^{-1}					4.3×10^{-1}					4.7×10^{-1}					1.8×10^{-1}	3.5×10^{-1}	3.8×10^{-1}
Pb	2.4×10^{-3}	1.3×10^{-3}	9.9×10^{-4}	4.3×10^{-3}	7	9.0×10^{-3}				1	3.0×10^{-2}				1		1.4×10^{-2}	7.9×10^{-2}
Po	2.4×10^{-3}				1												2.4×10^{-3}	
S	1.4×10^{-1}				1	6.1×10^{-2}	3.0×10^{-2}	3.5×10^{-2}	1.0×10^{-1}	4	2.3×10^{-1}						1.4×10^{-1}	2.7×10^{-1}
Sb	2.7×10^{-3}				1												2.7×10^{-3}	
Se	5.7×10^{-2}	4.5×10^{-2}	2.6×10^{-2}	1.5×10^{-1}	7	3.5×10^{-2}											4.6×10^{-2}	6.2×10^{-1}
Sr	2.3×10^{-2}	2.2×10^{-2}	5.0×10^{-3}	1.4×10^{-1}	43	4.4×10^{-2}	4.4×10^{-2}	1.6×10^{-2}	1.2×10^{-1}	5						4.4×10^{-2}	3.7×10^{-2}	5.2×10^{-1}
Te	8.0×10^{-3}		4.8×10^{-3}	1.1×10^{-2}	2	1.2×10^{-2}				1							1.0×10^{-2}	6.7×10^{-1}
U	5.0×10^{-3}																5.0×10^{-3}	
Zn	7.5×10^{-2}	1.6×10^{-2}	5.5×10^{-2}	9.5×10^{-2}	6	9.6×10^{-2}					1.2×10^{-1}					5.5×10^{-2}	8.7×10^{-2}	4.6×10^{-1}
Zr	1.4×10^{-5}				1	1.7×10^{-5}				1							1.5×10^{-5}	8.3×10^{-1}

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[1] Shaded cells denote *CR* values estimated from the stable element review.

In summarising F_f values we have only used results reported for pigs, sheep and goats of 6 months or older (where this information was given). For cattle, only data from animals aged 1 year or older were included whilst for poultry only data for animals older than 40 d were used.

However, if data for animals above these ages were totally lacking for an element, data for younger animals were used; in the case of mammals data were not used for animals of less than a few months in age. If data were not available for chicken but were for duck, these values were used instead (only relevant for Na and Co).

Data from experiments of less than 20 d duration were not used for sheep, goats, pigs or poultry whilst for cattle experiments of less than 60 d duration were not used (this greatly reduced the number of data for Cs).

Beresford et al. [14, 109] and Crout et al. [18] report data from single administration studies which they interpreted by fitting models to data for consecutive slaughter dates over *circa* one year. Subsequent predictions of F_f were made for differing periods of continuous administration; values incorporated into the database were the equilibrium or 1000 d predictions. If required to convert reported dry weight values to fresh weight, it was assumed that the dry matter content of meat for all animal types was 25%.

Tables 10-14 present F_f values for beef, sheep meat, goat meat, pork and poultry respectively in the same manner as the summaries presented for milk in Table 5-7. All data for F_f values below are in units of d kg^{-1} fresh weight. The tables include F_f values for a number of animal-element combinations for which values were not available in Technical Reports Series No. 364 [4]:

Beef – La and Th

Sheep – Cd, Na and Pb

Goat – Y

Pork – Ca and P

Poultry – Na and Po

As for milk, all data for Tc were for isotopes other than ^{99}Tc and consequently F_f values are not included in the tables below; values for the shorter lived Tc isotopes can be found in Jones [92] and Johnson et al. [89].

A Russian language study by Taucin and Svilane 1962 [110] was used as the basis for a number of values given in Technical Reports Series No. 364, specifically for Mn, Fe, Co and Zn for sheep, cattle, poultry and pigs, as it was a key data source used by Ng et al.[52]. We have carefully examined the source paper and compared the derived F_f values with other data. Seven of the F_f values derived using this paper are substantially (order of magnitude) higher than those of other sources. We have therefore decided not to include this paper in the database and consequently there are fewer F_f values given for these elements in the meat (and also subsequent egg) tables than in Technical Reports Series No. 364.

Where the values in the Tables 10-14 deviate from those in Technical Reports Series No. 364 [4] by approximately an order of magnitude or more, the reasons for the difference is discussed below.

TABLE 10. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO COW MEAT d kg^{-1}

Element	N	GM	GSD	AM	AMSD	Min	Max	Source
Am	1			5.0×10^{-4}				Database
Ba	2			1.4×10^{-4}		5.0×10^{-5}	2.3×10^{-4}	Database
Ca	3	1.3×10^{-2}	30.0	2.0×10^{-1}	3.5×10^{-1}	1.0×10^{-3}	6.1×10^{-1}	Database–includes stable data
Cd	8	5.8×10^{-3}	7.8	1.9×10^{-2}	2.4×10^{-2}	1.5×10^{-4}	6.0×10^{-2}	Database–includes stable data
Cl	1			1.7×10^{-2}				Database–stable data only
Co	4	4.3×10^{-4}	2.3	5.2×10^{-4}	3.0×10^{-4}	1.3×10^{-4}	8.4×10^{-4}	Database
Cs	58	2.2×10^{-2}	2.4	3.0×10^{-2}	2.3×10^{-2}	4.7×10^{-3}	9.6×10^{-2}	Database–includes stable data
Fe	4	1.4×10^{-2}	1.5	1.5×10^{-2}	6.7×10^{-3}	9.0×10^{-3}	2.5×10^{-2}	Database–stable data only
I	5	6.7×10^{-3}	3.2	1.2×10^{-2}	1.5×10^{-2}	2.0×10^{-3}	3.8×10^{-2}	Database
La	3	1.3×10^{-4}	1.2	1.3×10^{-4}	2.0×10^{-5}	1.1×10^{-4}	1.5×10^{-4}	Database–stable data only
Mn	2			6.0×10^{-4}		6.0×10^{-4}	6.0×10^{-4}	Database–stable data only
Mo	1			1.0×10^{-3}				Database
Na	2			1.5×10^{-2}		1.0×10^{-2}	2.0×10^{-2}	Database
Nb	1			2.6×10^{-7}				Database
P	1			5.5×10^{-2}				Database–stable data only
Pb	5	7.0×10^{-4}	2.5	9.3×10^{-4}	6.4×10^{-4}	2.0×10^{-4}	1.6×10^{-3}	Database–includes stable data
Pu	5	1.1×10^{-6}	24.8	6.0×10^{-5}	1.3×10^{-4}	8.8×10^{-8}	3.0×10^{-4}	Database
Ra	1			1.7×10^{-3}				Database
Ru	3	3.3×10^{-3}	1.8	3.7×10^{-3}	2.3×10^{-3}	2.2×10^{-3}	6.4×10^{-3}	Database
Sb	2			1.2×10^{-3}		1.1×10^{-3}	1.3×10^{-3}	Database–young animals
Sr	35	1.3×10^{-3}	2.9	2.1×10^{-3}	2.2×10^{-3}	2.0×10^{-4}	9.2×10^{-3}	Database
Te	1			7.0×10^{-3}				Database
Th	6	2.3×10^{-4}	2.9	3.5×10^{-4}	3.3×10^{-4}	4.0×10^{-5}	9.6×10^{-4}	Database
U	3	3.9×10^{-4}	1.6	4.2×10^{-4}	2.0×10^{-4}	2.5×10^{-4}	6.3×10^{-4}	Database
Zn	6	1.6×10^{-1}	3.2	2.6×10^{-1}	2.4×10^{-1}	4.0×10^{-2}	6.3×10^{-1}	Database–stable data only
Zr	1			1.2×10^{-6}				Database

TABLE 11. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO SHEEP MEAT d kg^{-1}

Element	N	GM	GSD	AM	SD	Min	Max	Source
Ag	1			4.8×10^{-4}				Database
Am	1			1.1×10^{-4}				Database
Cd	1			1.2×10^{-3}				Database
Ce	1			2.5×10^{-4}				Database
Co	2			1.2×10^{-2}		8.0×10^{-3}	1.6×10^{-2}	Database
Cs	41	1.9×10^{-1}	2.2	2.7×10^{-1}	2.6×10^{-1}	5.3×10^{-2}	1.3	Database
I	1			3.0×10^{-2}				Database
Mn	1			9.0×10^{-3}				Database–stable data only
Na	1			1.1×10^{-1}				Database
Pb	2			7.1×10^{-3}		4.0×10^{-3}	1.0×10^{-2}	Database–stable data only
Pu	2			5.3×10^{-5}		2.0×10^{-5}	8.5×10^{-5}	Database
Ru	2			2.1×10^{-3}		6.3×10^{-4}	3.6×10^{-3}	Database
S	3	1.7	1.3	1.7	4.7×10^{-1}	1.2	2.1	Database
Sr	25	1.5×10^{-3}	1.7	1.7×10^{-3}	7.5×10^{-4}	3.0×10^{-4}	4.0×10^{-3}	Database
Zn	6	4.5×10^{-2}	2.2	5.9×10^{-2}	4.7×10^{-2}	2.0×10^{-2}	1.4×10^{-1}	Database

TABLE 12. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO GOAT MEAT d kg^{-1}

Element	N	GM	GSD	AM	SD	Min	Max	Source
Ba	1			1.3×10^{-5}				Database
Cs	11	3.2×10^{-1}	2.5	4.8×10^{-1}	5.3×10^{-1}	1.2×10^{-1}	1.9	Database
Nb	1			6.0×10^{-5}				Database
Sr	8	2.9×10^{-3}	1.2	3.0×10^{-3}	5.1×10^{-4}	2.0×10^{-3}	3.7×10^{-3}	Database
Te	1			2.4×10^{-3}				Database
Y	1			5.4×10^{-2}				Database
Zr	1			2.0×10^{-5}				Database

TABLE 13. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO PIG MEAT d kg^{-1}

Element	N	GM	GSD	AM	SD	Min	Max	Source
Ca	1			2.0×10^{-3}				Database–stable data only
Cs	22	2.0×10^{-1}	1.5	2.2×10^{-1}	9.0×10^{-2}	1.2×10^{-1}	4.0×10^{-1}	Database
Fe	1			3.0×10^{-3}				Database – young animals
I	2			4.1×10^{-2}		1.5×10^{-2}	6.6×10^{-2}	Database
Mn	1			5.3×10^{-3}				Database–stable data only
P	1			2.7×10^{-2}				Database
Ru	1			3.0×10^{-3}				Database
Se	1			3.2×10^{-1}				Database–stable data only
Sr	12	2.5×10^{-3}	2.7	3.6×10^{-3}	2.7×10^{-3}	5.0×10^{-4}	8.0×10^{-3}	Database–includes stable data
U	2			4.4×10^{-2}		2.6×10^{-2}	6.2×10^{-2}	Database
Zn	2			1.7×10^{-1}		1.3×10^{-1}	2.0×10^{-1}	Database

TABLE 14. TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO POULTRY MEAT d kg^{-1}

Element	N	GM	GSD	AM	SD	Min	Max	Source
Ba	2			1.9×10^{-2}		9.2×10^{-3}	2.9×10^{-2}	Database
Ca	2			4.4×10^{-2}		4.4×10^{-2}	4.4×10^{-2}	Database–stable data only
Cd	2			1.7		1.7	1.8	Database–stable data only (includes duck)
Co	2			9.7×10^{-1}		3.0×10^{-2}	1.9	Database (includes duck)
Cs	13	2.7	1.6	3.0	1.3	1.2	5.6	Database (includes duck)
I	3	8.7×10^{-3}	2.0	1.0×10^{-2}	5.6×10^{-3}	4.0×10^{-3}	1.5×10^{-2}	Database
Mn	2			1.9×10^{-3}		1.0×10^{-3}	2.8×10^{-3}	Database
Mo	1			1.8×10^{-1}				Database
Na	1			7.0				Database–duck
Nb	1			3.0×10^{-4}				Database
Po	1			2.4				Database
Se	4	9.7	2.3	1.3×10^{1}	1.1×10^{1}	4.1	2.8×10^{1}	Database–stable data only
Sr	7	2.0×10^{-2}	1.8	2.3×10^{-2}	1.2×10^{-2}	7.0×10^{-3}	4.1×10^{-2}	Database (includes duck)
Te	1			6.0×10^{-1}				Database
U	2			7.5×10^{-1}		3.0×10^{-1}	1.2	Database
Zn	3	4.7×10^{-1}	1.2	4.7×10^{-1}	7.9×10^{-2}	3.8×10^{-1}	5.3×10^{-1}	Database
Zr	1			6.0×10^{-5}				Database

Beef

The value for Sb reported in Technical Reports Series No. 364 [4] (4×10^{-5} d kg^{-1}) was based upon a single administration study reported by van Bruwaene et al. [111] with meat being measured after c. 102 d and the F_f being estimated based upon this measurement and the total excretion of Sb. The higher value (1.2×10^{-3} d kg^{-1}) in Table 10 is based upon two continuous feeding studies reported by Sirotkin et al. [25] and Sirotkin [112] although the animals were either 6 months old or slightly younger. The latter value appears reasonable compared with the *CR* value, which can be estimated from Boyer et al. [113] and, given the method of calculation used by van Bruwaene et al. [111], we have excluded the latter from the database.

The F_f value for Cd in Technical Reports Series No. 364 [4] was based upon one stable element data set and is an order of magnitude lower (at 4×10^{-4} d kg^{-1}) than the mean in Table 10. The Technical Reports Series No. 364 [4] value was based upon Ng et al. [52] and we cannot replicate this value from the original paper [114]; a value of 5×10^{-3} d kg^{-1} is included in the database estimated from the data of Nelmes et al. [114].

The F_f value for Ru in Technical Reports Series No. 364 of 5.0×10^{-2} d kg^{-1} is almost 10 fold higher than that in Table 10 and comes from a review by Coughtrey [10]. Coughtrey quotes two Russian sources (Annenkov 73 [115] and Sirotkin et al. 70 [25] and also values used in various models. The geometric mean value of 3.3×10^{-3} d kg^{-1} in Table 10 is based on three original Russian data sources; the value of F_f for sheep meat in Table 11 is also lower than previous model based values.

The geometric mean F_f value in Table 10 for Ca is nearly an order of magnitude greater than that in Technical Reports Series No. 364 [4]. The Ca F_f value in Table 10 is based on a limited data set (3 studies) two of which report values of the order of 10^{-3} (which includes the study used in Technical Reports Series No. 364 [4]) and one a value in the order of 10^{-1}. Excluding the 10^{-1} value would result in a mean of 2.2×10^{-3} d kg^{-1} (similar to the Technical Reports Series No. 364 [4] value). However, this would be lower than the F_f for Sr which is based upon many data whereas we would expect the Ca value to be approximately an order of magnitude higher (note the Sr F_f advised in Technical Reports Series No. 364 [4] was higher than that for Ca). The comparative difference between the geometric mean values for Sr and Ca in Table 10 is reflective of the difference expected.

The F_f value for Am of 5×10^{-4} d kg^{-1} in Table 10 is based on Sumerling et al. [116] and is more than an order of magnitude higher than the Technical Reports Series No. 364 value of 4×10^{-5} which is a model-derived value for 100d from Coughtrey [10].

There is considerable variation in the values derived for Pu in Table 10. The F_f value for Pu in Technical Reports Series No. 364 [4] of 1.0×10^{-5} d kg^{-1} comes from the review by Coughtrey [10] and is approaching an order of magnitude higher than the value based on 5 data sources in Table 10. Coughtrey refers to values derived by extrapolation from experimental data reported by Stanley [117] which was not used in our database.

An F_f value for beef for W is recommended in Technical Reports Series No. 364 [4] based upon a value derived by Ng et al. [52] from unassociated data, which is not referenced and is not included in Table 10.

The Technical Reports Series No. 364 [4] value for Y was based on that suggested by Ng et al. [52] derived from the comparative *CR* values of ^{88}Y and ^{137}Cs for reindeer. Whilst a Y value of 1.0×10^{-6} d kg^{-1} is reported in the Sirotkin's 1991 review [15] (i.e. three orders of magnitude lower than that advised in Technical Reports Series No. 364 [4]) the source of this value could not be identified and hence no F_f value for Y is suggested in Table 10. Similarly, whilst Sirotkin [118] reports a beef F_f value of 1.0×10^{-6} for Ce its source and derivation could not be confirmed and it is not included here. The value of 2.0×10^{-5} recommended for Ce in Technical Reports Series No. 364 [4] appears to be a 100 d model prediction from CEC 1987 [119]. Consequently, Table 10 does not present a value for Ce.

Sheep meat

The F_f value for Ru derived here (2.1×10^{-3} d kg^{-1}) on the basis of two experimental studies is approximately two orders of magnitude lower than the previous Technical Reports Series No. 364 [4] value. The F_f value for Ru in Technical Reports Series No. 364 [4] was based upon a model prediction only [41].

The geometric mean value for the F_f for Sr is 1.5×10^{-3} d kg^{-1} for sheep meat in Table 11. The value is based on data from the Russian language reported by Fesenko et al. [108] although some data for short periods of Sr administration to sheep from time series experiment have been excluded. The value recommended here is more than an order of magnitude lower than that in Technical Reports Series No. 364 of 4×10^{-2} d kg^{-1} based on a model prediction for equilibrium from Coughtrey [10].

The F_f value for Zn to sheep meat in Technical Reports Series No. 364 [4] was 4.1 d kg^{-1} taken from Ng et al. [52] which was based upon stable data reported by Taucin and Svilane [110] which, as explained above, has not been used in this review. This is at least an order of magnitude greater than any values within the database compiled here from stable Zn measurements and radiotracer studies. The database value is considerably lower, with a geometric mean of 4.5×10^{-2} d kg^{-1}.

Values of sheep meat F_f's for Rb, Cu, Fe and Nb are recommended in Technical Reports Series No. 364 [4] but not in Table 11. Rb and Cu were not included in this review. The value for Nb in Technical Reports Series No. 364 [4] (3×10^{-4} d kg^{-1}) was based on a model prediction by Coughtrey [10] and we have not included such values in the database. The value for Fe (7.3×10^{-2} d kg^{-1}) in Technical Reports Series No. 364 was again taken from Ng et al. [52] who based it upon stable element data of Taucin and Svilane [110] which we have not used.

Goat meat

In the case of goat meat, F_f values in Table 6.12 are all similar to those in Technical Reports Series No. 364 [4]. In part, this is because both reviews base a number of values on the work of Johnson et al. [89]. However, the values for Sr and Cs are now based upon larger datasets.

Pork

The revised value for I in Table 13 is an order of magnitude higher than that in Technical Reports Series No. 364 whereas that for Sr is an order of magnitude lower than that in Technical Reports Series No. 364.

The value of F_f for Ru in Technical Reports Series No. 364 [4] was 6.6×10^{-1} d kg^{-1}, more than two orders of magnitude higher than the value based on two Russian language papers presented here in Table 13. The data in Table 13 are for young animals and would therefore be expected to be higher than for adults. The Technical Reports Series No. 364 [4] value was cited as coming from Ng et al. [51], however, this reference does not give a value for pork. Ng [49] gives a value of 6.8×10^{-3} d kg^{-1} (method of estimation unclear) which is similar to the values in Table 13. Furthermore, the value presented here for pork appears reasonable compared with those for other meats.

As for cow milk and poultry meat (see below) the value for U in the database appears rather high. However, the source data for the pork value from Prister [22] has been rechecked and appears valid; the Prister reference was also used to derive the Technical Reports Series No. 364 value.

Considerably more radionuclides have recommended F_f's in Technical Reports Series No. 364 [4] for pork than we present in Table 13. The values for Am and Nb in Technical Reports Series No. 364 [4] were both model derived and not based upon data. The values for Ag and Ce in Technical Reports Series No. 364 [4] were derived from data for sheep and cattle. The value for Pu in Technical Reports Series No. 364 [4] for pork is referenced as Ng et al. [52], however, we cannot find these data in the cited report. The recommended F_f for Co to pork in Technical Reports Series No. 364 [.4] was based upon a minimum detectable activity (i.e. a 'less than' value) from Voigt et al. [6.90] and has not been considered here. The F_f value for Fe to pork was taken from Ng et al. [6.52] based upon stable data reported in Taucin and Svilane [6.110] which we are not using. The value for Cd given in Technical Reports Series No. 364 [6.4] was again taken from Ng et al. [6.52] based upon data from Cousins et al. [6.120] but this study involved elevated dietary Cd concentration and hence has not been included in this review.

Poultry meat

There are no order of magnitude differences between the Technical Reports Series No. 364 values and those in Table 14.

The F_f value of 10 d kg^{-1} for Cs in Technical Reports Series No. 364 [4] is c. three fold higher than that derived from the database. The Technical Reports Series No. 364 [4] value refers to Coughtrey [41], but, in this reference, Coughtrey derives a value of 12 d kg^{-1} for 100 days of contamination, and the experimentally derived values (n=8) which he lists give values which range from 1.4-9.5. These higher values are more consistent with our database value of 2.7 which is now based on a comparatively large number of data (n=13).

The F_f value of 1.0 d kg^{-1} for Mo is based on Ennis et al. [121] and Johnson and Ward [122]. In the latter reference, the weight basis is not clear and the value is an order of magnitude higher than the other data source, so we have not used this value in the database, thereby deriving a value five fold lower.

The value for U in the database is rather high. The source data for this value [25] have been rechecked and appear valid; the same data were used for the recommended value in Technical Reports Series No. 364 [4].

Values provided for La, Pr, Nd, Pm and Y in Technical Reports Series No. 364 [4] were based on collateral data from Ng et al. [52] on the basis of data from a single dosing experiment by Mraz et al. [124] where the F_f values for these elements were derived based on normalising to that of Sr. The value of 8 d kg^{-1} for Ru in Technical Reports Series No. 364 [4] is much higher than that justified from the quoted reference of Coughtrey [41]. The only experimentally derived value of 7.0×10^{-3} d kg^{-1} listed by Coughtrey is, again, based on the single dosing experiment used by Ng et al. [52]. Thus all these values have a high uncertainty and have not been used in the database.

Values for Ag in Technical Reports Series No. 364 [4] were derived from model evaluations using retention equations. There are no experimentally derived values quoted for comparison with this data and the values are not used in the database. The value for Ce in Technical Reports Series No. 364 [4] can be traced back to referring to CEC [119]. However, there are no directly relevant data in this reference and the value has not been used in the database. The Fe value in Technical Reports Series No. 364 [4] is derived from Taucin and Svilane [110] which has not been used in the database. Technical Reports Series No. 364 [4] recommends an F_f value for Pu based on predominantly model predictions from CEC 1987 [119]. Whilst Ng [52] presents a value derived from the experimental data of Mullen et al. [123] this was calculated using meat activity concentrations measured 12 d after the cessation of daily administration of Pu and hence has not been used here.

6.3.2. Concentration ratio values for meat

As for milk in Table 9, we give values for CR for meat for a number of species in Table 15. The data are only extensive for cow meat, and therefore comparisons across species for the elements are fewer but still encouraging.

In addition to the data given, values for horse meat of 3.8×10^{-1} for Fe and 5.3×10^{-1} for Zn can be derived using stable data from the agricultural sources described above. These are in reasonable agreement with values for other species in Table 15.

6.4. Transfer to eggs

The F_f data for eggs have been compiled as described above for meat, including the use of data cited in Ng et al. [52]; it has been possible to verify much of the Russian language data cited by Ng et al. The compilation is largely for hens although there are a few data for duck. Data are reported for a variety of whole egg (including shell), egg contents and occasionally yolk only. By preference, egg contents data have been used although whole-egg (i.e. including the shell) and in a few cases yolk values have been considered if data are sparse.

Whole-egg data were excluded from the review of data for Sr, Ca and Ba because of the high content of these elements in shell. If F_f values were reported for yolk and egg-white separately the average has been used (on the basis of studies for which it is possible to compare to reported data on whole-egg contents, this approach gives reasonable estimates). Some publications report the fraction of daily intake per egg, these have been converted to F_f values using quoted egg weights; if egg weight was not available a value of 64 g has been assumed.

The F_f values for egg (data combined for all species) are presented in Table 16. Data for most radionuclides is relatively sparse (no more than seven values for any radionuclide). The values in Table 16 are generally similar to those proposed in Technical Reports Series No. 364 [4]. This is not surprising since many of the Technical Reports Series No. 364 values were based upon Ng et al. [52] which is the major source of data for this review. The value for U in the database is rather high. However, the source data for this value Prister [22] has been rechecked and appears valid.

TABLE 15. CONCENTRATION RATIOS FOR THE MEAT OF DIFFERENT ANIMALS

Element	Beef					Sheep					Pork					Generic
	CR	SD	Min	Max	N	CR	SD	Min	Max	N	CR	SD	Min	Max	N	
Ag						4.3×10^{-4}				1						4.3×10^{-4}
Am						1.1×10^{-4}				1						1.1×10^{-4}
Ca	2.3×10^{-2}		2.1×10^{-2}	2.6×10^{-2}	2	1.4×10^{-2}										1.9×10^{-2}
Cd	1.7×10^{-1}	1.5×10^{-1}	2.3×10^{-3}	3.5×10^{-1}	7	1.2×10^{-2}		1.3×10^{-3}	2.3×10^{-2}	2	1.3×10^{-1}				1	9.2×10^{-2}
Ce						2.2×10^{-4}				1						2.2×10^{-4}
Cl	2.4×10^{-1}		4.8×10^{-2}	4.3×10^{-1}	2											2.4×10^{-1}
Co	3.9×10^{-1}		7.2×10^{-3}	7.8×10^{-1}	2	2.3×10^{-1}										3.1×10^{-1}
Cs	2.3×10^{-1}	1.7×10^{-1}	2.2×10^{-2}	7.3×10^{-1}	17	6.4×10^{-1}	1.0	5.3×10^{-2}	7.5	51	9.2×10^{-2}	1.0×10^{-1}	8.3×10^{-3}	2.4×10^{-1}	4	3.9×10^{-1}*
Fe	2.2×10^{-1}	2.5×10^{-1}	6.0×10^{-2}	7.2×10^{-1}	6	2.7×10^{-1}				1						2.5×10^{-1}
I	9.5×10^{-2}	8.2×10^{-2}	3.2×10^{-2}	1.9×10^{-1}	3					1	9.3×10^{-2}		3.5×10^{-2}	1.5×10^{-1}	2	9.4×10^{-2}
La	1.6×10^{-3}	2.4×10^{-4}	1.3×10^{-3}	1.8×10^{-3}	3											1.6×10^{-3}
Mg	1.4×10^{-1}		9.4×10^{-2}	1.9×10^{-1}	2											1.4×10^{-1}
Mn	8.0×10^{-3}		4.6×10^{-3}	1.1×10^{-2}	2											8.0×10^{-3}
Mo	9.6×10^{-2}		2.5×10^{-2}	1.7×10^{-1}	2											9.6×10^{-2}
Na	9.7×10^{-1}				1											9.7×10^{-1}
Nb	6.5×10^{-6}				1											6.5×10^{-6}
Ni	8.0×10^{-2}				1											8.0×10^{-2}
P	1.3															1.3
Pb	7.7×10^{-2}	1.8×10^{-1}	1.0×10^{-3}	6.2×10^{-1}	11	1.2×10^{-2}	4.0×10^{-3}	9.2×10^{-3}	1.6×10^{-2}	3	6.6×10^{-1}		2.3×10^{-1}	1.1	2	2.5×10^{-1}
Po	1.4×10^{-1}	1.3×10^{-1}	3.7×10^{-2}	4.1×10^{-1}	7											1.4×10^{-1}
Pu						3.9×10^{-5}	2.4×10^{-5}	1.5×10^{-5}	6.3×10^{-5}	3						3.9×10^{-5}
Ra	1.8×10^{-1}	3.8×10^{-1}	1.3×10^{-3}	1.3	11											1.8×10^{-1}
Rb	3.0×10^{-1}				1											3.0×10^{-1}
Ru						5.7×10^{-4}				1						5.7×10^{-4}
S						5.0×10^{-1}										5.0×10^{-1}
Sb	2.7×10^{-1}				1											2.7×10^{-1}
Se											1.1				1	1.1
Te	1.8×10^{-1}				1											1.8×10^{-1}
Th	6.2×10^{-3}	5.0×10^{-3}	1.7×10^{-3}	1.2×10^{-2}	3											6.2×10^{-3}
U	3.3×10^{-1}	6.1×10^{-1}	3.0×10^{-3}	1.7	8											3.3×10^{-1}
Zn	1.7	1.1	4.7×10^{-1}	3.2	9	2.1		1.3	2.9	2						1.9

* Goat value of 6.2×10^{-1} for Cs (n=4) is included in the generic value; Shaded cells denote CR values estimated from the stable element review.

TABLE 16 TRANSFER COEFFICIENTS FOR RADIONUCLIDE TRANSFER TO EGG CONTENTS d kg^{-1}

Element	N	GM	GSD	AM	SD	Min	Max	Source
Am	1			3.0×10^{-3}				Database
Ba	1			8.7×10^{-1}				Database
Ca	1			4.4×10^{-1}				Database–stable data only
Ce	1			3.1×10^{-3}				Database
Co	2			3.3×10^{-2}		2.6×10^{-2}	4.0×10^{-2}	Database (duck)
Cs	11	4.0×10^{-1}	1.5	4.3×10^{-1}	1.6×10^{-1}	1.6×10^{-1}	7.1×10^{-1}	Database (includes duck)
Fe	2			1.8		8.5×10^{-1}	2.8	Database–includes stable data
I	4	2.4	1.3	2.4	5.7×10^{-1}	1.9	3.2	Database
Mn	3	4.2×10^{-2}	1.4	4.4×10^{-2}	1.6×10^{-2}	3.2×10^{-2}	6.2×10^{-2}	Database–stable data only
Mo	3	6.4×10^{-1}	1.3	6.5×10^{-1}	1.9×10^{-1}	5.2×10^{-1}	8.7×10^{-1}	Database–includes stable data
Na	2			4.0		1.9	6.0	Database–includes stable data (includes duck)
Nb	1			1.0×10^{-3}				Database
P	1			6.4×10^{-1}				Database
Po	1			3.1				Database
Pu	2			1.2×10^{-3}		9.9×10^{-6}	2.3×10^{-3}	Database
Ru	1			4.0×10^{-3}				Database
Se	4	1.6×10^{1}	1.9	1.8×10^{1}	1.0×10^{1}	8.8	2.8×10^{1}	Database–stable data only
Sr	9	3.5×10^{-1}	1.4	3.7×10^{-1}	$1.4 \text{ x } 10^{-1}$	2.2×10^{-1}	6.4×10^{-1}	Database–includes stable data (includes duck)
Te	1			5.1				Database
U	2			1.1		9.2×10^{-1}	1.2	Database
Zn	4	1.4	1.2	1.4	2.9×10^{-1}	1.2	1.9	Database–includes stable data
Zr	1			2.0×10^{-4}				Database

TABLE 17. SUMMARY OF AVAILABLE TRANSFER COEFFICIENT DATA–SHADING DENOTES NO AVAILABLE DATA

Element	Beef	Sheep meat	Goat meat	Pork	Poultry	Egg	Cow milk	Goat milk	Sheep milk
Ag		1							
Am	1	1				1	1	2	
Ba	2		1		2	1	15	3	1
Be							1		
Ca	3				2	1	15	12	St
Cd	8	1			2		8	1	1
Ce		1				1	6	1	
Cl	1								
Co	4	2			2	2	4	1	2
Cr							3	2	1
Cs	58	41	11	22	13	11	288	28	28
Fe	4			1		2	7	St	St
I	5	1		2	3	4	104	24	7
La	3								
Mn	2	1		1	2	3	4	St	1
Mo	1				1	3	7	4	
Na	2	1			1	2	7	St	1
Nb	1		1		1	1	1	1	
Ni							2	2	1
Np								1	
P	1			1		1	St	St	St
Pb	5	2					15		St
Po					1	1	4	2	
Pu	5	2				2	n/a		1
Ra	1						11		
Ru	3	2		1		1	6		
S		3					1	12	St
Sb	2						3		
Se				1	4	4	12	2	
Sr	35	25	8	12	7	9	154	21	4
Te	1		1		1	1	11	1	1
Th	6						3		
U	3			2	2	2	3	1	
W							7		
Y			1					1	
Zn	6	6		2	3	4	8	St	St
Zr	1		1		1	1	6	1	

St – recommended value based on unassociated stable element data; n/a whilst limited F_m values available for Pu transfer to cow milk recommended value is derived from review. NOTE: ^{14}C and ^{3}H are not considered above as specific activity models have been used (see paper by Davis et al. in this document).

7. CONCLUSIONS

Table 17 summarises the availability of data on the transfer coefficient of radionuclides to farm animal products. For approximately 50 % of the required radionuclide-animal product combinations no transfer coefficient data were available. Of those combinations for which values could be recommended less than 14 % were based upon more than five studies.

We have proposed that the *CR* would be a more robust and generic parameter than the transfer coefficient. For most radionuclides, the concentration ratio data compiled varies little between the species considered (sheep, goats, cattle, horses and poultry). Therefore, concentration ratios derived for one species could be applied to another. However, unfortunately many authors who report transfer coefficients do not provide the information required to estimate concentration ratios. Further data collation to derive *CR* values for meat from agricultural animals would be valuable to supplement the data in the Tables.

Currently, there is no equivalent critically evaluated international compilation of other parameter values which are also important for modelling radionuclide transfer to animals, notably on biological half-lives and transfer to tissue other than muscle.

A number of radionuclides have recently been identified for which there is currently inadequate information for modelling or assessment including ^{36}Cl, ^{237}Np, ^{99}Tc, U-isotopes, Th-isotopes, ^{241}Am, ^{59}Ni, ^{94}Nb and ^{60}Co. Some of these radionuclides have not been considered (e.g. ^{252}Cf, ^{192}Ir) within the revised IAEA handbook. .

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AGRICULTURAL ECOSYSTEMS: TRANSFER TO FRUITS

TRANSFER TO FRUITS

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Abstract

The paper describes the key transfer processes, concepts and conceptual models regarded as important for dose assessment, as well as relevant parameters for modelling radionuclide transfer in fruits. Information relate to fruit plants grown in agricultural ecosystems of temperate regions. The relative significance of each pathway after release of radionuclides depends upon the radionuclide, the kind of crop, the stage of plant development and the season at time of deposition. Fruit intended as a component of the human diet is borne by plants that are heterogeneous in habits, and morphological and physiological traits. Information on radionuclides in fruit systems has therefore been rationalised by characterising plants in three groups: woody trees, shrubs, herbaceous plants. Parameter values have been collected from open literature, conference proceedings, institutional reports, books and international databases. Given the scarce information available on the subject, the choice has been made to not reject any information. Data on root uptake are reported as transfer factor values related to fresh weight, being consumption data for fruits usually given in fresh weight. Recommendations are given for research and modelling.

1. INTRODUCTION

Botanically, fruit is the structure of angiosperms that develops from the ovary wall after fecundation as the enclosed seed or seeds mature. In this context, the term ‘‘fruit’’ refers not to a well defined botanical plant part, called pericarp, but in the horticultural sense, to a component of the human diet generally consumed as a dessert item [1].

Plants bearing fruits occur in a vast range of habits (size and shape), and morphological and physiological traits, and can be woody trees, bushes, or herbaceous plants, evergreen or deciduous, perennial or annual. Herbaceous annuals usually blossom once and then die, having set fruit and seed. Some perennials, like the monocarp banana and pineapple, also flower once and then die. However, woody fruit crops are polycarps, they bloom year after year. Some trees are deciduous; e.g. apples shed their leaves in winter. Others, like citrus, mango, avocado, are evergreen and retain their leaves for two or more years.

Fruit contamination following a release to atmosphere can be the result of various processes: (i) direct deposition to exposed fruit surfaces, absorption by the fruit skin and transport to the interior; (ii) deposition to exposed plant surfaces (directly from the atmosphere or as a result of resuspension), absorption to interior and translocation to fruit; (iii) for perennial plants: remobilisation of radionuclides from the leaves to the over-wintering organs prior to leaf drop, followed by retranslocation from storage organs to other plant components at the resumption of growth; (iv) deposition to soil, vertical migration in the soil profile, root uptake and transfer to the fruit. The main processes involved in the transfer of radionuclides to fruits are shown in Figure 1 [2].

The relative significance of each pathway after release of radionuclides depends upon the radionuclide, the kind of crop, the stage of plant development and the season at time of deposition. Furthermore, in agricultural ecosystems, the transfer of radionuclides to fruit is generally affected by human intervention, which includes horticultural practices, intended to modify physiology and translocation of nutrients, to achieve early cropping, high, early and sustained yield, and high fruit quality. The published literature that describes the phenological development of above- and below-ground organs of temperate tree fruit, and hence the potential for uptake of radionuclides directly from the atmosphere and via the soil, has been reviewed by Atkinson and Webster [3].

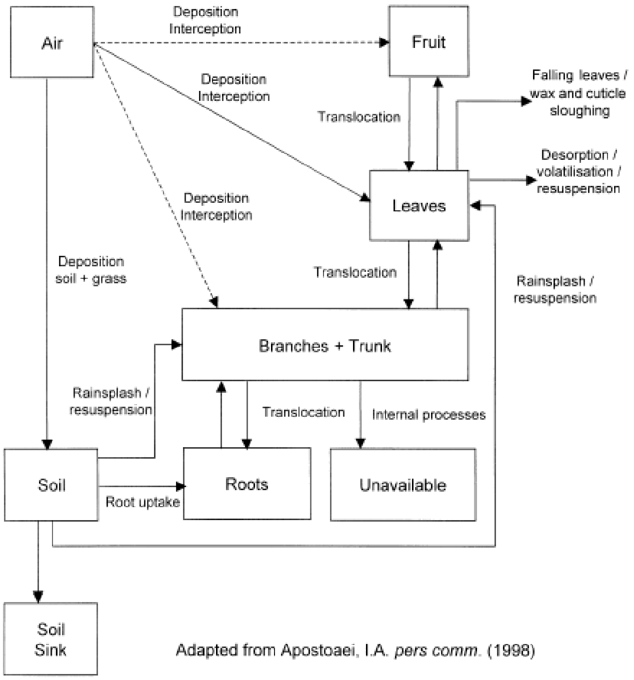

FIG. 1. Processes involved in the transfer of radionuclides to fruit.

2. FOLIAR UPTAKE

Among food-chain transfer processes, direct deposition on to fruit is recognised as important, particularly if this occurs close to ripening. Notwithstanding this, there is a paucity of data on fruit direct contamination and on skin adsorption, limited mainly to radioisotopes of Cs and Sr and to vine and apple systems [4, 5]. Direct deposition depends on the extent of protection exerted by leaves. The phenological stage of the plant at the time of deposition, as well as the anatomical features particular to species and cultivar and to human manipulation (shoot pruning), are important in determining the degree of direct contamination of the fruit.

The area of interception by fruits is usually less than that of the leaves during the growing season, but it may increase relatively at end of ripening, when the canopy biomass is reduced in size. Direct contamination also becomes the main process responsible for radionuclide uptake when the process of leaf to fruit translocation does not take place or is limited. Retention of intercepted activity by fruit is dependent in turn upon the fruit surface properties. Absorption by fruit skin occurs under favourable humidity conditions and depends on differences in wettability and roughness of the fruit surface. Under dry conditions, the fruit skin acts as a barrier to radionuclides. Even where there is no transfer of radionuclides from skin to pulp or juice, surface contamination can enter human food chains through transfer to internal tissues during peeling or in reconstituted juices and flavourings made from skins [6]. Also, of course, some fruits are eaten with their skins.

2.1. Deposition to the above-ground parts of the plant

Data on deposition to fruit plants are limited, particularly transfer rate constants for the movement of contaminants from particles to plants, as reported in two reviews on the current state of knowledge concerning gaseous [7] and non-gaseous (wet and dry deposited) radionuclides [8].

Perennial species in orchards do not form a continuous canopy, but consist either of round-headed trees with individual access to each from all sides or as vertical hedgerows with trees contiguous in rows separated by alleyways. For the same species, the plantation density can vary widely. Fruit trees in hedgerows can be trained to different shapes, with different spatial arrangements of leaves, so that trees with the same Leaf Area Index (*LAI*), but different growth shapes have different specific interception capacities [9]. Interception by fruit trees with discontinuous canopies has to take into account the leaf area distribution in both the horizontal and vertical dimensions over the entire orchard surface.

Studies comparing interception and deposition of ozone on peach orchards and soybean crops indicate that fruit trees of this type constitute a three-fold greater sink for gaseous pollutants than herbaceous crops, as a consequence of the structural and architectural characteristics and the biological activity of the vegetative surfaces of the two crops [10]. Other studies demonstrate that the deposition velocity calculated per unit plant mass (V_g^w : cm^3 g^{-1} s^{-1}) exhibits significant differences between different fruit plants (apple, strawberry, blackcurrant and raspberry) for deposition of gaseous $^{14}CO_2$ and $CO^{35}S$ [11]. However, the deposition velocities calculated per unit area of the plant (V_g^a : cm s^{-1}) were of the same magnitude as those observed for other crops. The authors suggested that a single deposition velocity for all crops should be used in assessment studies and that this should be the highest value, unless a probabilistic approach was being pursued. Other authors have proposed the use of canopy fractal dimension, the parameter which describes tree-crown structure including information on spatial distribution of the leaf area [12], instead of the LAI, to seek correlations with interception of airborne contaminants [8].

For non-gaseous airborne contaminants, experimental studies show that the water-storage capacity of plants is the major contributor to a plant's total wet deposited contaminant loading, following a single rainfall event [13]. The factors affecting the number of leaves and the total leaf area per tree, at least for top fruit -such as apple (*Malus*)- and temperate climates, are the scion species/cultivar, the rootstock, the age of the tree, environmental conditions, the level of cropping, and management practices such as pruning, irrigation and nutrition [14, 15 cited by Ref. [3].

Evergreen species, such as citrus and olives, are constantly exposed to the risk of contamination, whereas deciduous species have a reduced epigeous interception capacity during winter. For deciduous species, the phenological stage at time of fallout is the main parameter determining canopy interception. Growth curves, in particular times of anthesis and leafing, are therefore significant for modelling radionuclide intake from different fruit species. Values showing the time dependence of *LAI* are available for some fruit types [14-16].

2.2. Absorption

Absorption of soluble radionuclides in fruit plants can occur into leaves, blossom and branches. Blossom should behave like leaves, although experimental evidence for this suggestion is limited to cereals [17]. The bark of the trunk and branches probably has low permeability to radionuclides, although experiments on young apple trees have shown penetration under wet conditions [18].

Swietlik and Faust [19] reported models on diffusion through isolated cuticles. According to the authors, the most plausible model, developed by Schönherr and Huber [20], on the basis of citrus, pear and apricot studies, assumed that above pH 3 cuticles are more permeable to cations, whereas below pH 3 they are more permeable to anions. The rate of penetration of cations through the cuticle is inversely related to the radius of the hydrated ion and occurs in the following order: $Cs^+ > Rb^+ > K^+ > Na^+ > Li^+ > Mg^{2+} > Sr^{2+} > Ca^{2+}$ [21-23 cited by Ref. 19].

Absorption is rarely represented in literature as a net flux of contaminants from external to internal leaf, but rather as data on the releasable fraction upon rinsing. Data are often reported as translocation, which includes absorption and loss processes. In a few cases, it has been possible to derive absorption from the published data as the sum of 'not removable from leaves + translocated to other plant components + already absorbed, but lost through weathering and spontaneous shedding of superficial particles' [4]. Data are reported in Table 1, expressed as percentage of the applied activity, and relate to foliar application of ^{134}Cs and ^{85}Sr in soluble form on apples, strawberries and grapevines.

The rate of foliar absorption depends on radionuclide, plant species and even cultivar. Data reported in Table 1 indicate that adsorption is larger for ^{134}Cs than for ^{85}Sr and that it follows the order: apple ≥ strawberry > grapevine. In general, comparative data on the absorption rates of different species are lacking. Results collected on absorption in various fruit trees following foliar application of fertilizers show that *Prunus* leaves are less efficient in foliar absorption than are apples or citrus leaves [19].

Available data indicate that foliar absorption is time dependent: the longer a contaminant is retained on a leaf surface the more likely it is to be absorbed. Therefore, weathering processes, in addition to spontaneous shedding of cuticular wax, will affect absorption due to reduction in the pool of external radionuclide available for absorption. Translocation to other plant organs removes absorbed activity from the leaf and may be of significance for a high rate of absorption. Temperature and humidity, as well as foliar irrigation, influence absorption. Foliar irrigation by sprinkling over or under the canopy is a common practice in orchards: in winter-spring to increase the freeze resistance of fruit trees and protect the buds from the frost, in summer to reduce the temperature of the canopy.

2.3 Translocation to fruit

The above-ground part to fruit translocation factor is an aggregated parameter that describes a net flux from leaf to fruit as a result of all the processes of deposition, interception, retention, absorption and translocation, measured at a single point in time.

TABLE 1. ACTIVITY ABSORBED BY APPLES, STRAWBERRIES AND GRAPEVINES AT VARIOUS TIMES AFTER FOLIAR APPLICATION OF ^{134}Cs AND ^{85}Sr

Fruit	Element	Number of entries, N	Year	Time after contamination, d	Absorbed activity (% of the applied)	Reference
Apple	Cs	1		1	60	[24]
		1		41	76	[24]
		1		84	65	[24]
Grapevine	Cs	2	1991	30	29	[25]
		2		45	31	[25]
		2		60	47	[25]
		2		75	34	[25]
		2		90	29	[25]
		2		120	40	[25]
		2		150	36	[25]
	Cs	2	1992	30	34	[25]
		2		45	38	[25]
		2		60	40	[25]
		2		75	39	[25]
		2		90	45	[25]
		2		105	48	[25]
		2		135	60	[25]
Strawberry	Cs	3		1	34	[26]
		3		7	58	[26]
		3		21	48	[26]
		3		35	71	[26]
		3		49	77	[26]
		3		56	67	[26]
Apple	Sr	1		1	51	[24]
		1		41	60	[24]
		1		84	52	[24]
Grapevine	Sr	2	1990	30	3	[25]
		2		60	3	[25]
		2		90	7	[25]
		2		120	8	[25]
Strawberry	Sr	3		1	18	[26]
		3		7	26	[26]
		3		21	22	[26]
		3		35	27	[26]
		3		49	36	[26]
		3		56	37	[26]

2.3.1 Data from experiments

In the literature, there is no uniformity in the mathematical description of the process of above-ground part to fruit translocation, due to the lack of a generally adopted experimental protocol: data have been produced in a variety of experimental conditions, fruit plants have been contaminated at various phenological stages, and different methodologies of radionuclide application have been used. Therefore, published data are not always comparable.

Much of the information on non-gaseous radionuclides concerns caesium and strontium. Experiments have generally been carried out simulating wet deposition, except for the data of Madoz-Escande et al. [27], where dry deposition of ^{137}Cs and ^{90}Sr has been studied. The activity of fruits for non-gaseous radionuclides has been calculated from fractions of recovered activity and expressed as fraction of the applied or intercepted activity [4, 5]. Data are summarised in Tables 2 for Cs, 3 for Sr and 4 for Ca.

Information on gaseous radionuclides relates to $^{14}CO_2$, $CO^{35}S$ and HTO (tritiated water). Data produced by Ref. [11] are expressed as % of the activity in the whole plant at harvest and are reported in Table 5.

Data reported in Tables 2, 3 and 4 are not directly comparable with data in Table 5. However some general conclusions can be drawn.

Translocation, like absorption, continues with time and depends on the chemical element. In general, radiocaesium is readily translocated to fruits, whereas absorbed radiostrontium and calcium, on the basis of the few available data, remain at the site of application or are translocated to an extent that is one order of magnitude less than radiocaesium. For gaseous radionuclides, when the contamination process occurs at flowering, ^{35}S and ^{3}H are subsequently translocated to fruits, 27 and 32% respectively, whereas very little ^{14}C, 7%, is remobilised from leaves to fruits late in the growing season.

There are large differences in the degree of translocation of radionuclides to fruits of different species, depending upon physiological characteristics of the plants. The highest experimental values for radiocaesium are observed in apple, strawberry and pear, and the lowest in orange. Among the shrubs, gooseberry shows the highest translocation and blueberry the lowest [31]. Differences in leaf-to-fruit translocation depend not only on the scion species, but also on the cultivar, as discussed by Ref. [3] and demonstrated for different varieties of apples by Ref. [18].

Translocation of radionuclides from leaf to fruit (or to other parts of the plant) depends on the phenological stage of the plant at the time of deposition. The highest values in fruit at harvest occur following deposition between fruit development and beginning of ripening. This interval is regarded by horticulturists as the stage of the higher demand of fruits for photosynthetic products from leaves, and that could explain the higher transport of C and of Cs, analogous to potassium, from leaves to fruits [11, 27, 36, 37].

TABLE 2. FRUIT ACTIVITY OF Cs RESULTING FROM TRANSLOCATION AFTER SUPERFICIAL CONTAMINATION

Fruit species (Variety)	Experimental conditions	Phenological stage at time of contamination	Time between contamination and harvest, d	Fruit activity (% of applied/intercepted)	Ref.
Apple *(Golden)*	greenhouse	-	-	17 (of recovered)	[28]
Apple *(Golden)*	greenhouse	-	-	0.15 L/kg w.w.	[29]
Apple *(Gloster)*	leaf droplet, precipitation-free	beginning of fruit development	70 (fruit maturity)	39	[18]
Apple *(Jonagold)*				42	[18]
Apple *(Gloster)*	foliar shoot dipping, precipitation-free			19	[18]
Apple *(Jonagold)*				29	[18]
Apple *(Jonagold)*	bark, precipitation-free	beginning of fruit development	49	0.2¯1.8 (of the amount absorbed by the bark)	[18]
Apple *(Aroma)*	precipitation-free	mature fruit, mass 73 g	1	1	[24]
Apple *(Aroma)*		fruit mass 46.5 g	41	16	[24]
Apple *(Aroma)*		green fruit, mass 8.4 g	84	29	[24]
Apple *(Golden Delicious)*	open field	green fruit	50	47	[30]
Blueberry	greenhouse	-	-	0.7	[31]
Gooseberry	greenhouse	-	-	3.4	[31]
Grapevine *(Riesling)*	greenhouse	-	150	8	[25]
		-	365	1.5	[25]
Grapevine *(Pinot Blanc)*	open field	beginning of ripening	30	1.8	[32]
Grapevine *(Pinot Blanc)*	open field	beginning of ripening	365	0.2	[32]
Grapevine *(Pinot Blanc)*	open field	beginning of ripening	35	3.7	[33]
Grapevine *(Pinot Blanc)*	open field	beginning of ripening	30	9.6	[34]
Grapevine *(Chardonnay)*	open field	green fruit	50	5.5	[30]
Grapevine	greenhouse	late flowering	30	2.9	[27]
Grapevine	greenhouse	beginning of ripening	2	3.0	[27]
Grapevine	greenhouse	beginning of ripening	7	3.8	[27]
Grapevine	greenhouse	beginning of ripening	20	3.8	[27]
Grapevine	greenhouse	beginning of ripening	30	6.9	[27]
Orange	open field	-	-	0.1	[35]
Pear *(Conference)*	open field	green fruit	50	1.28	[30]
Redcurrant	greenhouse	-	-	2	[31]
Strawberry	greenhouse	-	-	20	[31]
Strawberry	greenhouse	-	21-105	20-36	[26]
Strawberry *(Miss)*	tunnel	anthesis	22 – 48	15	[36]
Strawberry *(Miss)*	tunnel	ripening	1 – 27	6.5	[36]

TABLE 3. FRUIT ACTIVITY OF Sr BY TRANSLOCATION AFTER SUPERFICIAL CONTAMINATION, EXPRESSED AS FRACTION OF THE APPLIED OR INTERCEPTED ACTIVITY

Fruit species (Variety)	Experimental conditions	Phenological stage at time of contamination	Time between contamination and harvest, d	Fruit activity by translocation (% of applied/intercepted)	Ref.
Apple (*Golden*)	greenhouse	-	-	2.2 of recovered activity in plant	[28]
Apple (*Aroma*)	precipitation-free	mature fruit, mass 73 g	1	0	[24]
		fruit mass 46.5 g	41	0	
		green fruit, mass 8.4 g	84	0	
Apple (*Golden Delicious*)	open field	green fruit	50	0.80	[30]
Blueberry	greenhouse	-	-	0.02	[31]
Gooseberry	greenhouse	-	-	2.1	[31]
Grapevine (*Riesling Sylvaner: Müller-Thurgau*)	greenhouse	-	120	<2.5	[25]
Grapevine (*Pinot Blanc*)	open field	beginning of ripening	30	0.73	[34]
Grapevine (*Chardonnay*)	open field	green fruit	50	0.09	[30]
Grapevine	greenhouse	late flowering	30	1.1	[27]
Grapevine	greenhouse	beginning of ripening	2	3.7	[27]
Grapevine	greenhouse	beginning of ripening	7	3.0	[27]
Grapevine	greenhouse	beginning of ripening	20	2.7	[27]
Grapevine	greenhouse	beginning of ripening	30	4.0	[27]
Orange	open field	-	-	0.004	[35]
Pear (*Conference*)	open field	green fruit	50	0.98	[30]
Redcurrant	greenhouse	-	-	0.003	[31]
Strawberry	greenhouse	-	-	0.3	[31]
Strawberry (*Miss*)	tunnel	anthesis	22 – 48	1.6	[36]
		Ripening	1 – 27	2.2	[36]

TABLE 4. FRUIT ACTIVITY OF Ca BY TRANSLOCATION AFTER SUPERFICIAL CONTAMINATION, EXPRESSED AS FRACTION OF THE APPLIED OR INTERCEPTED ACTIVITY [31]

Fruit species (Variety)	Experimental conditions	Phenological stage at time of contamination	Time between contamination and harvest, d	Fruit activity by translocation (% of applied/intercepted)
Blueberry	greenhouse	-	-	0.02
Gooseberry	greenhouse	-	-	2.7
Redcurrant	greenhouse	-	-	0.2

TABLE 5. FRUIT ACTIVITY AFTER SUPERFICIAL CONTAMINATION OF GASEOUS $^{14}CO_2$, $CO^{35}S$ AND HTO, EXPRESSED AS % OF THE ACTIVITY IN THE WHOLE PLANT AT HARVEST TIME. ALL THE DATA ARE FOR APPLE CONTAMINATED IN A WIND TUNNEL [11]

Radionuclide	Phenological stage at time of contamination	Time between contamination and harvest, d	Fruit activity (%)
^{14}C	flowering	127	7
^{14}C	fruitlet formation	84	21
^{14}C	fruit development	32	61
^{14}C	fruit ripening	5	38
^{3}H	flowering	127	32
^{3}H	fruitlet formation	84	18
^{3}H	fruit development	32	43
^{3}H	fruit ripening	5	42
^{35}S	flowering	127	27
^{35}S	fruitlet formation	84	9
^{35}S	fruit development	32	25
^{35}S	fruit ripening	5	6

The fruit activity at harvest after an acute release is also determined by growth: the concentration of a radionuclide in fruit can increase or decrease during the growth period, depending upon the rate of translocation relative to the growth rate. The growth effect has been studied for radioisotopes of Cs, Sr and Ca in redcurrants, gooseberries and blueberries [31], and for Cs in grapes [25] and in apples [24]. Due to the growth effect, a reduced yield due to a period of drought can result in a higher concentrations of radionuclides in fruits and vice versa, as ascertained for Cs and Sr in strawberries [36] and in grapevines [25, 33].

2.3.2 Chernobyl data

When radionuclides are introduced into an agricultural system it is often difficult to evaluate the initial deposited activity. After Chernobyl deposition, some authors assumed the deposition on leaves to be the same as on soil and expressed the 'translocation coefficient' as the ratio between the radionuclide concentration in fruit and the total deposition to the ground (Table 6) [38].

Other authors calculated a fruit: leaf activity ratio (Table 7) [9; 4 calculated from Ref. 39].

The high translocation coefficient for hazelnuts (Table 6) is explained by the large quantity of leaves on hazelnut trees at the time of deposition and the low moisture content of hazelnut kernels. More generally, the contamination of fruits is correlated to the ratio between the leaf area and the corresponding mass of fruit. The higher the ratio, the higher the radionuclide activity concentration is.

Data in Tables 6-7 confirm the dependence of translocation on physiological properties of different plant species as discussed above for Cs (Table 2). The lower translocation for both ^{137}Cs and ^{103}Ru in apple and pear as compared with peach and grapevine can be explained by the more active metabolism of the latter two species [9, 39].

TABLE 6. ESTIMATED ^{137}Cs TRANSLOCATION COEFFICIENTS (DEPOSITION → FRUIT) [38]

Fruit species	^{137}Cs translocation coefficient (m^2 kg^{-1} w.w.)
Hazel-nut	3.2×10^{-2}
Chestnut	2.4×10^{-3}
	$<5 \times 10^{-3}$
	2.5×10^{-3}
Olive	$<3.8 \times 10^{-3}$
Walnut	5.8×10^{-3}
Apple	2.1×10^{-3}

TABLE 7. FRUIT: LEAF RATIO OF ACTIVITY CONCENTRATION ON A DRY WEIGHT BASIS FOR ^{137}Cs AND ^{103}Ru. DATA FROM REF. [39] ARE EXPRESSED AS ARITHMETIC MEANS AND STANDARD ERRORS

Fruit species	Time between contamination and harvest, d	^{137}Cs	^{103}Ru	Reference
Apple	54	4.3×10^{-1}	3×10^{-2}	[9]
Apple	150	$(8.9 \pm 1.3) \times 10^{-2}$	-	[39]
Cherry	45	$(1.8 \pm 0.5) \times 10^{-1}$	-	[39]
Grapevine	54	1.9	5.6×10^{-1}	[9]
Pear	54	1.1×10^{-1}	1×10^{-1}	[9]
Pear	150	$(3.5 \pm 0.3) \times 10^{-2}$	-	[39]
Peach	54	1.3	1.1×10^{-1}	[9]
Peach	54	1.1 ± 0.3	-	[39]

2.4 Remobilisation in perennial plants

Generally speaking, when the scenario is a single deposition event, the residual activity in the plant in the years following deposition is regarded as deriving from soil as the donor compartment, through the processes of soil to plant transfer and/or resuspension and splash. Although this may hold true for annual plants, it is not always so for perennial plants such as fruit trees.

A fraction of the radionuclides intercepted by, and absorbed into, the leaves in the year of deposition is remobilised and translocated to the storage organs, mainly wood and roots, before leaf drop. Remobilisation is highly selective for mineral elements and depends on their concentration in the fully expanded leaves [40], cited by Ref. [41]. At the resumption of growth, a portion of the plant reservoir can be remobilised from the inventory of the plant to fruits, leaves and shoots. This process is assumed to be responsible for residual fruit contamination in perennial fruit trees in the first few years after contamination, a hypothesis supported by data from various authors [9, 42-44].

Therefore, the components involved in the contamination of fruit in the years following a single, acute deposition are both the soil reservoir and the plant reservoir. The relative importance of these reservoirs will change with time. However, their respective contributions have not yet been clearly defined. Nevertheless, it is known that root uptake is negligible in the first few years after contamination, with most of the activity coming from translocation from the sinks of aerial deposition [45]. The process of translocation of Cs from foliage to perennial organs such as wood and bark and subsequent reallocation to the foliage has also been investigated in various studies on evergreen forest trees (*Pinus sylvestris* L.) contaminated by the Chernobyl accident [46].

3. ROOT UPTAKE

2.1 Derived transfer factor values

Data presented in this section (Tables 8-16) relate to fruit plants grown in agricultural ecosystems of temperate regions. Data on fruits that grow in tropical and sub-tropical environments are reported in the paper by Velasco and Juri Ayub presented in this document[11]. Data have been collected from open literature, conference proceedings, institutional reports, books and international databases. Given the scarce information available on the subject, the choice has been made to not reject any information. More details on the information sources are reported in [47].

Data on root uptake are reported as F_v values related to fresh weight, because consumption data for fruits are usually given in fresh weight:

$$F_v = \frac{Bq\ fresh\ fruit\ weight}{Bq\ dry\ soil\ weight}$$

Where F_v values have been expressed in the literature on dry weight basis they have been converted into fresh weight according to the water content reported by the authors. In the absence of this information, a literature value has been used[12] or an average water content of 80% has been assumed, as proposed by Ref. [48].

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[11] See Velasco and Juri Ayub 'Root uptake: tropical and sub-tropical environments' in this publication.

[12] See Fesenko et al. 'Radioecological definitions, soil, plant classifications and reference ecological data for radiological assessments' in this publication.

TABLE 8. SOIL-TO-PLANT (F_v) FOR Cs

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	All	15	1.5×10^{-2}	2.2×10^{-2}	5.8×10^{-3}	1.5×10^{0}	8.6×10^{-4}	8.0×10^{-2}	[11]
		Clay	2	1.1×10^{-3}	3.7×10^{-4}	1.1×10^{-3}	3.3×10^{-1}	8.8×10^{-4}	1.4×10^{-3}	[1]
		Loam	5	4.5×10^{-3}	3.2×10^{-3}	3.5×10^{-3}	8.8×10^{-1}	9.4×10^{-4}	9.2×10^{-3}	[3]
		Sand	4	3.2×10^{-2}	3.5×10^{-2}	1.5×10^{-2}	1.6×10^{0}	1.9×10^{-3}	8.0×10^{-2}	[3]
		Organic	1	3.7×10^{-2}						[1]
		Others	3	1.1×10^{-2}	9.3×10^{-3}	6.0×10^{-3}	1.7×10^{0}	8.6×10^{-4}	1.9×10^{-2}	[3]
	Fruit skin	C,S,O*	3	8.9×10^{-3}	1.4×10^{-2}	2.7×10^{-3}	1.9×10^{0}	7.8×10^{-4}	2.5×10^{-2}	[2]
Shrubs	Fruits	All	6	2.6×10^{-3}	1.9×10^{-3}	2.1×10^{-3}	8.1×10^{-1}	6.9×10^{-4}	5.7×10^{-3}	[3]
		Clay	2	2.2×10^{-3}	1.7×10^{-3}	1.8×10^{-3}	8.7×10^{-1}	9.8×10^{-4}	3.3×10^{-3}	[1]
		Loam	2	3.8×10^{-3}	2.8×10^{-3}	3.2×10^{-3}	8.2×10^{-1}	1.8×10^{-3}	5.7×10^{-3}	[1]
		Others	2	2.0×10^{-3}	1.9×10^{-3}	1.5×10^{-3}	1.1×10^{0}	6.9×10^{-4}	3.3×10^{-3}	[1]
Herbaceous plants	Fruits	All	8	2.9×10^{-3}	3.3×10^{-3}	1.5×10^{-3}	1.2×10^{0}	4.1×10^{-4}	8.9×10^{-3}	[6]
		Loam	1	9.0×10^{-4}						[1]
		Sand	1	4.2×10^{-3}						[1]
		Organic	1	6.4×10^{-3}						[1]
		Others	5	2.3×10^{-3}	3.7×10^{-3}	1.0×10^{-3}	1.3×10^{0}	4.1×10^{-4}	8.9×10^{-3}	[3]

* Fruit skin: the soils are Clay, Sand, Others.

TABLE 9. SOIL-TO-PLANT F_v FOR Sr

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	All	18	2.5×10^{-2}	1.9×10^{-2}	1.7×10^{-2}	9.7×10^{-1}	1.2×10^{-3}	7.0×10^{-2}	[10]
		Loam	4	4.7×10^{-2}	2.7×10^{-2}	3.9×10^{-2}	8.2×10^{-1}	1.2×10^{-2}	7.0×10^{-2}	[3]
		Sand	1	2.5×10^{-2}						[1]
		Organic	1	1.2×10^{-3}						[1]
		Others	12	1.9×10^{-2}	9.8×10^{-3}	1.6×10^{-2}	6.1×10^{-1}	4.3×10^{-3}	3.4×10^{-2}	[5]
Shrubs	Fruits	All	9	5.5×10^{-2}	3.7×10^{-2}	4.4×10^{-2}	7.6×10^{-1}	1.4×10^{-2}	1.1×10^{-1}	[4]
		Clay	2	5.4×10^{-2}	3.9×10^{-2}	4.6×10^{-2}	8.0×10^{-1}	2.6×10^{-2}	8.1×10^{-2}	[1]
		Loam	2	3.6×10^{-2}	2.7×10^{-2}	3.1×10^{-2}	8.3×10^{-1}	1.7×10^{-2}	5.5×10^{-2}	[1]
		Others	5	6.4×10^{-2}	4.4×10^{-2}	5.0×10^{-2}	8.5×10^{-1}	1.4×10^{-2}	1.1×10^{-1}	[2]
Herbaceous plants	Fruits	All	8	5.5×10^{-2}	6.9×10^{-2}	3.3×10^{-2}	1.0	1.2×10^{-2}	2.1×10^{-1}	[6]
		Loam	1	1.0×10^{-1}						[1]
		Sand	1	2.1×10^{-1}						[1]
		Organic	1	1.2×10^{-2}						[1]
		Others	5	2.3×10^{-2}	9.7×10^{-3}	2.2×10^{-2}	3.7×10^{-1}	1.5×10^{-2}	4.0×10^{-2}	[3]

TABLE 10. SOIL-TO-PLANT F_v FOR Pu

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	All	10	2.6×10^{-3}	6.6×10^{-3}	1.4×10^{-4}	2.9×10^{0}	1.3×10^{-6}	2.1×10^{-2}	[7]
		Loam	1	8.0×10^{-6}						[1]
		Sand	1	2.0×10^{-5}						[1]
		Organic	1	1.0×10^{-6}						[1]
		Others	7	3.7×10^{-3}	7.7×10^{-3}	5.5×10^{-4}	2.2×10^{0}	2.8×10^{-5}	2.1×10^{-2}	[4]
Shrubs	Fruits	Others	2	1.7×10^{-4}	1.5×10^{-4}	1.3×10^{-4}	1.0×10^{0}	6.4×10^{-5}	2.7×10^{-4}	[1]
Herbaceous plants	Fruits	All	9	2.4×10^{-4}	3.1×10^{-4}	1.2×10^{-4}	1.2×10^{0}	2.7×10^{-5}	8.3×10^{-4}	[7]
		Loam	1	8.8×10^{-5}						[1]
		Sand	1	1.6×10^{-4}						[1]
		Organic	1	7.3×10^{-5}						[1]
		Others	6	3.0×10^{-4}	3.8×10^{-4}	1.3×10^{-4}	1.5×10^{0}	2.7×10^{-5}	8.3×10^{-4}	[4]

TABLE 11. SOIL-TO-PLANT F_v FOR Am

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruitss	All	6	1.8×10^{-4}	2.7×10^{-4}	3.1×10^{-5}	2.4×10^{0}	1.3×10^{-6}	6.2×10^{-4}	[5]
		Loam	1	8.0×10^{-6}						[1]
		Sand	1	1.5×10^{-5}						[1]
		Organic	1	1.3×10^{-6}						[1]
		Others	3	3.6×10^{-4}	3.1×10^{-4}	1.8×10^{-4}	1.8×10^{0}	2.2×10^{-5}	6.2×10^{-4}	[2]
Shrubs	Fruits	Others	2	1.5×10^{-4}	1.2×10^{-4}	1.2×10^{-4}	8.9×10^{-1}	6.5×10^{-5}	2.3×10^{-4}	[1]
Herbaceous plants	Fruits	All	8	1.8×10^{-4}	2.3×10^{-4}	1.1×10^{-4}	1.0×10^{0}	4.1×10^{-5}	7.2×10^{-4}	[6]
		Loam	1	7.3×10^{-5}						[1]
		Sand	1	1.7×10^{-4}						[1]
		Organic	1	6.8×10^{-5}						[1]
		Others	5	2.3×10^{-4}	2.9×10^{-4}	1.2×10^{-4}	1.3×10^{0}	4.1×10^{-5}	7.2×10^{-4}	[3]

TABLE 12. SOIL-TO PLANT F_v FOR I

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	Others	5	1.2×10^{-2}	1.2×10^{-2}	6.3×10^{-3}	1.6×10^{0}	4.1×10^{-4}	3.1×10^{-2}	[2]
Herbaceous plants	Fruits	Others	1	1.5×10^{-2}						[1]

TABLE 13. SOIL-TO PLANT F_v FOR Ce

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	Others	2	5.3×10^{-4}	1.3×10^{-4}			4.4×10^{-4}	6.2×10^{-4}	[1]
Herbaceous plants	Fruits	Others	1	3.0×10^{-4}						[1]

TABLE 14. SOIL-TO PLANT F_v FOR Cm

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	Others	2	5.3×10^{-4}	1.3×10^{-4}			4.4×10^{-4}	6.2×10^{-4}	[1]
Herbaceous plants	Fruits	Others	1	3.0×10^{-4}						[1]

TABLE 15. SOIL-TO PLANT F_v FOR Ru

Plant group	Plant compartment	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody trees	Fruits	Others	2	1.3×10^{-3}	3.3×10^{-4}			1.1×10^{-3}	1.6×10^{-3}	[1]
Herbaceous plants	Fruits	Others	1	7.4×10^{-4}						[1]

TABLE 16. SOIL-TO-PLANT F_v FOR Co, Cu, Mn, Na

Plant group	Plant compartment	Element	Soil group	N	AM	SD	GM	GSD	Min	Max	Reference
Woody tree	Fruits	Co	Loam	1	4.8×10^{-3}						[1]
Herbaceous plant	Fruits	Cu	Others	1	6.6×10^{-5}						[1]
Fruits	Fruits	Mn	Others	1	3.9×10^{0}						[1]
Woody tree	Fruits	Na	Loam	1	2.4×10^{-2}						[1]

Plants bearing fruits have been distinguished into three groups, on the basis of their gross morphology and physiology i.e. woody trees, both deciduous and evergreen ones, shrubs and herbaceous plants. Data reported under the heading 'woody trees' include apple, pear, peach, apricot, grapevine, olive and orange. Data reported as 'shrubs' include gooseberry, blackcurrant, red raspberry and redcurrant, whereas those reported as 'herbaceous plants' include strawberry, melon, watermelon and rhubarb.

A qualitative analysis of the data derived (Tables 8-16) shows the following trends. F_v values for caesium cover two orders of magnitude, from 10^{-4} to 10^{-2} (Table 8). F_v values for fruits of woody trees range from 8.6×10^{-4} to 8.0×10^{-2}. A narrower range is obtained for fruits of shrubs, i.e. from 6.9×10^{-4} to 5.7×10^{-3}, a similar range to that found in fruits from herbaceous plants i.e. 4.1×10^{-4} to 8.9×10^{-3}. More generally, the variability in F_v values reflects primarily differences in soil characteristics rather than differences among plants. The highest values are for light textured or organic soils.

F_v values for strontium are generally one order of magnitude higher than for caesium, ranging from 1.2×10^{-3} to 2.1×10^{-1} (Table 9). Values reported for woody trees range from 1.2×10^{-3} to 7.0×10^{-2}. F_v values for shrubs are, in general, about one order of magnitude higher than those for woody trees, falling within the range of 1.4×10^{-2} to 1.1×10^{-1}. The same applies to values for herbaceous plants, with values ranging from 1.2×10^{-2} to 2.1×10^{-1}.

Soil characteristics are fundamental in determining the amount of strontium transferred from soil to fruit. Uptake from organic soil is one order of magnitude less than that from loam and sand.There is some evidence that the genotype of a plant has a weight similar to soil characteristics in determining the uptake of radiostrontium: strawberry and blackcurrant show the highest F_v values (2.1×10^{-1} and 1.1×10^{-1} respectively [49]), whereas there are small differences between values for the other fruits.

Values for plutonium range from 1.3×10^{-6} to 2.1×10^{-2} (Table 10). Removing a few specific regional data on apple from Cumbria, supposedly affected by external contamination, the range is reduced to: 1.3×10^{-6} to 9.2×10^{-4}. Data for americium range from 1.3×10^{-6} to 7.2×10^{-4} (Table 11). Plutonium and americium appear to behave in a similar way in the soil-plant system and F_v values follow approximately the same trend for each class of fruit.

Data on F_v for iodine are scarce and show a range from 4.1×10^{-4} to 3.1×10^{-2} (Table 12). Some of the data in the literature have been obtained from weapons fallout in field conditions. F_v values for cerium (Table 13) and curium (Table 14) are all in the order of 10^{-4}, whereas those for ruthenium (Table 15) range from 7.4×10^{-4} to 1.6×10^{-3}. The few data available for Mn, Na, Co and Cu are gathered in Table 16.

CONCLUSIONS

Conclusions and recommendations reported here are the results of the work carried out by many experts in the IAEA BIOMASS (BIOsphere Modelling and ASSessment) Fruits Working Group. For a summary of the activities of the Group the reader is referred to [50, 51] and for more detailed information to [2-5, 7-8, 47, 52-59].

Fruit intended as a component of the human diet is borne by plants that are heterogeneous in habits, and morphological and physiological traits. The collection of information on radionuclides in fruit systems has therefore been rationalised by characterising plants in three groups: woody trees, shrubs, herbaceous plants.

Main finding from the data reported here can be summarized as follows:

- There are large differences in the degree of translocation of radionuclides from the aboveground part to fruit of different species, depending upon physiological habits of the plants. Differences also occur between different varieties.
- Translocation occurs to a greater extent during the time from fruit development to beginning of ripening for those radionuclides mobile in the phloem.
- Direct deposition to fruit is connected with the phenological stage of the plant at the time of deposition, with the extent of protection exerted by the leaves on the fruits, and indirectly with the ability of pollutants to translocate from leaves to fruits.
- Soil-to-fruit transfer is radionuclide specific. The variability in transfer factors for a given radionuclide is attributable primarily to the different properties of soils. For example, the highest transfer factors for caesium are specific to peat or light textured soils. The lowest transfer factors for strontium are specific to organic soils, such as peat, and to soils with high calcium content. The transfer of both plutonium and americium is lower in loam, organic and calcareous soils.
- A smaller contribution to the variability of transfer factors depends on the type of plant. Given the paucity of data, it is difficult to determine which class or species generally has the largest soil-to-fruit transfer of radionuclides.
- The contamination of fruits borne by woody trees in the years following the initial deposition can occur by remobilisation of reserves from the storage organs of the tree. However, the relative importance of the processes of transfer from soil to plant and retranslocation from storage organs has not yet been well determined.
- Generally the activity in fruit in the years following that of deposition shows a decrease of various orders of magnitude, depending not only on the kind of radionuclide and the kind of plant, but also, presumably, on different human interventions in the soil-plant system.
- Some of the information sources used in this review are for the temperate climate of Northern Europe, a relatively minor fruit-producing area. Large quantities of fruit and related products may be imported. It is important to take the contribution of imported fruit into account when considering the radiation exposure of a population due to contaminated fruit. It is also important that consideration should be given to the full range of fruits grown in all countries; in particular, grapes, olives, citrus fruits and bananas.
- Radionuclide concentrations in fruit depend on the yield. Low yield correlates with high concentrations of radionuclides. The radionuclide concentration in fruit varies with time to ripening. It may increase because of leaf-to-fruit translocation or soil-to-fruit transfer, decrease because of growth dilution, and then increase again towards ripening because of water loss by aging.
- The uptake and transfer of radionuclides to fruit crops are influenced by the chemical form of the radionuclide (speciation). Current information on the effects of speciation is limited in regard to the uptake and retention of radionuclides in fruit crops. For ^{14}C, ^{129,131}I, ^{3}H and ^{35}S, there is some relevant information on speciation with respect to exposure of fruit via the atmosphere. For exposure via soil pathways there is also some information but this is generally not relevant to current models. At present, however, there are no published studies on the consequences of radionuclide speciation for foliar or root uptake of radionuclides by fruit.

- Although the behaviour of radionuclides in fruit can be predicted from current knowledge of the behaviour of radionuclides in other crops, there are many associated uncertainties and much scope for validation of models and principles of interpretation of data for fruit specifically.

There is a need to standardise the methodology of carrying out experiments on foliar uptake and to standardise the mathematical description of the processes.

There is also a need for research on the behaviour of radionuclides in fruit crops to drive model development, not simply to parameterise existing models. Research should focus on understanding the key processes. Experiments undertaken to validate existing models should be directed towards the provision of time-dependent data on the distribution of radionuclides in fruit crops rather than to the provision of data sets comprising single end-points such as concentration in fruit.

In particular, for modelling the transfer of radionuclides to fruit, there is a general need for composite data sets that provide the following information: deposition/interception, distribution in the plant-soil system, and change in distribution with time. Also, the age and growth stage of fruit at the time of contamination is an important factor in modelling the transfer of radionuclides to fruit.

In addition, there is a need to compile a database of growth details for important fruit. The database should include leaf area index and herbage density over a growing season.

There are specific needs for data on shrub-type fruits and for research on the deposition and transfer of ^{129}I to food crops, including fruits.

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NATURAL AND SEMI NATURAL ECOSYSTEMS

RADIONUCLIDE TRANSFER IN FOREST ECOSYSTEMS

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Abstract

The behaviour of radionuclides in forest ecosystems differs substantially from the other ecosystems. The contamination of various forest products is commonly quantified using the Aggregated Transfer Factor (T_{ag} in m^2 kg^{-1}) which integrates various environmental parameters including soil and plant type, root distribution as well as nature and vertical distribution of the deposits. This review aims at compiling the most relevant quantitative information on radionuclide transfers to forest biota including trees, understorey vegetation, mushrooms, berries and game animals. For both radiocaesium and radiostrontium in trees, the order of magnitude of mean T_{ag} values is 10^{-3} $m^2.kg^{-1}$(dry weight). The transfer of radionuclides to mushrooms and berries is high, in comparison with foodstuffs grown in agricultural systems. Concerning caesium uptake by mushrooms, the transfer is characterized by a very large variability of T_{ag}, from 10^{-3} to 10^{1} $m^2.kg^{-1}$(dry weight). For berries, typical values are around 0.01 to 0.1 $m^2.kg^{-1}$ (dry weight). Transfer of radioactive caesium to game animals and reindeer and the rate of activity reduction, quantified as an ecological half-life, reflect the soil and pasture conditions at individual locations. Even if, the importance of radioactive contamination of forests as a significant source of the population exposure is recognized, most of the data refer to caesium and to a lesser extent, strontium. Data for other radionuclides are rather limited.

1. INTRODUCTION

The behaviour of radionuclides in forest ecosystems has to be given specific consideration because such ecosystems differ substantially in radionuclide biogeochemistry and exposure pathways from the agricultural ecosystems that are more generally considered in assessments. In particular, radionuclides can be efficiently trapped and recycled in forests, implying long residence times and the potential for enhanced external and internal exposures over timescales of decades to centuries.

The importance of radioactive contamination of forests as a significant source of the population exposure was recognized after at least two major radiation accidents–the Kyshtym accident, Ural, USSR (now Russian Federation), in 1957 and the Chernobyl accident, USSR (now Ukraine), in 1986. A substantial amount of the recent research on radionuclide behaviour in semi-natural ecosystems has been undertaken in those contexts. While the Chernobyl signal was mainly ^{137}Cs and that of Kyshtym, ^{90}Sr, other local ecological insults have been the source of forest contamination by other radionuclides (e.g. $^{239,240}Pu$, ^{99}Tc) but the database on parameters that can be used for the modelling of contaminated forests is, as yet, substantially incomplete. In particular, most of the data that are available relate to the behaviour of radioisotopes of caesium and of strontium to a lesser extent, and this is reflected in the emphasis of the material presented below. However, the more limited information on radioisotopes of other elements is also discussed, as appropriate.

2. TRANSFER TO TREES

2.1. Processes and dynamics of tree contamination

In addition to the forest zone contaminated following the Kyshtym accident, the vast extent of semi-natural areas affected by the Chernobyl deposits and the corresponding radiological impact on various local populations has initiated a new interest in forest radioecology. In the two last decades in particular, numerous studies of radiocaesium and radiostrontium transfers have been conducted in a wide range of forest types located at various sites in the CIS and in Western Europe. This extensive set of research activities has led to the accumulation of a considerable amount of new data and increased understanding of processes of relevance.

Following deposition of atmospheric radioactive fallout, the primary source of tree contamination is direct dry or wet interception of aerosol-derived radionuclides by the canopy, followed by further translocation from foliar surfaces to structural components of the tree. Further changes in tree contamination after the initial fallout are due to two main processes. The first of these is a dominant self-decontamination process of the tree canopy, affected by weathering of intercepted radioactive material, through fall and litter fall. These processes are followed by or accompanied by root uptake which is the predominant route of contamination over the longer term. In terms of dynamics of contamination of the system, two stages can be distinguished: 1. the "*early' phase* lasting 4-5 years and characterized by a rapid redistribution of the initial deposits between the soil and the trees, 2. a *'steady state' phase* characterized by slow changes in biological availability, with root uptake determining the degree of contamination of the trees.

2.1.1 The early phase

Because of high biomass density of forest canopy, interception of atmospheric radioactive fallout by the forest cover may be very efficient as well but depends on a great number of local factors: vegetation biomass per unit area, plant surface nature, radioactive particle dispersion, meteorological conditions. A summary of the information on radionuclide interception by forest canopies from the research performed before the Chernobyl accident is given in Table 1. Subsequent to the interception of radioactivity, variations in the intensity of tree crown self clearing may be large; the main influencing factors are as follows: species composition, age, physiological state and density of stocking, as well as the season when radioactive fallout happens. The ecological half life of clearing after the deposition event and in the course of active growth of trees varies from 3–4 weeks to 3 months depending on the type and age of trees. In the phase of physiological dormancy (autumn and winter), the half-life is 4–6 months [2].

As illustrated in Figure 1, ^{90}Sr concentrations in crowns of deciduous forests can decrease to several per cent of the total amount present in the forest within a year after a single deposition event. In coniferous forests, the process of crown self-clearing is much slower; it may take 3–4 years [1].

TABLE 1. INTERCEPTION FRACTIONS FOR DIFFERENT TYPES OF FORESTS [1].

Forest type	Deposition type	Interception, %
Pine forest, age 6-10 years	Artificial injection of ^{89}Sr in a water-soluble form into the crowns of trees	90-100
Pine forest, age 60 years	Deposition of radioactive particles with the size less than 50 µm	80-100
Pine forest, age 25 years	Deposition of radioactive particles with the size less than 100 µm	70-90
Pine forest, age 30 years	Deposition of resuspended radioactive particles.	40-60
Birch forest, age 40 years, winter period	Deposition of resuspended radioactive particles.	20-25
Birch forest, age 35-40 years, summer period	Global fallout	20-60
Pine forest, age 50-60 years	Global fallout	50-90
Tropical rain forest	Global fallout	100

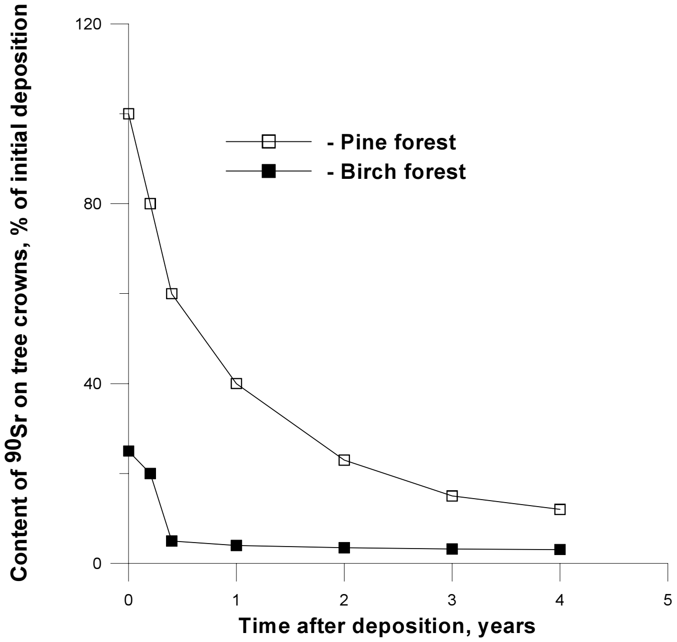

FIG. 1. Self-clearing of overground part of tree layers of pine and birch phytocenoses after a single artificial introduction of ^{90}*Sr in a water-soluble form into the crowns late in summer [1].*

Over the first few days of radioactive discharges from the Chernobyl NPP, about 70–80% of all the radioactive fallout was retained by the aboveground parts of trees. Over this period, coniferous trees trapped radioactivity 2–3 times as effectively as did deciduous forests and 7–10 times more than other types of natural cenoses (meadow, mire) [3].

The size and solubility of the Chernobyl fallout particles were the foremost factors influencing the initial distribution of Cs between tree components. The acute phase lasting 4-5 years was further a period of great differences in Cs recycling in the system as a function of the distance from the accident. In the near zone (< 30 km), characterized by greater particle size in the deposits, limited initial contamination of pine trees was associated with rapid wash-off of the intercepted contamination followed by a progressive increase in root uptake and tree contamination due to the solubilisation of the particles in the soil. In the remote zone

(> 30 km), the higher solubility of the sub-micron particles that were preferentially transported as aerosols over these greater distances was the cause of a higher degree of foliar adsorption and initial tree contamination. Afterwards a reduction in the biological availability of Cs resulted mainly from the migration and 'irreversible' fixation of Cs in the soil, as determined by site-specific properties.

2.1.2. The steady state phase

In the initial period decontamination processes dominate and result mainly in decreasing contamination of crown-layer components. Later on as a result of root accumulation of radionuclides, their content in the aboveground biomass increases until a quasi-equilibrium state is achieved. Just as in the case of its nutrient analogue potassium, the rate of radiocaesium cycling within forests is rather rapid and a quasi-equilibrium applicable to its distribution in trees as well as to its biological availability in soil may be reached at 5 to 10 years after atmospheric fallout. For radiostrontium, the period when this plateau in all tree compartments is achieved is 10 -15 years (Fig. 2).

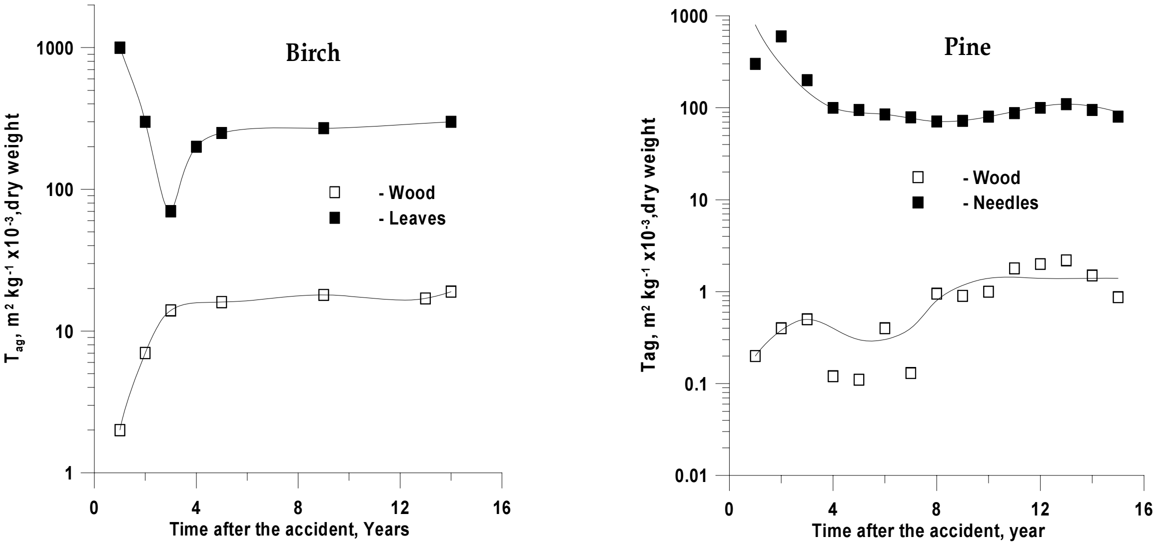

FIG 2. Transfer of ^{90}Sr to wood and needles/leaves of trees after the Kyshtym accident.

Beyond that time, inter-annual changes in tree contamination are expected to mainly relate to tree development and biomass accumulation rather than to significant changes in the biological availability of the radionuclide in soil.

The type of forest soil and its water regime are also important factors affecting the extent of radioactive contaminant availability to trees and for ecosystem cycling [2, 4] (Fig. 3). For soils associated with a high biological availability of radionuclides, the equilibrium state is usually achieved faster and at a higher level [2]. The quasi-equilibrium content of ^{90}Sr in the overground biomass of trees amounted to 1% of the total content of this radionuclide in soils and vegetation for pine forests located in the area of Kyshtym accident. In the 30 km zone of Chernobyl accident in 1991 the corresponding value was 3% for automorphic landscapes (sandy turf-podzolic) and about 11 % for hydromorphic landscapes with sandy soil characterized by a thick organic layer [2].

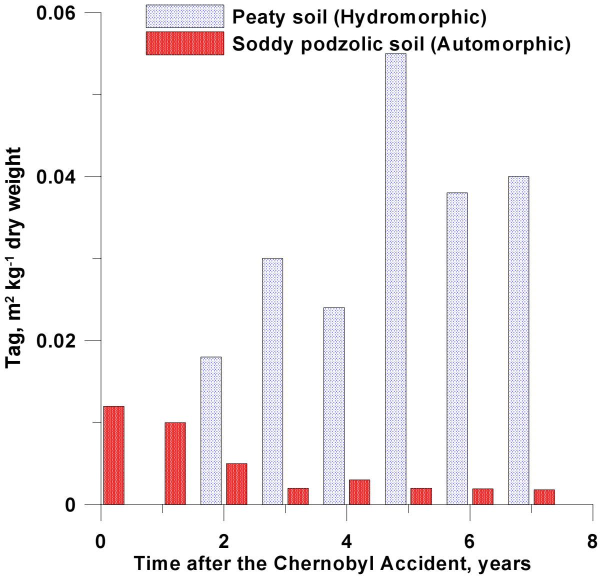

FIG. 3. Transfer of ^{137}Cs to pine wood on soils with contrasting properties

2.2 Long term effect of ecological factors on aggregated transfer factor

In the long term, the soil is the dominant radionuclide reservoir in forest systems and root uptake governs further accumulation in standing biomass at a rate that depends on a combination of several abiotic and biotic factors like soil type, moisture regime, stand composition, stand age and tree species. A tentative hierarchy of the mentioned environmental factors was proposed by [4] based on their respective significance on the potential variability in the average radiocaesium accumulation by trees (Table 2). The indicative significance of each factor is illustrated in Table 2 by a ‘variability’ index which reflects the possible magnitude of observed differences in tree contamination between two sites due to one specific factor) [4].

TABLE 2. ENVIRONMENTAL FACTORS GOVERNING THE EXTENT OF TREE CONTAMINATION BY RADIOCAESIUM

Influencing factors	Index	Examples of T_{ag} hierarchy for trees
Soil type	100 (10-200)	peat-gley > peat-podzolic > soddy-podzolic > podzolized chernozems
Moisture regime	10 (3-70)	central depression > terrace basement > terrace slope > slope upper part > watershed top
Stand composition	4 (5-10)	Monospecific coniferous stand > mixed coniferous-deciduous forest
Stand age	4 (3-8)	0-30 > 30-60 > 60-90 > +90
Tree species	2 (2-3)	aspen > oak > birch > pine > lime > spruce

Since certain combinations of influential properties of the forest system (soil and humus type, soil profile development, vegetation association, moisture regime) are inherent to a limited number of forest eco-types, a preliminary basic ecological classification of forest ecosystems can also contribute to the minimization of the variability of T_{ag} coefficients for trees in those particular forest systems.

Examples given by [4] and [5] are shown in Tables 3 and 4, respectively, while additional information on the relationship between the degrees of contamination of different tree organs can be extracted from the same tables.

TABLE 3. IMPACT OF MOISTURE CONDITIONS ON VARIATIONS IN RADIOCAESIUM AGGREGATED TRANSFER FACTORS (T_{ag} in 10^{-3} m^2 kg^{-1}, DW) TO FOREST TREES.

Tree species	Moisture conditions	Assimilative organs		One-year-old shoots	Stem bark	Wood
		One-year-old needles (leaves)	Two-year-old needles			
Pine	Automorphic	9.8	4.1	1.1×10^{1}	7.8	3.2×10^{-1}
	Semi-hydromorphic	3.8×10^{1}	1.4×10^{1}	3.2×10^{1}	-	1.2
Birch	Automorphic	1.6	-	2.2	6.5	3.5×10^{-1}
	Semi-hydromorphic	3.6×10^{1}	-	3.2×10^{1}	-	1.4
Aspen	Automorphic	-	-	-	-	4.8×10^{-1}
	Semi-hydromorphic	7.5	-	7.8	1.0×10^{1}	1.7
Alder	Not specified	7.7	-	7.2	1.2×10^{1}	1.4
Oak	Not specified	8.4	-	8.8	1.6×10^{1}	2.3

TABLE 4. VARIATIONS IN RADIOCAESIUM AGGREGATED TRANSFER FACTORS (T_{ag} in 10^{-3} m^2 kg^{-1}, DW) TO DIFFERENT PINE AND BIRCH COMPARTMENTS (REPRODUCED FROM [8.5])

Eco-type	Tree organs					
	Wood	Bark		Branches		Foliage
		Inner	Outer	Large	Small	Current/older
Pine						
Automorphic	7×10^{-2}	5.6×10^{-1}	1.1	2.6×10^{-1}	4×10^{-1}	$1.1 \times 10^{0}/1.7 \times 10^{-1}$
Hydromorphic	4.0	4.4×10^{1}	1.1×10^{1}	8.0	1.8×10^{1}	$4.9\ 10^{1}/1.5 \times 10^{1}$
Birch						
Automorphic	1.1×10^{-1}	3.8×10^{-1}	6.0	9.5×10^{-1}	1.5	1.7
Hydromorphic	7.4	2.3×10^{1}	8.5	1.0×10^{1}	3.4×10^{1}	6.7×10^{1}

2.3 Radiocaesium and radiostrontium transfer to foliage and stem wood of forest trees

It is of relevance to define those components and tissues that can be considered most representative of radionuclide accumulation in trees. For radiocaesium, the best indicative organs are usually the most physiologically active ones such as leaves or 1 year-old needles, since they show the best correlation with radiocaesium concentrations in the other components.

Stem wood, which is the largest pool of aboveground tree biomass, was identified in many situations as the main long-term reservoir for radiocaesium in forest vegetation. Table 5 make a list of the T_{ag} values for ^{137}Cs in foliage and wood as recorded in different ecological conditions for various ages and species of trees, while Table 6 similarly presents T_{ag} values across some contaminated regions for ^{90}Sr.

It can be seen from Tables 2-6 [6-21] that deciduous trees are characterised by a higher absorption of ^{90}Sr and ^{137}Cs than are coniferous ones [21]. Among the woody plants the maximum transfer factors were observed in aspen both in areas close to and far from the Chernobyl accident. This agrees with a high demand of aspen for potassium as compared with other species of woody plants. Because of higher nutrient demand, higher concentrations of ^{137}Cs are also usually observed in organs of young plants as compared with those of middle-aged and mature trees [1, 4]. Most analyses of radionuclide distributions in trees show that they are accumulated to the greatest extent in roots [1]. The exception is ^{90}Sr, which actively moves from roots into the over-ground parts of trees and is accumulated predominantly in vegetative parts of forest herbaceous and woody plants [21].

TABLE 5. RADIOCAESIUM AGGREGATED TRANSFER FACTORS (T_{ag} in 10^{-3} m^2 kg^{-1}, DW) TO FOREST TREES, MEASURED IN APPARENT STEADY STATE CONDITIONS [6]

Site	Species	Age(y)	Soil type	T_{ag} (10^{-3} m^2 kg^{-1}, DW)		
				Wood	Needles/Leaves	
					Current	Old
Lady Wood (UK)	Spruce	40	Juvenile iron pan podzol	2.8×10^{-1}	5.7×10^{-1}	
Clogheen Wood (Ireland)	Pine	60	Peaty podzol on sandstone	1.4	2.1×10^{1}	
Shanrahan Wood (Ireland)	Pine	50	Peaty podzol on sandstone	3.2×10^{-1}	1.0×10^{1}	
Ballyporeen 1976 (Ireland)	Pine	33	Peaty podzol on sandstone	1.4	5.1	
Ballyporeen 1965 (Ireland)	Pine	22	Peaty podzol on sandstone	2.3	5.2	
Roundwood (Ireland)	Spruce	40	Peaty podzol on schist	9.3×10^{-1}	5.0	
Tarvisio (Italy)	Spruce	60-100	Brown earth on calcareous moraine	1.7	9.1	
Novaggio (Switzerland)	Oak	na	Crypto podzol	3.8	12	
Weinsberger (Austria)	Beech	na	Dystric cambisol	1.3	2.3	
Kobermausser (Austria)	Spruce	60-70	Dystric cambisol	2.1	1.7×10^{1}	
Kobermausser (Austria)	Beech	na	Dystric cambisol	1.6	2.7	

TABLE 5. RADIOCAESIUM AGGREGATED TRANSFER FACTORS (T_{ag} in 10^{-3} m^2 kg^{-1}, DW) TO FOREST TREES, MEASURED IN APPARENT STEADY STATE CONDITIONS (Cont.)

Site	Species	Age(y)	Soil type	T_{ag} (10^{-3} m^2 kg^{-1}, DW)			Refer.
				Wood	Needles/Leaves		
					Current	Old	
Bryansk (Russian Federation)	Pine	11-50	Soddy-podzolic loamy sand (automorphic)	5.9×10^{-1}	1.9×10^{1}	4.3 (2-3 y)[a]	[7]
Bryansk (Russian Federation)	Pine	45-55	Humic podzolic gley loamy sand (semi-hydromorphic)	5.6×10^{-1}	1.9×10^{1}	3.6 (2-3 y)	[7]
Bryansk (Russian Federation)	Pine	10-30	Humic-peat (hydromorphic)	5.2	6.4×10^{1}	6.4 (2-3 y)	[7]
Bryansk (Russian Federation)	Birch	8-40	Humic podzolic gley loamy sand (semi-hydromorphic)	9.0×10^{-1}	3.0×10^{1}		[7]
Bryansk (Russian Federation)	Birch	30-40	Soddy-podzolic loamy sand (automorphic)	2.4×10^{-1}	2.8		[7]
Zaborie (Russian Federation)	Pine	45	Sandy	1.95 (1m)[b]	1.9×10^{1}	4.2 (3 y)	[8]
Zaborie (Russian Federation)	Pine	10	Sandy (deep ploughing)	1.1×10^{-1} (1m)	1.7	-	[8]
Novo Bobovichi (Russian Federation)	Pine	21	Sandy	7.5×10^{-1} (1m)	1.1	2.4×10^{-1}	[8]
Guta Muravinka (Russian Federation)	Oak	13	Sandy	1.53 (1m)	1.1×10^{1}		[8]
Mol (Belgium)	Willow	1	Orthic podzol (sandy)	2.4×10^{-2}	-		[9]
Mol (Belgium)	Willow	1	Orthic luvisol (loamy)	1.0×10^{-2}	-		[9]
Vetka (Belarus)	Pine	17	Dystric cambisol (sandy)	3.5	2.5×10^{1}	1.1×10^{1}	[10]
Vetka (Belarus)	Pine	37	Dystric cambisol (sandy)	1.6	1.0×10^{1}	4.1	[10]
Vetka (Belarus)	Pine	57	Dystric cambisol (sandy)	2.0	1.2×10^{1}	5.0	[10]
Belarus polesye	Pine	0-20	Soddy podzolic soil (sandy)	3.46± 1.03[c]	-	-	[4]
Belarus polesye	Pine	21-40	Soddy podzolic soil (sandy)	3.00± 0.48	-	-	[4]

TABLE 5. RADIOCAESIUM AGGREGATED TRANSFER FACTORS (T_{ag} in 10^{-3} m^2 kg^{-1}, DW) TO FOREST TREES, MEASURED IN APPARENT STEADY STATE CONDITIONS (Cont.)

Site	Species	Age(y)	Soil type	T_{ag} (10^{-3} m^2 kg^{-1}, DW)			Refer.
				Wood	Needles/Leaves		
					Current	Old	
Belarus polesye	Pine	41-60	Soddy podzolic soil (sandy)	1.42± 0.35	-	-	[4]
Belarus polesye	Pine	61-80	Soddy podzolic soil (sandy)	0.46± 0.10	-	-	[4]
Duboka (Croatia)	Beech	96	-	1.8×10^{-1}	-	-	[11]
Spacva (Croatia)	Oak	96	-	1.1×10^{-1}	-	-	[11]
Gorski (Croatia)	Fir	26	-	1.2×10^{-1}	-	-	[11]
Parkano (Finland)	Pine	50-80	Peat	2.1×10^{1}	9.2×10^{1}		[12]
Sweden-central part	Spruce	100	Podzol (sandy)	3.9	2.6×10^{1}		[13]
Sweden-central part	Pine	100	Podzol (sandy)	3.5	1.9×10^{1}		[13]
Sweden-central part	Birch	100	Podzol (sandy)	3.8	7.8		[13]
Sweden-central part	Spruce	50	Podzol	2.7	5.2×10^{1}	1.5×10^{1}	[14]
Sweden-central part	Pine	50	Podzol	2.7	3.3×10^{1}	1.2×10^{1}	[14]
Pot experiment (Sweden)	Willow	1	Decomposed fen peat	7.8 (wood+leaves)			[15]
Tisvilde (Denmark)	Pine	mature	Podzol (sandy)	2.0	6.4×10^{1} (needles+twigs)		[16]
Weinsberger (Austria)	Spruce	60-70	Distric cambisol (clay)	1.5	8.4	3.6	[17]
Uppsala (Sweden)	Willow	2	Loam	6.8×10^{-2}	2.1×10^{-1}		[18]
Aheden (Sweden)	Pine	40-70	Regosol (loamy sand)	3.5	2.0×10^{1}		[19]
30 km zone of the ChNPP	Pine	40-65	Soddy-podzolic (sandy)	5.0±0.47	64.7±5.8	17.7±1.8	[20]

na = not available.

[a] Age of sample; [b] height of sampling; [c] values are averages ± standard deviations.

TABLE 6. T_{ag} VALUES FOR RADIOSTRONTIUM TRANSFER TO FOLIAGE AND WOOD OF DIFFERENT FOREST TREE SPECIES. MEASURED FOLLOWING THE CHERNOBYL (1991-1992) AND KYSHTYM (1966-1972) ACCIDENTS

Site	Species	Age(y)	Soil type	T_{ag} (10^{-3} m^2 kg^{-1}, DW)			Refer.
				Wood	Needles/Leaves		
					Current	Old	
Tula (Russian Federation)	Pine	50-55	Chernozem	5.7 x 10^{-1}	1.5	3.3	[20]
Bryansk (Russian Federation)	Pine	40-65	Soddy-podzolic (sandy)	2.0±0.9[1]	4.4±2.3	3.3±1.5	[20]
Kiev (Ukraine)	Pine	55-65	Soddy-podzolic (sandy)	1.20±1.08	2.25±3.23	2.60±2.75	[20]
Kiev (Ukraine)	Pine	55-65	Humic podzolic (hydromorphic)	10.3± 2.4	21.2	29.5± 6.3	[20]
Kyshtym (Russian Federation)	Pine	55-70	Chernozem	0.7±0.4	6.2±3.2		[21]
Tula (Russian Federation)	Birch	50-55	Chernozem	3.7	6.8		[20]
Bryansk (Russian Federation)	Birch	40-60	Soddy-podzolic (sandy)	6.2±5.5	44.0±20		[20]
Kiev (Ukraine)	Birch	35-55	Humic podzolic (semi-hydromorphic)	2.19±0.08	19.2±0.9		[20]
Kiev (Ukraine)	Birch	50-55	Podzol (sandy)	0.58±0.13	4.3±1.6		[20]
Kyshtym (Russian Federation)	Birch	55-70	Chernozem	2.8±1.7	78±34		[21]
Tula (Russian Federation)	Oak	45-50	Chernozen (Loam)	1.71	4		[20]
Bryansk (Russian Federation)	Oak	35-45	Soddy-podzolic (sandy)	2.78±0.05	9.96±1.75		[20]
Kiev (Ukraine)	Oak	40-45	Podzol (sandy)	0.47±0.17	1.85±0.79		[20]
Bryansk (Russian Federation)	Fur-tree	50-55	Podzol (sandy)	4.39	12.9	13.4	[20]
Kiev (Ukraine)	Alder	40-55	Podzol (sandy)	0.946	5.73		[20]
Kiev (Ukraine)	Aspen	40-55	Podzol (sandy)	2.1±1.4	17.3±11.9		[20]

[1] Values are arithmetic means ± standard deviations.

2.4 Recommendations on use of T_{ag}

During the early phase, varying dynamics of initial redistribution processes mainly influenced by the nature of radioactive deposit and the type of forest greatly reduce the correlation between the radionuclide content in plant and soil. Even in the long-term (steady state phase) the long-lasting radionuclide recycling in forest ecosystems is far from implying a simple relationship between concentrations in soil and tree. These complexities can affect the usefulness of T_{ag} for a holistic spatial and temporal ranking of forest systems in terms of risks of radionuclide transfer. The most realistic use of the T_{ag} coefficient is for forest systems where radionuclide fluxes have stabilized and in that case, it remains a satisfactory tool for simple screening models.

3. TRANSFER TO UNDERSTOREY VEGETATION

The understorey in forests is composed of fungi and plants growing beneath the tree canopy. The understorey vegetation can be divided into layers according to the height of plants [22]:

- shrub layer (only ligneous, including climbers) > 0.5 m height
- herb layer (all non-ligneous, and ligneous < 0.5 m height)
- moss layer (i.e. terricolous bryophytes and lichens).

The carbon content of understorey vegetation in a boreal subarctic forest is typically 0.5 tons per hectare, corresponding to 1.4 percent of the carbon content of forest vegetation, when both below-ground and above-ground biomass are considered [23]. Species composition and biomass of understorey vegetation are related to the fertility and humidity of forest soil and also to the structure of the tree stand. The species may also vary by season; in spring the composition is different from that in midsummer.

Forest types have traditionally been defined through species of trees and dominant or indicator plants of ground vegetation in a forest ready for felling, thus avoiding characterisation during temporary disturbances of the understorey. Rich grass and herb vegetation is typical of, for instance, a spruce forest of *Oxalis–Maianthemum* type. In dry, nutrient poor heath forests of *Cladonia* type (ClT), dominant species are lichens, their coverage being more than 75% [24]. For selected forest types, such as beech forests and eastern mesophytic deciduous forests, the type definition in a current European system may be detailed also from floristic point of view and the herb layer can be an additional but relevant criterion for forest type discrimination [25].

The development stage of a tree stand has an effect on the nutrient dynamics and growth conditions of understorey plants. During forest regeneration and in the stage of emergence of seedlings, the soil contains nutrients and water in abundance and surface vegetation is lush. At this stage, the understorey vegetation will reduce the washout of nutrients from the soil. When the biomass of trees increases, the amount of surface vegetation may decrease and nutrients become available for the growth of trees [23], particularly in spruce and broad-leaved forests. After crown closure, if tree canopies are dense, lack of light at the ground layer may reduce or eliminate the understorey growth in certain oak, hornbeam and spruce forests.

During atmospheric deposition of radioactive material, radionuclides are partially intercepted by the understorey plants, either in dry conditions or during rain. Maximum interception of radionuclides deposited wet has been suggested to occur when rainfall is of the order of the saturation capacity of the plant [26]. Both dry and wet deposition result in primary contamination of vegetation. Weathering is same removal process of intercepted radionuclides as for tree canopies. Radionuclides on surfaces of plants, if in water-soluble form penetrate surfaces of leaves and enter the plant. Metabolised radionuclides are translocated to different parts of a plant, for instance to the fruit or berries. Of importance for human ingestion dose is the fraction of intercepted activity that is found in edible parts during harvest.

After radionuclide contamination of the forest floor, a fraction of deposited activity is retained in moss and lichen patches. Such retention will delay the contamination of soil below bottom vegetation, and contribute to the uneven pattern of surface activity density [26]. In natural conditions, i.e. in undisturbed forest soils, the water-soluble fraction of deposited radionuclides is effectively incorporated into the nutrient pools of mineral elements. Generally the response of understorey plants to additional radionuclide contamination of the forest floor is fast.

Certain understorey species are used as indicator organisms for environmental radionuclides. For instance, mosses and lichens can indicate deposition of airborne radionuclides, and ferns and ericoid species, radionuclides accumulated in soil.

Concerning dose pathways from forest to people, understorey vegetation and fungi transfer radionuclides through wild food to man. Contributions of animal products to ingestion dose are received from ruminants grazing in forests, and big and small mammals and terrestrial birds taken as game. Wild berries, herbs and mushrooms can also be considerable dietary sources of radionuclides.

On two natural coniferous sites located in south Bavaria and showing a rich flora, ^{137}Cs and ^{134}Cs uptake by understorey plants was studied by Wirth et al. in 1991 [27]. Stands on these mineral soil sites were 120 year old mixed Scots pine and Norway spruce, and a 100 years old Norway spruce stand. Soil horizons and mineral contents showed some differences. Plants take up nutrients and radioactive caesium from O-horizons mainly, and the caesium in deeper mineral layers A and B was strongly adsorbed in those layers. On a site where a mineral component was present in the O-horizon, a considerably lower uptake of ^{134}Cs and ^{137}Cs in leaves and stems was found than on a site without a mineral component in the O-horizon. The concentration ratios (for dry weights) of plants and forest litter horizons O_f plus O_h were 4- and 17- fold larger in dwarf shrubs bearing berries and ferns on the site where these layers were not mixed with the underlying mineral horizon compared with another site where these layers were mixed [27]. In general, in the boreal forest vegetation zone, adsorption of radioactive caesium to soil minerals is mostly lower than in the temperate zone, due to lack of clay.

Significant differences in uptake of radionuclides were found between understorey species, sites and soil types, when ^{137}Cs concentrations in annual herbs, grasses and perennial dwarf shrubs were compared on seven sites in northern and central Europe. Plant samples were taken twelve years after the Chernobyl accident [28]. The aggregated transfer factors (m^2 kg^{-1} dry weight) for vascular plants grown in forests on mineral soils were generally lower than in a peat-land forest or for plants grown on a thick layer of organic soil. Both perennial and annual plants differed by species in uptake of ^{137}Cs. The highest values of T_{ag} were found for *Dryopteris carthusiana* (0.090-0.434, two sites), *Calluna vulgaris* (0.025-0.241, four sites), and *Melampyrum pratense* (0.091-0.125, one site).

The contribution of mycorrhizal fungi in the uptake, bioavailability and recycling of radionuclides by understorey plants is essential, but can not be quantified at the moment [29, 30, 31]. Also, the wide variety of forest types in various climatic conditions necessitates diverse analyses of radionuclide dynamics and substantial differences in quantification of radionuclide uptake by understorey plants.

3.1. Transfer to mushrooms

The increased interest in forest ecosystems following the Chernobyl accident has been associated with a greater emphasis on the radiological impact of naturally occurring foods that are collected in such ecosystems. In particular, it has been recognized that mushrooms are collected in relatively large quantities and can accumulate radionuclides to a significant degree. As in the case of forest trees, the following discussion relates primarily to radioisotopes of Cs, as these have been subject to the most intense study following the Chernobyl accident. The caesium transfer factor to mushrooms is widely variable (3 to 4 orders of magnitude). This variability arises for several reasons:

- The species plays a role of prime importance. The high transfers of caesium in some mushrooms species could be related to the existence in the carpophores of substances with a great affinity for caesium.
- The mycelium depth plays a crucial role in the contamination chronology. Mushroom species with surface mycelia can be contaminated immediately after deposition, whereas mushroom species with deeper mycelia will be contaminated later. For example, *Boletus edulis* which is a symbiotic mushroom with deep mycelia was found to be at its maximum of contamination two or three years after the Chernobyl accident. For some saprophytic species of mushrooms, it is possible to specify the feeding source: litter, humus or dead wood. This information gives an idea on the mycelium depth of these saprophytic mushrooms.
- The nutritional type of mushroom species can affect the degree of caesium transfer. **Saprophytic** mushrooms develop on decomposing materials in the surface layers of a soil, so these kinds of mushrooms will be first contaminated following deposition. Transfer factors will subsequently decrease as the deposit migrates deeper into the soil. **Symbiotic or Mycorrhizal** mushrooms live in a mutually beneficial association with trees. Using their extended mycelium, mushrooms bring minerals to the trees, and trees provide mushrooms with organic matter originating in photosynthesis. **Parasitic** mushrooms develop at the expense of the trees. Very few are edible and their radionuclide concentration is dependant on the degree of host tree contamination, and they tend to be characterized by low transfer factors. Most of the edible mushrooms are symbiotic ones and can be the most contaminated in the medium- and long-term after deposition.

As for other forest compartments, the aggregated transfer factor concept is mainly used for characterisation of artificial radionuclide transfer to mushrooms (Tables 7-9). The majority of available information relates to ^{137}Cs (Table 8). However, more limited data are also available for some other long-lived radionuclides (Tables 7, 9). In these tables, it is assumed that the average dry matter content of mushrooms is equal to 10%. More accurately, the dry matter content of mushrooms varies from around 5 to 15%, depending on species and weather conditions [32]. The Tables 7-9 present the data for the aggregated transfer factors to mushrooms [33-53]

3.1.1. Use of the transfer factor for mushrooms

Although very useful for describing transfers of artificial radionuclides, the approach based on aggregated transfer factors is hardly ever applicable for natural radionuclides. Therefore, the transfer factor as defined for agricultural plants is also in a wide use to characterise the transfer of natural radionuclides to mushrooms (Table 10).

TABLE 7. AGGREGATED TRANSFER FACTORS (T_{ag}) TO MUSHROOMS FOR ^{90}Sr, m^2 kg^{-1}, DW [52]

Mushroom species	Edibility and life mode of mushrooms	^{90}Sr transfer factor
		Arithmetic mean
Boletus edulis	Edible. Symbiotic.	6 × 10^{-3}
Boletus appendiculatus	Edible. Symbiotic.	5 × 10^{-3}
Cantharellus cibarius	Edible. Symbiotic.	6 × 10^{-3}

TABLE 8. AGGREGATED TRANSFER FACTORS TO MUSHROOMS FOR ^{137}Cs, m^{2}·kg^{-1}, DW

Mushroom species	Edibility and life mode	Caesium transfer factor (m^2.kg^{-1} dry weight)			
	of mushrooms	GM(1)	Min.	Max	Reference
Agaricus arvensis	Edible. Humus saprophytic	5 × 10^{-3}	6 × 10^{-4}	1 × 10^{-2}	[33, 34]
Agaricus campestris	Edible. Humus saprophytic	6 × 10^{-3}	5 × 10^{-4}	8 × 10^{-3}	[34, 35]
Agaricus silvatica	Edible. Humus saprophytic	4 × 10^{-3}	-		[33]
Agrocybe aegerita	Edible. Saprophytic	1 × 10^{-1}	-		[35]
Amanita citrina	Not edible. Symbiotic	-	1.4	2.3	[36]
Amanita fulva	Not edible. Symbiotic	5	-		[37]
Amanita muscaria	Not edible. Symbiotic	-	1 × 10^{-1}	2 × 10^{-1}	[36, 38]
Amanita porphyria	Not edible. Symbiotic	-	9	1.5 × 10^{1}	[36]
Amanita rubescens	Edible. Symbiotic	2 × 10^{-1}	3 × 10^{-2}	4	[33, 37, 38, 39]
Amanita vaginata	Not edible. Symbiotic	5	-		[37]
Armillaria mellea	Edible. Parasitic/Xylophyte saprophytic	4 × 10^{-2}	1 × 10^{-4}	1 × 10^{-1}	[34, 35, 40, 41]
Boletinus cavipes	Not edible. Symbiotic	-	3 × 10^{-1}	1.4	[33, 35]
Boletus aestivalis	Edible. Symbiotic	-	9 × 10^{-2}	1 × 10^{-1}	[33, 42]
Boletus appendiculatus	Edible. Symbiotic	2 × 10^{-2}	-		[35]
Boletus edulis	Edible. Symbiotic	9 × 10^{-2}	4 × 10^{-3}	1.4	[33, 34, 35, 38, 39, 41, 43, 44, 45, 46]
Cantharellus cibarius	Edible. Symbiotic	2 × 10^{-1}	1.5 × 10^{-2}	7 × 10^{-1}	[33, 35, 40, 41, 42, 43, 44, 45, 46]
Cantharellus lutescens	Edible. Symbiotic	5 × 10^{-1}	-		[35]
Cantharellus pallens	Edible. Symbiotic	2 × 10^{-1}	-		[42]
Cantharellus tubaeformis	Edible. Symbiotic	9 × 10^{-1}	6 × 10^{-1}	1.5	[38, 43, 45]
Clitocybe clavipes	Not edible. Litter saprophytic	-	4	1.1 × 10^{1}	[36, 38]
Clitocybe gibba (or infundibuliformis)	Edible. Litter saprophytic	6 × 10^{-1}	-		[35]
Clitocybe nebularis	Not edible. Litter saprophytic	2 × 10^{-1}	-		[38]
Collybia butyracea	Not edible. Litter saprophytic	-	1 × 10^{-1}	2 × 10^{-1}	[36]
Collybia confluens	Not edible. Saprophytic	2 × 10^{-1}	-		[38]
Collybia dryophila	Not edible. Litter saprophytic	3 × 10^{-1}	-		[39]
Collybia maculata	Not edible. Litter saprophytic	-	2 × 10^{-1}	3 × 10^{-1}	[36, 38]
Collybia peronata	Not edible. Saprophytic	3 × 10^{-1}	-		[38]
Coprinus comatus	Edible. Saprophytic	5 × 10^{-3}	4 × 10^{-4}	1.5 × 10^{-2}	[34]

TABLE 8. AGGREGATED TRANSFER FACTORS TO MUSHROOMS FOR ^{137}Cs, m^2.kg^{-1}, DW (Cont.)

Mushroom species	Edibility and life mode of mushrooms	Caesium transfer factor (m^2.kg^{-1} DW)			
		GM(^{1})	Min.	Max	Reference
Cortinarius sp.	Symbiotic	6	-		[47]
Cortinarius alboviolaceus	Not edible. Symbiotic	6	-		[38]
Cortinarius armillatus	Not edible. Symbiotic	3.8	1.2	12	[33, 37, 38, 42, 48]
Cortinarius brunneus	Not edible. Symbiotic	6	-		[38]
Cortinarius cinnamomeus	Not edible. Symbiotic	-	1	9	[49]
Cortinarius delibutus	Not edible. Symbiotic	-	2.6	3.0	[33, 38]
Cortinarius glandicolor	Not edible. Symbiotic	9	-		[33]
Cortinarius mucifluus	Not edible. Symbiotic	2×10^{-2}	-		[33]
Cortinarius praestans	Edible. Symbiotic	2×10^{-2}	-		[48]
Cortinarius traganus	Not edible. Symbiotic	-	5×10^{-1}	1.4	[33, 48]
Craterellus cornucopioides	Edible. Symbiotic	3×10^{-2}	-		[35]
Dermocybe sp.	Not edible. Symbiotic	-	4	1.0×10^{1}	[36, 38]
Hebeloma sp.	Not edible. Symbiotic	-	1.5×10^{1}	2.5×10^{1}	[36]
Hydnum repandum	Edible. Symbiotic	4×10^{-1}	-		[38]
Hygrophorus sp.	Symbiotic	2	-		[44]
Hygrophorus olivaceoalbus	Not edible. Symbiotic	5	-		[38]
Hypholoma fasciculare	Not edible. Humus saprophytic	8×10^{-2}	-		[39]
Kuehneromyces mutabilis	Edible. Saprophytic	3×10^{-1}	-		[38]
Laccaria amethystea	Edible. Symbiotic/Humus saprophytic	4.9	2.1	8.1	[33, 38, 42, 48]
Laccaria laccata	Edible. Symbiotic/Humus saprophytic	7	5.2	8	[33, 38]
Laccaria proxima	Edible. Symbiotic/Humus saprophytic	-	2	4	[36]
Lactarius sp.	Symbiotic	3.9	5×10^{-1}	9	[47, 49]
Lactarius camphoratus	Not edible. Symbiotic	2	-		[33]
Lactarius deliciosus	Edible. Symbiotic	2×10^{-1}	8×10^{-4}	5×10^{-1}	[34, 46]
Lactarius deterrimus	Edible. Symbiotic	-	1×10^{-1}	4×10^{-1}	[33, 37]
Lactarius helvus	Not edible. Symbiotic	3×10^{-1}	-		[33]
Lactarius lignyotus	Edible. Symbiotic	9×10^{-1}	-		[33]
Lactarius necator or turpis	Edible. Symbiotic	1.5	7×10^{-1}	3	[33, 38, 40]
Lactarius odoratus	Not edible. Symbiotic	1×10^{-1}	-		[33]
Lactarius picinus	Not edible. Symbiotic	3×10^{-1}	-		[33]
Lactarius porninsis	Edible. Symbiotic	6.0	-		[33]
Lactarius quietus	Not edible. Symbiotic	8×10^{-1}	-		[33]
Lactarius rufus	Not edible. Symbiotic	1.5	6×10^{-1}	7	[33, 36, 38, 40, 42, 43]

TABLE 8. AGGREGATED TRANSFER FACTORS TO MUSHROOMS FOR ^{137}Cs, $m^2.kg^{-1}$, DW (Cont.)

Mushroom species	Edibility and life mode	Caesium transfer factor ($m^2.kg^{-1}$ DW)			
	of mushrooms	GM(*)	Range		Reference
Lactarius theiogalus	Not edible. Symbiotic	11	-		[38]
Lactarius torminosus	Edible. Symbiotic	-	4×10^{-1}	8×10^{-1}	[37, 43]
Lactarius trivialis	Not edible. Symbiotic	-	1.3	1.8	[43, 45]
Leccinum sp.	Symbiotic	4×10^{-1}	5×10^{-3}	7×10^{-1}	[47, 49]
Leccinum aurantiacum	Edible. Symbiotic	2×10^{-2}	-		[33]
Leccinum rotundifoliae	Edible. Symbiotic	3×10^{-1}	-		[37]
Leccinum scabrum	Edible. Symbiotic	3×10^{-1}	8×10^{-4}	1.1	[33, 34, 37, 38, 41, 42, 45, 46]
Leccinum versipelle	Edible. Symbiotic	9×10^{-2}	7×10^{-2}	1.2×10^{-1}	[37, 43, 45]
Leucoagaricus leucothites or Lepiota naucina	Not edible. Humus saprophytic	1×10^{-1}	-		[35]
Macrolepiota procera	Edible. Humus saprophytic	8×10^{-3}	7×10^{-5}	4×10^{-2}	[34, 35, 39]
Macrolepiota rhacodes	Edible. Humus saprophytic	3×10^{-3}	3×10^{-4}	1×10^{-2}	[34]
Lepista nuda	Edible. Litter saprophytic	1×10^{-2}	2.5×10^{-4}	1×10^{-1}	[33, 34, 38]
Lepista saeva	Edible. Litter saprophytic	1×10^{-2}	-		[33]
Lycoperdon perlatum	Edible. Humus saprophytic	-	3×10^{-3}	7×10^{-2}	[35, 39]
Oudemansiella sp.		1×10^{-1}	-		[39]
Oudemansiella radicata	Not edible.	2×10^{-1}	-		[39]
Paxillus atrotomentosus	Not edible. Xylophyte saprophytic	1×10^{-1}	-		[33]
Paxillus involutus	Not edible. Symbiotic/Humus saprophytic	2.1	6×10^{-1}	13	[33, 36, 38, 39, 40, 42, 46, 50]
Rozites caperatus	Edible. Symbiotic	2.3	4×10^{-1}	8	[33, 36, 37, 45, 47, 48, 49]
Russula sp.	Symbiotic	6×10^{-1}	3×10^{-2}	4.2	[39, 41, 46, 47, 49]
Russula badia	Not edible. Symbiotic	-	2	3	[36]
Russula decolorans	Not edible. Symbiotic	4×10^{-1}	-		[44]
Russula erythropoda	Edible. Symbiotic	-	2	3	[36]
Russula nigricans	Not edible. Symbiotic	7×10^{-1}	-		[38]
Russula ochroleuca	Not edible. Symbiotic	-	1.4	2	[36, 38]
Russula turci	Not edible. Symbiotic	-	8×10^{-1}	1.3	[36]
Sarcodon imbricatum	Edible. Symbiotic	3×10^{-2}	-		[35]
Suillus bovinus	Not edible. Symbiotic	-	2	3	[36]
Suillus elegans or S. grevillei	Edible. Symbiotic	4×10^{-1}	7×10^{-2}	9×10^{-1}	[33, 35, 42]

TABLE 8. AGGREGATED TRANSFER FACTORS TO MUSHROOMS FOR ^{137}Cs, $m^2.kg^{-1}$, DW (Cont)

Mushroom species	Edibility and life mode of mushrooms	Caesium transfer factor ($m^2.kg^{-1}$ DW)			
		GM(1)	Min.	Max.	Reference
Suillus granulatus	Not edible. Symbiotic	-	1	2	[36]
Suillus luteus	Edible. Symbiotic	-	8×10^{-1}	1.4	[36, 41]
Suillus variegatus	Edible. Symbiotic	9×10^{-1}	5×10^{-1}	3	[36, 43, 44]
Tricholoma album	Not edible. Symbiotic	7	-		[47]
Tricholoma auratum	Not edible. Symbiotic	-	4×10^{-1}	6×10^{-1}	[36]
Tricholoma fulvum	Not edible. Symbiotic	2	-		[47]
Tricholoma imbricatum	Not edible. Symbiotic	1×10^{-1}	-		[47]
Tricholoma pessundatum	Not edible. Symbiotic	1×10^{-1}	-		[47]
Tricholoma saponaceum	Not edible. Symbiotic	1×10^{-1}	-		[39, 47]
Trichomolopsis rutilans	Not edible. Xylophyte saprophytic	-	7×10^{-1}	1.0	[36]
Tylopilus felleus	Not edible. Symbiotic	2.5	8×10^{-1}	8	[33, 36, 38, 42]
Xerocomus badius	Edible. Symbiotic	1.3	2×10^{-3}	7	[33, 34, 36, 38, 39, 40, 41, 42, 45, 48, 50, 51]
Xerocomus chrysenteron	Edible. Symbiotic	1.4	3×10^{-1}	5	[33, 38, 39, 40, 42, 45, 51]
Xerocomus subtomentosus	Edible. Symbiotic	4×10^{-1}	2×10^{-1}	1.8	[37, 38, 40]

[1] In case of only one value, it is given as a best estimate, if two values, a range is presented, in case of three values or more, a geometric mean is calculated.

TABLE 9. AGGREGATED TRANSFER FACTORS TO MUSHROOMS FOR Pu, m^2 kg^{-1}, DW [53]

Mushroom species	Edibility and life mode of mushrooms	$^{239+240}$Pu transfer factor			
		N	AM	Min.	Max
Armillaria mellea	Edible. Parasitic/Xylophyte saprophytic	1	9×10^{-5}	-	
Boletus edulis	Edible. Symbiotic	4	3×10^{-4}	1.4×10^{-4}	4.5×10^{-4}
Cantharellus cibarius	Edible. Symbiotic	1	2×10^{-2}	-	
Lactarius vellereus	Not edible. Symbiotic	3	3×10^{-3}	2.7×10^{-3}	4×10^{-3}
Macrolepiota procera	Edible. Humus saprophytic	2	4×10^{-4}	3.2×10^{-4}	5.7×10^{-4}
Paxillus involutus	Not edible. Symbiotic/Humus saprophytic	1	8×10^{-4}	-	
Suillus luteus	Edible. Symbiotic	1	9×10^{-4}	-	
Xerocomus badius	Edible. Symbiotic	6	1×10^{-3}	8×10^{-5}	3.8×10^{-2}

TABLE 10. TRANSFER FACTORS FROM FOREST SOIL TO MUSHROOMS FOR NATURALLY OCCURING RADIONUCLIDES (Bq kg^{-1})/ (Bq kg^{-1}), DW [54]

Mushroom species[1]	^{234}U		^{238}U		^{228}Th		^{230}Th		^{232}Th		^{226}Ra	
	AM	SD	AM	SD	AM	SD	AM	SD	AM	SD	AM	SD
Amanita curtipes (M, NE)											7×10^{-1}	4×10^{-1}
Amanita muscaria (M, NE)	3.6×10^{-1}	5×10^{-2}	3.3×10^{-1}	5×10^{-2}	2.3×10^{-1}	4×10^{-2}	1.2×10^{-1}	3×10^{-2}	3.2×10^{-1}	5×10^{-2}	1.1	1.0
Hebeloma cylindrosporum (M, NE)	4.9×10^{-1}	6×10^{-2}	4.6×10^{-1}	5×10^{-2}	3.8×10^{-1}	2×10^{-2}	2.2×10^{-1}	2×10^{-2}	6.2×10^{-1}	4×10^{-2}	2.2×10^{-1}	1×10^{-1}
Tricholoma equestre (M, E)	1.4×10^{-1}	1×10^{-2}	1.3×10^{-1}	1×10^{-2}	1.4×10^{-1}	1×10^{-2}	5×10^{-2}	3×10^{-2}	1.3×10^{-1}	1×10^{-2}	2.5×10^{-2}	7×10^{-3}
Tricholoma pessandatum (M, NE)	2.4×10^{-1}	3×10^{-2}	2.3×10^{-1}	3×10^{-2}	1.5×10^{-1}	2×10^{-2}	8×10^{-2}	1×10^{-2}	2.2×10^{-1}	2×10^{-2}		
Tricholoma terreum (M, E)	2.4×10^{-1}	6×10^{-2}	2.7×10^{-1}	7×10^{-2}	3.9×10^{-1}	1×10^{-2}	1.9×10^{-1}	3×10^{-2}	4.7×10^{-1}	7×10^{-2}		
Lactarius deliciosus (M, E)	6×10^{-2}	2×10^{-2}	6×10^{-2}	2×10^{-2}	3×10^{-2}	4×10^{-3}	3×10^{-2}	2×10^{-2}	3×10^{-2}	1×10^{-2}		
Rhizopogon roseoulus (M, E)	4×10^{-2}	1×10^{-2}	4×10^{-2}	1×10^{-2}	3×10^{-2}	1×10^{-2}	3×10^{-2}	2×10^{-2}	3×10^{-2}	2×10^{-2}		
Russula cessans (M, NE)	3.4×10^{-1}	7×10^{-2}	3.1×10^{-1}	6×10^{-2}	1.8×10^{-1}	4×10^{-2}	2.1×10^{-1}	5×10^{-2}	1.7×10^{-1}	4×10^{-2}	1.4	9×10^{-1}

[1] E=edible, NE=not edible, (P) = parasitic , (S) = saprophytic, (M) = mycorrhizal or symbiotic.

TABLE 11. TRANSFER FACTORS FROM FOREST SOIL TO MUSHROOMS FOR SOME ARTIFICIAL RADIONUCLIDES (Bq kg^{-1})/ (Bq kg^{-1}), DW [55, 56]

Mushroom species[1]	^{90}Sr		^{137}Cs		$^{239/240}$Pu		^{241}Am	
	AM	SD	AM	SD	AM	SD	AM	SD
Amanita muscaria (NE, M)	3.9×10^{-1}	1.1×10^{-1}	4.3×10^{-1}	5×10^{-2}	5.1×10^{-1}	8×10^{-2}	1.3×10^{-1}	3×10^{-2}
Clitocybe sp.	3.6	5×10^{-1}	4.0	6×10^{-1}	2.8×10^{-1}	4×10^{-2}	6×10^{-1}	1.6×10^{-1}
Hebeloma cylindrosporum (NE, M)	1.5	1.9×10^{-1}	5.5×10^{1}	5	5.9×10^{-1}	7×10^{-2}	5.8×10^{-1}	1.1×10^{-1}
Lactarius deliciosus (E, M)	4.2×10^{-1}	1.1×10^{-1}	1.7	1.3×10^{-1}	1.6×10^{-1}	2×10^{-2}	1.4×10^{-1}	3×10^{-2}
Lycoperdon perlatum(E, HS)	5.3×10^{-1}	0.09	8.1×10^{-1}	2×10^{-1}	2.8×10^{-1}	4×10^{-2}	5.4×10^{-1}	1.2×10^{-1}
Rhizopogon roseoulus (E, M)	1.9	3×10^{-1}	1.31		1.1×10^{-1}	2×10^{-2}	1.8×10^{-1}	5×10^{-2}

[1] E = edible, NE = not edible, (HS) = humus saprophytic, (M) = mycorrhizal or symbiotic.

It is noted that information on transfer factors to mushrooms is also available for some artificial radionuclides giving the opportunity for a screening evaluation when only information on radionuclide activity concentrations in forest soil is available (Table 11).

Mycelia of some mushroom species exploit specific soil layers for their nutrition and the dynamics of contamination of such species has been related to the contamination of these specific layers [46, 57]. Contamination of wood degrading mushrooms has been related to radionuclide activity concentrations in 'living wood'. These distinctions in nutrient substrate imply different patterns of contamination of different mushrooms species after a deposition. To illustrate the above affect, some examples for contamination of selected mushroom species by ^{137}Cs sampled in a pine forest located in the Zhytomyr region (Ukraine) at a distance of around 137 km SW from the ChNPP are shown in Fig.4

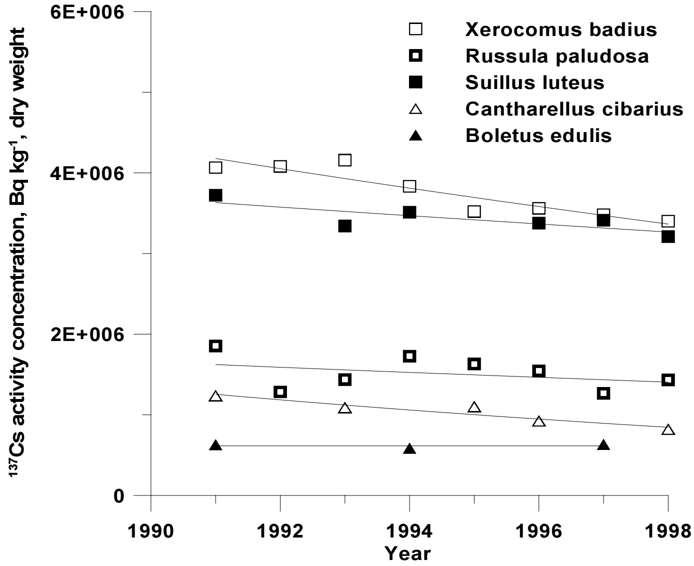

FIG. 4. Cs-137 activity concentrations (Bq/kg DW) in selected mushroom species. Cs-137 soil deposition at the site in 1986 was 555 kBq m^{-2} [58].

Changes with time in the contamination of mushrooms reflect the bioavailability of ^{137}Cs in the various relevant nutrient sources utilised by different species and Fig 4 indicates the tendency for a slow decrease in mushroom contamination during the 1990s and associated ecological half-lives can be ranked as follows: *Cantharellus cibarius ~ Xerocomus badius (22.4 y) < Russula paludosa (34 y) < Suillus luteus (46.2 y)< Boletus edulis (~800 y).*

2.2 Transfer to berries

As with mushrooms, berries are an important natural food collected in forest ecosystems. Uptake of radiocaesium by forest berries is high in comparison with foodstuffs grown in agricultural systems. Aggregated transfer factors (dry weight) of around 1 to 10 × 10^{-2} m^2.kg^{-1} were typically observed for various species of forest berries [59]. T_{ag} values for radiocaesium in different berry species have been reviewed [60, 61]. A summary of T_{ag} values collated during the course of this review is presented by individual species in Table 12. The different species appear to fall into one of the three categories on the basis of their T_{ag} values (although there is considerable within-species variability):

- Predominantly non-forest berries – *F. vesca*, *R. idaeus* and *R. fruticosus* – with a mean T_{ag} value of 7.5×10^{-3} $m^2.kg^{-1}$ (DW)
- Berries that grow in wet (and forest) ecosystems – *R. chamaemorus* and *V. oxycoccus* – with a mean T_{ag} value of 1.0×10^{-1} $m^2.kg^{-1}$ (DW)
- Other *Vaccinium* species growing within forests – *V. myrtillus* and *V. vitis-idaea* – with a mean T_{ag} value of 5.5×10^{-2} $m^2.kg^{-1}$ (DW)

After the Chernobyl accident, the evolution of the ^{137}Cs content in all plant organs of all berry species shows a clear decreasing trend, as seen in Figures 5 and 6.

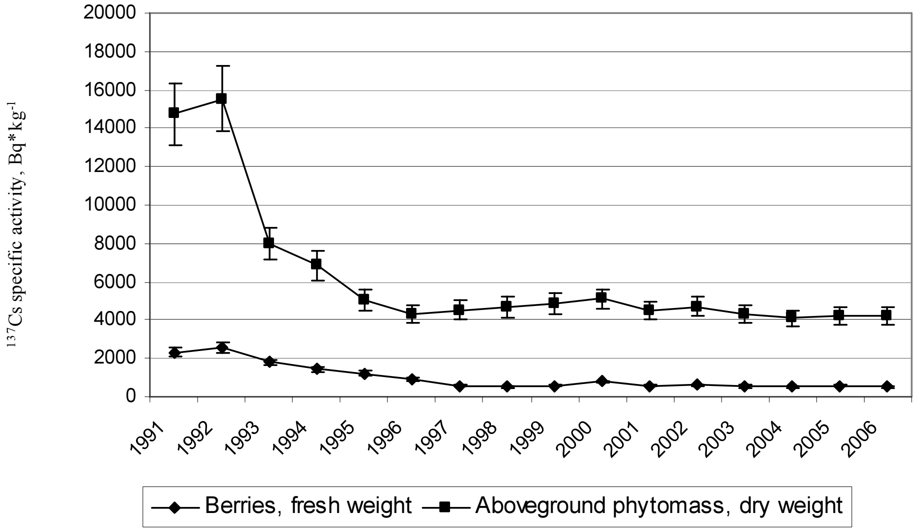

FIG. 5. Multiyear dynamics of ^{137}Cs activity concentrations in cranberry phytomass on an experimental plot in an automorphic landscape (density of ^{137}Cs ground deposition was 75 kBq m^{-2} in 1991) [62].

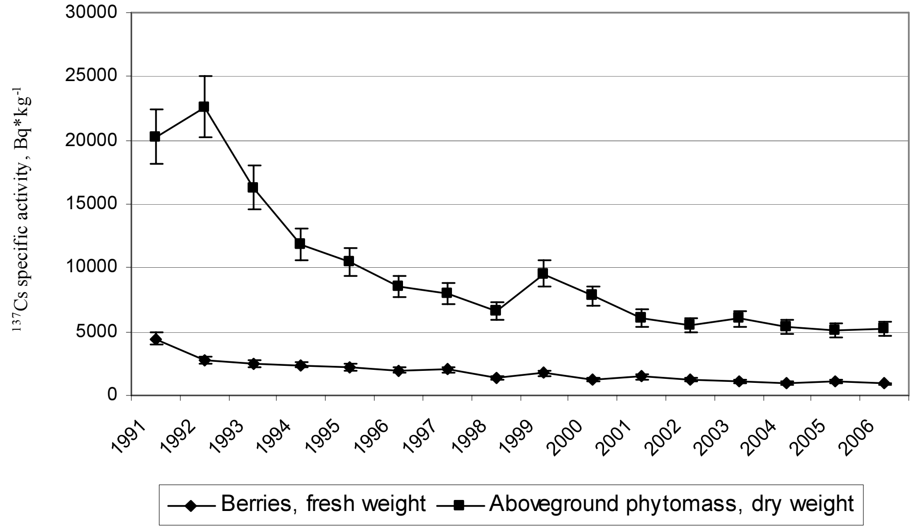

FIG. 6. Multiyear dynamics of ^{137}Cs activity concentrations in bilberry phytomass on an experimental plot in a hydromorphic landscape (density of ^{137}Cs ground deposition was 250 kBq m^{-2} in 1991) [62].

TABLE 12. AGGREGATED TRANSFER FACTOR FOR Cs IN BERRIES, $m^2 kg^{-1}$, DW [60]

Berries	N	AM	Min	Max
Bilberry (*Vaccinium myrtillus*)	952	5×10^{-2}	2×10^{-3}	3×10^{-1}
Cowberry (*Vaccinium vitis-idaea*)	170	3×10^{-2}	5×10^{-3}	1×10^{-1}
Cranberry (*Vaccinium oxycoccus*)	65	1.2×10^{-1}	3×10^{-3}	2×10^{-1}
Cloud berry (*Rubus chamaemorus*)	45	1×10^{-1}	8×10^{-3}	1.5×10^{-1}
Raspberry (*Rubus idaeus*)	241	3×10^{-2}	5×10^{-3}	1×10^{-1}
Blackberry (*Rubus fruticosus*)	686	2×10^{-2}	5×10^{-3}	7×10^{-2}
Wild strawberry (*Fragaria vesca*)	466	4×10^{-3}	2×10^{-3}	7×10^{-3}

Relatively fast decrease of the ^{137}Cs specific activity in the aboveground biomass, as well as in berries of the main berry species, in the initial period after deposition (1986-1995) ; and then a very slow decrease in the more recent period (after 1995) was recorded based on long-term monitoring data after the Chernobyl accident.. These data allow the derivation of effective half-life values, which are shown in Table 13.

As it can be seen, for the period from 1991 to 2006, an effective half-life ($T_{1/2}^{eff}$.) for ^{137}Cs is close to 10 years for most of the berry species. For bilberry and cowberry, lower values of $T_{1/2}^{eff}$ are typical in automorphic landscapes in comparison with semi-hydromorphic ones. For cranberry, $T_{1/2}^{eff}$ is higher in semi-hydromorphic landscapes in comparison with hydromorphic ones. Data on ^{90}Sr transfer to berries in areas affected by the Chernobyl accident are much scarcer than those for ^{137}Cs. The only available information was given by Ipatyev [4] for 1992-1993, 1999 who reported $T_{1/2}^{eff}$ values of $(7.1\pm4.1) \times 10^{-3}$ $m^2 kg^{-1}$ and $(9.2\pm3.0) \times 10^{-2}$ $m^2 kg^{-1}$ for bilberry and wild strawberry, respectively.

Various other radionuclides (^{60}Co, ^{106}Ru, ^{125}Sb, ^{137}Cs, ^{144}Ce, ^{154}Eu, ^{239}Pu) have been measured in different types of understorey vegetation by Lux [64]. Derived transfer factors show that mean degree of bioaccumulation for most radionuclides is at least a factor of ten less than that observed for ^{137}Cs, except for ^{90}Sr, for which the bioaccumulation was close to that of ^{137}Cs. Concerning plutonium, it seems that measurements could have been influenced by adsorption of soil particles to plant surfaces. Consequently, no firm conclusions could be drawn for this element [59].

The data of Table 12 are given for dry weight. As data for dose assessments they are often required in fresh weight, and Table 14 gives the dry matter content of some berries [65], so that appropriate conversions can be made.

4. TRANSFER TO GAME

The radionuclide concentration in animal meat depends strongly on the feeding habits of the animal. Variability in game contamination can have three main origins:

- Heterogeneous deposition causes variability in feedstuff contamination;
- Dietary composition and feeding behaviour differs between game species;
- Seasonal variations in diet and/or feeding behaviour can be identified for some species of game (e.g. roe deer, wild boar, reindeer (cf. section about Arctic environment).

TABLE 13. EFFECTIVE HALF-LIVES ($T_{1/2}^{eff}$, YEARS) OF ^{137}CS IN BERRIES OF THE MAIN BERRY SPECIES OF BOREAL FORESTS IN UKRAINE IN THE YEARS FROM 1991 TO 2006 [62, 63]

Berries	Type of forest landscape	AM	$T_{1/2}^{eff}$	
			Min.	Max.
Bilberry (*Vaccinium myrtillus*)	automorphic	7.5	5.4	8.5
	semi-hydromorphic	12.6	10.8	14.6
Cowberry (*Vaccinium vitis-idaea*)	automorphic	7.1	4.7	9.8
	semi-hydromorphic	11.4	11.1	11.7
Cranberry (*Oxycoccus palustris*)	semi-hydromorphic	7.7	6.2	10.1
	hydromorphic	5.5	4.6	6.6

TABLE 14. DRY MATTER CONTENT OF SOME WILD BERRIES, %

English name	Latin name	N	AM	SD	Max	Min
Bilberry	*Vaccinium myrtillus*	307	13.2	1.1×10^{-1}	21	8.6
Lingonberry	*Vaccinium vitis-idaea*	254	14.1	7.8×10^{-2}	18.8	11.3
Cranberry	*Vaccinium oxycoccus*	16	10.8	2.3×10^{-1}	12.1	9.3
Bog bilberry	*Vaccinium uliginosum*	6	12.1	4.5×10^{-1}	13.5	10.5
Black crowberry	*Empetrum nigrum*	1	7.4		-	-
Cloudberry	*Rubus chamaemorus*	26	14.0	3.2×10^{-1}	18	9
Wild raspberry	*Rubus idaeus*	21	17.3	4.0×10^{-1}	21.9	14.4
Wild strawberry	*Fragaria vesca*	1	15.4		-	-

This variability is emphasized for caesium because of the great variability in its transfers to plants and mushrooms. With radionuclides other than caesium, such large variations have not been reported.

As to modelling transfers to game, the variability originating from various factors for caesium, means that the use of a single aggregated transfer factor may substantially under- or over- estimate game contamination at some periods and in some ecological contexts. In order to address the issue of temporal variability, two aggregated transfer factors specific to two different feeding seasons can be adopted. The use of a more elaborate model, taking into account a time-varying diet with a limited number of typical feedstuffs, allows more accurate estimation of the contamination of each species of game. This improvement depends, of course, on a suitable knowledge of the feeding habits of game (i.e. rates of consumption of each feedstuff and their average radionuclide concentrations).

Early reports described relationships between radionuclide concentrations or amounts in precipitation, soil and animals [66, 67] and their intake by humans [68]. A review of studies conducted before and after the Chernobyl accident on the transfer of radiocaesium to ruminants provides an overview of such information for that radionuclide [69]. As background, a comparative review of the digestive system of ruminants is given in [70]. Parameter values for radionuclide transfers from soil to game are collected in [45, 69]. The biological transport of radiocaesium in invertebrates and small (non-game) mammals is covered by [71, 72].

Once an animal is no longer eating a contaminated diet, the biological half-life T_{bio} is the time required for the radionuclide content in an animal (either in the whole body or in a specific organ or tissue) to be reduced by one half, excluding the effects of radioactive decay.

Particularly in the first year after the deposition, the feed of herbivorous game animals may contain plant material contaminated directly from the initial deposition. The aggregated transfer factors for game animals compiled here are sometimes derived from samples of animal tissues taken in years of non-negligible atmospheric deposition from an accident or from nuclear weapon tests.

4.1 Transfer to humans: assessment of radionuclide intake through game meat and offal

For assessment of ingestion doses received by people through meat and other tissues of game animals, concentrations of radionuclides in edible parts of carcasses are needed. Regionally, the amounts of consumed game meat can be derived from the annual game bag, when edible fractions of live or carcass weights are available.

The annual game bag is related to the density of animal populations; which tends to have annual fluctuations and substantial regional variations. Part of the annual variation is due to the licensing of big game hunting, where the objectives of game management are considered. Annual hunting statistics rather than population densities, not published annually, are applicable for assessments for human intake of game meat. Individual or *per capita* doses can also be based on a regional consumption survey.

The intensity of hunting varies in different countries, as well as in different regions of a single country and with time. Such variations in intensity have an effect on the degree of radionuclide transfer to humans after radionuclide contamination of the animals. The numbers of some game animals harvested per year in some European countries and Canada are listed in Table 15. [73-80]

Information on various game animal species regarding description, classification, feed habits, geographic distribution etc. is found for instance on Animal Diversity Web, an online database of the University of Michigan[1].

In Central Europe, the mean weight (eviscerated animal with head) of roe deer is 12.5 kg, red deer 65 kg, wild boar 41 kg and chamois 15 kg [77, 81]. This weight has to be multiplied by about a factor 0.6 to get approximately the weight of the meat to be consumed. Also, often used is carcass weight, defined for various groups of animals. For certain big mammals it includes weight of the slaughtered animal's cold body after having been bled, skinned and eviscerated, and after removal of the head, tail, udders and part of the limbs. Kidneys, kidney fats and butcher's fat are included in the carcass. Slaughter grease is not [81]. The fraction of edible meat included in carcass weight can be derived by subtraction of the weight of bones and other not edible parts from carcass weights (Table 16) [82-84].

Tissues other than muscle, such as liver, kidney and heart, and also blood, have traditionally been used for human food by for instance reindeer-herding populations in Fennoscandia [80], aboriginals of northern America [80] and hunters elsewhere [68]. Currently, the contamination by toxic heavy metals, particularly cadmium, restricts the use of liver and kidney of big game animals for food in some regions.

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[1] Conditions of use, copyright owners and recommendations for citing are given in http://animaldiversity.ummz.umich.edu/site/about/overview.html.

TABLE 15. NUMBER OF GAME ANIMALS HARVESTED PER YEAR[1]

Country	Game animal	Year	Killed per year	Reference
Sweden	Roe deer	1986/95	1.2×10^5-3.8×10^5	[73, 74]
	Roe deer	2004	1.5×10^5	
	Moose	2004	9.7×10^4	
	Red deer	2004	2.8×10^3	
	Fallow deer	2004	1.7×10^4	
	Wild boar	2004	1.9×10^4	
	Arctic hare	2004	3.5×10^4	
	Brown hare	2004	5.6×10^4	
	Goldeneye	2004	9×10^3	
	Pheasant	2004	2.8×10^4	
	Black grouse	2004	2.6×10^4	
	Wood pigeon	2004	8.1×10^4	
Norway	Moose	2004	3.7×10^4	[75]
	Red deer	2004	2.6×10^4	
	Roe deer	2003	2.9×10^4	
	Wild reindeer	2004	4×10^3	
	Hare	2003/04	2.9×10^4	
	Grouse	2003/04	4.4×10^5	
	Wood pigeon	2003/04	5.9×10^4	
	Ducks	2003/04	5.7×10^4	
	Geese	2003/04	1.5×10^4	
Finland	Moose	2001/05	$(6.7\text{-}7.6) \times 10^4$	[76]
	White-tailed deer	2001/05	$(1.7\text{-}2.3) \times 10^4$	
	Roe deer	2001/04	$(1.1\text{-}3.4) \times 10^3$	
	Brown hare	2001/02	$(6.2\text{-}7.2) \times 10^4$	
	Arctic hare	2001/02	$(1.9\text{-}2.0) \times 10^5$	
	Grouse	2001/02	$(3.0\text{-}3.2) \times 10^5$	
	Waterfowl	2001/02	$(4.4\text{-}5.5) \times 10^5$	
	Wood pigeon	2001/02	1.4×10^5	
	Pheasant	2001/02	$(2.4\text{-}3.1) \times 10^4$	
Germany	Roe deer	1987/88	8.8×10^5	[77]
		2003/2004	8.9×10^5	
	Red deer		6.0×10^4	
	Wild boar		4.6×10^5	
	Chamois		5.9×10^3	
Baden-Württemberg	Roe deer	1987/88	1.5×10^5	[78]
Czech Republic	Roe deer and deer	1992	$>3.0 \times 10^{6\,(1)}$	[40]
Austria	Roe deer	2004/2005	2.8×10^5	[79]
	Red deer		4.9×10^4	
	Wild boar		3.3×10^4	
	Chamois		2.5×10^4	
Canada	Caribou	1986	$\approx 10^5$	[80][a]

[1] Kilograms.

TABLE 16. CARCASS WEIGHT AND MEAT FRACTION FOR GAME ANIMALS

Species of animal	Carcass weight, kg	Fraction of meat in carcass weight
Moose, adult	1.9×10^2	0.80
Moose, calf	8.3×10^1	0.78
White-tailed deer	5.0×10^1	0.78
Fallow deer	3.3×10^1	0.78
Roe deer	1.8×10^{1}[a]	0.78
Brown hare	2.4	0.90
Arctic hare	1.8	0.90
Capercaillie	1.9	0.90
Black grouse	6.6×10^{-1}	0.90
Hazel grouse	2.4×10^{-1}	0.90
Willow grouse	3.6×10^{-1}	0.90
Partridge	2.4×10^{-1}	0.90
Pheasant	6.9×10^{-1}	0.90
Goose	2.3	0.90
Eider	1.3	0.90
Long-tailed duck	3.8×10^{-1}	0.90
Mallard	6.6×10^{-1}	0.90
Goldeneye	4.5×10^{-1}	0.90
Teal	1.8×10^{-1}	0.90

[a] Roe deer gains more weight in Northern than in Central Europe.

Wild terrestrial food chains provide information on the uptake and distribution of radionuclides in large mammals. Various soft tissues and bone of deer have been extensively studied for their radionuclide contents in North America during the nuclear fallout era. Concentrations of ^{137}Cs in white-tailed deer decreased in the order: tongue or body muscle and kidney ≥ heart and spleen ≥ liver [85]. Liver contained on average fifty per cent of the ^{137}Cs concentration found in the body muscle of caribou, whereas kidney, heart and muscle did not differ substantially in a compilation of long-term data [80]. Soft tissues also concentrate ^{22}Na, ^{55}Fe, ^{65}Zn, ^{110m}Ag and ^{210}Po. In contrast bone is the main site of deposition and accumulation of ^{54}Mn, ^{90}Sr, ^{210}Pb and ^{226}Ra [86]. ^{90}Sr has been analysed in bones, and ^{144}Ce, ^{137}Cs, ^{54}Mn and ^{106}Ru in liver of mule deer by [68]. Plutonium ($^{239,\ 240}$Pu, ^{238}Pu) concentrations in soft tissues and bone of white-tailed deer decreased in the order: bone ≥ liver ≥ lungs ≥ muscle [87].

Seasonal changes in bone, muscle and rumen contents of ^{210}Po, ^{210}Pb, ^{226}Ra and ^{137}Cs in Alaskan reindeer and caribou were studied by [88]. They found variations in muscle concentrations of ^{210}Po with season; a fourfold increase from fall to spring, which is similar to the seasonal variation of ^{137}Cs concentrations. Concentrations of ^{226}Ra in bone were lower than concentrations of other radionuclides studied, and muscle contents were hardly detectable. The results clearly showed the contribution of lichen to both rumen and body content of ^{137}Cs, ^{210}Pb and ^{210}Po, whereas most of the ^{226}Ra originated in vascular plants. The radionuclide content and composition of feed were compared with ^{90}Sr in bone and ^{137}Cs in muscle of reindeer, caribou and moose by [67]. They analysed also rumen content and showed the significance of availability of lichen and the use of old growth of sedges as winter feed for ^{90}Sr and ^{137}Cs concentrations in animals. They concluded that absence of lichen and possibly plant litter in the diet of moose are likely reasons for the lower ^{90}Sr and ^{137}Cs burdens.

4.2 Transfer to roe deer (Capreolus capreolus)

Roe deer consume a wide variety of herbs, grasses and also fungi when they are available. Radiocaesium levels in roe deer peak in August and September when fungi are abundant. The population density is 25–50 animals per km² in Central Europe and, because of management of hunting, 0.03 animals per km² in Ukraine, Zhitomir region [58, 89, 90].

The spectrum of dietary plants and the seasonality of the diet are described in [91, 92, 93, 94]. Herbs form 80% of the food in spring and summer, and about 65 % in winter. Leaf buds, and small twigs of trees and shrubs form 11-12 % in spring and early summer and 23 % in autumn and winter. Fungi and lichens are found in analyses of rumen content–about 1.4 % dry mass as the highest fraction in autumn. No differences were found in the quality of food consumed by young and adult animals [95]. In 114 rumen of roe deer, an average fraction of fungi of 3.3 % was found for the period June to December, and a maximum of 15 % for October [93]. In some individual roe deer [91], the fraction of fungi in the rumen was found to be up to 76 % of the total content.

Fern, blackberry, bilberry, and raspberry are relevant as grazing plants of roe deer [93]. Intakes by roe deer were found to be 350 g dry mass intake per day in winter and up to 600 g dry mass intake per day in summer [92, 96]. ^{137}Cs activity concentrations and seasonality of intakes are described in [90, 97 – 108]. There is quantitative evidence that the observed increase in summer and autumn of the ^{137}Cs activity concentration in roe deer meat is due to higher rates of ingestion of mushrooms during this period of the year [109, 110, 111].

The availability of mushrooms can depend on the amount of rain just before the mushroom season [90, 100, 112]. Data concerning the time-dependency of T_{ag} are given in Table 17. All T_{ag} values are calculated with respect to fresh mass of game meat.

Table 18 gives the ecological half-lives for samples from forest ecosystems in Germany and Austria. The ecological half-lives given were calculated for time series of aggregated transfer factors (ratio of the activity concentration [Bq kg^{-1}] in a product and the total radionuclide deposition [Bq m^{-2}] of the area where the product has been sampled).

As can be seen for Ochsenhausen, the time-dependence of T_{ag} during the whole year can be described using 1 exponential function with $T_{1/2}^{eco}$ = (5.7 ± 0.13) years. In the case of roe deer in Pfrunger Ried, the time-dependence of T_{ag} during the whole year can be described by 1 exponential function with $T_{1/2}^{eco}$ = (13 ± 1.8) years [108, 116].

For comparison, the other ecological half-lives were derived from data sets giving activity concentrations in the samples. Table 18 also contains data sets from the Ukraine. Although there is considerable variability and from some of the data sets ecological half-lives ($T_{1/2}^{eco}$) could be derived with large statistical uncertainties only, some general trends are apparent from Table 18.

Ecological half-lives of roe deer meat vary between 6 and 13 years. It can be seen from Table 18 that soil properties provide a significant influence (e.g. organic matter fraction in peat or spruce forest), and that with increasing length of the time interval of observation $T_{1/2}^{eco}$ values are larger. This is an indication that different processes determine the availability of ^{137}Cs for roe deer as compared with those processes acting soon after the start of the contamination in 1986 (e.g. fixation, migration into the rooting zone).Thus, use of a single exponential may be an over-simplification in long-term assessment studies.

TABLE 17. AGGREGATED TRANSFER FACTOR VALUES FOR ^{137}Cs TO MEAT OF ROE DEER (m² kg^{-1} FW)

Site	Years after deposition	Period, month	N	GM	GSD	Site description	Reference
General				5.0×10^{-2}			[45]
Harbo/Sweden	3	1-6	35	2.6×10^{-2}		Coniferous forest; mean deposition 35 – 40 kBq/m² 1.05.1986	[109]
	4		59	2.8×10^{-2}			
	5		65	1.8×10^{-2}			
	6		49	2.0×10^{-2}			
	7		35	3.0×10^{-2}			
	8		36	2.3×10^{-2}			
Ochsenhausen/ Germany	2	1-6	36	1.7×10^{-2}	1.9	Spruce forest; mean deposition 39 kBq/m² 1.05.1986	[108, 112, 113]
	3		57	1.7×10^{-2}	1.8		
	4		48	1.0×10^{-2}	2.0		
	5		59	7.6×10^{-3}	2.4		
	6		55	8.2×10^{-3}	1.9		
	7		59	5.7×10^{-3}	1.9		
	8		63	5.2×10^{-3}	1.9		
	9		53	4.8×10^{-3}	2.0		
Pfrunger Ried Germany	2	1-6	6	8.1×10^{-2}	1.1	Peat bog, pine; mean deposition 22 kBq/m² 1.05.1986	
	3		20	2.3×10^{-2}	3.5		
	4		16	1.6×10^{-2}	3.2		
	5		29	2.5×10^{-2}	3.9		
	6		14	2.4×10^{-2}	3.4		
	7		14	4.0×10^{-2}	2.1		
	8		10	4.0×10^{-2}	1.6		
	9		15	2.3×10^{-2}	3.9		
	10		17	2.5×10^{-2}	3.0		
	11		17	2.9×10^{-2}	2.1		
	12		25	2.2×10^{-2}	2.2		
	13		23	2.6×10^{-2}	1.9		
	14		31	2.0×10^{-2}	2.8		
	15		40	1.2×10^{-2}	3.0		
	16		32	1.8×10^{-2}	3.1		
	17		25	2.7×10^{-2}	2.0		
	18		25	2.3×10^{-2}	1.8		
Bodenmais. Germany	3/4			2.7×10^{-2}		Spruce, mean deposition 98 kBq/m², 1.05.1986	[114]
Sumava Czech Rep.	0/5			2.7×10^{-2}		Spruce, forest. mean deposition 10.5 kBq/m², June 1986	[40]
Temelin Czech Rep.	0/5			3.8×10^{-2}		Spruce/agricultural land, mean deposition 3 kBq/m², 1991/93	[40]
Weinsberger Forest Austria	0	1-12	5	4.6×10^{-2}	1.8	Spruce forest, mean deposition (median) 52.2 kBq/m² at n = 218 sites, 1.05.1986	[115]
	1		9	3.9×10^{-2}	2.4		
	2		31	4.0×10^{-2}	2.4		
	3		44	2.7×10^{-2}	2.7		
	4		48	2.3×10^{-2}	2.3		
	5		177	2.0×10^{-2}	2.5		
	6		143	1.2×10^{-2}	2.5		
	7		87	1.6×10^{-2}	2.2		
	8		91	1.8×10^{-2}	2.7		
	9		17	2.0×10^{-2}	2.4		
	10		25	1.3×10^{-2}	1.7		
	11		25	1.7×10^{-2}	2.0		
	12		19	9×10^{-3}	2.4		
	13		12	1.0×10^{-2}	1.6		
	14		11	1.1×10^{-2}	1.7		
	15		27	1.4×10^{-2}	2.4		
	16		7	1.1×10^{-2}	1.8		
	17		12	8×10^{-3}	2.1		

TABLE 17. AGGREGATED TRANSFER FACTOR VALUES FOR ^{137}Cs TO MEAT OF ROE DEER (m² kg^{-1} FW)

Site	Years after deposition	Period, month	N	GM	GSD	Site description	Reference
Kobernhauser Forest. Austria	0		1	9.4×10^{-2}		Spruce Forest, mean deposition (median) 48.5 kBq/m² at n = 20 sites, 1.05.1986	
	1		8	6.0×10^{-2}	2.1		
	2		20	5.3×10^{-2}	2.3		
	3		13	4.4×10^{-2}	2.2		
	4		11	3.0×10^{-2}	1.9		
	5		10	3.6×10^{-2}	2.0		
	6		9	2.2×10^{-2}	2.1		
	7		2	4.9×10^{-2}	1.1		
	8		3	2.2×10^{-2}	1.4		
	9		7	3.1×10^{-2}	2.3		
	10		6	1.6×10^{-2}	2.3		
	11		1	1.9×10^{-2}			
	14		8	1.6×10^{-2}	1.6		

TABLE 18. ECOLOGICAL HALF-LIFES ($T_{1/2}^{eco}$) OF ^{137}Cs IN ROE DEER IN FOREST AND PEAT BOG, DERIVED FROM DATA FITTED BY A SINGLE EXPONENTIAL [116].

Medium	Ecosystem	N	$T_{1/2}^{eco}$, y		Site	Period	Reference
			AM	SD			
Roe deer, muscle, ^{137}Cs	Spruce forest	3501	5.7	1.3×10^{-1}	Ochsenhausen (D)	1987-2004	[108, 116]
Jan-Dec.,	Peat bog	897	$1.3. \times 10^{1}$	1.8	Pfrunger Ried (D)	1987-2004	
Roe deer, muscle, ^{137}Cs	Peat bog	784	1.2×10^{1}	1.8	Pfrunger, Ried, (D)	1989-2002	[107, 112, 113]
	Spruce forest	3309	5.7	2×10^{-1}	Ochsenhausen (D)		
		1429	6.3	4×10^{-1}	Bodenmais (D)	1987-2001	[93, 94, 114, 117]
		1505	8.5	6×10^{-1}		1987-2002	
		756	8.4	1.2	Weinsb.Wald (A)	1986-2001	[103-106, 115]
		722	5.8	7×10^{-1}	Eisenstein (D)	1987-1999	[118- 121]
		872	8.3	1.2	Zwiesel (D)		
	Forest/Fields	11613	1.2×10^{1}	9×10^{-1}	Bavaria (D)		
	Mixed		2.2×10^{1}			1986-1996	[119, 120]
	Spruce forest	1663	9			1987-2004	[117]
	Forest	5	1.3×10^{2}	1.2×10^{2a}	Ukr, Zhitomir	1991-1997	[58]

^{a}This value of $T_{1/2}^{eco}$ is quite uncertain, because $T_{1/2}^{eff}$ and T_r have similar values.

Radiocaesium concentrations in leaves of plants grazed by roe deer usually decrease with ecological half-lives of 1 to 5 years, but occasionally longer half-lives up to about 11 years have been derived [113, 116, 122]. However, only the longest ecological half-lives in grazed leaves correspond to the typical ecological half lives in the animals. As the biological half-lives in the animals are much shorter, this implies that a significant factor in maintaining radionuclide concentrations in animals is a component of diet with a longer ecological half-life than the leaves of grazed plants.

Biological half-lives ($T_{1/2}^{bio}$) of radiocaesium in roe deer are collected in Table 19, and values of 10 d <. $T_{1/2}^{bio} < 35$ d are identified. All mammals exhibit fast and slow components of retention. A typical value for the fast component is 1 day for all mammalian species, but typically relates to only about 10 percent of the retained activity. It was found that the inter-species variation in the half-life of the long term component of retention is determined by body mass, as discussed, for example, in [123].

TABLE 19. BIOLOGICAL HALF-LIFE OF RADIOCAESIUM IN ROE DEER

Game	Nuclide	N	$T_{1/2}^{bio}$ (d)		Reference
			AM	SD	
Roe deer	^{134}Cs	8	2.8×10^1	-	[124]
	^{137}Cs		1.2×10^1	-	[109]
			1.0×10^1	-	[114]
		108	2.2×10^1	2.2	[125]
			3.5×10^1	-	[104]

4.3. Transfer to red deer (Cervus elaphus)

Only few data exist concerning the time-dependence of the aggregated transfer factor T_{ag} with respect to ^{137}Cs in red deer. Ecological half-lives of ^{137}Cs and ^{90}Sr in red deer are given in Table 20, and the values are similar to those of the other deer and chamois. With respect to T_{ag} a value of 0.03 m^2/kg (FW) is given in [45]. Also, T_{ag} = 0.01 m²/kg (FW) during 1999-2000 in Bodenmais, Germany [93], T_{ag} = 0.028 m²/kg (FW) in 1988 in Austria (3 samples, GSD = 1.4 [115]) and T_{ag} = 0.05 m²/kg FW over the period 1986-1991 in the Czech Republic [40].

TABLE 20. ECOLOGICAL HALF-LIFE OF ^{137}Cs AND ^{90}Sr IN RED DEER IN FOREST AREAS, DERIVED FROM DATA FITTED BY A SINGLE EXPONENTIAL [113]

Medium	Ecosystem	N	$T_{1/2}^{eco}$ y		Site	Period	Reference
			AM	SD			
Red deer, muscle, ^{137}Cs	Forest/Fields	2619	6.1	4×10^{-1}	Bavaria (D)	1987-1999	[118 – 121]
		385	1.8×10^1			1986-1996	[119, 120]
	Spruce	205	5.4			1987-2004	[117]
Red deer, antlers; ^{90}Sr	Spruce	187	2.2×10^1	4	(A) **	1946-1990	[126]

** Langnau (CH), Trofaiach, Eisenkappel, Bludenz, Gerlos.

4.4. Transfer to chamois (Rupicapra rupicapra)

Values of T_{ag} for chamois are not reported in the literature. Ecological half-lives of ^{137}Cs in the muscle of chamois (7.7 ± 1.4) y, [119, 120]), comparable to the values for roe deer, are given in Table 21.

TABLE 21. ECOLOGICAL HALF-LIFE OF ^{137}CS IN THE MUSCLE OF CHAMOIS FROM FORESTED AREAS, DERIVED FROM DATA FITTED BY A SINGLE EXPONENTIAL [116]

Medium	Ecosystem	N	$T_{1/2}^{eco}$ (y)		Site	Period	Reference
			AM	SD			
Chamois, muscle, ^{137}Cs	Forest	1059	7.7	1.4	Bavaria (D)	1987-1999	[118-120]
	Spruce		2.6×10^1			1986-1996	[119, 120]

4.5 Transfer to reindeer and caribou

Domesticated reindeer (*Rangifer tarandus)* belongs to the deer family *(Cervidae)*. The live weight of reindeer varies; females over three years weigh around 60–100 kilos and males over three years as well as castrated males weigh about 90–180 kilos [127]. The size of reindeer stock varies by country (Table 22). The percentage of the surface area under reindeer herding ranges in Fennoscandian countries from 33% to 40% [128]. The annual per capita consumption of reindeer meat may be low, 0.2 kg in Sweden in the late 1990's [129], 0.4 kg in Norway in reindeer management year 2005/06 [127] and 0.5 kg per year in Finland in 2004 [130], whereas a minor population group based on reindeer husbandry may consume reindeer meat daily. In the Kola region, RU, the reindeer breeders consumed 0.3 kg reindeer meat per day in 1998-1999 [131].

TABLE 22. THE DOMESTICATED REINDEER STOCK IN SOME COUNTRIES [128]

Country	Size of the stock /year	Pasture area, km^2/per percent of the surface area of the country
Alaska	1.9×10^4/1999	
Finland	1.9×10^5/2000	1.1×10^5 /36%
Norway	1.7×10^5/2001	1.4×10^5 /40%
Sweden	2.3×10^5/1998	1.6×10^5 /34%
Mongolia	7.0×10^2/2001	Wide areas of (forest) tundra and taiga
Russian Federation	1.2×10^6/2001	

Reindeer uses wild feed, particularly lichen in winter and green vascular plants and possibly fungi [132] during the growth period. Reindeer lichens are *Cladonia stellaris*, *Cladonia rangiferina, Bryoria spp., Alectoria spp.*) [133] and others. Herded reindeer find their natural feed in large forested or mountainous areas, including the tundra and taiga of Fennoscandia, Russian Federation, Canada and Alaska throughout the year. The spectrum of wild vascular plants in the feed of reindeer is broad and includes some 450 species of feeding plants. They are dwarf shrubs (17%–68%) *Vaccinium myrtillus, Empetrum hermaphroditum, Calluna vulgaris;* Graminids: wavy hairgrass; Water horsetail; and buds, shoots and leaves from birches and willows [133 to 138]. In some of the pasture areas in Fennoscandia, supplementary feeding is needed and provided in winter. Feeding stuffs can be, for instance, grain-based fodders and grass silage or hay, and also bunches of leaf fodder and common reed (*Phragmites australis*).

Because of the nomadic behaviour of reindeer it is, in principle, difficult to derive an aggregated transfer parameter for reindeer meat (Table 23). The VAMP report [139] mentions

^{137}Cs T_{ag} values of (0.6-1.1) m²·kg^{-1} during the first winter after deposition (1986) whereas for August 1987 a value of 0.5 m²/kg (FW) is reported. The radiocaesium concentration in reindeer meat during summer and early autumn (northern hemisphere) is less than 10 or 20 % of the winter concentration. Lower values of T_{ag} than those listed above were obtained in winter 1979 when the source of contamination was accumulated nuclear test fallout. A low annual deposition rate for ^{137}Cs and accumulated activity of 2040 Bq/m^2 [140] resulted T_{ag} values with a geometric mean of 0.24 m^2 kg^{-1} (FW) (Table 24) for meat samples from three locations in Northern Finland [141].

TABLE 23. Cs-137 AGGREGATED TRANSFER FACTOR T_{ag} (m^2/kg FW) TO MEAT OF SEMI-DOMESTICATED REINDEER IN FENNOSCANDIA.

Site	Latitude °N	Deposition^1 kBq/m²	Deposition^2 kBq/m²	T_{ag}			Ref.
				Sept 1986	Nov-Dec 1986	Jan-Apr 1987	
Konkämä, Lainiovuoma SE	68-69	1.8	3×10^{-1}	4×10^{-2}	1.5×10^{-1}	1.5×10^{-1}	[143]
Stakke, Östra Kikkejaure SE	65-66	2.0	6×10^{-1}	1.8×10^{-1}	3.2×10^{-1}	3.6×10^{-1}	
Vilhelmina norra SE	64-65	2.3	3.6×10^{1}	1×10^{-1}	3.4×10^{-1}	7.8×10^{-1}	
Jiingevaerie SE	63-64	2.4	1.5×10^{1}	1.2×10^{-1}	5.2×10^{-1}	8.4×10^{-1}	
Tännäs, Idre SE	61-62	2.2	2.7	3.5×10^{-1}	5.5×10^{-1}	5.7×10^{-1}	
Paistunturi, FIN	69-70	1.7 (1986)	2×10^{-1}	Oct 2001-Apr 2002: 1.1×10^{-1}			[136, 146]
Ivalo, FIN	68-69	1.7 (1986)	2×10^{-1}	Oct 2001-Apr 2002: 1.1×10^{-1}			
Kemin Sompio, FIN	67-68	1.8 (1986)	3×10^{-1}	Oct 2001-Apr 2002: 1.9×10^{-1}			
Finnish Lapland, three sites	66-69	2.0 (1979)	–	Dec 1979: 2.4×10^{-1}			[141]
Alta NO	68-70			Nov 1998: 1.6×10^{-2}			
Varanger NO				Nov 1998: 7.1×10^{-2}			
Karlsøy NO				Nov 1998: 5.7×10^{-2}			
Northern Norway				Nov 1998: 1.1×10^{-1} Nov 1998, mean: 6×10^{-2}			[148]
Lovozero, Murmansk Oblast, RU;	66-69	1.6 (1998)		Aug 1998: 4.4×10^{-2} Mar 1999^3: 6.9×10^{-2} 1995-1998, winter: 9.1×10^{-2}			[147, 148, 131]
Kola region, RU							[149]

^{1}Global fallout; ^{2}Chernobyl fallout, mean values, $^3T_{ag}$ for ^{90}Sr: 3.6×10^{-4}.

TABLE 24. ECOLOGICAL ($T_{1/2}^{eco}$) AND EFFECTIVE ($T_{1/2}^{eff}$) HALF-LIFES OF RADIOCAESIUM IN SEMI-DOMESTIC REINDEER *(Derived from data on ^{137}Cs activity concentrations in meat. Short and long components are given as available.)*

Ecosystem	$T_{1/2}^{eco}$ (y)	$T_{1/2}^{eff}$ (y)	Site	Period	Reference
	6 to 15	5-10			[45]
Spruce forest/ mountains	5.6		Sweden	1986/2000	[143]
	$9.3 < T_{eco} < 17.4$		Northern		
	$4 < T_{eco} < 4.4$		Other		
		6	Finnish Lapland	1966/77	[144]
		Short: 1 to 3; long: 6 to 8	Finnish Lapland, three pasture areas	1986/2000	[136]
	3.95 ± 0.56		Vaga (NO)	1986/95[1] autumn	[148]
	3.63 ± 0.15			1986/98[1] winter	
	6.0 ± 1.7		Ostre Namdal (NO)	1986/95[1] autumn	[148]
	3.13 ± 0.27			1986/95[1] winter	
	Short: 1.2; long: 18	Short 1.2; long 11	Kautokeino NO		[73, 149]
	Short: 2.0; long 18	Short 1.2; long 18	Kola Peninsula, RU		
	Short: 1.8; long: 16	Short 1.5; long 10	Nenets AO, RU[2]		

[1] No significant decline after the specified period; [2]Autonomous Okrug.

The bioavailability of caesium depends on local conditions. The effective half-life typically varies from 5 to 10 years, but both lower and higher values have been reported [139]. Activity concentrations of ^{137}Cs and seasonality are described in [80, 136, 137, 142-144]. The effect of seasonality on T_{ag} for ^{137}Cs in the meat of reindeer is shown in Table 23**.** Ecological half-lives of ^{137}Cs in reindeer are given in Table 24. Values for biological half-life of radiocaesium in reindeer are given in Table 25. Data for semi-domesticated reindeer show variability in $T_{1/2}^{eco}$ that reveals a significant dependence on the type and condition of pastures, and also the need to consider seasons with their specific values of T_{ag}.

Caribou *(Rangifer tarandus)* is the same species as reindeer, and there are seven subspecies. Caribou lives wild in the arctic tundra, mountain tundra, and northern forests of North America, Russian Federation, and Scandinavia. The world stock is about 5 million [145]. Caribou is extensively hunted by aboriginals in the central Arctic of northern Canada. Data for ^{137}Cs in the muscle samples of caribou of major herds living in Alaska and northern Canada were compiled from 1960's to late 1980's [80]. The effective half-life for a single injection of ^{137}Cs to atmosphere was estimated as 6 years. The observed ecological half-life of 8 years was reported to include a contribution from the ongoing fallout over the study period. Chernobyl-derived ^{137}Cs contributed 6 to 40% of activity concentrations in muscle of caribou after spring 1986, varying by region. A best fit for a long-term rate of uptake normalised to the deposition was 0.68 $m^2 \cdot a^{-1} \cdot kg^{-1}$ wet weight.

TABLE 25. BIOLOGICAL HALF-LIFE OF RADIOCAESIUM IN REINDEER, DAYS

Nuclide	$T_{1/2}^{bio}$ (d)				Comment	Reference
	AM[1]	SD	Min.	Max.		
^{134}Cs	8.4×10^{-1}	1.2×10^{-1}	-	-	Fast component	[150]
^{134}Cs	1.7×10^{1}	1.4	-	-	Slow component	
^{137}Cs	-	-	2.0×10^{1}	3.3×10^{1}	Slow component	[151,152]

[1] Reduced 2–3 fold during summer months as compared to autumn-winter.

4.6 Transfer to moose (Alces alces) and white-tailed deer (Odocoileus virginianus)

Moose, the largest of the cervids, is typical of boreal and mixed deciduous forests of the Northern Hemisphere in temperate and sub-arctic environments. Moose has seven subtypes, one of them is European elk, or moose (*Alces alces alces*); four types of moose live in North America, and two in northeast Asia. In the Fennoscandian countries, Norway, Sweden, and Finland, are currently found the most productive and heavily harvested moose populations in the world; about 200 000 animals were killed in this area in 2000 [153]. Also, there are substantial stocks in various Baltic countries and Russian Federation.

Moose is the largest mammal in Northern Europe. It stands 2 m high at the withers. Its length can be about 2.8 m. The live weight of an adult buck can be almost 700 kg and of a cow about 350 kg [83].

In Finland, the number of hunted moose per year varied between 60 000 and 80 000 in the early 2000's. In addition, 20 000 white-tailed deer, a medium-sized cervid, were harvested. Moose and other wild deer species contribute annually 10 million kg of game meat to the diet of the Finnish people, corresponding to 2 kg *per capita* consumption and 90% of the consumption of game meat [76]. In Sweden, 1.3 kg moose meat and 0.4 kg roe deer meat was consumed per person [129]. Outside the Fennoscandian countries, the intensity of hunting and consumption of game meat are considerably lower.

For moose and other deer, the availability of feed plants, dry matter and nutrient content of the diet, and metabolism change with season. Accumulation of a radionuclide in meat and other tissues varies by intake rate, composition of feed, uptake from the gut and secretion from the body, and most of these factors are not constant throughout the year, particularly when arctic or sub-arctic conditions prevail.

The digestive system of the moose shows physiological changes that are adaptive to the period of rapid and non-selective intake of non-fragmented mature forage [154]. The effect of season should be considered and data specific to the hunting season used when assessments are made of activity concentrations in meat for human consumption.

Moose calves need considerable amounts of fresh plants during their first summer when the body weight is increasing from 8-13 kg up to 120-140 kg in a few months [155]. In winter, the demand for feed by a moose is 10-15 kg of branches or twigs per day. The most heavily consumed species are pine *(Pinus sp.)* and birch *(Betula sp.)*, but the moose favours most mountain ash *(Sorbus rowan),* osier (*Salix sp.)*, aspen *(Populus tremula)*, and juniper *(Juniperus).* In summer, the moose eats mostly leaves of trees and bushes, various grasses, and also large amounts of aquatic plants and plants from mires (horsetail *(Equisetum palustre)*, yellow water lily *(Nuphar luteum)*, species of the pondweed family *(Potamogetonaceae)*, bogbean *(Menyanthes trifoliata)*.

When plenty of green feed is available in summer, the moose can eat several tens of kilograms during a day. In autumn, the main part of the diet is composed of green parts of bilberry *(Vaccinium myrtillys),* lingonberry and dwarf birch. Mushrooms can be a seasonal component of the diet of the moose. In addition, the moose may visit oat fields and consume oats and other cultivated plants. White-tailed deer is occasionally given supplementary feeding in sub-arctic conditions in winter.

In Mid-Sweden, during the period July to October, three plant species – rosebay willow herb *(Chamaerion angustifolium)*, birch and bilberry – constitute more than 70 % of the rumen content. The daily intake of food is estimated as 7.5 kg dry weight during July and August, and 6 kg dry weight during September [156]. During fall, deciduous trees, conifers, dwarf shrubs and fungi (in October) provide ^{137}Cs [97].

Because of the varying composition of feed, the ^{137}Cs level in moose throughout the year varies slightly with time and may be highest in autumn due to availability of mushrooms or feeding on mineral-rich wetland vegetation. The aggregated transfer factor T_{ag} with respect to ^{137}Cs in Technical Reports Series No. 364 was $T_{ag} = 0.02$ m^2/kg, FW [45].

In Table 26, data from Fennoscandia suggest a lower value during the hunting season, about 0.01 m^2/kg (FW) for adult moose and 0.013-0.02 m^2/kg (FW) for calves [43. 65, 157; 158]. The bioavailability of radiocaesium in ecosystems grazed by moose seems to be rather constant and, therefore, the physical half-life determines the effective half-life [97, 98, 158, 43].

TABLE 26. ^{137}Cs AGGREGATED TRANSFER FACTOR T_{ag} (m² kg^{-1} FW) TO MEAT OF MOOSE *(Samples were taken during the hunting seasons, if months are not given.)*

Site	Year/months of sampling	Number of samples	T_{ag}, Mean/ST deviation or range, as indicated	Reference
Northern Sweden	May1986 to Apr. 1987	3661	1×10^{-2} [a] adults 2×10^{-2} [a] calves	[157, 158]
Central Sweden, Harbo	1986-1991	1307	2.1×10^{-2} [b], $(1.8–2.4 \times 10^{-2})$ [c] Calculated as annual means	[97, 160]
Finland, seven sites	1979	121	1.2×10^{-2} [d]; adults 2.0 [d] calves	[141]
Finland, nationwide sampling	1986-1996			[65, 43]
	Adults	1051	9.5×10^{-3} [b]; 5.8×10^{-3} [e]	
	Calves	365	1.3×10^{-2} [b]; 8.5×10^{-3} [e]	
–	–	–	2×10^{-2} [b]; $(6 \times 10^{-3} – 3 \times 10^{-2})$ [c]	[45]

[a] Derived with linear regression between ground contamination and activity concentration of caesium-137 in moose; [b] Arithmetic mean; [c] Range; [d] Geometric mean and standard deviation (GSD); [e] Standard deviation of arithmetic mean

However, data for moose, or elk, from Poland show a gradual decline of ^{137}Cs concentrations during 1986–1991, thus implying processes that reduce the bioavailability of ^{137}Cs in soil in regions south of the boreal forest zone [159]. Values of T_{ag} for white-tailed deer are often higher than those for moose (Table 27).

TABLE 27. AGGREGATED TRANSFER FACTORS FROM SOIL TO MEAT OF WHITE-TAILED DEER, $m^2\ kg^{-1}$ FW

Year/Month	Site	N	T_{ag}		Reference
			GM	GSD	
1986					
Jun-Aug	South of	18	8.4×10^{-3}	1.7	[43, 82]
September	Finland	9	2.6×10^{-2}	1.5	

4.7 Transfer to wild boar (Sus scrofa)

Wild boar is characterized by a very large area of feeding. Wild boar can cover a distance of 80 kilometres a day to feed and come back to its usual rest area in the evening. Omnivorous, the wild boar changes its diet with the seasons. Nearly totally herbivorous in spring and summer, it behaves mainly as a burrower when grass is rare in winter and feeds on roots, tubers, larva, earthworms, etc. for which the transfers for caesium are much higher. Hence increased levels of contamination (by of the order of 50%) are usually observed from October to March.

Over the first 18 years after the accident in Chernobyl, wild boar exhibited the highest ^{137}Cs contamination of any type of game in Bavaria, Germany; as compared with concentrations in other game: wild boar > roe deer > chamois > deer [121].

In wild boar from Bavaria, Germany, during the years 1986 to 1999, the median activity concentration for each year decreased with $T_{1/2}^{eco}$ = (10.5 ± 1.6) years, but the variability within the year increased to about four orders of magnitude e. g. in 1998 the values varied between 80 kBq kg^{-1} fresh meat and 2 Bq kg^{-1} [108].

In Croatia, over the period of 2000 to 2002, in areas with approximately equal contamination levels, ^{137}Cs concentrations in wild boar meat varied by over two orders of magnitude [161]. Hecht [121] and Fielitz [117] have reported on the fungus deer truffle, *Elaphomyces granulatus Fr.*, which substantially accumulates ^{137}Cs and is dug out from the forest soil by wild boar.

This fungus belongs to *Ascomycetes* and is considered to be inedible for people, but it was found [93] in the rumens of highly contaminated wild boar from the Bavarian Forest. This fungus was also considered responsible for the high degree of ^{137}Cs contamination in wild boar of southern Germany [162].

In Germany 123,165 wild boars were shot during the hunting year 1982/83, whereas in 2001/02 about 498,640 were shot and the trend is for further increases [117]. Values of T_{ag} for ^{137}Cs from soil to meat of wild boar are listed in Table 28. Ecological half-lives are given in Table 29.

The concentrations of ^{137}Cs in wild boar increased with time from 1987 to 2004 because of its special feeding habits [117, 121] as discussed above.

TABLE 28. ^{137}Cs AGGREGATED TRANSFER FACTOR T_{ag} TO MEAT OF WILD BOAR (m^2 kg^{-1} FW)

Site	Years after fallout	N	GM	GSD	Site description	Reference
Bodenmais (G)	13/14		2.1×10^{-1}		Spruce forest, mean deposition 98 kBq m^{-2} at 1.05.1986	[94]
Göttingen (G)	13/14		5×10^{-4}		Beech forest	
Sumava Temelin (Czech R.)	0/5		9.8×10^{-3}		Spruce/mountain Spruce/agriculture	[40]
Weinsberger forest (A)	1	1	4×10^{-3}		Spruce forest, median deposition 52.2 kBq m^{-2} at n = 218 sites, 1.05.1986	[115]
	2	4	8×10^{-3}	4.6		
	4	5	3.5×10^{-2}	2.4		
	5	12	1.3×10^{-2}	1.8		
	6	9	3.9×10^{-2}	2.2		
	7	19	2.9×10^{-2}	3.3		
	8	15	2.0×10^{-2}	3.2		
	9	10	3.7×10^{-2}	3.1		
	10	2	4.6×10^{-2}	1.7		
	11	5	2.7×10^{-2}	3.9		
	12	26	1.4×10^{-2}	7.0		
	13	7	2.8×10^{-2}	3.4		
	16	11	3.7×10^{-2}	3.4		
	17	12	4.2×10^{-2}	1.9		
Kobernausser forest (A)	2	13	6.7×10^{-2}	2.8	Spruce forest, median deposition 48.5 kBq m^{-2} at n = 20 sites, 1.05.1986	
	3	2	4.9×10^{-2}	3.0		
	6	1	1.6×10^{-1}			
	14	1	3.1×10^{-2}			

TABLE 29. ECOLOGICAL HALF-LIFE OF ^{137}Cs IN WILD BOAR MUSCLES IN FOREST AREAS DERIVED FROM DATA FITTED BY A SINGLE EXPONENTIAL [116].

Ecosystem	$T_{1/2}^{eco}$ y	N	Site	Period	Reference
Forest/Fields	10.5 ± 1.6	4077	Bavaria (G)	1987-1999	[108, 118, 121]
	no decrease	314		1986-1996	[119, 120]
Spruce	no decrease	321		1987-2004	[117]

4.8 Transfer to brown bear (Ursus arctos)

Bear stocks have decreased with time and the species has disappeared from many European countries, although considerable stocks are still found in the coniferous forest zone of the Northern hemisphere. The largest stocks are in Russian Federation (36,000), Alaska and Canada. The bear stock in Finland (1000) is densest in the eastern and south-eastern part of the country [76, 163]. Bear meat has been surveyed for ^{137}Cs since the mid-1980's [144, 146, 164]. The T_{ag} values from deposited radioactivity to meat have been derived using these data (Table 30).

The brown bear is a species of bear that can reach masses of 130–700 kg. Brown bears have a large hump of muscle over their shoulders, which give strength to the forelimbs for digging. The normal range of physical dimensions for a brown bear is a head-and-body length of 1.7 to 2.8 m and a shoulder height 90 to 150 cm.

Being omnivores, brown bear feed on a variety of plants and berries including roots or sprouts and fungi, as well as insects and small mammals; what is eaten depends largely on the time of year and precise location. The larger bears have been known to prey on large mammals such as moose, sheep, and caribou. In preparation for winter, bears will gain hundreds of kilograms of fat before going into a state of false hibernation.

TABLE 30. AGGREGATED TRANSFER FACTORS FOR ^{137}Cs FROM DEPOSITED RADIOACTIVITY TO MEAT OF BEAR (T_{ag}, m^2 kg^{-1} FW) IN NORTHERNMOST FINLAND

Year of sampling	N	GM	GSD	Reference
September1984; May 1985	6	4.3×10^{-2}	2.2	[144]
May-September 1986	9	7.1×10^{-2}	1.8	[146]
Jun.-September 1987	8	6.6×10^{-2}	2.4	[164]
May-November 1988-2001	61	4.5×10^{-2}	1.8	

[a] Recalculated from primary data.

4.9 Transfer to small game

Activity concentrations and seasonality with respect to ^{137}Cs concentrations are described in [98]. Information on the time-dependence of T_{ag} for ^{137}Cs from soil to meat (or, for waterfowl, from deposited activity to meat) of small game is given in Table 31. A comparison with values for roe deer in Table 17 shows a faster decline for the T_{ag} of ^{137}Cs with time in small game, particularly in waterfowl. Activity contents of ^{137}Cs in waterfowl follow the dynamics of ^{137}Cs in aquatic ecosystems. Variation in the values of T_{ag} for ^{137}Cs in waterfowl may be larger than with other birds because of migration of birds from their summer habitat after the beginning of hunting in August. Despite the fact that the birds were shot in inland locations, some birds from the coast of Gulf of Finland or Gulf of Bothnia may be represented in the data. Brackish water ecosystems result in lower uptake of ^{137}Cs to aquatic birds than is the case for freshwater ecosystems.

There is a distinct difference in uptake of caesium between herbivores feeding mostly in forest and those visiting agricultural fields for feed. Examples that exhibit lower intakes due to partly or mostly depending on cultivated feed are brown hare and pheasant.

The ecological half-time of ^{137}Cs was determined for red grouse *(Lagopus lagopus scoticus)* living in a heather dominated upland ecosystem [165]. A value of 10-11 days was obtained for captive birds.

Transfer of radioactive caesium to game animals and reindeer and the rate of activity reduction, given as ecological half-life, reflect the soil and pasture conditions. In arctic and subarctic regions there is practically no ecological decrease in activity concentration of ^{137}Cs in terrestrial game.

TABLE 31. AGGREGATED TRANSFER FACTORS T_{ag} (m^2 kg^{-1} FW) FOR ^{137}Cs FROM SOIL TO MUSCLES OF SMALL GAME *(Samples were taken in Finland during hunting seasons. For certain species, the hunting year may last until Jan-Feb in the next calendar year)*

Species or group Year	Site Number of samples	T_{ag}, (m^2 kg^{-1} FW)			Ref.
		AM	Min.	Max	
Arctic hare	–	3×10^{-2}	9×10^{-3}	1×10^{-1}	[45]
Brown hare	–	4×10^{-3}	2×10^{-3}	5×10^{-2}	
Willow grouse *(Lagopus lagopus)*	Northern Finland				
Aug.-Sep. 1986	31	2.7×10^{-2}	1.1×10^{-2}	4.6×10^{-2}	[144]
Black grouse *(Tetrao tetrix)*	Northern Finland				
2003	2	–	1.1×10^{-2}	1.3×10^{-2}	[164]
Brown hare *(Lepus europaeus)*	Southern Finland				
1986	10	1.1×10^{-2}	1.6×10^{-2} (SD)		[43]
1988	2	9.1×10^{-4}	5.4×10^{-4}	1.3×10^{-3}	
		GM	GSD		
Arctic hare *(Lepus timidus)*	Southern Finland				
1986	11	2.9×10^{-2}	1.9		
1988	20	2.4×10^{-2}	2.4		
1989	29	3.2×10^{-2}	2.3		
Terrestrial birds[1]	Finland				
1986	11	8.9×10^{-3}	2.4		
1988	16	1.6×10^{-2}	1.7		
1989	23	1.2×10^{-2}	2.1		
Pheasant *(Phasianus colchicus)*					
1989	3	3.2×10^{-4}	1.3		
Waterfowl[2]	Southern Finland				
1986	31	1.3×10^{-2}	3.8		
1988	6	5.3×10^{-3}	5.2		
1989	14	2.4×10^{-3}	2.8		
Willow grouse	Northern Finland				
1996-2004	26	1.5×10^{-2}	2.1		[164]

[1] Species in decreasing order of numbers of samples: black grouse *(Tetrao tetrix)*, hazelhen *(Bonasia bonasia)*, wood pigeon *(Columba palumbus)*, willow grouse *(Lagopus lagopus)*, capercaillie *(Tetrao urogallus)*; [2] Species: mallard *(Anas platyrhynchos)* (33 samples) and goldeneye *(Bucephala clangula)* (18 samples).

Forests in temperate and boreal regions differ by soil type and vegetation, and a faster decline of muscle activity concentration of deer animals occurs in temperate zone. In wild boar the caesium activity concentration shows no decline because of its special feeding habits. In waterfowl a relatively fast decline in uptake of ^{137}Cs has been found, determined by the dynamics of ^{137}Cs in aquatic ecosystems.

5. CONCLUSIONS

The parameter most widely used to quantify radionuclide transfers to wild foodstuffs in forests is the aggregated transfer coefficient (T_{ag}). This simplistic approach is adopted because of the complexity of transfer pathways in forests. Unlike domestic animals, game animals have complex diets which are difficult to identify because they vary between locations and between seasons.

Edible fungi are an ecologically complex group of organisms which obtain their nutrition from widely different locations and with highly variable rates. Similarly, understorey plants producing edible berries can have complex root distributions which make it difficult to understand exactly where they absorb radionuclides in the soil profile.

The simplicity of the T_{ag} approach may lead to inappropriate application of T_{ag} values in dose assessment calculations. T_{ag} values should only be used in calculations for forest systems in which radionuclide fluxes have stabilized, in the medium to long term. Where effective or ecological half life data are available, T_{ag} values should not be considered to be constant with time. Finally, in all cases T_{ag} values should be considered as a means of carrying out screening calculations, rather than providing a definitive method of calculating transfers in forests under all conditions. Detailed, site-specific dose assessments will require careful consideration of local transfer processes, pathways and rates than T_{ag} values can provide.

The behaviour of radionuclides in forest ecosystems is specific and differs substantially from the agricultural ecosystems. Even if, the importance of radioactive contamination of forests as a significant source of the population exposure is now recognized, most of the data refer to caesium and strontium to a lesser extent. Some limited information is also available for plutonium and other radionuclide. But, practically no other data are available on other radionuclides.

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RADIONUCLIDE TRANSFER IN ARCTIC ECOSYSTEMS

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Abstract

Arctic ecosystems show a considerable variability in radionuclide transfer parameters, which can be explained by site-scpecific factors. The paper provides basic modelling approaches based on usage of agrigated transfer factors that are in used for radiological assessments in Arctic environments. Parameter values that can be used in environmental models (T_{ag}s and ecological half lives) are provided for ^{90}Sr and ^{137}Cs transfer to lichen and reindeer and to locally produced food products.

1. INTRODUCTION

There are some environments, in particular, the Arctic and Alpine ecosystems where the behaviour of radionuclides differs from that in a temperate environment and both of them are very vulnerable to radioactive contamination due to the specific environmental conditions.

Numerous studies have shown that, in terms of radiocaesium transfer to man, the Arctic ecosystems are much more vulnerable to fallout than temperate ecosystems [1-3]). This is due to generally higher early, as well as long-term concentrations in the vegetation per unit fallout than prevalent south of the arctic zones. The rate of litter decomposition is slower in arctic ecosystems than in ecosystems of temperate regions. Biogeochemical processes are generally slower, too; hence, a contaminant resides for a longer period in the biota of an arctic ecosystem. Moreover, high utilisation of semi-natural ecosystems for foodstuffs and local dietary habits will lead to cumulative doses with significant contributions persisting for a long time after the initial environmental contamination.

The food chain *lichen–reindeer–man* has been the main object of study within terrestrial arctic radioecological research. The high interception by lichen of radionuclides, particularly radiocaesium, is most important, and is one of the key factors contributing to probably the most vulnerable arctic food pathway. Lichen represents the main accessible reservoir of radionuclides in the Arctic environment and 65% of the overall radionuclide burden is accumulated in the top three cm of lichen which is consumed by reindeer [4]). This results in the elevated concentrations of ^{137}Cs and ^{90}Sr in the food chain 'lichen–reindeer–reindeer-herder' and this phenomenon has been intensively studied in different Arctic regions since the 1960s [1, 5-7]).

Usually ammonium, which is formed during decomposition of organic matter, is oxidised to nitrates, but in arctic regions ammonium may not be completely oxidised, leaving it available for competition with radionuclide cations. Ammonium has a five fold higher affinity than potassium for illitic adsorption sites; this influences the fixation of caesium in Arctic ecosystems [2].

Some of these factors provide also great impact on behaviour of radionuclides in Alpine ecosystems. Alpine ecosystems show a very distinct behaviour with respect to nutrient cycling and also are very sensitive against contamination due to the specific prevailing climatic and environmental conditions. So, high altitudes typical for alpine ecosystems, provide rather cold climatic conditions with high amounts of precipitation, a long-lasting snow cover and well frozen soils in winter. This results in slow decomposition of soil organic matter, leading to low pH-values in soil, nutrient deficiency and very specific composition of plant species [8].

2. ENVIRONMENTAL FACTOR GOVERNING BEHAVIOUR OF RADIONUCLIDES IN ARCTIC ECOSYSTEMS

The Arctic is more vulnerable to radioactive contamination than temperate regions [1, 9, 10]). The rates of biogeochemical processes are generally slower in the Arctic; hence, a contaminant can reside for a longer period in the biota of an arctic ecosystem. High utilisation of semi-natural ecosystems for foodstuffs is typical in many arctic areas, and agricultural practises are less intensive than in temperate regions. Local dietary habits can lead to relatively high doses accumulating over a long time after the initial environmental contamination. The Arctic is also characterised by a pronounced diversity in environmental conditions, types of land use and dietary habits. E.g., although reindeer/caribou meat may be the most vulnerable food product to contamination following a radioactive release, contamination in cow milk and lamb meat may be a larger problem in some Arctic areas. Mushrooms, freshwater fish and berries may also be important exposure pathways. Therefore, care must be taken not to assume that results from one region must be applicable for another. The concentration in foodstuffs can depend on many factors such as local deposition rates, soil and vegetation types and availability, dietary preferences of animals and the presence of forests. Production of some foodstuffs can be highly skewed toward a comparatively small and atypical part of the region as a whole.

The mechanisms of uptake and transfer of radionuclides to animals and plants in the Arctic are generally similar to those in other regions. Information on transfer to wild berry species which are collected in large quantities also in the subarctic and arctic regions is given earlier. This paper gives a general introduction to the most important food products in the Arctic, reindeer meat, lamb meat and milk, and focuses on aspects of transfer of radionuclides to products that are specific to the Arctic region.

For example, the considerable range in observed concentration values of ^{137}Cs in food products in the Arctic is given in the AMAP reports [1, 9]. The Nordic countries (Sweden, Norway, Finland, Iceland and Denmark, including the Faroe Islands) have also cooperated for many years on radioecological studies within the framework of NKS (Nordic Nuclear Safety Research). This research has included coordinated studies on the transfer of radionuclides to cow milk and sheep meat [11, 12].

3. TRANSFER COEFFICIENTS AND TIME TRENDS

Evaluation of time-dependent transfer coefficients requires information on possible changes in contamination during the period of interest (e.g. impact of the Chernobyl accident) and agricultural practices that may result in spatial variations of the data, such as countermeasures which might be applied in some areas and changes in the farming practices.

The aggregated transfer factor (T_{ag}) is normally used to quantify the ratio between the total deposition density (Bq m^{-2}) and the concentration in a given (food) product (Bq kg^{-1}). As pointed out in the AMAP (1998) compilation [1], T_{ag} values are highly variable between seasons and years. Similarly, using a single T_{ag} value for lamb neglects the seasonal pattern of sheep grazing in Norway (sheep are stabled in the winter and fed upon stored feed) and assumes an exclusive consumption of locally-produced feed. However, the slaughter of reindeer and sheep typically occurs during the autumn/winter in a given year. These temporal variations make it difficult to compare T_{ag} values between sites in different years. To predict changes with time, T_{ag} values need to be combined with effective ecological half-lives.

When making assessments in the arctic, e.g. using T_{ag} values, major sources of variability must be taken into account, and in particular:

a) Variation with time

b) Seasonal variability

c) Spatial variability and dietary variability of grazing animals (as already discussed)

The reduction of concentration of radionuclides in a system is commonly described as an exponential decrease with an effective half life $T_{1/2}^{eff}$.

Effective half lives of ^{137}Cs in most food products (other than reindeer) is related to the rate of depletion of the available pool, either due to fixation by soil or due to removal by runoff or leaching down the soil profile. Therefore, the contribution to the total ^{137}Cs intake from those products with the longest T_{eff}, namely those such as fungi and lamb meat produced on pasture with organic soils, can become proportionately greater with time. This fact, combined with the relatively high transfer factor values and wide use of semi-natural food products is a special feature of the Arctic ecosystems.

The decrease of radionuclide concentrations in environmental compartments is commonly described based on the effective or ecological half lives of radionuclide in the media of interest. Arctic data (as well as data from other regions) often show such a decline in parameter values (e.g. T_{ag} values, concentrations) which cannot be described by a single term exponential function. Often two exponential models are needed to describe the data adequately, a fast loss component T_{eff1} and a slow loss component T_{eff2}. The time dependency of the concentration (or other quantities such as the aggregated transfer coefficient, T_{ag}) then can be expressed as:

$$T_{ag}(t) = T_{ag}(0)\cdot\left(a_1\cdot e^{-\frac{\ln 2}{T_{eff1}}\cdot t} + (1-a_1)\cdot e^{-\frac{\ln 2}{T_{eff2}}\cdot t}\right) = T_{ag}(0)\cdot e^{-\frac{\ln 2}{T_r}\cdot t}\cdot\left(a_1\cdot e^{-\frac{\ln 2}{T_{eff1}}\cdot t} + (1-a_1)\cdot e^{-\frac{\ln 2}{T_{eff2}}\cdot t}\right)$$

where $T_{ag}(0)$ is the initial concentration and a_1 is the initial fraction of the concentration associated with the fast loss term (numbered 1). The estimates for the fast loss term depend on the definition of time zero and care must thus be taken when comparing results from different studies.

The value of a_1 determines if and how long the fast term is dominant in the decrease. The value of a_1 needs to be greater than 0.5 for the fast term to become dominating and the following equation can be used for calculating how long it will be dominating:

$$t_{eq} = \frac{\ln\left(\frac{a_1}{1-a_1}\right)}{\frac{\ln(2)}{T_{1/2}^1} - \frac{\ln(2)}{T_{1/2}^2}}$$

It does not matter in the equation whether the fast and slow loss components are described in terms of the effective half life or the ecological one, as the result only depends on the difference between the fast and the slow removal rates.

It can be difficult to get reliable estimates for the environmental half lives, and parameters found to be applicable at one region under given conditions need not be applicable elsewhere and/or under other conditions.

3.1. Radionuclide transfer to lichen and reindeer

Reindeer and caribou are the same species (*Rangifer tarandus)* and together they have a circumpolar distribution. As described in Chapter 8, there is a potential for high radionuclide concentrations in reindeer, and high consumption of reindeer meat by indigenous population groups in Europe, Asia and Northern America make these people vulnerable to radioactive contamination. General aspects of uptake and metabolism of radionuclides in animals are given in Chapter 6.

Following radioactive contamination of their pastures reindeer and caribou attain higher radionuclide concentrations than other game species because of their extensive intake of lichens during winter. The autumn change towards a lichen diet deficient in mineral elements like potassium is accompanied by a 2-3 fold increase in biological half-life of radiocaesium from about 7 to about 20 days [13-16]. This increase in biological half-life will also contribute significantly to increasing radiocaesium concentrations in reindeer.

3.3.1 Lichens

Lichens have no rooting system and take up nutrients and contamination from air and precipitation. The deposited radionuclides are retained by the lichen and the contamination level is thereafter reduced through dilution by fresh lichen growth, removal by grazing or leaching. Depending on the physical and chemical properties radionuclides may also be translocated to fresh growth. Sr-90 has been found to be more mobile than ^{137}Cs in lichens, and is washed out from lichens more rapidly than ^{137}Cs [17].

Table 1 gives examples of mass interception fractions (f_l) estimated for lichens at the Kola Peninsula on the basis of annual data during the period 1961-1999 (see Fig. 1) [18]. In a situation with deposition onto snow (in winter), lichen will only become contaminated during snowmelt, with some of the deposited radioactivity lost via runoff and effectively decreasing the lichen interception fraction. In contrast, if radionuclide deposition occurs as a single pulse dry deposition then the interception fraction may exceed the annual values. If the season when deposition occurs and the type of deposition (dry or wet) are not considered, significant underestimation and overestimation, respectively, can influence radiological assessments [18].

Post-Chernobyl studies of radiocaesium in lichens indicate effective half-lives of 3-6 years [19-22]. These studies do not report decreasing concentrations comparable to a double exponential process. The estimated half-lives in lichens after the Chernobyl fallout are in agreement with estimated half-lives for radiocaesium in reindeer in the same period (see below). The results in Table 1 may therefore not be generally applicable and have to be interpreted with some caution.

TABLE 1. ^{137}Cs AND ^{90}Sr LICHEN MASS INTERCEPTION FRACTIONS (f_L, DRY WEIGHT) FOR THE KOLA PENINSULA AND ECOLOGICAL HALF LIVES (T_1, T_2) FOR THE LICHEN [18]

Radionuclide	f_l, m^2 kg^{-1}	a_1[1]	T_{eff1}, y	T_{eff2}, y
^{137}Cs	1.4	8.0×10^{-1}	2.0	20
^{90}Sr	7×10^{-1}	7.2×10^{-1}	1.0	20

[1] a_1 gives the fraction of the initial concentration in lichen declining with the short half-life T_1.

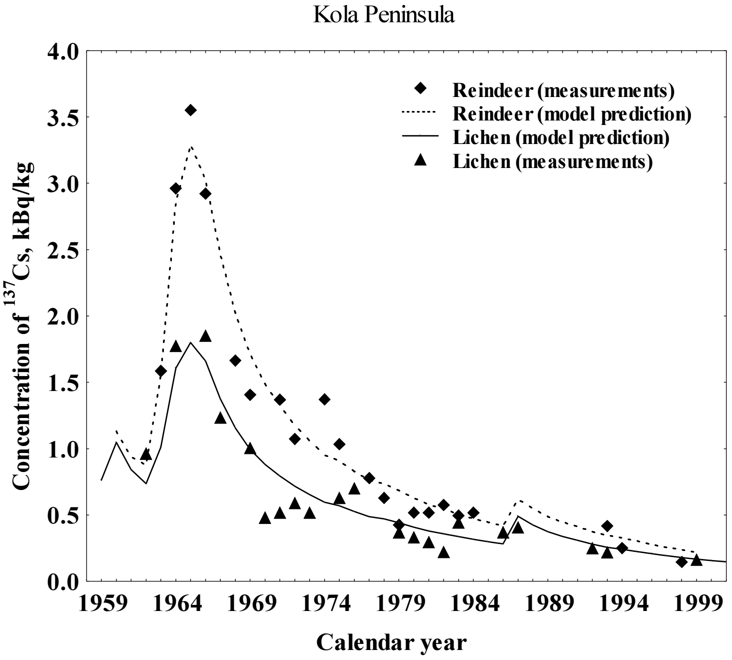

FIG. 1. Time variation of ^{137}Cs concentrations in lichen and reindeer meat in the Kola peninsula (Russian Federation). the curve is model prediction and symbols are measured values [18].

3.3.2. Reindeer

The reindeer diet change from a summer diet of a wide range of plants to a lichen based diet during winter[1]. This generally causes seasonal variability in contamination levels in reindeer with maximum concentrations during winter. However, the seasonal variability also depends on the variable deposition levels in the various seasonal grazing areas of these nomadic animals. The seasonal variability will also depend on the amount of lichens in the winter diet, which depends on the lichen abundance and availability in the grazing area. Furthermore, in autumn reindeer can eat large quantities of mushrooms and attain radiocaesium levels comparable to those during winter [22, 23]. Estimated T_{ag} values for reindeer are therefore expected to be highly site and season specific.

A particular example is the low transfer to reindeer in Iceland due to consumption of herbaceous vegetation and a species of lichen (*Cetaria islandica*) less efficient at trapping and retaining radiocaesium than e.g. *Cladonia* species preferred by reindeer in other regions [24]. Care must also be taken when estimating concentration of radionuclides in free ranging flocks of reindeer moving in a heterogeneous environment.

A study in Iceland showed no significant reduction in ^{137}Cs concentration beyond physical decay during a three decade period, but more than an order of magnitude reduction in concentration in samples from a flock grazing in a different environment one autumn (all samples below 1 Bq kg^{-1} FW) [9, 24]. In case of relatively uniform deposition, like the nuclear weapons tests fallout, there may be less variability in lichen – reindeer transfer than for food-chains involving variable soil – plant transfer [1]. When assessing initial T_{ag} values for reindeer after single deposition events the above information on variable interception in lichen should also be properly considered.

A review of radionuclide contamination levels in reindeer and caribou due to the nuclear weapons tests fallout is given by AMAP [1]. Maximum ^{137}Cs activity concentrations in reindeer in all Arctic countries were reached in the middle of the 1960s (see example in Fig. 1), and were generally much lower than those observed in central Sweden and Norway due to the higher Chernobyl fallout. Figure 2 gives an example on seasonal variability and long-term trends in ^{137}Cs concentrations in a Swedish reindeer herd in the years after the Chernobyl fallout, whereas Fig. 3 gives similar information for a Norwegian herd.

Reindeer consume lichens during all seasons and reductions in contamination levels in reindeer will therefore follow the reduction of ^{137}Cs in lichens if lichens are much more contaminated than other parts of the reindeer's diet. Post-Chernobyl observations of faster decline in radiocaesium levels in lichens than in plants lead Gaare and Staaland [3] to hypothesize that the seasonal difference in radiocaesium concentrations in reindeer would decline, and that the concentrations in reindeer during winter would ultimately become lower than during summer. Observations in Norway, exemplified in Fig. 3, illustrates that the seasonal difference has faded out in some reindeer herds. In the Swedish herds studied by Åhman [13] there are still pronounced seasonal variations.

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[1] See paper 'Radionuclide transfer in forest ecosystems' by Calmon et al. in this publication.

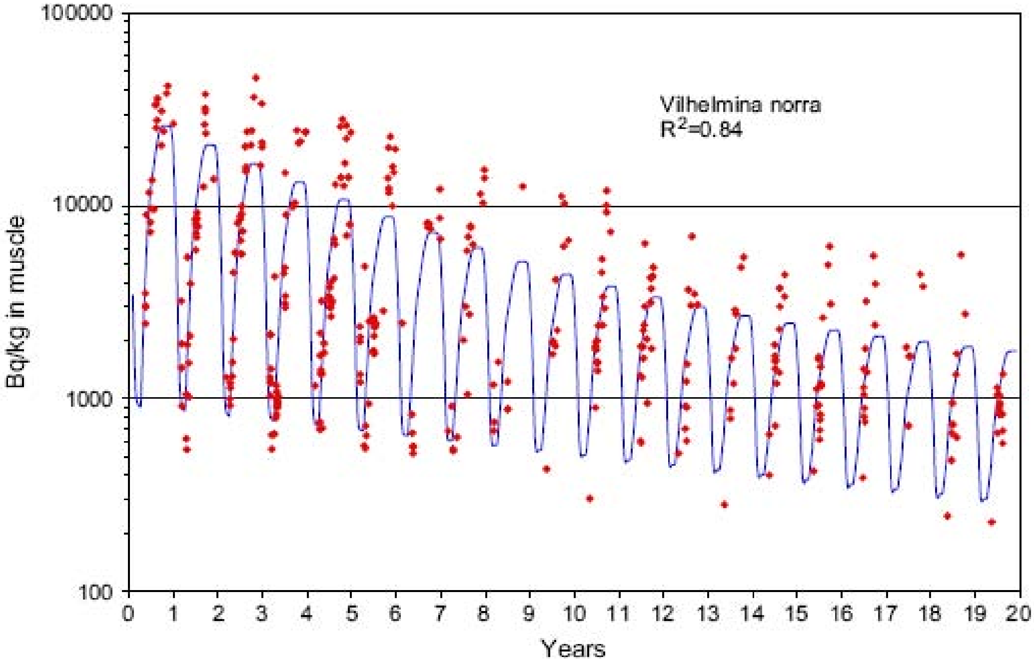

FIG. 2. Comparison of simulated and observed activity concentrations of ^{137}Cs in reindeer from Vilhelmina norra reindeer herding district in Sweden. Dots are observed values (each representing 10 or more, mostly ≥30, individual reindeer) and the line is a modelled curve [13].

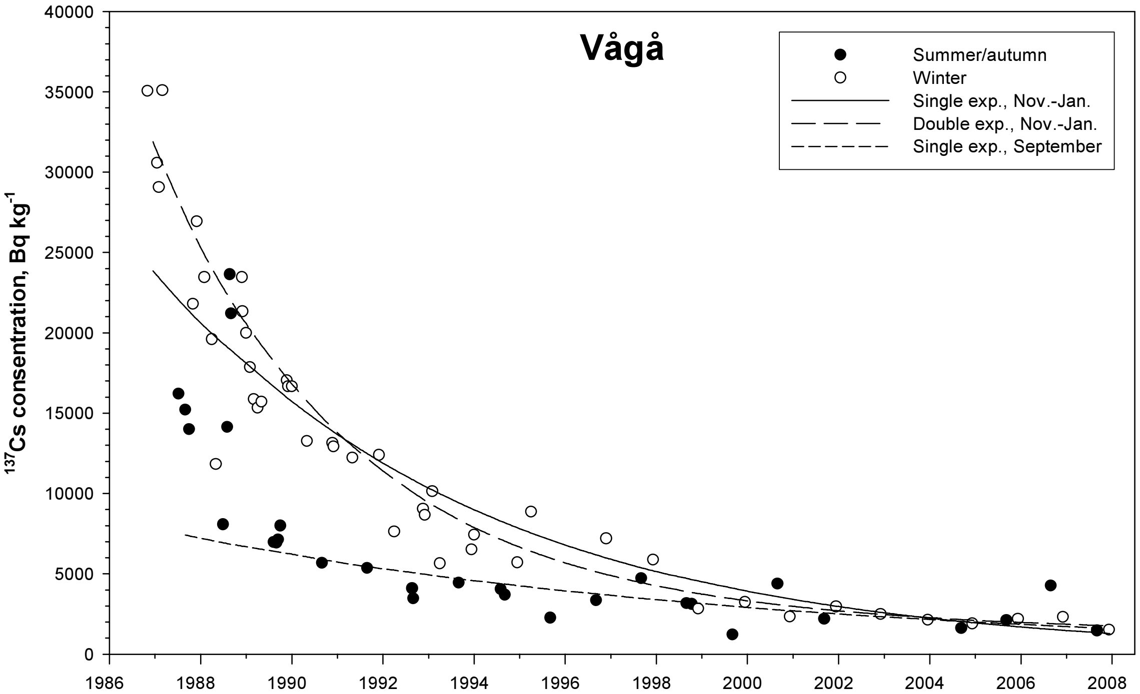

FIG. 3. Observed activity concentrations of ^{137}Cs in reindeer from Vågå reindeer herding districts in Norway. • indicates observed concentrations during July-October; o indicates concentrations observed during November-May. the curves are fitted single and double exponential models to values observed during September and November-January, respectively (see text below and table 3) (Skuterud, unpublished. results for the period up to 2003 were presented in Skuterud et al. [23]).

Golikov et al. [18] analysed ^{137}Cs in reindeer during winter at the Kola Peninsula (Russian Federation) and northern Norway from the 1960s onwards and suggested that the decline in radiocaesium concentrations can be described by a double exponential model with short and long term effective half-lives of 1-2 and 10-11 years respectively (Table 2). These were in agreement with the estimates for ^{137}Cs in lichens (Table 1). It may be tempting to discuss the similarity in these estimates and the general half-life estimates suggested by Smith et al. [25]. However, as long as contamination levels in lichens govern concentrations in reindeer, the processes discussed by Smith et al. [25] do not apply to lichen and reindeer.

TABLE 2. INITIAL ^{137}Cs AGGREGATED TRANSFER FACTORS (T_{ag}) FOR REINDEER MEAT (FRESH WEIGHT) AND ECOLOGICAL (T_1, T_2)/EFFECTIVE (T_{eff1}, $Teff_2$) HALF-LIVES IN MUSCLES OF REINDEER, [18]

Area	^{137}Cs					
	$T_{ag}(0)$, $m^2\ kg^{-1}$	a_1^1	T_1, y	T_2, y	T_1^{eff}, y	T_2^{eff}, y
Kola Peninsula	1.7	8.2×10^{-1}	2.0	1.8×10^1	1.9	1.1×10^1
Nenets Autonomous Okrug	1.2	8.1×10^{-1}	1.8	1.6×10^1	1.5	1.0×10^1
Kautokeino (Norway)	1.8	8.9×10^{-1}	1.2	1.8×10^1	1.2	1.1×10^1

^{1}The factor a_1 gives the fraction of the initial concentration in lichen declining with the short half-life T_1.

Observed ^{137}Cs concentrations in reindeer in central Sweden and Norway also indicate that concentrations declined faster during the initial period after the Chernobyl fallout than later [13, 23], as suggested by a double exponential model. However, due to the variability in the data it has not been possible to identify statistically significant long-term components of such models. An example is shown in Fig. 3: The double exponential model gives a better fit to the data, but the long-term half-life estimate of 27 years has a standard error of 84 years. As an alternative approach Åhman [13] divided the time period into the first 10 years and the last 10 years (year 10 – 20). Analysis of data from all herds together showed that the effective half life during the first period was considerably shorter (about 3.5 years for observations in Nov.-Dec.) than in the latter period (about 7 years for Nov.-Dec.) [13]. Results from three sites studied in detail by Åhman are given in Table 3 together with results from two Norwegian sites. Observed concentrations and fitted models are shown in Figs. 2 and 3 for the Vilhelmina norra and Vågå herds, respectively. According to the results in Table 3 the estimated effective half-lives during November-December in the Swedish herds correspond to those in the Norwegian herds during November-January, whereas the decline in September may be more rapid in the Swedish herds.

Tables 23 and 24 of the paper on radionuclide behaviour in forest ecosystems presented in the current IAEA-TECDOC[1] give overviews of various aggregated transfer factors and ecological and effective half-lives for radiocaesium in reindeer from the literature and gives reference to studies of ^{210}Po, ^{210}Pb, ^{226}Ra in reindeer and caribou in Alaska. There are relatively few long-term studies of ^{90}Sr available. In the Vågå herd in Norway concentration in reindeer calves indicate that ^{90}Sr concentrations in the reindeer diet decreased by a half-life of about 9 years during 1988-2002 [26]. In a feeding experiment the transfer of Sr to reindeer milk was estimated to be about 0.022 day kg^{-1}, much lower than the corresponding figure for Cs of 0.12 day kg^{-1} [26].

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[1] See paper 'Radionuclide transfer in forest ecosystems' by Calmon et al. in thispublication.

TABLE 3. Cs-137 AGGREGATED TRANSFER FACTORS FOR ^{137}CS IN REINDEER MEAT IN THE FIRST YEAR AFTER FALLOUT (T_{agi}) AND EFFECTIVE HALF-LIVES (T_{eff}) FOR DIFFERENT PERIODS AFTER THE CHERNOBYL ACCIDENT (after Åhman [13] and Skuterud (unpublished data)).

Country and herd	Season	T_{ag} (1986-1987) ($m^2\ kg^{-1}$)	T_{eff} (1-10) (y)	T_{eff} (10-20) (y)	T_{eff} (all years) (y)
Sweden: Vilhelmina norra, Ubmeje, Ran	September	$(1.1\text{-}2.4) \times 10^{-1}$	2.5-3.1	7.6- no decline	4.5-6.7
	October^b	$(2.7\text{-}3.9) \times 10^{-1}$	2.1-2.5	11.4-20.6	5.8-7.9
	Nov-Dec	$(4.7\text{-}8.1) \times 10^{-1}$	2.8-4.8	4.9-6.9	5.0-6.6
	Jan-Apr	0.92-1.2	4.5-7.0	7.5-10.4	5.1-6.8
Norway: Østre Namdal, Vågå	September		4.1-4.9	No decline	9.2-12.4
	Nov-Jan		3.9-4.1	6.6^c	4.8-5.0

^{a} Ranges are given for the individual herds; ^{b} In October there are no observations from the Ran site; ^{c} The estimate is 6.6 year for both sites, with standard errors of 0.8 and 1.5 year.

3.3.3 Transfer to milk, meat and other food products

Other food products in the Arctic have not been the subject of as intensive studies as reindeer meat. Even though T_{ag} values are appropriate for reporting radionuclide transfer for semi-natural food products, the activity concentrations of products from Arctic ecosystems are often not reported with relevant deposition data. As with reindeer meat, the values decrease rapidly in the first years after deposition and then more slowly and the decay is sometimes modeled with a double exponential function or by looking at the early phase and late phase separately. This was clearly illustrated in the AMAP compilation (Fig 4).

In most time series with an adequate sampling frequency, strong seasonal variations can be observed with higher ^{137}Cs and ^{90}Sr activity concentrations in the summer, when cows are put out to pasture or fed fresh grass. In some cases, the completeness of directly comparable time series has been affected by dairies closing down and consequent changes in the collection areas for those remaining. The results of the study showed two orders of magnitude difference between the lowest and highest values of the ^{137}Cs aggregated transfer factor (T_{ag}), from 0.4 at the Danish site to $4.7 \times 10^{-2}\ m^2\ kg^{-1}$ at the Swedish site. No evident time trend could be demonstrated during the 8 year study period, 1990 – 1997 (in Iceland a study site in a wet area was used in 1991-1993 wheras a dry area was used during 1995-1997, resulting in much lower T_{ag} values) [12].

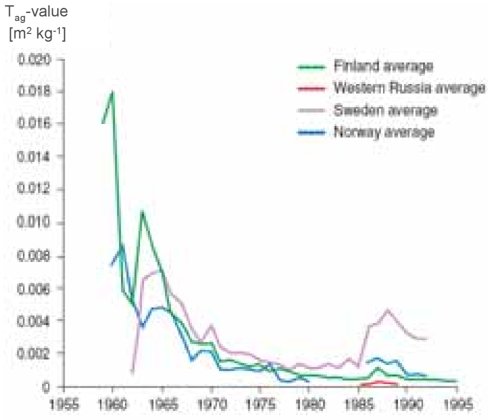

FIG. 4. Temporal variation in cow milk T_{ag} values in Sweden, Norway and Finland (after AMAP [1]).

Unlike for reindeer, the dominant mechanism for lowering the rate for transfer to milk is fixation of radiocaesium in the soil, type of fodder and its sources can also be important sources. Much of the fodder is likely to be produced locally, concentrates, however, may have been imported.

Tables 4 and 5 give available data on T_{ag} values (for early and late periods after the depositions) and appropriate information on variability of the effective ecological half lives in milk.

Similar effective half lives were also found in a Nordic NKS study using ^{137}Cs and ^{90}Sr time series (the oldest starting in 1959) from Denmark, the Faroe Islands and Finland [27]. The fast component of the effective half life was around 1 year for both radionuclides. The slow component for ^{137}Cs was 6-7 years for the nuclear weapons testing fallout whereas there was considerable geographical variation in the post-Chernobyl period: around 7 years for the Danish Islands to 13 years for western Finland. The slow component for ^{90}Sr showed also considerable variation, from around 4 years in the Faroe Islands to around 12 years in Finland (Kursu).

A recent study in Iceland using data from 2001-2004 gave ^{137}Cs geometric mean T_{ag} values to milk, ranging from 0.00073 – 0.0012 m^2 kg^{-1} [28]. Estimated values for 1966 were, however, in the range 0.0024 – 0.01 m^2 kg^{-1}, which in agreement with values recommend in Table 2 for $T_{ag}(0)$.

Although, it is not feasible to estimate $T_{ag}(0)$ and effective half lives from these values, they do not contradict the estimates for the parameters given earlier.

TABLE 4.SUMMARY OF T_{ag} VALUES FOR COW MILK, m^2 kg^{-1} FW

Phase	Region	Year(s)	T_{ag}	Reference
Early period ($T_{ag}(0)$)				
	Fennoscandia and NW Russian Federation		1.0×10^{-2}	[29]
	Arctic regions (esp. Norway)		1.0×10^{-2}	[30]
	Finnmark (Norway)		2.0×10^{-2}	[1]
	Troms (Norway)		9.0×10^{-3}	[1]
	Nordland (Norway)		1.4×10^{-2}	[1]
	Iceland	1965	7.6×10^{-3}	(Palsson, unpublished)
Late period				
	Lovozero (Russian Federation)	1998-1999	0.24×10^{-3}	[31]
	Kola region (Russian Federation)	1998-1999	0.15×10^{-3}	[31]
	Kola region (Russian Federation)	1974-1978	0.14×10^{-3}	[2]
	Nenets AO (Russian Federation)	1974-1978	0.12×10^{-3}	[2]
	Kola region (Russian Federation)	1978-1985	0.082×10^{-3}	[2]
	Nenets AO (Russian Federation)	1978-1985	0.062×10^{-3}	[2]
	Iceland	2001-2004	$1.1 x 10^{-3}$	(Palsson, unpublished)

TABLE 5. EFFECTIVE HALF-LIVES VALUES (YEARS, AM±SD) FOR ^{137}Cs AND ^{90}Sr ACTIVITY CONC. IN MILK FROM VARIOUS ARCTIC AREAS ([9])

Area/Source	Nuclide		Sites							
Faroe Islands			Klaksvik		Tỏrshavn		Tvǿroyri			
			AM	SD	AM	SD	AM	SD		
Global fallout	^{137}Cs	T_{eff1}	1.5	1×10^{-1}	1.0	1×10^{-1}	1.8	2×10^{-1}		
	^{137}Cs	T_{eff2}	7.1	5×10^{-1}	6.5	4×10^{-1}	8.8	7×10^{-1}		
Chernobyl fallout	^{137}Cs	T_{eff}	1.3	1×10^{-1}	1.8	1×10^{-1}	1.8	2×10^{-1}		
	^{90}Sr	T_{eff1}	1.0	1×10^{-1}	1.4	1×10^{-1}				
	^{90}Sr	T_{eff2}	5.2	1×10^{-1}	5.5	1×10^{-1}				
Finland			Apukka		Kusu		Rovaniemi			
			AM	SD	AM	SD	AM	SD		
Global fallout	^{137}Cs	T_{eff1}			1.0	1×10^{-1}				
	^{137}Cs	T_{eff2}			4.5	7×10^{-1}				
Chernobyl fallout	^{137}Cs	T_{eff}	0.7	1×10^{-1}			3.4	1×10^{-1}		
	^{90}Sr	T_{eff1}			1.3	1×10^{-1}				
	^{90}Sr	T_{eff2}			8.4	3×10^{-1}				
Norway			Bodǿ		Vadsǿ		Mǎlselv		Kautokeino	
			AM	SD	AM	SD	AM	SD	AM	SD
Global fallout	^{137}Cs	T_{eff1}	1.9	6×10^{-1}	1.6	4×10^{-1}	1.5	3×10^{-1}	1.1	3×10^{-1}
	^{137}Cs	T_{eff2}	4.5	1.2	5.1	1.3	6.1	1.2	6.0	2.0
Chernobyl fallout	^{137}Cs	T_{eff}								
	^{90}Sr	T_{eff1}	1.8	6×10^{-1}	n.a.[1]		n.a.		1.5	4×10^{-1}
	^{90}Sr	T_{eff2}	4.0	1.0	n.a.		n.a.		4.6	1.3
Sweden			Tärnaby		Vittiangi					
			AM	SD	AM	SD				
Global fallout	^{137}Cs	T_{eff1}	1.4	4×10^{-1}	1.8	6×10^{-1}				
	^{137}Cs	T_{eff2}	9.1	1.1	6.2	1.0				
Chernobyl fallout	^{137}Cs	T_{eff}								
	^{90}Sr	T_{eff1}	3.0	1.0	1.4	3×10^{-1}				
	^{90}Sr	T_{eff2}	9.0	2.0	8.5	1.0				

[1] n.a. – no statistically valid half-life could be calculated.

Information on some other food products, based on extensive literature search is given in Table 6 [30]. Transfer values and effective half-lives were preferentially based on Arctic data, and especially on data derived from Norway.

The reference values given in Table 6 for cow milk suggest that the slow removal process is associated with 20% of the initial value. The values in Table 4 are generally lower and Table 4 indicates faster removal processes. The corresponding reference value for sheep meat suggests that all the reduction is associated with a relatively fast process of a 7.6 year effective ecological half life and this seems to agree well with results from some of the study sites in the NKS study, whereas other (e.g. in Sweden and Iceland) clearly seem to have a slower removal process. A compilation of T_{ag} values for lamb meat is shown in Fig. 5 and average values for the first years are given in Table 7.

TABLE 6. Cs-137 and ^{90}Sr SUMMARY OF AGGREGATED TRANSFER FACTORS AND EFFECTIVE HALF LIVES IN SOME PRODUCTS DERIVED IN ARCTIC AREAS.

Product	T_{ag} (m^2 kg^{-1})	a_1	T_{eff1}, years	T_{eef}, years
^{137}Cs				
Milk and cheese	2.0×10^{-2}	0.8	2	20
Potato	5.0×10^{-4}	0.8	2	20
Beef	6.0×10^{-3}	0.8	2	20
Sheep and goat meat	1.5×10^{-1}	1.0	7.6	
Rindeer	1.4×10^{-0}	0.12	1	10
Freshwater fish	5.0×10^{-2}	0.8	1	10
Mushrooms	2.9×10^{-1}	1	30.2	
Berries	3.0×10^{-2}	1	30.2	
^{90}Sr				
Milk	3.0×10^{-3}	0.5	0.05	3.7
Potato	1.0×10^{-2}	1.0	8.7	
Beef	4.0×10^{-5}	1.0	6.2	
Sheep and goat meat	4.0×10^{-4}	1.0	6.2	
Rindeer	3.0×10^{-3}	1.0	8.3	
Freshwater fish	1.0×10^{-3}	1.0	9.75	
Mushrooms	2.0×10^{-4}	1.0	28.1	
Berries	6.0×10^{-3}	1.0	28.1	

TABLE 7. SUMMARY OF INITIAL AND LATE PHASE T_{ag} VALUES FOR LAMB (SHEEP) MEAT

Phase	Region	Year(s)	T_{ag} [m^2 kg^{-1}]	Reference
Early ($T_{ag}(0)$)				
	Fennoscandia and Northwest Russia		3.8×10^{-1}	[29]
	Arctic regions (esp. Norway)		1.5×10^{-1}	[30]
	Finnmark (Norway)		1.6×10^{-1}	[1]
	Troms (Norway)		6.3×10^{-1}	[1]
	Nordland (Norway)		1.4×10^{-1}	[1]
Late				
	Northern Sweden	1990-1997	4.7×10^{-2}	[1]
	Faroe Islands	1990-1997	$(5.5\text{-}2.5) \times 10^{-3}$	[32]
	Finland	1990-1993	0.83×10^{-3}	[20]
	Iceland	1990-1993	1.5×10^{-2}	[20])
	Norway	1990-1993	3.9×10^{-2}	[20])

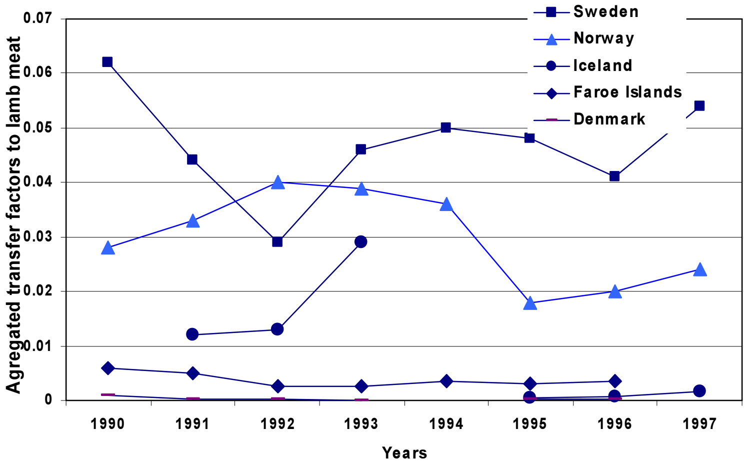

FIG. 5. Aggregated transfer factors [m^{2} kg^{-1}] of ^{7}Cs from soil to meat at the different study sites. (after Bergan [12]).

The results showed two orders of magnitude difference between the lowest and highest values of the ^{137}Cs aggregated transfer factor (T_{ag}), from0.4 at the Danish site to 4.7x10^{-2} m^{2} kg^{-1} at the Swedish site. No evident time trend could be found during the 8 year study period, 1990 – 1997 (in Iceland a study site in a wet area was used in 1991-1993 wheras a dry area was used during 1995-1997, resulting in much lower T_{ag} values) [12]. The study in the Faroe Islands showed T_{ag} values ranging from 0.001 – 0.01 m^{2} kg^{-1} [32], and Icelandic monitoring data have shown a similar range [28].

Even though the Faroe Islands cover a small geographical area, there was considerable spatial and temporal variation in the transfer of ^{137}Cs from soil to both grass and lamb meat. Owing to this high variability it was concluded that it is inappropriate to use a single T_{ag} value for either grass pasture or lamb meat, even for such a well defined region as the Faroe Islands.

In other countries, there is generally greater variation in the key soil characteristics influencing radiocaesium uptake than was measured at these sites. Even higher variability could thus be expected in other countries and country-wide generalizations about transfer are open to considerable error. Compilation of the data referred to in this paper shows that under many conditions the data can be modelled using a double term exponential model for the decline with time, but taking seasonal variability into account as appropriate.

4. RADIONUCLIDES IN ARCTIC LAKES AND RIVERS

Although many of the processes governing radionuclide behaviour in Arctic and northern alpine freshwaters are similar to those at lower latitudes, there are some differences. Snow and ice characterizes Arctic ecosystems and fallout during winter will remain on lake and river ice and give rise to a marked pulse of contamination during the spring snow melt [33]. Discharge and water renewal at this time are high, such that a major part of mobile radionuclides such as ^{90}Sr are likely to be transported downstream. In contrast radionuclides associated with particles, such as caesium isotopes, will be retained to a greater extent in lakes and slow flowing rivers [34, 35], although they may quickly sediment out, often resulting in a shorter effective half-life in river waters compared to strontium [9, 36]. Many terrestrial areas in the Arctic have a high percentage of organic boggy soils and radionuclide runoff to lakes and rivers from such areas is higher than from catchment with other soil types. In addition, contaminated riparian vegetation is an important food source for aquatic organisms as primary production within arctic water bodies is severely limited by low nutrient status, low temperatures and the short ice free period [34].

Lakes and rivers at northern latitudes are typically nutrient poor and have a low biomass. This increases the uptake of radionuclides in the food chain, compared to more eutrophic lowland freshwaters. This has been particularly apparent with caesium and strontium, analogues of potassium and calcium, respectively. This is reflected in the inverse relationship between radioactivity activity concentrations in fish and the respective concentrations of potassium and calcium in lake waters [37, 38]. Thus, radionuclide activity concentrations can be higher and effective half-lives longer in the Arctic and northern alpine areas [9, 34, 39, 40].

In addition to atmospheric fallout, the headwaters of many Arctic rivers, such as the Yenisey, Ob and Lena have been directly contaminated by nuclear processing facilities and levels well above background are still being measured even near the river mouth [9].

5. CONCLUSIONS

Arctic ecosystems show a considerable variability of radioecological parameters, which can be explained by seasonal effects, small-scale heterogeneity in soil and climate parameters, and specific habits of free-ranging animals. Nevertheless, the use of aggregated transfer factors seems to be the most practical approach for the prediction of contamination levels in food-products from such environments. Also, the long-term development of radionuclide concentration in food-stuffs can be estimated by two-fold exponential models.

In Arctic environments, the greatest amount of presently accumulated data for ^{90}Sr and ^{137}Cs belongs to the period of global deposition, i.e. deposition that lasted for many years though with time-varying intensity. In spite of the fact that significant levels of the naturally-occurring radioactive isotopes such as ^{210}Po and ^{210}Pb were found in northern Canada, Alaska and Russian Federation, information for radionuclides other than ^{90}Sr and ^{137}Cs is rather sparse and cannot readily be used for predictive purposes.

According to current literature, arctic ecosystems are especially sensitive to climate change phenomena. Increase of temperature will also influence all other driving factors like the organic matter decay velocities, vertical migration of radionuclides, plant uptake etc. Therefore, the radioecological future of arctic regions cannot be predicted with high accuracy and further investigation of processes for the distribution of radionuclides in food-chains of such ecosystems is recommended.

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RADIONUCLIDE TRANSFER IN ALPINE ECOSYSTEMS

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Abstract

Although alpine ecosystems are only of limited extent in Europe and the world, they serve as an important socio-economic basis for food production and tourism for the local population. The specific climate conditions and agricultural practice do have influence on radionuclide transfer in Alpine regions. The paper describes basic proceses governing radionuclide behaviour in Alpine ecosystems and gives parameters for vertical migration of radionuclide in soil, for soil-plant transfer as well as transfer of radionuclide to cow milk.

1. ENVIRONMENTAL FACTOR GOVERNING BEHAVIOUR OF RADIONUCLIDES IN ALPINE ECOSYSTEMS

Alpine ecosystems show a very distinct behaviour with respect to nutrient cycling and are very sensitive against contamination due to the specific prevailing climatic and environmental conditions. In general Alpine ecosystems are characterized by:

- high altitudes leading to extreme climatic conditions with high amounts of precipitation, (at least in Europe) a long-lasting snow cover and frozen soils in winter. When melting, big amounts of water rush down the slope immediately (too much for infiltration, leading to erosion, wash off) and a short vegetation period
- derived from these conditions slow soil building processes and slow decomposition of soil organic matter leading to low pH-values in soil, litter accumulation
- nutrient deficiency (no fertilisation, input of organic matter only via browsing animals, slow litter decay)
- plant species/communities adapted to such conditions with deep or widely extended rooting systems, nutrient storage in the root or litter layer, nutrient recycling from decaying plant parts
- shallow soil profiles on slopes and disturbed soil profiles with accumulation of eroded material in foothill positions; variable soil texture composition dependent on the position at the slope (fine material accumulated at foot-hills)
- driven by soil properties, alpine soils have a very specific soil microbiology with high amount of fungal biomass (in acid soils), high capacity to store Cs in superficial soil layers [1]

All these factors favour a retarded migration of radionuclides into deeper soil layers and a higher plant uptake of radionuclides and longer ecological half-lives in the food-chain in comparison to lowland ecosystems [2, 3].

Nevertheless, alpine regions have high economic value. They serve as basis for extensive agriculture. In summer meadows are used as browsing area for cows (milk and meat production) and for the production of winter feed for animals. Moreover these regions are prominent tourist attractions and serve as recreation areas for humans. Not much data are available from alpine ecosystems, a compilation of published information is given below.

2. VERTICAL MIGRATION OF RADIONUCLIDES IN SOILS OF ALPINE ECOSYSTEMS

The same approach as for agricultural soils[1] were applied for Alpine soil to characterize vertical migration of radionuclidesl, and CDE-approach is recommended to describe migration processes. Parameters of the Convection-Dispersion-Equation are [4]: *D*: effective (or apparent) dispersion coefficient (cm² a^{-1}) and *v*: convection velocity (cm a^{-1}). These values can be used in the convection-dispersion equation for a chosen time t to produce a vertical profile of the radionuclide.

Due to the small data base (only two references from peer-reviewed journals could be found) a comparison of migration parameters from alpine soils with lowland soils does not seem feasible. Nevertheless, the parameters in Table 1 are quite representative, because of a large number of field data used by Bossew and Kirchner [4].

TABLE 1. CDE–PARAMETERS *v* (DISPERSION VELOCITY) AND *D* (DISPERSION COEFFICIENT) AND MIGRATION RATES TO DESCRIBE VERTICAL MIGRATION IN ALPINE SOIL PROFILES

Parameter	N	GM	GSD$^{\pm1}$	AM	Med^{1}	CV%^{2}	SD	# Ref.
^{137}Cs, v(cm a^{-1}) (Gf^{3})	140			8.6×10^{-2}	8.5×10^{-2}	47	4.1×10^{-2}	1[4]
^{137}Cs, D (cm^{2} a^{-1}) (Gf)	150	4.5×10^{-2}	3.2		4.3×10^{-2}			1[4]
^{134}Cs, v (cm a^{-1}) (Ch^{4})	133			2.1×10^{-1}	1.5×10^{-1}	104	2.2×10^{-1}	1[4]
^{134}Cs, D (cm^{2} a^{-1}) (Ch)	223	1.7×10^{-1}	2.7		1.5×10^{-1}			1[4]
^{137}Cs, v (cm a^{-1}) (Ch)	100				1×10^{-3}			2[4, 5]
^{137}Cs, D (cm^{2} a^{-1}) (Ch)	100				4.8×10^{-1}			2[4, 5]
^{106}Ru, v (cm a^{-1}) (Ch)	55			3.5×10^{-1}	3.5×10^{-1}	76	3.1×10^{-1}	1[4]
^{106}Ru, D (cm^{2} a^{-1}) (Ch)	105	2.6×10^{-1}	2.7		2.4×10^{-1}			1[4]
^{125}Sb, v (cm a^{-1}) (Ch)	53			3.9×10^{-1}	2.9×10^{-1}	69	2.7×10^{-1}	1[4]
^{125}Sb, D (cm^{2} a^{-1}) (Ch)	87	2.6×10^{-1}	3.0		2.6×10^{-1}			1[4]
^{110m}Ag, v (cm a^{-1}) (Ch)	4			9.4×10^{-1}	8.4×10^{-1}	50	4.7×10^{-1}	1[4]
^{110m}Ag, D (cm^{2} a^{-1}) (Ch)	10	3.1×10^{-1}	3.5		1.9×10^{-1}			1[4]
^{144}Ce, v (cm a^{-1}) (Ch)	3			8.1×10^{-1}	6.8×10^{-1}	103	8.4×10^{-1}	1[4]
^{144}Ce, D (cm^{2} a^{-1}) (Ch)	4	7.6×10^{-1}	10		8.5×10^{-1}			1[4]
^{137}Cs (Ch) migration rate (cm a^{-1})	100				2.1×10^{-1}	range: 2.0×10^{-2} – 7.8×10^{-1}		1[5

^{1}Median value; ^{2}Coefficient of variance. ^{3}Global fallout; ^{4}Chernobyl fallout.

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[1] See paper by Strebl et al 'Vertical migration of radionuclides in undisturbed soils' in this publication.

3. SOIL-PLANT TRANSFER IN ALPINE ECOSYSTEMS

Similar like for other semi-natural systems the use of aggregated transfer factors has become popular for alpine ecosystems. This is due to the fact that variability of transfer factors is extremely high. Influencing factors are e.g. high variability of microclimatic conditions, small-scale variability of soil properties, and changing hydrological conditions. Available peer-reviewed data are given in Table 2.

TABLE 2. AGGREGATED TRANSFER FACTORS (T_{ag}; m^2 kg^{-1}) FOR ^{137}Cs AND ^{90}Sr FROM SOIL TO GRASSLAND VEGETATION IN ALPINE ECOSYSTEMS

Soil type	N	GM	GSD	AM	SD	Min	Max	Ref.
					^{137}Cs			
Sandy	8	1.4×10^{-2}	3.1	2.1×10^{-2}	1.5×10^{-2}	2×10^{-3}	4.3×10^{-2}	[1, 6, 7])
Loamy	4	3×10^{-3}	2.9	4×10^{-3}	6×10^{-3}	1×10^{-3}	1.3×10^{-2}	[6, 8, 9]
Unspec.	1	6×10^{-3}						[8]
All soils	13	8×10^{-3}	3.7	1.5×10^{-2}	1.5×10^{-2}	1×10^{-3}	4.3×10^{-2}	_
					^{90}Sr			
All soils	3	2.6×10^{-2}	2.1	3.0×10^{-2}	1.8×10^{-2}	1.1×10^{-2}	4.7×10^{-2}	[6, 10]

TABLE 3 MILK FEED TRANSFER COEFFICIENTS F_m [d l^{-1}] FOR ^{137}Cs AND ^{90}Sr.

Site	AM	SD[a]	Min	Max	Reference
			^{137}Cs		
Lowland	9×10^{-4}	8×10^{-4}	-	-	[12]
Silicate bedrock	7.1×10^{-3}	9×10^{-4}	3.5×10^{-3}	1.1×10^{-2}	[12, 13]
Calcareous bedrock	6.9×10^{-3}	1.3×10^{-3}	2.5×10^{-3}	2×10^{-2}	[11-13]
			^{90}Sr		
Lowland	8×10^{-4}	3×10^{-4}	-	-	[12]
Silicate bedrock	1.1×10^{-3}	4×10^{-4}	5×10^{-4}	1.7×10^{-3}	[12, 13]
Calcareous bedrock	1.0×10^{-3}	8×10^{-4}	5×10^{-4}	1.0×10^{-3}	[12, 13]

[a] SD = standard deviation derived from error propagation.

4. FOOD-CHAIN TRANSPORT IN ALPINE ECOSYSTEMS

In the peer reviewed literature available data for the food chain transport of radionuclides in alpine ecosystems are restricted to the transfer from plant-to-milk, and only three (3) peer reviewed articles on the topic were available at the time of the data assessment.

For the transfer of radionuclides from feed to milk, the parameter F_m, (d l^{-1}) is used, which is the *equilibrium* ratio of the activity concentration in milk related to the daily dietary radionuclide intake. Feed transfer coefficient were determined in milk from semi-natural extensive production regions with calcareous and silicate bedrock [11-13]), the two basically different rock units, and in one intensively productive lowland region for comparison [12]. Concerning the time trends of ^{137}Cs contamination of cow milk from Alpine pastures, considerably longer ecological (or effective) half-lives have been observed in comparison to lowland production sites. Similar to the Arctic ecosystems, also here a two-component exponential decay function can be applied for the description of the decrease in concentrations over the years. For the period 1988 – 2006 Lettner et al. [14] derived ecological half-lives of 0.7/1.4 years for the fast initial period and 9.3/12.7 for the long-term decay component of ^{137}Cs concentration in cow milk. Keeping into account the difference between ecological and effective half-lives (which includes physical decay of radionuclides), these values are very similar compared to the results from Arctic environments (see Table 5). The feed transfer coefficients for ^{137}Cs in lowland region is significantly lower than on alpine production sites, however for ^{90}Sr no difference could be found. Also no significant differences for both radionuclides could be determined between milk feed transfer factors on silicate and calcareous bedrock.

5. CONCLUSIONS

Like in Arctic environments, radionuclide behaviour in Alpine ecosystems is strongly dependent on site-specific conditions, indicating a considerable variability of radionuclide transfer parameters and their high sensitivity to climate changes. The melting of glaciers caused by temperature increase will remobilize a considerable amount of radionuclide inventories stored in permanent ice (both from global fallout, but also Chernobyl fallout in alpine regions). The same as for Arctic ecosystems, long-term predictions of radionuclides redistributions in Alpine regions are associated with high uncertainty and further investigation of processes for the migration of radionuclides in food-chains of such ecosystems is necessary.

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RADIONUCLIDE TRANSFERS IN FRESHWATER ECOSYSTEMS

Radionuclides dispersed in the environment can be deposited into water surface or on surface of the watershed, which can represent in this case a long-term source of radionuclides for the freshwater ecosystems. The first pathway represents direct source of radionuclides to the freshwater ecosystem and can be easily simulated by the specification of the relevant input model parameters. The second pathway includes a set of processes and radionuclides can be transferred to water body being dissolved in flowing water or with soil particles adhered radionuclides. In water radionuclides can be adsorbed by the solid particles, portioning between the solid and water. The solid particles can settle out to the bottom of the like or river and be removed from the water column. Radionuclides dissolved in water can also be adsorbed by the bottom sediments and be transferring after that to the deep sediment layers. At the same time, adsorbed radionuclides can be remobilised, becoming available again for uptake by freshwater biota.

TRANSFER BY WASH-OFF FROM WATERSHEDS

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Abstract

Radionuclide transport with water from contaminated watersheds is an intermittent flux – also called watershed wash-off – which causes radionuclide redistribution in terrestrial ecosystems, and more critically radionuclide delivery to downstream water bodies. Mean wash-off fluxes and their evolution with time can be generally quantified via two types of parameters: liquid and solid entrainment coefficients and transfer functions, which can be seen as transfer factors varying with time. The paper provides an extensive review covering various radionuclides, source terms, processes, time and space scales reported in the literature after 1960.

1. INTRODUCTION

Radionuclide wash-off from watersheds is a secondary pathway of contamination of ecosystems. It is both a term of radionuclide redistribution in terrestrial ecosystems [1-2], and a delivery term for aquatic ecosystems [3]. From a terrestrial point of view, wash-off is not an effective process of decontamination. The removed amounts and removal rates are not significant enough to be considered in food chain models [4]. But from an aquatic point of view, this way of contamination is important and persistent because it collects fluxes from wide potentially contaminated areas that remain affected by precipitations [5]. These transfers have been observed since the beginning of atmospheric bomb tests [3], and became much more evident after the Chernobyl accident [6].

2. MAIN PROCESSES

Radionuclide wash-off is the transport of contamination by flowing water over the soil surface (run-off). Actually two pathways can be distinguished:

- liquid wash-off: when radionuclide is transported in soluble phase with water;
- solid wash-off: when radionuclide is transported in particulate form with suspended matter.

Wash-off is particularly intense during and within few weeks after deposition. This stage is referred as short-term wash-off. Wash-off is significantly lower for larger delays after deposition, but is still active years and decades after deposition. This stage is referred as long-term wash-off.

Short-term liquid wash-off originates in:

- direct deposition on surface water without passage in the soil store [7];
- non-equilibrium sorption on the soil surface during wet deposition (substantial rainfall and high fallout input). The excess contamination is rapidly exported with runoff [8-9].

Its duration over few weeks is due to transit times of water in watersheds and extended deposit time to soil by vegetation wash-off [10].

Long-term liquid wash-off results from dilution of soil water with runoff water, and desorption from a thin soil layer [11]. Desorption seems to occur at equilibrium [12], for both ^{137}Cs and ^{90}Sr, except when runoff of ^{90}Sr is intense like in rills and gullies [13]. Availability of the radionuclide to liquid wash-off generally decreases with time, because of migration for strontium [13], and essentially ageing for caesium [14].

Short-term solid wash-off [2, 15] is a direct transfer from rainfall to suspended matter by adsorption. It is active during the deposition period or while plant leaching is significant [10]. Soil disaggregating by contaminated rain explains the rapid mobilization of contaminated particles [16].

Long-term solid wash-off is caused by detachment and transport of soil particles by the combined action of rain and runoff [17]. During this stage, rainfall water is uncontaminated and conveys soil particles previously contaminated.

Major co-factors influencing wash-off processes are:

- the radionuclide: most of the operating processes are chemical dependent;
- the type of contamination: for a given radionuclide, the nature of the contamination can affect its availability to wash-off, especially for liquid wash-off. Important factors are: initial exchangeable fraction of the contamination, nature of the source (bomb, Chernobyl);
- the pathway: if the considered flux is the liquid, solid, or total wash-off;
- space and time scales: if the flux integrates the radionuclide entrainment at the plot/rainfall scale or at the watershed/season scale;
- the advection fluxes: wash-off stems from the movement of water at the soil surface. The climate (arid ,humid, with/without snow cover [18]) and soil properties (erodibility, vegetation cover [19]) are determinant. The abundance of surface water influences the rapid flush [7] and more generally the dilution between deposition and water concentration [20];
- the nature of the soils in the watersheds: An important literature has focused on the higher wash-off rates of Caesium in (organic) peaty soils [21].

3. CONVENTIONAL PARAMETERS

Usual parameters quantify wash-off fluxes (Bq/s) exported from watersheds. Two complementary approaches appear in the literature:

- Entrainment coefficients, at small time scales (e.g. rainfall, season);
- Transfer functions, at larger time scales (e.g. months, years). Wash-off is here described as continuous though it is intermittent by nature.

3.1. Entrainment coefficients

Liquid/solid normalized entrainment coefficients (noted K_l^*, K_s^*) are transfer factors between mean soil contamination and mean liquid/solid contamination of water [12, 22]. For a given radionuclide M, their expression is:

$$K_l^* = \frac{\overline{[M]_{wat}}}{\overline{[M]_{soil}}} \qquad (\text{m}^{-1}) \qquad (1)$$

$$K_s^* = \frac{\overline{[M]_{sm}}}{\overline{[M]_{soil}}} \qquad (\text{m}^2/\text{g}) \qquad (2)$$

where $\overline{[M]_{soil}}$ (Bq/m^2) is the mean initial contamination density of the soil, $\overline{[M]_{wat}}$ (Bq/L) is the mean soluble M concentration in runoff water, $\overline{[M]_{sm}}$ (Bq/g) is the mean M concentration in suspended matter.

Normalized entrainment coefficients span approximately one order of magnitude for ^{137}Cs and ^{90}Sr (Table 1). At a given site, liquid wash-off coefficients are generally one order of magnitude higher for strontium than for caesium. The situation is opposite with these two radionuclides for solid wash-off.

TABLE 1. REPORTED VALUES OF NORMALIZED ENTRAINMENT COEFFICIENTS FOR LIQUID AND SOLID WASH-OFF

Process	Coefficient	Nucl.	Min.	Max.	Units	References
Liquid wash-off	K_l^*	^{144}Ce	2.0×10^{-3}	6.0×10^{-3}	m^{-1}	[22]
Liquid wash-off	K_l^*	^{134}Cs	3.0×10^{-3}	3.5×10^{-2}	m^{-1}	[22]
Liquid wash-off	K_l^*	^{137}Cs	1.9×10^{-3}	1.2×10^{-1}	m^{-1}	[4, 12, 22]
Liquid wash-off	K_l^*	^{103}Ru	7.0×10^{-3}	1.6×10^{-2}	m^{-1}	[22]
Liquid wash-off	K_l^*	^{90}Sr	1.9×10^{-3}	1.8×10^{-1}	m^{-1}	[4, 12, 22-23]
Solid wash-off	K_s^*	^{137}Cs	1.6×10^{-5}	6.7×10^{-4}	m^2/g	[4, 12]
Solid wash-off	K_s^*	^{90}Sr	6.5×10^{-6}	3.1×10^{-4}	m^2/g	|4, 12]

Most of the values are derived from experimental plots submitted to one or two rainfalls. These values thus stress the intrinsic variability of wash-off episodes within a year. Just few rainfalls can lead to the most important inventory losses of a whole season [10].

The reported values are very specific but seem pessimistic. They concern essentially areas of Ukraine close to Chernobyl, where soils are not cultivated and have a light texture (loamy-sand or sandy-loam). All sites are in the exclusion zone, where hot particles deposited. The experiments were conducted with long and intense rainfalls to estimate maximum entrainment coefficients.

These coefficients can be used to estimate radionuclide wash-off due to non-contaminating rainfalls. We will consider a watershed with initial soil contamination $\overline{[M]_{soil}}$ (Bq/m^2).

Concentrations. Liquid and solid radionuclide concentrations at the outlet are:

$$\overline{[M]_{wat}} = \overline{[M]_{soil}} \times K_l^* \qquad \text{(Bq/m}^3\text{)} \qquad (3)$$

$$\overline{[M]_{sm}} = \overline{[M]_{soil}} \times K_s^* \qquad \text{(Bq/g)} \qquad (4)$$

Fluxes. For a water flow rate Q_{wat} (m^3/s) and a suspended matter flow rate Q_{sm} (g/s) at the outlet, liquid and solid wash-off fluxes are:

$$Q_{wat} \times \overline{[M]_{soil}} \times K_l^* \qquad \text{(Bq/s)} \qquad (5)$$

$$Q_{sm} \times \overline{[M]_{soil}} \times K_s^* \qquad \text{(Bq/s)} \qquad (6)$$

The effective value of coefficients K_l^* and K_s^* can be replaced by reported values of Table 1, with the usual limits for such extrapolations.

3.2. Transfer functions

The wash-off transfer function f of a watershed is its characteristic wash-off flux as a function of time after a unit pulse of contamination by atmospheric deposition. It is expressed as a fraction of the initial deposit exported per unit of time (s^{-1}).

Wash-off transfer functions depend on the considered watershed, radionuclide and wash-off process (liquid, solid or total).

The following transfer function synthesizes the data reported for global [24-26] and Chernobyl fallout [27, 28]:

$$f(t) = f_0\lambda_0 \cdot \exp\left[-(\lambda_0 + \lambda)t\right] + f_l\lambda_1 \cdot \exp\left[-(\lambda_1 + \lambda)t\right] \qquad \text{(s}^{-1}\text{)} \qquad (7)$$

Where f_0 is the fraction of deposited contamination affected by short-term (rapid) wash-off; f_1 is the fraction of deposited contamination affected by long-term (slow) wash-off; λ (s^{-1}) is the radioactive decay rate; λ_0 (s^{-1}) is the rapid wash-off rate constant (few weeks); λ_1 (s^{-1}) is the slow wash-off rate constant (few years). This analytical formulation distinguishes two characteristic time scales for wash-off: within a year and years.

Short-term wash-off (few weeks) can represent more than a percent of the deposit for very soluble radionuclides like ^{90}Sr, ^{103}Ru, ^{131}I and ^{132}Te (Table 2).

For the other reported radionuclides, extreme values of f_0 generally corresponds to estimations at small space scale. For watershed scenarios, f_0 should be considered of the order of a percent. The different forms of wash-off (total, liquid, solid) were distinguished when enough data were available.

TABLE 2. FRACTION OF DEPOSITED CONTAMINATION AFFECTED BY SHORT-TERM WASH-OFF (f_0), %

Process	Radionuclide	Min.	Max.	References	#ref
Liquid wash-off	^{137}Cs	0.007	1.7	[10, 24, 26, 27, 29-32]	8
Liquid wash-off	^{131}I	4	10	[32]	1
Liquid wash-off	^{103}Ru	4	10	[32]	1
Liquid wash-off	^{90}Sr	0.63	10	[25, 26]	2
Liquid wash-off	^{132}Te	5	10	[32]	1
Solid wash-off	^{7}Be	0.6	2.3	[32]	1
Total wash-off	^{137}Cs	0.2	7.4	[6, 28, 34, 35]	4
Total wash-off	^{131}I	0.14		[6]	1
Total wash-off	^{103}Ru	1.15		[6]	1
Total wash-off	^{90}Sr	0.10	7.2	[3, 35, 51, 53]	4

The major part (around 95%) of rapid wash-off has occurred approximately T_0=45 days after deposition (e.g. [19, 30, 31]). It can thus be recommended:

$$\lambda_0 = -\frac{\ln(1-0.95)}{T_0} \approx 24 \qquad (\mathrm{yr}^{-1}) \qquad (8)$$

Long-term wash-off (years). A pessimistic (in terms of the radionuclide transfer to water bodies) assessment consists in assuming that all the watershed contamination is accessible to wash-off:

$$f_1 = 1 - f_0 \qquad (-) \qquad (9)$$

Slow wash-off rate constant λ_1 generally represents less than a percent per year of the initial deposit (Table 3). Variability of ^{137}Cs and ^{90}Sr coefficients is partly explained by the nature of the soils. Organic soils significantly increase radionuclide availability for wash-off.

Wash-off fluxes after an accidental deposit. This approach supposes that wash-off is the same for similar events happening at different times (*time invariance principle*), and is proportional to the deposited amount (*proportionality principle*). After an accidental deposit $dA(t_0)$ (Bq/m^2) occurring at time t_0, the wash-off flow rate f (Bq/s) at the outlet is then:

$$\Phi(t) = S \cdot dA(t_0) \cdot f(t - t_0) \qquad (\mathrm{Bq/s}) \qquad (10)$$

Where t (yr) is the time; S (m^2) is the surface of the watershed; f (/s) is the transfer function of the watershed, modelling its wash-off response.

Wash-off fluxes after a chronic deposit. This approach also assumes that wash-off adds up the responses to individual deposition events (*superposition principle*).

If a watershed is submitted to a varying atmospheric deposition rate D (Bq/m^2/s), the wash-off flow rate $\Phi(t)$ (Bq/s) at the outlet is then:

$$\Phi(t) = S \cdot \int D(\tau) \cdot f(t - \tau) \cdot d\tau \qquad (\mathrm{Bq/s}) \qquad (11)$$

TABLE 3. SLOW RATE CONSTANT OF RADIONUCLIDE WASH-OFF λ_1 , a^{-1}

Process	Radionuclide	Min.	Max.	References	#ref
Solid wash-off	^{241}Am	1.0×10^{-4}	2.0×10^{-4}	[36, 37]	2
Solid wash-off	^{7}Be	2.0×10^{-2}		[37]	1
Liquid wash-off	^{137}Cs	6.7×10^{-5}	2×10^{-2}	[10, 26, 29-30, 36, 38-39, 55]	7
Solid wash-off	^{137}Cs	8.9×10^{-5}	1.2×10^{-1}	[1, 7, 10, 33, 36, 37, 40, 41]	8
Total wash-off	^{137}Cs	3.8×10^{-5}	1×10^{-2}	[6, 18, 28, 34, 35, 36, 40, 43, 45, 48, 49, 54]	11
Total wash-off	^{131}I	1.1×10^{-1}		[45]	2
Solid wash-off	^{210}Pb	3.3×10^{-4}	7.1×10^{-4}	[33, 37]	2
Total wash-off	^{210}Pb	3.3×10^{-4}	1.3×10^{-3}	[46,47]	2
Solid wash-off	^{210}Pu	2×10^{-5}	6×10^{-3}	[36, 37, 7, 40, 44]	5
Total wash-off	^{210}Pu	1×10^{-4}	1.2×10^{-3}	[24, 42, 43]	3
Total wash-off	^{103}Ru	2×10^{-2}		[6]	1
Liquid wash-off	^{90}Sr	6.7×10^{-4}	5×10^{-2}	[13, 24-26, 35, 50, 52]	7
Total wash-off	^{90}Sr	1.1×10^{-3}	2×10^{-2}	[24, 49, 51]	3

In the expression of f (Eq. 7), the effective value of coefficients f_0 and λ_1 can be replaced by reported values of Tables 2 and 3, with the usual limits of such extrapolations.

Concentrations. For a water flow rate Q_{wat} (m^3/s) and a suspended matter flow rate Q_{sm} (g/s) at the outlet, mean soluble and particulate radionuclide concentrations are:

$$\overline{[M]_{wat}} = \Phi_l(t)/Q_{wat} \qquad \text{(Bq/m}^3\text{)} \tag{12}$$

$$\overline{[M]_{sm}} = \Phi_s(t)/Q_{sm} \qquad \text{(Bq/g)} \tag{13}$$

where Φ_l and Φ_s correspond to liquid and solid wash-off fluxes. These estimated concentrations are representative of average variations over months and years. For short term assessments (below the month), use entrainment coefficients.

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APPENDIX

list of publication used for evaluation of parameters of wash-off from watersheds

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FRESHWATER ECOSYSTEMS

PHYSICAL PROCESSES IN FRESHWATER ECOSYSTEMS

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Abstract

In the present paper the main methodological approaches to model the physical processes controlling the migration of radionuclides through water ecosystems are presented and discussed. These processes include the dispersion and the transport through the water and the migration from and to the bottom sediments caused by the settling and the re-suspension of contaminated particles of suspended matter. The equations that control the above processes and that are used by most existing state-of-the-art models are listed and briefly discussed and motivated. Values of the parameters in the equations have been selected following a review of information available from the scientific literature.

1. PROCESSES INVOLVED

Modelling the behaviour of contaminants in aquatic ecosystems requires the quantitative assessment of phenomena of hydraulic, geochemical, sedimentological, ecological and anthropogenic nature.

Hydraulic processes are mainly responsible of the transport of toxic substances by water, whereas geochemical processes influence the interaction of dissolved radionuclides with suspended matter and bottom sediments. Sedimentation and resuspension are of importance for controlling the two-way migration of radionuclides from the water column to the bottom sediments and *vice versa*. Thus, the main physical processes controlling the migration of radionuclides in water bodies are [1]:

1) Diffusion and dispersion caused by gradients of concentration and the turbulent motion of water;
2) Transport caused by the water current;
3) Exchanges of contaminants between the dissolved and solid phases;
4) Deposition and re-mobilisation of the radionuclide following the interaction with suspended matter and bottom sediments;

Processes 1) and 2) are independent of the radionuclide characteristics, whereas processes 3) and 4), being related to physical and chemical properties of the pollutant, depend on the radionuclide.

Recently models for predicting the migration of radionuclides through the abiotic components of the freshwater ecosystem have been analysed and assessed [1-3]. This paper describes the physical processes that are accounted for by state-of-the-art models to predict the behaviour of radioactive substances in water bodies and provides a review of values of the parameters in the equations describing such processes.

2. DIFFUSION AND DISPERSION PROCESSES

Diffusion is the process that implies a movement of molecules from regions of space of high concentration to regions of low concentration. The radionuclide flux, F (Bq m^{-2} s^{-1}), due to diffusion is related to the concentration gradient (Fick's first law):

$$F = -E \cdot grad(C) \tag{1}$$

where *grad* is the gradient of C ($grad(C) = i\frac{\partial}{\partial x}C + j\frac{\partial}{\partial y}C + k\frac{\partial}{\partial z}C$), C is the radionuclide concentration in water, and *i, j* and *k* are the unit vectors along the coordinate axes × , y and z. E is a 3x3 component symmetric tensor (the diffusion tensor).

The values of the diffusion tensor components for molecular diffusion depend on the temperature and the mass of particles. Consequently, they depend on the specific features of radionuclide. The order of magnitude of E in water ranges from 10^{-10} to 10^{-6} m^2 s^{-1}.

Diffusion originates from a thermal motion of molecules (molecular diffusion). However, the diffusion of a pollutant due to disordered, turbulent motion of water is usually also modelled by equation (1) by analogy with Fick's first law.

In rivers, the water current is not uniform across the water column, having a maximum value at the surface and decreasing with depth due to friction against the bottom. This velocity profile is the cause of shear dispersion. If a radionuclide is released at the surface, it is partially transported downwards in the water column due to diffusion. However, radionuclides at the surface are advected faster than radionuclides at deeper locations since the velocity at the surface is greater. Thus, the volume of contamination suffers a deformation as indicated in Figure 1, along the direction of the current in the upper part of the water column.

The overall result is an enhanced dispersion in the current direction. Shear dispersion is included in 3-dimensional diffusion models, since they solve the vertical current profile, but not in the case of 2-dimensional, depth-averaged models. Horizontal current shear also produces shear dispersion. In the case of a river, currents are stronger at the centre than along shores due to friction. The shear effect will enhance mixing along the channel. In 2-dimensional diffusion models this horizontal shear is included since the hydrodynamic model automatically produces the current distribution outlined above. However, if a 1-dimensional model is applied, the effective diffusion coefficient must be increased to account for this horizontal shear. The effective dispersion coefficient depends on the spatial grid size used in the numerical solution of the diffusion equation: coarse grids require larger diffusion coefficients because larger eddies cannot be solved and their effects have to be described as turbulent mixing.

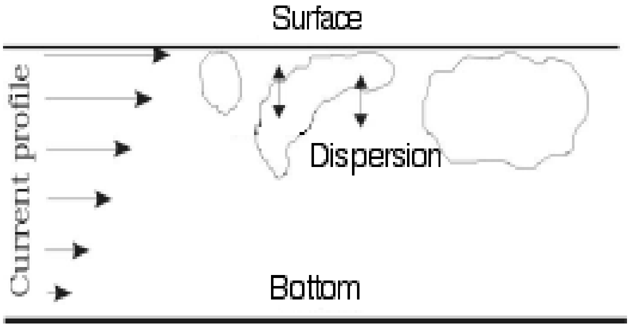

FIG. 1. Illustration of shear effect.

In conclusion, the water turbulences and the effect of bottom surfaces, shores and obstacles are responsible of a disordered motion of pollutant that is usually assumed to be controlled by equation (1) [12]. From now on, to be consistent with [12], we will call this comprehensive process “contaminant dispersion”, whereas the ordered motion due to the transport by water currents will be called simply ‘contaminant advection’. The description of the theoretical foundations of the diffusion and dispersion concepts is out of the scope of the present section. Interested readers can find more details in the specialised literature [4].

The molecular diffusion process is negligible compared with the turbulent motion of water. Consequently, the values of the overall dispersion coefficient are independent of the characteristics of the radionuclide.

Dispersion is, in principle, an anisotropic processes occurring in the three dimensions. Anisotropic dispersion can be important in deep and large water bodies. The radionuclide fluxes in longitudinal (×), transversal (y) and vertical directions (z) are as follows:

$$-E_x\frac{\partial C}{\partial x};\quad -E_y\frac{\partial C}{\partial y};\quad -E_z\frac{\partial C}{\partial z}.$$

2.1. Vertical dispersion

Dispersion in the vertical direction is influenced by the stratification of water bodies. In shallow or well-mixed water bodies, vertical dispersion is a rapid process. Consequently, it is commonly assumed that the concentration of a radionuclide in the water column quickly reaches a homogeneous vertical profile. The vertical dispersion coefficient in a river can be assessed by the following approximate equation [4]:

$$E_z = 6.7\cdot 10^{-2}\cdot h\cdot u^* \approx 6.7\cdot 10^{-3}\cdot h\cdot U \tag{2}$$

where h is the water depth (m), u^* is the friction velocity (m.s^{-1}) that we have assumed to be 1/10 of U, the average flow velocity (m.s^{-1}). Assuming, for instance, a current velocity of the order of 1 m s^{-1} and a depth of few meters, the order of magnitude of the vertical diffusion coefficient would be 10^{-2}m^{2}s^{-1}.

In deep water bodies, different densities of water layers may induce water stratification that will give low values of the dispersion coefficient E_z. A typical example is the thermal stratification of water characterised by the formation of the thermocline, a sharp vertical gradient in temperature, in an intermediate layer of the water column. Thermal stratification is a seasonal process very common in deep lakes. Different salinity levels of marine and fresh waters in coastal areas can also give rise to stratification. In such cases, a halocline forms.

Similarly, a chemocline is defined as a water layer showing a significant gradient of a dissolved chemical substance and a lutocline is a layer showing a gradient in suspended matter concentrations.

Several empirical estimates of the vertical dispersion coefficient in lakes are available from the literature (Table 1). It should be noted that the term diffusion is frequently used to denote this process as, in the absence of shear stresses in lakes, turbulences are the main responsible of disordered water mixing.

The values in Table 1 were generally obtained by calibration of the diffusion equation for predicting the distribution of substances dissolved in the water column. A selection of values was previously reported in [8] from an analysis of literature data of parameters used in ecological models. The values for the epilimnion (the upper well-mixed water layer in stratified lakes) were of the order of 10^{-2}–10^{-4} m^2 s^{-1}. Such high values are induced by mechanical processes of water turbulence, such as wind action, and by the daily cycles of water warming and cooling that are common to the different kinds of water bodies.

The vertical dispersion coefficient through the thermocline (or chemocline/halocline), during the water stratification, is orders of magnitude lower ($6. \times 10^{-9}$–10^{-6} m^2 s^{-1}). The minimum values of E_z in [9] were measured at specific hours of the day. In general, the daily maximum values were at least one order of magnitude higher. Deep lakes show higher values of the vertical dispersion coefficient in the epilimnion than shallow lakes. Consequently, from the values here reported, it is possible to conclude that the contaminant is homogeneously distributed in the epilimnion of a lake after a few days. Intermediate values of E_z were obtained for the deep layers below the thermocline (hypolimnion).

Stratification in rivers is less frequent than in lakes due to the mechanical mixing processes induced by the water current, the obstacles and the varying morphometric characteristics that can cause water turbulence along the river course. Nevertheless, the water column of deep, gently meandering rivers can be affected by thermal stratification processes. E_z may also be calculated using turbulence models [11].

2.2. Transversal and longitudinal dispersion

Horizontal transport in lake surface water is caused by currents and also induced by the effect of wind. Changes in wind intensity and direction can induce disorganised water movements and mixing that can be modelled, on the long term, by the diffusion equation . The horizontal dispersion coefficient in lakes can be several orders of magnitude higher than E_z through the thermocline of stratified waters. Values ranging from 10^{-3} to 10^{-2} m^2 s^{-1} were measured by Quay, et al. [5].

TABLE 1. SELECTED VALUES OF THE VERTICAL DISPERSION COEFFICIENT IN LAKES

Values ($m^2\ s^{-1}$)	Conditions	Methodology	Reference
10^{-9}-10^{-8}	Lakes, thermocline	Injection of tritiated water in two small temperate dimictic lakes.	[5]
1.5×10^{-7}	Diffusion across chemocline in stagnant lake	Calibration of a model	[6]
$1. \times 10^{-6}$	Diffusion across thermocline during stratification. Lakes in central Italy.	Calibration of a model to predict the diffusion of dissolved ^{134}Cs of Chernobyl origin in water	[7]
10^{-2}	Lakes, epilimnion	Collection of literature data	[8]
$6. \times 10^{-5}$ -$1. \times 10^{-2}$	Epilimnion, lakes Erie and Ontario (Canada)	H_2O_2 used as a natural, photochemically produced tracer. Values were measured at specific hours of the day (minimum value at 15 h, maximum at 12 h).	[9]
$5. \times 10^{-6}$ -$5. \times 10^{-4}$	Epilimnion, Jacks Lake (Canada)	H_2O_2 used as a natural, photochemically produced tracer. Values were measured at specific hours of the day (minimum value at 17 h, maximum at 14 h).	[9]
$6. \times 10^{-4}$	Lakes in central Italy. From epilimnion to deep water during water mixing	Calibration of a model to predict the diffusion of dissolved ^{134}Cs of Chernobyl origin in water	[7]
10^{-7}–10^{-5}	Diffusion in deep water layer. Lake Baikal	Calibration of a model to predict diffusion of substances	[10]
10^{-7}-10^{-6}	Lakes, hypolimnion	Injection of tritiated water in two small temperate dimictic lakes	[5]
$1. \times 10^{-4}$	Diffusion in deep layers below thermocline.	Collection of data from a literature search	[8]
		Suggested ranges and values	
Range ($m^2\ s^{-1}$)	Value ($m^2\ s^{-1}$)	Water layer	
$< 10^{-6}$	10^{-7}	Through thermocline (during stratification)	
$> 10^{-4}$	10^{-3}	Epilimnion	
10^{-7} – 10^{-4}	10^{-4} (accounting for the different stratification conditions during the year)	Hypolimnion	

Evaluations of longitudinal and transversal dispersion coefficients in rivers are available from the literature. IAEA suggests values ranging from $7.6\ 10^{-1}$ to $1.5\ 10^{3}$ $m^2\ s^{-1}$ and from $4.8\ 10^{-3}$ to 1.1 $m^2\ s^{-1}$ for, respectively, the longitudinal (E_x) and the transverse (E_y) dispersion coefficients in several North America rivers [12]. It is generally recognised that values of these parameters are higher in large rivers. Moreover, the values of the dispersion coefficients in a river are strongly influenced by the water velocity. Turbulence effects imply that, in a given river, the dispersion coefficient increases with the water flux. As an example, evaluations derived from dye tracer experiments suggested the following quadratic relationship between the longitudinal dispersion coefficient (m^2s^{-1}) and the water flux Q (m^3s^{-1}) $E_x = 0.0148Q^2 + 0.33Q$ in the river Thames [13]. An approximate estimate of E_x can be obtained by the following formula [4]:

$$E_x \approx \frac{0.11 \cdot U \cdot L^2}{h} \tag{3}$$

where L is the width of the river. There is no theoretical relationship for transverse diffusivity in rivers because this parameter depends on the characteristics of the river. Nevertheless, for rivers with low meandering, it was suggested that E_y can be approximated by a semi-empirical formula such as taken from [12, 14]:

$$E_y \approx 0{,}6 \cdot h \cdot u^* \approx 6 \cdot 10^{-2} \cdot h \cdot U \qquad (4)$$

giving values of the order of $10^{-2} - 10^{-1}$ m^2 s^{-1}.

It should be noted that, on the basis of the values of diffusion coefficients, in the order of a few hours or days are required for achieving relatively complete transverse mixing. In the case of steady state (the contaminant input rate is a constant independent of time) point-source conditions, complete transverse mixing in a river is achieved when $\frac{x}{L} >> \sqrt{\frac{E_x}{E_y}}$, where × is the distance from the contaminant source. The use of formulae (16) and (17) gives reason of the usual assumption that a complete mixing in lateral direction is approximately achieved when $x > 3\frac{L^2}{h}$ [14].

3. TRANSPORT BY THE CURRENT

Whereas dispersion is due to the turbulent components of the water flow, advection (the transport by the water current) is related to the average, ordered water movement.

The radionuclide flux (F) per unit surface of the water body section (Bq m^{-2} s^{-1}) due to water advection is related to the concentration C of radionuclide (Bq m^{-3}) and to the water velocity v (m s^{-1}) by the following equation:

$$F = v \cdot C \qquad (5)$$

Formula (19) was obtained by dividing the total amount of substance flowing, per unit time, through the surface S, by the surface area.

It is quite obvious that parameter v in the pollutant advection equation is of hydrological nature and is independent of the pollutant. Ordinarily, v in rivers ranges from few decimetres per second to > 1 m s^{-1}, depending on the river characteristics and on the hydraulic regime. Dispersion and advection are processes common to any kind of substance in dissolved form or attached to suspended matter as the contaminant movement is only caused by the water movements. The focus now shifts to assessing those processes that depend on radionuclide specific characteristics.

4. INTERACTION BETWEEN DISSOLVED AND SOLIDS PHASES OF RADIONUCLIDES

Although the interaction between dissolved and solids phases of contaminant is the subject of a specific section, here a brief outline is given of some concepts that can be useful for understanding the contents of the present section. This process relates to both the exchange between the radionuclides dissolved in the water column and those absorbed on the suspended matter and the exchanges between the radionuclides dissolved in the sediment interstitial water and those absorbed on the sedimented material.

The process of interaction of dissolved radionuclides with solids particles in suspension or deposited, is usually modelled according to the 'K_d concept', where K_d is the partition coefficient 'particulate form/dissolved form' based on the hypothesis of a reversible and rapid equilibration between the dissolved (C_w, Bq m^{-3}) and the adsorbed phases (C_s, Bq kg^{-1}) of radionuclide

$$K_d = \frac{C_s}{C_w} \tag{6}$$

However, this is not generally and rigorously true for every contaminant substance. The equilibrium between the concentrations of the dissolved and the attached phases may be not instantaneously achieved and the adsorption and desorption processes are not always rapidly reversible [15-17].

Thus, the most simple model to simulate the time behaviour of the radionuclide interaction with solid particles is based on a first order kinetics:

$$\frac{dC_s}{dt} = b \cdot (K_d \cdot C_w - C_s) \tag{7}$$

where b is the rate (s^{-1}) of radionuclide desorption between the attached and the dissolved phases. If b is large, the equilibrium condition is rapidly attained. Moreover, if b is significantly larger than the time constants of other pollutant migration processes occurring in the aquatic environment, the time behaviour of the radionuclide interaction with solid particles is of negligible significance in relation to the expected time resolution of migration models. It is worthwhile to note that the processes of interaction of radionuclides with particulate matter can be very complicated. Neither equation (6) nor equation (7) simulate in their entirety such complex processes [18] that is analysed, in detail, in paper Ciffroy et al. 'Distribution of radionuclides between solid and liquid phases in freshwaters' of this TecDoc.

The interaction of radionuclides with bottom sediments and suspended particles is also controlled by complex environmental processes such as the sedimentation and the resuspension. The aim of the next section 'Migration to and from sediments' is to describe the approaches commonly used to simulate these processes in models for predicting the behaviour of radionuclides in fresh water systems.

Migration to and from sediments

Conceptually, three main active compartments can be assumed to be involved in the processes of radionuclide interactions with sediments and suspended matter (Figure 2). Such a schematic approach is used by the most state-of-the-art models to evaluate the radionuclide migration to and from sediments (suspended matter) by including, in a pragmatic way, three active compartments and seven main 'aggregated' radionuclide transformation/transfer processes from water to sediments or suspended matter and *vice versa.*

The active compartments are (Figure 2.):

- Dissolved radionuclide in water (C);
- Particulate radionuclide -rapid exchange component (D_f);
- Particulate radionuclide–slow exchange component (D_s);

A fourth 'passive compartment' (deep sediment) is used to represent the radionuclide subject to non-reversible removal processes from the active sediments. C, D_f and D_s are the total amount of radionuclides in the three active compartments. The fluxes (Bq s^{-1}) from a compartment can be calculated as the product of the total amount of radionuclide in the compartment (Bq) multiplied by the 'rates' (s^{-1}) of migration (K_{wf}, K_{fw}, K_{sf}, K_{fs}, K_{ws}, K_{sw}, K_s).

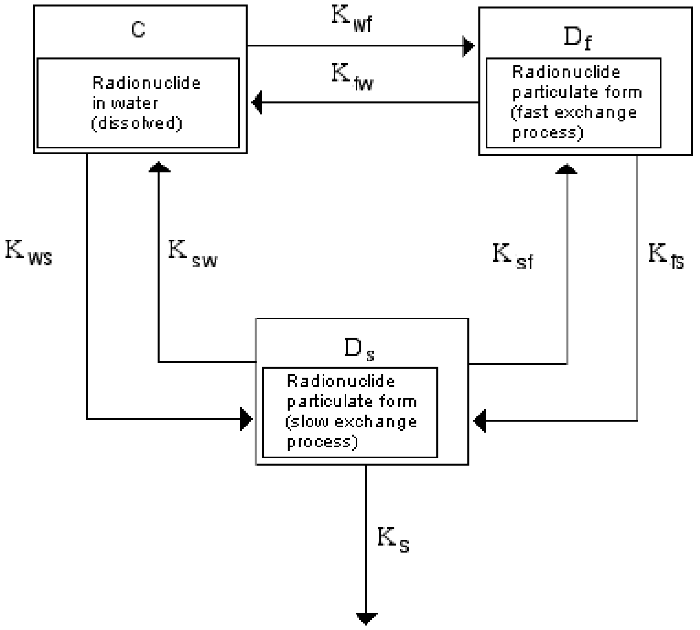

FIG. 2. Schematic structure of contaminant fluxes and environmental compartments involved in the processes of radionuclide migration to and from bottom sediment.

The seven radionuclide fluxes can be schematised as follows:

a) Radionuclide fluxes from dissolved form to particulate form and *vice versa* – rapid exchange processes ($K_{wf} \cdot C$ and $K_{fw} \cdot D_f$);

b) Radionuclide fluxes from D_f to D_s and vice-versa ($K_{fs} \cdot D_f$ and $K_{sf} \cdot D_s$);

c) Radionuclide fluxes from water to D_s and vice-versa ($K_{ws} \cdot C$ and $K_{sw} \cdot D_s$);

d) Radionuclide irreversible burial in inactive sediments ($K_s \cdot D_s$).

The compartments and the transfer/transformation processes should be regarded as 'functional' factors that reproduce the overall effects of complicated pollutant migration mechanisms in the system 'water-suspended matter-bottom sediment'. To emphasise the concept, the fluxes can be schematised as being caused by physical processes such as:

- Fast processes of interaction of dissolved radionuclides with suspended matter and with the bottom sediment interface layer;
- Slow processes of interaction of radionuclides with suspended matter and the bottom sediment interface layer;
- Sedimentation and the re-mobilisation/re-suspension of radionuclides;
- Any process responsible for the irreversible removal of radionuclides from the water column.

The 'fast and slow exchange processes', in principle, aggregate, among the others, all the processes involving the adhesion and the resuspension of radionuclides from any kind of more or less firm surface in contact with water.

If we hypothesise that the fast processes of interaction occurs between the dissolved and the particulate phases in water, from (21) we obtain the relationships relating b and K_d to K_{fw} and K_{wf}:

$$K_{fw} = b$$

$$K_{wf} = b \cdot K_d \cdot S_m$$

where S_m (kg m^{-3}) is the suspended matter concentration in water.

Table 2 shows examples of values of the migration rates available from the literature. The order of magnitude of K_{wf} and K_{fw} for suspended particulate matter in fresh water systems suggest that, few hours or few days are sufficient for the concentrations of radionuclides in water and in suspended particulate (fast exchange process) attain the equilibrium. The corresponding values for bottom sediments are higher. Nevertheless, the equilibrium is attained in few days or, exceptionally, in few weeks.

It is worthwhile to note that the measures of K_{wf} an K_{fw} for bottom sediments are affected by intrinsic uncertainties due to the difficulty of a univocal identification of the 'fast' and the 'slow' components for complex time dependent processes like those occurring in sediments. Moreover, the values in Table 2 refer to radionuclide fluxes due to the sorption and desorption processes, whereas migration processes due to particle sedimentation and re-suspension are significant in the fresh water environment.

As the equilibrium between the radionuclide dissolved in water and the radionuclide in rapid exchange component (D_f) is quickly achieved, we can write $\frac{D_f}{C} = A$ where A is a constant. The flux (F_{fs}) per square metre (Bq m^{-2} s^{-1}) of radionuclide migrating from compartment D_f to compartment D$_s$ is $F_{fs} = \frac{K_{fs} \cdot D_f}{S} = \frac{K_{fs} \cdot A \cdot C}{S} = \frac{K_{fs} \cdot A \cdot V \cdot C_w}{S} = K_{fs} \cdot A \cdot h \cdot C_w$ where V and S are the volume and the surface of the water body, respectively. The product $v_S = K_{fs} \cdot A \cdot h$ has the dimension of a velocity (the radionuclide sedimentation velocity defined in relation to the dissolved radionuclide concentration). Finally we can write:

$$F_{fs} = v_s \cdot C_w \tag{8}$$

The above result was obtained from the sole hypothesis of quick equilibrium between the radionuclide dissolved in water and the radionuclide in the rapid exchange compartment. It is instructive to note that a similar formula can be obtained considering that the flux of radionuclide migrating to sediment is due to sedimentation of particulate matter:

$$F_{fs} = S_r \cdot C_s \tag{9}$$

where S_r is the average sedimentation rate (kg m^{-2} s^{-1}) that accounts for the dynamic balance of the particle settling to bottom sediment and of particle resuspension from sediment to water and C$_s$ is the concentration of the radionuclide in suspended matter (Bq kg^{-1}).

TABLE 2. SELECTION OF VALUES OF KINETIC RATES

Location	SPM[1] (ppm)	Nuclide	K_{wf} (s^{-1})	K_{fw} (s^{-1})	K_{fs} (s^{-1})	K_{sf} (s^{-1})	K_{ws} (s^{-1})	K_{sw} (s^{-1})	Reference
Savannah River	1000	^{85}Sr	5.0×10^{-5}	5.0×10^{-5}					[19]
Guadalquivir River	172	^{241}Am	1.3×10^{-2}	2.8×10^{-3}	8.6×10^{-5}	3.6×10^{-5}			[20]
Gualdalquivir River	13	Pu	3.4×10^{-3}	4.7×10^{-3}			2.5×10^{-5}	5.6×10^{-5}	[21]
Reservoir, Spain	4	^{241}Am	2.4×10^{-4}	4.7×10^{-3}			1.9×10^{-5}	1.4×10^{-4}	[22]
Tamar River, UK	200	Cu	1.1×10^{-5}	4.3×10^{-6}					[23]
Tamar River, UK	200	Ni	8.6×10^{-6}	3.9×10^{-6}					[23]

[1] SPM is the suspended particulate matter.

If the K_d concept can be applied and one-step kinetics is assumed, from equations (20) and (23) we obtain equation (22) where $v_S = S_r \cdot K_d$ .The flux of dissolved radionuclide in water that is adsorbed to bed sediment per second and per square metre is frequently assumed to be approximately proportional to the concentration of dissolved radionuclide in water. Therefore, the total radionuclide flux from the water column to sediment is:

$$F_{ws} = (v + v_s) \cdot C_w \tag{10}$$

where v is the 'migration velocity' of the dissolved phase that accounts for the direct interaction of the dissolved radionuclide with the bottom sediments.

The following formulas can be used to evaluate the main radionuclide fluxes:

- Flux from water to bottom sediment: $F_{ws} = v_{ws} \cdot C_w$
- Flux from bottom sediment to water: $F_{sw} = K^*_{sw} \cdot D^*_s$;
- Flux from bottom sediment to passive sediment: $F_{ds} = K_s \cdot D^*_s$.

$v_{ws} = v + v_s$ (m s^{-1}) is the radionuclide migration velocity from water to bottom sediment, $K^*_{sw} = K_{sw} + K_{sf}$ (s^{-1}) and K_s (s^{-1}) are, respectively, the rate of radionuclide migration from sediment to water and to passive sediment and D^*_s is the radionuclide deposit per square metre in the active layer of the bottom sediment (Bq m^{-2}) where the slow exchange processes of radionuclide are supposed to occur

Many models suggest suitable formulae that allow one to calculate v as functions of physical and chemical characteristics of the sediment, for instance:

$$v = r \cdot (1 - e) \cdot a_{ws} \cdot d \cdot K_d \tag{11}$$

where r is the sediment density, e is the sediment porosity, d is the 'effective thickness' of the contaminated upper layer of bottom sediment, a_{ws} is the rate of radionuclide exchange 'water-bottom sediment' and K_d is the partition coefficient [24]. Similarly, in several models the rates are related to specific physical and chemical characteristics of sediments or to the prevailing *environmental* features of the water system [25]. For radiocaesium, detailed modelling of diffusion in freshwater sediments has been carried out by Smith and co-workers [26-27]. It should be noted that for many of the parameters in the quoted formulae, site-specific evaluations are necessary for the proper application of the relevant models.

Following the analysis of data from severe accidents in recent decades, evaluations of aggregated parameters v_{ws}, K_{sw} and K_s are available for ^{90}Sr and ^{137}Cs, two long-lived radionuclides of relevance for the aquatic environment.

Table 3 gives indicative ranges of values of migration parameters for strontium and caesium. These values were obtained by assessing the data reported by Smith, et al. [28, 29]; Monte et al. [1, 30], IAEA [31] and Shukla [32]. Evaluations of radionuclide deposition velocity for plutonium are available for the Great Lakes in North America. The relevant range was from 1.0 10^{-6} to 3.5 10^{-6} m s^{-1} [32]. Similar data for other radionuclides are not available. However, it should be noted that radiocaesium and radiostrontium, due to their different chemical characteristics, can be considered as reference radionuclides covering a wide range of mobility in the aquatic environment.

TABLE 3. INDICATIVE RANGES OF MODEL PARAMETER VALUES FOR ^{137}Cs AND ^{90}Sr IN FRESHWATER ENVIRONMENT

	^{90}Sr				^{137}Cs			
Parameter	Min.	Max.	GM	GSD	Min.	Max.	GM	GSD
v_{ws} (m s^{-1})	6.4×10^{-8}	1.4×10^{-7}	9.6×10^{-8}	1.4	6.8×10^{-7}	2.9×10^{-6}	9.8×10^{-7}	2
K_{sw} (s^{-1})	5.7×10^{-10}	3.0×10^{-8}	5.0×10^{-9}	2.6	1.1×10^{-8}	1.4×10^{-7}	3.4×10^{-8}	2.9
K_s (s^{-1})	≈ 0	8.8×10^{-10}	$<1.0 \times 10^{-9}$	1.5	5.8×10^{-9}	1.6×10^{-8}	8.0×10^{-9}	1.6

5. CONCLUSIONS

Most of the available state-of-the-art models for predicting the behaviour of radioactive substances in the abiotic compartments of freshwater systems show a common overall structure (Figure 2) and are based on the assessment of radionuclide fluxes in the balance equation. Although simplifications and approximations are possible and are used to different degrees by modellers, there are no significant conceptual differences among the theoretical formulations of the models at this basic stage. As shown above, models make use of different hypotheses to approach the complex problem of modelling the physical and chemical behaviour of radioactive substances in water bodies. The equations that are used by models represent more or less coarse approximations of complex processes that, in principle, depend on a variety of environmental, hydraulic, hydrochemical, etc. characteristics of the water bodies. The models essentially differ in the values of the transfer parameters or in the sub-models used to assess these. Moreover, different formulations for evaluating the fluxes of radionuclides are possible depending on the assumed hypotheses. This gives rise to a number of different existing models. There is no convincing evidence of successful 'crucial' tests to select univocally a single model or a set of transfer parameters. Therefore, the use of different modelling approaches is deemed advantageous for the interpretation of the model results in practical applications.

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APPENDIX

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ADHESION OF SUSPENDED MATTER TO THE EXTERNAL PLANT SURFACE

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Abstract

The retention of suspended particles transported by river flow on freshwater plant surfaces is a potentially important process in the contamination of aquatic biota and can significantly increase the apparent concentration of radionuclides above their endogenous value, leading to an overestimation of the uptake rate and concentration ratios and may significantly influence analytical results of aquatic biota In the case of contaminants that have a high affinity for soil particles, like caesium that interacts strongly with the micaceous clay minerals in natural particulate matter, the concentration of these contaminants in suspended particles is generally higher than the endogenous concentration of the same contaminant in biota, thus even small quantities of particles adhering to the plant can significantly increase the apparent concentration of elements above their endogenous value, leading to wrong conclusions. The mass of suspended particles retained on the surfaces of freshwater plants and the plant biomass could be determined by comparing the scandium content of suspended material with that in the fresh plant. Scandium, a predominantly soil-associated trace element, is a useful "tracer" for the inorganic component of suspended particles, because this element is geologically ubiquitous in soils and sediment, and it is recognized that it cannot cross natural membranes of plants and other organisms.

1. INTRODUCTION

Many contaminants have a high affinity for soil particles and exhibit systematic correlation with the clay mineral fraction. Soil particles are thus both the physical support and the carrier of numerous contaminants in natural waterways. They can act as a sink: the settling of suspended material is clearly one of the most important processes, which effectively removes contaminants adsorbed to the particles from the water column. Suspended matter may also become a source of short- or long-term contamination: erosion and resuspension due to flood events, transportation by the flow and biological processes have a central role in the fate and mobility of contaminants associated with soil particles in natural waters.

One of the factors that could explain part of the variability exhibited by aquatic biota tissues is the adhesion of organic or inorganic contaminated particles to external plant surface [1-3]. The concentration of many contaminants in suspended particles is generally higher than the endogenous concentration in biota thus even small quantities of particles adhering to the plant can significantly increase the apparent concentration of trace elements above their endogenous value, leading to erroneous conclusions concerning plant uptake. Thus in comparing data from different areas or for assessing temporal and spatial variations of the contamination of freshwater plants, the effects of the adhesion of suspended particles to the plant surface should be considered [4].

When contaminants are released into waterways they become more or less associated with the clay mineral fraction of suspended particles, depending upon their individual chemical properties. Under conditions of strong association, the transport and fate of the pollutant is highly controlled by the fate and mobility of particles in natural waters. In these conditions, the association of contaminants with suspended material can play an important role in controlling their transfer through the ecosystem. Available data on caesium transfer to aquatic organisms suggest that aquatic plants accumulate ^{137}Cs by both absorption and adsorption [5-7], but these parameters were not quantitatively assessed. In addition, several studies [1-3] showed that external adhesion of suspended particles on submerged biological materials may significantly influence analytical results of aquatic biota. Scandium, a predominantly soil-associated trace element, has been used as a 'tracer' for the inorganic component of suspended particles [8-10], because this element is geologically ubiquitous in soils, and it is assumed that it cannot cross natural membranes of plants and other organisms [1-3, 9, 11]. For these reasons, the ratio between the amount of Sc in aquatic plants and the amount of Sc in suspended particles provides a good estimate of the relative mass of inorganic particles adhered to aquatic plants.

Materials coating submerged material can accumulate very high levels of metals [12]. Such materials, defined as periphyton or aufwuchs, have been widely used to monitor metal bioavailability in contaminated environments [1-3, 13-19]. These materials are a complex mixture of biotic matter that is consumed by many species of scrappers and grazers [12]. Periphytic algal communities have been extensively studied because of interest in their role in the ecosystem and, from an applied viewpoint, because of their tendency to concentrate materials from the surrounding water. Despite their trophically significant role, the association of metals with those materials coating submerged surface and the abiotic components of sexton has been generally ignored in the past. Table 1 gives the mean values for Sc measured in the periphyton samples (n=20) scrapped off from the artificial substrates collected at intervals of 16, 21, 25 and 30 days (Sc = 4.00±0.99 mg kg^{-1}) [3].

TABLE 1 VARIATION WITH TIME IN Sc CONCENTRATIONS IN THE PERIPHYTON MAT

Days of exposure	Sc (mg kg^{-1})	
	AM	SD
16	5.18	3.6×10^{-1}
21	4.72	1.9×10^{-1}
25	3.35	5.4×10^{-1}
30	2.75	4.8×10^{-1}

Comparison of the scandium content of suspended material with that in algal communities collected at intervals of 16, 21, 25 and 30 days after placement of the artificial substrates, allows the quantification of the contribution of suspended material to the total caesium contamination of the algal samples:

$$\frac{[Sc]_{per}}{[Sc]_{par}} = \textit{relative amount of inorganic suspended particles adhering to periphyton}$$

where $[Sc]_{par}$ is the Sc concentration in periphyton (mg kg^{-1}) and $[Sc]_{per}$ is the Sc concentration in suspended particles (mg kg^{-1})

The relative mass of suspended particles adhering to the total mass of periphyton scraped off from the artificial sheets at intervals of 16, 21, 25 and 30 days after placement in the river water are shown on Figure 1.

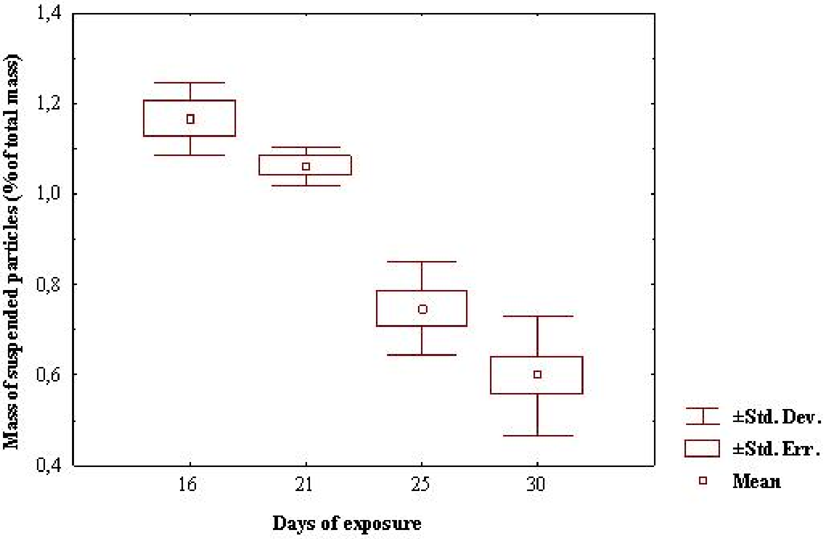

FIG. 1. Mass of suspended particles adhering to mass unit of periphyton.

The results indicated that the suspended particles mass adhering to the periphyton contributes between 55 and 100% of the total apparent periphyton mass. The uncertainty associated in the assessment of the extremely low quantities of periphyton biomass and suspended particles mass adhering to the biota at the beginning of the ecological succession explains the mean values higher than 100% found until the 21st days of the exposure of the artificial substrates in the river water.

Figure 2 illustrates the bio volumes of the main species composing the periphyton mat, expressed as the total volume of all individuals (specific cell volume determined from mean dimensions of the cells, multiplied by the specific cell density) during the growing period. As expected the biovolume per surface unit increases with time. The growth curve reaches a maximum at day 25 and then decreases due to the limited growing area on the artificial substrate.

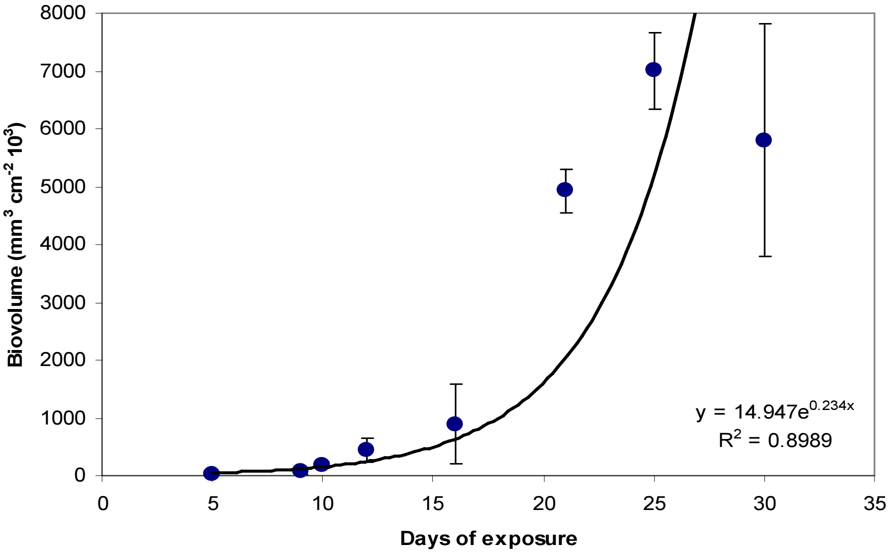

FIG. 2. Variation with time in biomass of main species composing the periphyton mat [2].

Quantitatively, it is well known that uptake of materials may be linear over a certain range of biomass but as the biomass increases, proportional accumulation diminishes. This 'weight dilution' effect, apparently applies to adhering particles as well, and is apparent in Figures 1 and 2, where the mass of suspended particles tend to be lowest in the zone of maximum biomass and highest in the area of low biomass. The data show a higher suspended particles adhesion capacity of the cells at the beginning of the ecological succession and we hypothesise that this effect could be dependent on the cells' surface exposed to the particles. At the beginning of the ecological succession the total surface of all individual small cells is high and we expect a diminution of the total surface along the ecological succession.

2. CONCLUSIONS

In the case of contaminants that have a high affinity for soil particles, like caesium that interacts strongly with the micaceous clay minerals in natural particulate matter [20-25], the concentration of these contaminants in suspended particles is generally higher than the endogenous concentration of the same contaminant in biota, thus even small quantities of particles adhering to the plant can significantly increase the apparent concentration of elements above their endogenous value, leading to wrong conclusions.

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DISTRIBUTION OF RADIONUCLIDES BETWEEN SOLID AND LIQUID PHASES IN FRESHWATERS

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Abstract

The paper provides operational Best Estimate values, as well as Probability Density Functions (PDFs), for distribution coefficients (K_ds) in freshwater, representing the partition of radionuclides between the particulate and the dissolved phases respectively. A literature review allowed to highlight the most important environmental factors influencing the distribution of radionuclides at the interface particles/water is presented. The construction of PDFs for 8 elements (Ag, Am, Co, Cs, I, Mn, Pu and Sr) was established according the following procedure: (i) construction of a comprehensive database where K_ds values obtained under various environments and parametric conditions were collected; (ii) scoring procedure to account for the 'quality' of each data point (according to several criteria such as the presentation of data (e.g. raw data vs mean with or without replicates), contact time, pH, solid-to-liquid ratio, expert judgement) in the construction of the PDF; (iii) weighted bootstrapping procedure to build the PDFs, in order to give more importance to the most relevant data points. For 7 other radionuclides (Ba, Be, Ce, Ra, Ru, Sb and Th), a simplified procedure was adopted because of the scarcity of data: only non-conditional PDFs were built, without incorporating a scoring procedure

1. K_d CONCEPT FOR FRESHWATER ECOSYSTEMS

The residence time of radionuclides in freshwater streams is strongly affected by their interactions with suspended particulate matter (SPM) and settlement in sedimentation zones of a water system. Besides, the uptake of radionuclides by aquatic organisms depends on the concentration and on the speciation of radionuclides remaining in the dissolved phase. Partitioning of radionuclides between water and suspended matter is often described in terms of distribution coefficients (K_ds), expressed as the concentration ratio between the particulate phase and the dissolved phase under equilibrium conditions (in Bq.kg^{-1} of SPM per Bq L^{-1}).

2. FACTORS AFFECTING K_d

2.1. Kinetics aspects

Numerous studies have shown that the sorption of radioactive or metallic trace elements on natural particles resulted from several kinetic processes, involving rapid, but also slow processes (e.g. oxidation processes, inner sphere complexation and migration of cation in the clay structure) [1-15]. Some examples of studies that demonstrated the existence of several kinetic processes for some radionuclides are given in Figures 1 and 2.

Referenced studies showed that kinetics in the interactions of radionuclides at the interface water-SPM obviously depend on the element of concern, but also on other environmental co-factors, such as:

- The season, and the associated biological activity in the river. For some radionuclides (e.g. radioisotopes of Co or Mn), oxidation processes, that are partly microbially mediated, govern their slow uptake onto SPM. Seasonal differences may then reflect strong seasonal variation in the abundance of oxidizing bacteria [11-12, 16].

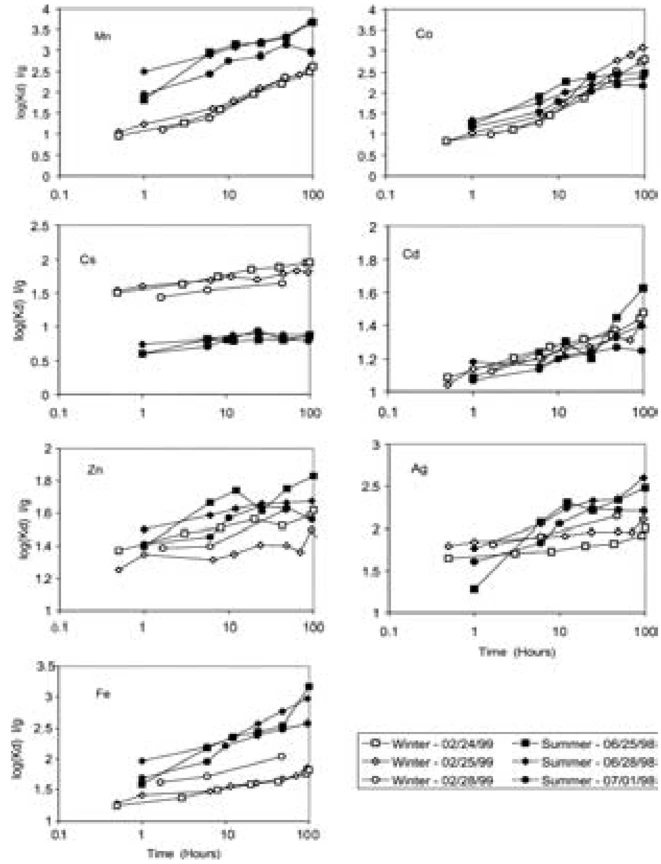


FIG. 1. Sorption kinetics onto freshwater particles from the Loire river over 100 h–experimental data obtained under winter and summer conditions [124].

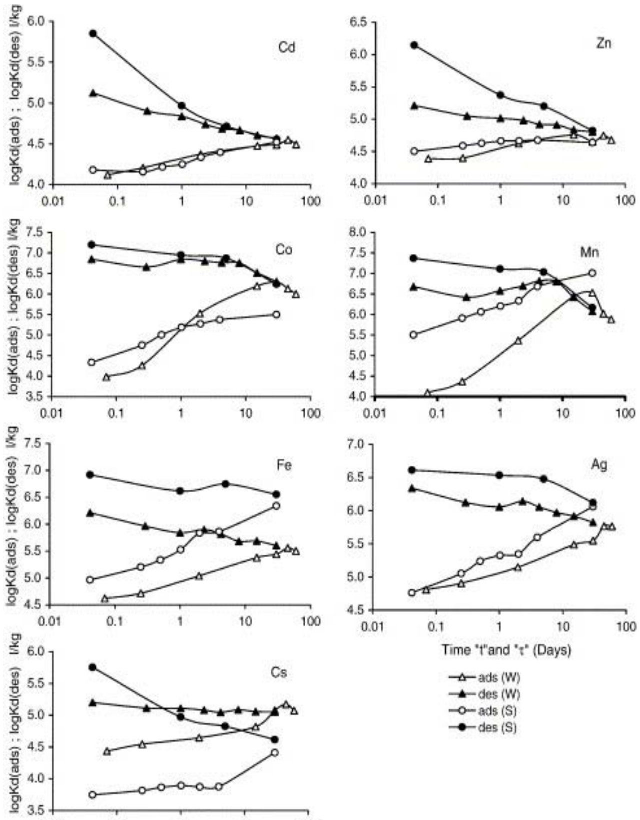


FIG. 2. Sorption (ads) and desorption (des) kinetics of trace elements on SPM in winter and in summer conditions over 30 days [127].

- SPM concentration. Benes et al. [5-7] studied the uptake kinetics of Cs over a wide range of SPM concentration (20-2000 mg.l^{-1}) and observed that time to reach equilibrium is more longer for low SPM concentration, reflecting that the ratio between binding sites and available radionuclide in the solution partly governs the kinetics of transfer.
- The ionic strength. Even if this document focuses on freshwaters, experiments performed in estuarine systems characterized by an ionic strength gradient can indicate the influence of this parameter on uptake and release kinetics. For some radionuclides (e.g. radioisotopes of Cs, Co, Mn), it was shown that the velocity of rapid processes is more or less constant over a wide range of salinity, while velocity of slow processes is highly affected by salinity [12]. Rapid processes likely correspond to binding/release or ionic exchange with easily accessible sites that are not significantly modified by chemical composition in water, while slow processes likely correspond to the change of speciation of bound elements (oxidation, migration to less accessible sites and inner sphere complex formation) that may be salinity-limited.
- age of the contamination (called ageing effect). The desorption kinetics may be highly governed by the inner speciation of bound radionuclides (i.e. distribution among easily and less accessible binding sites respectively). For example, long time kinetic adsorption–desorption experiments (over 30 days) [7, 15] indicate that elements freshly associated with particles present less stable complexes and are released more rapidly than 'old-contaminated' particles.Thus, kinetics in the interaction of radionuclides at the interface water-SPM may highly depend on experimental designs used for their determination and are highly site-specific.

Site-specific data can be generated through in vitro experiments conducted with freshly collected natural water and spiked radioactive solutions, according to protocols that are described in detail by several authors [4-6, 10-13, 15]. When such site-specific data are available, uptake and release kinetics can be simulated by compartmental models (one-, two, three- or more box models with successive or parallel reactions) summarized in Figure 3 (from Barros [17]). In general, all these compartmental models provide multi-exponential-type solutions. However, Barros et al. [17} showed the mathematical equivalence between these models and concluded that it is a practical fact that no difference between models occurs under certain environmental conditions.

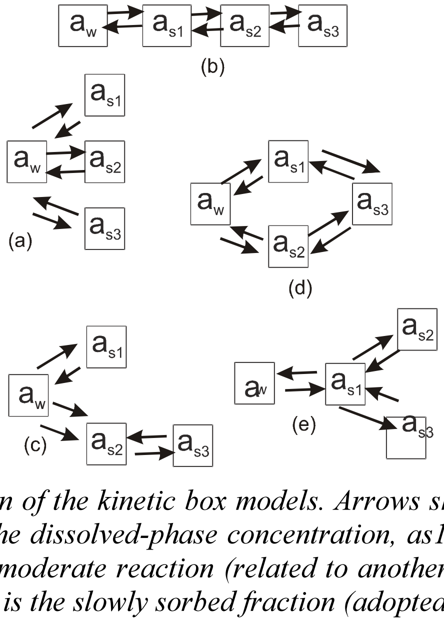


FIG. 3. Schematic representation of the kinetic box models. Arrows show the sorption and desorption processes, compartment aw is the dissolved-phase concentration, as1 is the concentration associated to the fastest reaction, as2 is a moderate reaction (related to another mechanism or due to the delay caused by accessibility) and as3 is the slowly sorbed fraction (adopted from [17]).

2.2. pH

pH was demonstrated to be a predominant factor for several radionuclides, such as Co [18-22], because it modifies the speciation of solid surface sites and the speciation of competitive ions in water. Radionuclides adsorption to SPM can generally be represented by a sigmoid curve, with sudden increase in adsorption occurring over a narrow range of 1 to 2 pH units (see examples provided in Figure 4 [22]). The observed radionuclides uptake behaviour, qualitatively similar to results for simpler surfaces (e.g. metal oxides [23]), can be interpreted in terms of interaction of dissolved metals with deprotonated sites on the surfaces of the SPM. The importance of pH in the choice of relevant K_d is particularly crucial for elements that show steep slopes in the range of natural pH found in freshwaters (typically 6-7), such as Co or Mn.

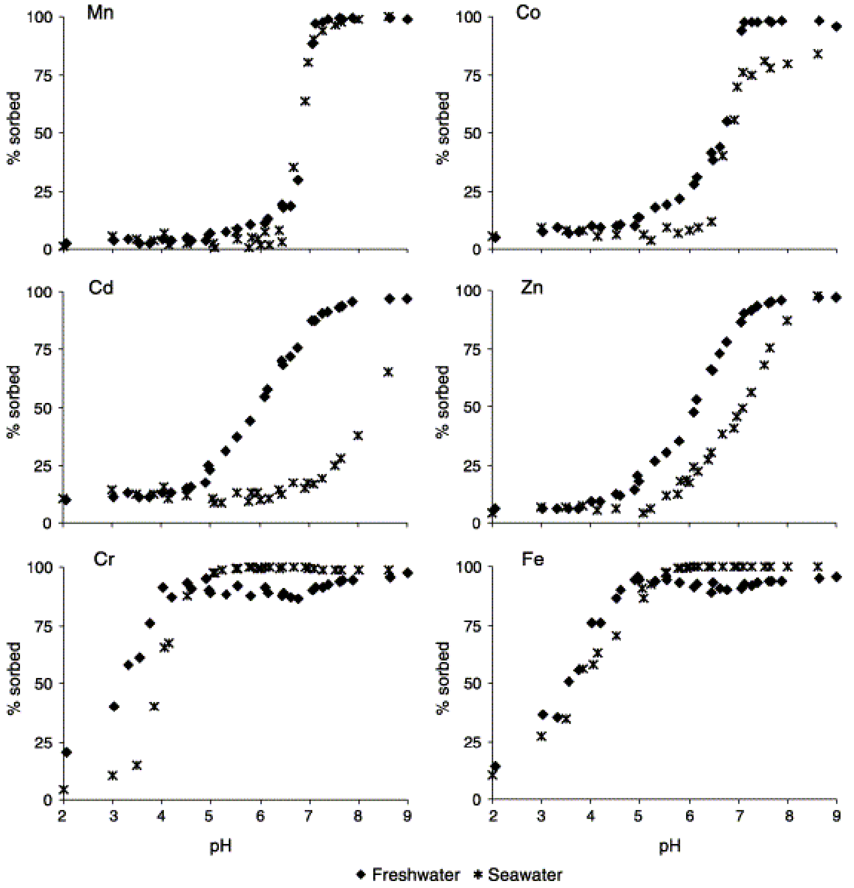


FIG. 4. Adsorption edges for ^{109}Cd, ^{60}Co, ^{51}Cr, ^{59}Fe, ^{54}Mn, AND ^{65}Zn onto suspended particulate matter in seawater (salinity 31.6) and freshwater [22]

2.3. Suspended particulate matter concentration and colloids effects

The so-called 'particle concentration effect' (i.e. the effect of the particle concentration on the distribution coefficient) was showed by several studies [4, 7, 18-20, 22, 24-26]. The 'particle concentration effect' is an unexpected decline in partition coefficients (K_d) as suspended particulate matter (SPM) increases. This observation has been attributed to a variety of causes that were listed by Benoit and Rozan [27]: sorption kinetics, irreversible adsorption or incomplete desorption, qualitative variations in surface chemistry, filtration artefacts, particle-particle interactions, the presence of a non-aqueous dissolved phase, and the contribution of trace elements bound to colloids included in the filtrate fraction. However, this effect remains controversial. Indeed, according to McKinley and Jenne [28], examination of published data accounts of the solid concentration effect revealed either inappropriate experimental designs or errors of data interpretation. In particular, the colloidal phase (intermediate phase between the 'true' dissolved phase and the particulate phase), that is included (in error) within the so-called 'dissolved' phase, may play a major role: according to Benoit and Rozan [27], when corrections are applied for both colloids and large particles, K_d values become truly constant.

Nevertheless, on a practical point of view, such corrections (that need data on both 'truly dissolved' and colloidal forms) can be applied only if site-specific data are collected. The contribution of colloids to the distribution within the so-called dissolved phase was studied by several authors through ultrafiltration techniques [29-33] – see some examples in Figs. 5 and 6).

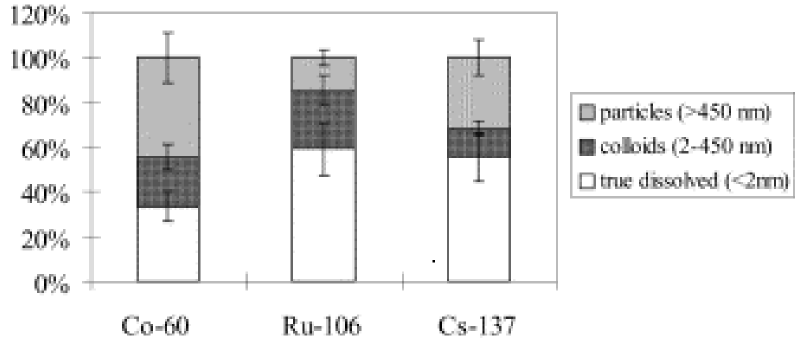
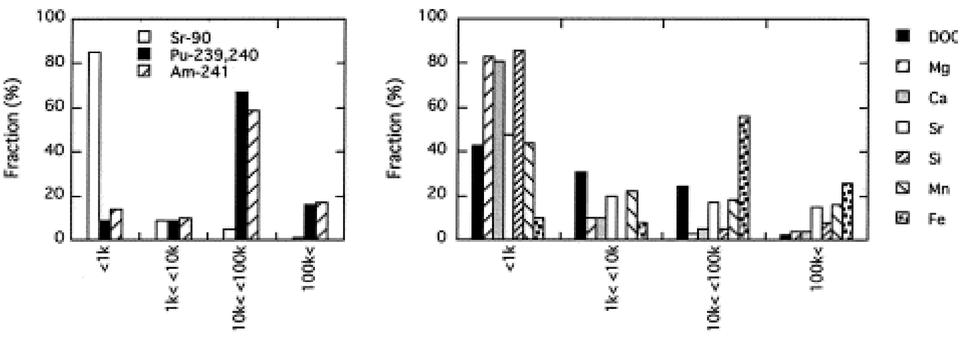
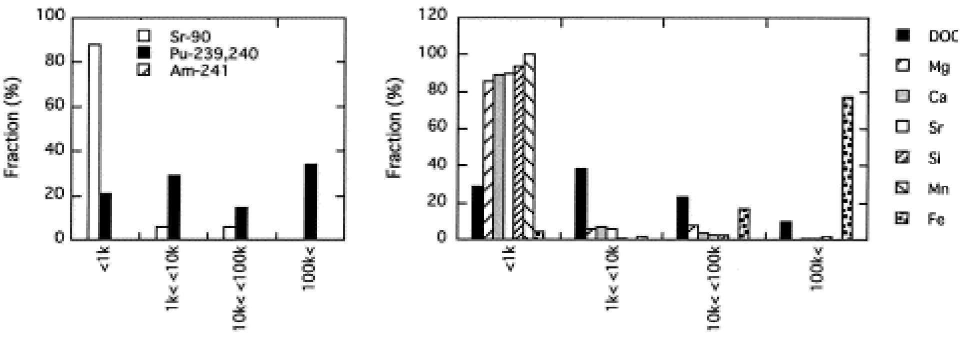


FIG.5. Distribution of ^{60}Co, ^{106}Ru and ^{137}Cs, among particulate (>450 nm), true dissolved (<2 nm) and colloidal (2–450 nm) phases [30].



(1) Sahan River (April, 10, 1998)



(2) Sahan River (Sept. 30, 1998)

FIG. 6. Fractional distributions of radionuclides and stable elements in 4-step ultrafiltration in the Chernobyl zone [32].

It was observed that the contribution of colloidal forms to the ‘dissolved’ phase may be quite significant for some radionuclides (e.g. ^{106}Ru, ^{239,240}Pu, ^{241}Am) and that their transformation (e.g. aggregation, coagulation) may be a source of variability in the determination of K_d values.

2.4. Ionic strength

The concentration of complexing ligands and/or competitors in the dissolved phase (e.g. analogue ions or dissolved organic matter) largely influences the K_d values. Indeed, the competitive complexation between particulate surface sites and inorganic dissolved ligands depends on the concentration of dissolved ligands and on the complexation constant of these ligands with each element. Experiments, conducted in estuarine waters where significant salinity gradients were observed, demonstrated the effect of ionic strength on K_d values for most of the studied radionuclides [11, 12, 24, 35]. Even if the variability of ionic strength is moderate in freshwaters compared to estuaries, the effect of competitive ions can be significant for some elements. For example, competition processes between ions were extensively studied and demonstrated for Cs and Sr that compete with monovalent (K^+ or NH_4^+) and divalent (Ca^{2+} or Mg^{2+}) ions respectively [36-41]. Some models were proposed for Cs and Sr to account for the concentration or competitive ions in the calculation of K_d values. Thus, Hilton et al [40] proposed the relationship that represents the decrease of K_d values with increasing competing ions concentrations:

- the sites able to uptake Cs are supposed to be located at the broken edges of illite mineral fragments called ‘frayed edge sites’ (FES), and thus:
- $K_d^{Cs} = \frac{K_c(Cs/K).FES}{\left([K^+] + K_c(Cs/K).[Cs^+]\right)}$, where $K_c(Cs/K)$ is the selectivity coefficient for homovalent competition (e.g. about 1000 for illite according to de Preter [42]).
- for bivalent Sr, a similar relationship was proposed: $K_d^{Sr} = \frac{K_c(Sr/Ca).CEC}{\left([Ca^{2+}] + K_c(Sr/Ca).[Sr^{2+}]\right)}$, where CEC is the cation exchange capacity (in meq.kg^{-1}) and $K_c(Sr/Ca)$ is the selectivity coefficient for homovalent competition (e.g. 1 according to Bruggenwert et al [43]).

2.5. Nature of particles and extend of reversibility

Suspended matter gathers contrasted type of particles such as clay, carbonates, oxides and hydroxides, organic matter including living cells, such as phytoplankton. The partitioning of a given radionuclide is highly dependant on the composition of the particulate pool. The degree of reversibility also highly depends on the distribution of the radionuclide of concern among these different types of particulate ligands. One way to investigate the degree of reversibility is to conduct sequential extractions, with successive extractants that theoretically allow the selective release of radionuclides (e.g. [44]: extraction of five fractions corresponding to five types of sites with various exchange characteristics (exchangeable, acid soluble, reducible, oxidizable, residual); [45-47]). The example presented in Figure 7 shows that the association of radionuclides to particles highly depends on the element of concern, but also on the particulate pool.

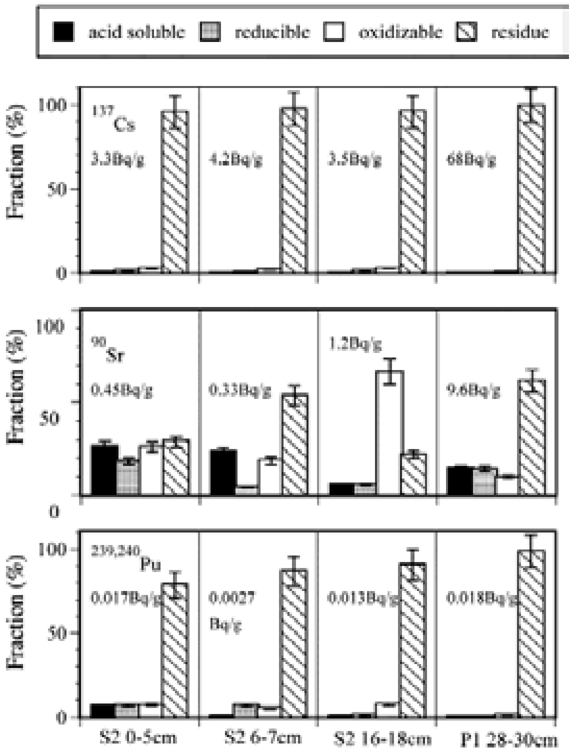


FIG. 7. Fractions of radionuclides over different phases segregated by sequential selective extraction applied to selected samples of the river bottom sediments in the Chernobyl zone [47].

3. DERIVED VALUES

Data given in sections *2.2-2.4* demonstrate that the K_ds can be highly uncertain. Thus, the K_d variability should be considered in uncertainty analysis of risk assessment models. Useful method to account for parametric uncertainty is to represent the investigated parameter by a Probability Density Function (*PDF*) rather than by a best estimate value. Indeed, such *PDF*s both give information on the most probable value of the parameter, and also account for all the range of potential values, with an indication of their probability of occurrence. Thus, such *PDF*s can further be used to propagate uncertainty in risk assessment models. The use of such *PDF*s in extensive uncertainty/sensitivity analysis of risk assessment models can be illustrated in several studies [48-49].

3.1. K_d values for radionuclides with large databases

For eight elements (Ag, Am, Co, Cs, I, Mn, Pu and Sr), an extended database of K_d values available in the literature was built for calculating non-conditional and conditional *PDF*s. Non-conditional *PDF*s can be used when no specific information is known on the site under investigation and conditional *PDF*s can be used when some specific information is available (e.g. on pH, distance from the source point, solid-to-liquid ratio, etc).

3.1.1. Freshwater K_d database

The database contains published K_d values for Ag, Am, Co, Cs, I, Mn, Pu and Sr obtained on natural freshwaters (rivers and lakes) from suspended particulate matter (SPM) or superficial sediment (first 0-5 cm). 73 references, essentially originated from peer-reviewed journals, have been studied (Appendix 1). The spreading of publications in function of the elements is : 8 for Ag, 8 for Am, 27 for Co, 32 for Cs, 11 for I, 18 for Mn, 8 for Pu and 12 for Sr, providing

K_d values in freshwater. K_d values came from different river and lake water systems (5 for Ag, 15 for Am, 19 for Co, 57 for Cs, 13 for I, 13 for Mn, 16 for Pu and 16 for Sr). In addition, information on potential co-factors and on criteria allowing estimating the quality of each referenced data was collected, especially about: pH, suspended matter concentration, contact time between water and particles, method for the determination of K_d values (in situ measurements or laboratory experiments under adsorption or desorption conditions with spiked solutions), redox conditions (especially for I).

In many studies, raw data were not published, but only mean values, eventually associated with information on standard deviation and the number of measurements (e.g. replicates). In the present study (aiming at providing information on K_d uncertainty), it was chosen to use the information regarding the dispersion of raw data (indicated through the standard dispersion), instead of using only the mean value. Thus, 'calculated raw data' were generated through a Monte Carlo procedure (Latin Hypercube procedure) in a way to respect the mean value, the standard deviation and the number of originally raw data.

3.1.2. Data quality evaluation

In the frame of the statistical analysis presented in this paper, it appeared necessary to affect a score describing the quality of the K_d values found in the literature. Indeed, the quality of each K_d values depend for example on the method chosen for its determination, the number of replicates, the quality and quantity of information given on the physico-chemical characteristics of water, etc. Some objective criteria were defined in order to determine for each data a score describing its quality according to the objective of this study (Table 1):

1. Criterion 1: presentation of data. Presentation and accessibility of data in published papers may be quite different according to the publication (all raw data available, or mean value with or without standard deviation and number of replicates, raw data or mean values presented on a Figure, etc). The score linked to the 'presentation of data' depended on the information provided (e.g. raw data are preferred to a mean value) and whether data are easily legible or not (data from a table are preferred to data in a figure).
2. Criterion 2: contact time. As mentioned earlier, the K_d theoretically describe interactions of radionuclides at the water/particles interface at equilibrium. Among the published papers, some of them included kinetic aspects. To account for kinetics, K_d values obtained under rather long-term experiments were better scored (also because most concrete applications concern long-term interactions, except for zones situated just downstream of release points). Such information allowed us to define five 'contact time' classes (see Table 1). For long contact time (over 120 hours), equilibrium is expected to be achieved but the in vitro experimental medium may be submitted to physico-chemical modifications (e.g. phytoplankton death, organic matter degradation) leading to less relevant K_d values, as shown by Millward and Liu [14] for example. This is the reason why a lower score was affected. Moreover, K_d values directly measured in the field are supposed to be equilibrated.
3. Criterion 3: pH range. pH may play a significant influence on complexation processes and then on the K_d. Thus, it was necessary to discriminate K_d values obtained under natural conditions with or without any indication on the pH value, and K_d values obtained under extreme pH values. Consequently, different scores were affected to these different categories, considering that pH of the natural freshwaters is generally between 5 and 9 [50].

TABLE 1. SCORE TABLE FOR THE FIVE SELECTED CRITERIA (the minimum and maximum scores for a given criteria are 1 and 100 respectively)

Criteria	Criteria score	Comments
Criterion 1 : Presentation of data		
raw data with replicates > 2	100	raw data from a table with more than 2 replicates
raw data without replicates	30	raw data from a table without replicates
data from figure with dispersion	30	data from a figure with indication of the dispersion
data from a figure without dispersion	10	data from a figure without indication of the dispersion
Criterion 2 : Contact time (c.t)		
120 h < c.t	50	long contact time but possibility of modification of the experimental system
48 h ≤ c.t ≤ 120 h	100	contact time where equilibrium is generally achieved
24 h ≤ c.t < 48 h	30	short contact time but equilibrium may be achieved
2 h ≤ c.t < 24 h	5	short contact time (useful for the impact of release near nuclear power plant)
c.t < 2 h	1	very short contact time (useful for the impact of release near nuclear power plant)
in situ	100	data determined with in situ measurement (assumed to be at equilibrium)
Criterion 3 : pH		
pH ≤ 5	1	not natural pH for most of the freshwaters
5 < pH ≤ 6	50	possible pH in freshwaters
6 < pH ≤ 8.2	100	most probable pH in freshwaters
8.2 < pH ≤ 9	50	possible pH in freshwaters
pH > 9	1	not natural pH for most of the freshwaters
pH nat	30	pH not mentioned but measurement in natural freshwaters
Criterion 4 : solid-to-liquid ratio (m/V in mg/L)		
m/V ≤ 500 mg/L	100	[SPM] in freshwater under normal conditions
500 < m/V ≤ 2000 mg/L	10	[SPM] in freshwater under flooding conditions
2000 < m/V ≤ 10000 mg/L	5	not natural [SPM], close to sediment condition
m/V > 10000 mg/L	1	not natural [SPM], close to sediment condition
superficial sediment	5	m/V ratio not mentioned but in situ measurement in sediment
Criterion 5: Expert judgement		
	100	K_d determination is the objective of the article with a good experimental protocol
	80	K_d determination is one of the objectives but quality of the dataset is decreasing
	60	K_d determination is one of the objectives but fundamental information is lacking
	40	K_d determination is one of the objectives but not peer-reviewed
	20	K_d determination is a secondary information or data are from a report

4. Criterion 4: solid-to-liquid ratio. A scoring criteria was defined for this parameter because some experiments were performed in conditions (batch experiments with very high solid-to-liquid ratio) that are poorly relevant for most of natural situations. From the works of Håkanson et al [51], who modelled and fitted SPM in temperate rivers from extended databases, it can be shown that SPM is, most of the time, lower than 100 mg/L, sometimes (during flood events) in the range 100-1000 mg/L and rarely higher than 1000 mg/L. Thus, experiments performed with high SPM (e.g. more than 10000 mg/L) are poorly relevant for the majority of usual situations and a lower score was affected to them.
5. Criterion 5: expert judgement. A complementary score was added to take into account the 'expert judgement' criteria when building the database. The expert judgement was defined according to several points: (i) the impact factor of the journal; (ii) the quality of data found in the publication. For given experimental conditions, replicated leading to scattered data were less scored than data with a good standard deviation; (iii) the objective of the article (e.g. articles specifically focused on the determination of K_d values, with well-defined experimental protocols). The global score of each data point is the mean of the individual scores obtained for the criteria 1 to 5.

3.1.3. Statistical analysis

The previous stages allowed us to build a database containing K_d values, each of them being associated with a score. The generation of PDFs was then provided through a direct weighted bootstrap procedure. The weighted bootstrap method is used to construct samples in which the proportions of data among referenced studies correspond to those desired. We therefore carry out a non-equiprobable sampling with replacement from raw data available in the database: the probability of drawing each data point corresponds to the score previously defined for each of them. Thus, 'relevant' data points are more represented in the bootstrap sample than poorly relevant data points.

Each bootstrap sample is then fitted to log-normal PDF or log-triangular PDF. The goodness of fit was tested by a Kolmogorov-Smirnov test with a Dallal-Wilkinson approach and by the multiple R-square coefficient (R^2) between theoretical and empirical distribution. The calibration allows to determine the characteristics of the distribution, such as the mode (50^{th} percentile), and the 5^{th} and 95^{th} percentiles.

For the non-conditional PDFs, all the data entered in the database are taken into account. On the contrary, for the conditional PDFs, the same procedure was used, but the database was previously filtered in order to keep only data respecting a given constraint (e.g. data points obtained under a given range of pH, or a given range of contact time, etc).

Figure 8 gives an example of the PDF that can be obtained according to this procedure (case of Cobalt under adsorption conditions and with a contact time between 48 and 120 hours) while the others PDFs derived based in collected information is given in Appendix 2 (Figs. A2.1-A.2.6.).

In such a Figure, the following information are reported: (i) the origin of each data point and its score; (ii) the characteristics of the theoretical distribution (mean and standard deviation for log-normal PDFs; minimum, mode and maximum for log-triangular PDFs); (iii) the goodness of fit (Kolmogorov-Smirnov test an R^2 coefficient between theoretical and empirical distribution); (iv) the confidence interval of the theoretical PDFs (extreme PDFs among the 1000 bootstrap generated samples).

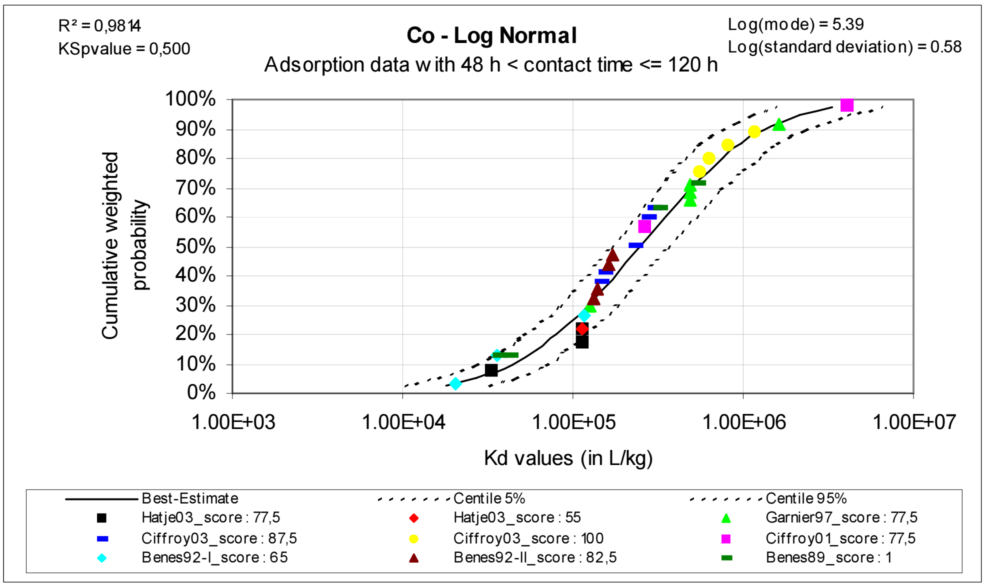


FIG. 8. Example of PDFs obtained by the weighted bootstrap procedure.

3.1.4. Results: operational PDFs

Two types of operational PDFs were determined (non-conditional and conditional PDFs). The non-conditional PDFs were built from three different sources of data: (i) in situ data. K_d values calculated from field measurements. (ii) adsorption data. K_d values calculated from laboratory experiment of adsorption of spiked radionuclide from dissolved phase to suspended matter. (iii) desorption data. K_d values calculated from laboratory experiment of desorption of spiked radionuclide from suspended matter to dissolved phase. The conditional PDFs are subclasses of the laboratory experiments (adsorption and desorption data) giving more accurate information about the influence of parameters (contact time, pH, solid-to-liquid ratio or oxic/anoxic conditions). Such conditional PDFs could not be determined from in situ data since few details about the parameters previously cited were given in most of the references.

The aim of this study was to provide to end-users (risk assessors) who need K_d values in freshwater for their own assessments (e.g. transport models, risk assessment of the aquatic food-chain, for which the distribution dissolved/particulate is needed) operational PDFs allowing further uncertainty/sensitivity analysis. Non-conditional PDFs can be used when no specific information is known on the site under investigation. Such non-conditional PDFs gather a wide range of parametric conditions studied in laboratory. However, the scoring procedure and further bootstrapping method allowed providing more importance to conditions that can typically be met in natural environments. When more detailed information is known about the context of investigation, conditional PDFs can be used, and thus uncertainty can be reduced.

The various global and conditional PDFs that were thus generated were synthesised in Tables 2-10 where the following information were collected: (i) the number of data (N), the number of references (#ref) and the number of freshwater systems of concern (NE) used for the construction of the PDF; (ii) the mode, as well as the 5th and 95th percentiles of the distribution. When N < 10, #ref < 2 or NE < 2, the mention ‘irrelevant data’ was written. The latter disposition was taken to present only relevant K_d values coming from different water systems that can be considered not is to be linked to special local conditions.

TABLE 2. OPERATIONAL PDFS OF K_d VALUES ($l.kg^{-1}$) FOR Ag

Parameter	N	NE	GM	GSD	#ref
ads	81	4	8.5×10^4	2.3	7
ads_-2h	13	2	3.8×10^4	1.8	3
ads_+2h-24h	31	4	6.6×10^4	1.8	7
ads_+24h-48h	12	2	8.9×10^4	2.0	3
ads_+48h-120h	16	2	1.0×10^5	2.0	3
ads_+120h	9	3	irrelevant data		3
ads_pH+7-8	20	2	8.7×10^4	2.3	4
ads_pH+8-9	61	4	8.3×10^4	2.2	5
ads_mV-50	58	4	8.3×10^4	2.6	7
ads_mV+50-500	23	2	9.1×10^4	1.5	2
des	41	2	4.4×10^5	1.7	2
des_-2h	18	2	4.8×10^5	1.6	2
des_+2h-24h	14	2	4.0×10^5	2.0	2
des_+24h-48h	5	1	irrelevant data		1
des_+48h-120h	4	1	irrelevant data		1
des_pH+7-8	17	2	5.2×10^5	1.4	2
des_pH+8-9	24	2	3.6×10^5	1.8	2
des_mV-50	31	2	3.7×10^5	1.7	2
des_mV+50-500	10	1	irrelevant data		1
in situ	2	1	irrelevant data		1

TABLE 3. OPERATIONAL PDFS OF K_d VALUES ($l.kg^{-1}$) FOR Am

Parameter	N	NE	GM	GSD	#ref
ads	88	9	2.1×10^5	3.7	4
ads_-2h	3	1	irrelevant data		1
ads_+2h-24h	36	4	1.4×10^5	5.4	3
ads_+48h-120h	28	4	2.6×10^5	3.3	3
ads_+120h	3	1	irrelevant data		1
ads_pH-5	8	3	irrelevant data		2
ads_pH+5-6	7	2	irrelevant data		1
ads_pH+6-7	13	3	3.2×10^5	5.1	2
ads_pH+7-8	34	6	3.3×10^5	2.5	3
ads_pH+8-9	4	3	irrelevant data		2
ads_pH+9	2	2	irrelevant data		2
ads_mV-50	48	5	1.5×10^5	6.1	2
ads_mV+50-500	33	5	2.7×10^5	2.1	2
ads_mV+500-2000	7	1	irrelevant data		1
in situ	42	5	1.2×10^5	5.7	3

TABLE 4. OPERATIONAL PDFS OF K_d VALUES (L kg^{-1}) FOR Co

Parameter	N	NE	GM	GSD	#ref
Ads	534	17	4.3×10^4	9.5	23
ads_-2h	108	8	6.6×10^3	2.8	8
ads_+2h-24h	186	13	4.1×10^4	5.2	17
ads_+24h-48h	66	7	4.3×10^4	1.3×10^1	10
ads_+48h-120h	31	5	2.5×10^5	3.8	7
ads_+120h	101	9	1.5×10^5	8.8	11
ads_pH-5	15	3	1.0×10^2	2.1	3
ads_pH+5-6	8	3			3
ads_pH+6-7	19	4	1.9×10^3	4.9	4
ads_pH+7-8	270	12	4.1×10^4	7.2	18
ads_pH+8-9	134	6	1.7×10^5	5.2	8
ads_pH+9	4	1	irrelevant data		1
ads_mV-50	140	12	6.6×10^4	7.9	17
ads_mV+50-500	246	10	3.0×10^4	1.0×10^1	12
ads_mV+500-2000	74	2	2.5×10^4	5.5	5
ads_mV+2000	58	2	2.4×10^4	7.9	2
Des	74	5	4.8×10^5	4.9	5
des_-2h	21	3	4.4×10^5	6.1	3
des_+2h-24h	27	4	4.6×10^5	6.7	4
des_+24h-48h	7	2	irrelevant data		2
des_+48h-120h	4	2	irrelevant data		2
des_+120h	14	2	3.9 × 105	2.9	2
des_pH+6-7	12	1	irrelevant data		1
des_pH+7-8	23	3	2.5×10^5	4.7	3
des_pH+8-9	39	3	7.1×10^5	4.1	3
des_mV-50	32	3	7.6×10^5	6.4	3
des_mV+50-500	30	3	2.9×10^5	4.0	3
des_mV+2000	12	1	irrelevant data		1
in situ	29	3	4.4×10^4	3.8	5
ads	569	28	8.5×10^3	6.7	25
ads_-2h	60	8	2.2×10^3	1.0×10^1	8
ads_+2h-24h	225	19	1.1×10^4	3.3	19
ads_+24h-48h	23	6	8.3×10^3	9.2	7
ads_+48h-120h	53	12	1.2×10^4	3.3	10
ads_+120h	132	13	1.6×10^4	8.1	14
ads_pH-5	64	7	3.4×10^2	1.6×10^1	5

TABLE 5. OPERATIONAL PDFS OF K_d VALUES (L kg^{-1}) FOR Cs

Parameter	N	NE	GM	GSD	#ref
ads_pH+5-6	39	6	5.4×10^3	0.0	4
ads_pH+6-7	29	6	1.8×10^3	1.3×10^1	5
ads_pH+7-8	212	17	1.6×10^4	5.3	17
ads_pH+8-9	66	9	6.6×10^3	3.3	9
ads_pH+9	17	4	8.5×10^2	9.7	3
ads_mV-50	167	12	1.7×10^4	2.4	14
ads_mV+50-500	250	18	6.5×10^3	9.9	14
ads_mV+500-2000	54	2	1.8×10^4	2.1	3
ads_mV+2000	75	5	1.6×10^3	1.2×10^1	5
des	119	6	2.8×10^4	2.4	6
des_-2h	32	3	3.8×10^4	2.0	4
des_+2h-24h	52	5	1.8×10^4	2.5	5
des_+24h-48h	5	1	irrelevant data		1
des_+48h-120h	7	2	irrelevant data		2
des_+120h	23	2	5.1×10^4	1.4	2
des_pH+6-7	10	1	irrelevant data		1
des_pH+7-8	69	4	3.4×10^4	2.5	5
des_pH+8-9	40	3	1.5×10^4	2.3	3
des_mV-50	63	3	3.0×10^4	2.3	3
des_mV+50-500	46	4	3.1×10^4	2.5	4
des_mV+2000	10	1	irrelevant data		1
in situ	219	30	2.9×10^4	5.9	17

TABLE 6. OPERATIONAL PDFs OF K_d VALUES (L kg^{-1}) FOR I

Parameters	N	NE	GM	GSD	#ref
ads	158	12	2.2×10^2	$3.2 \times 10^1 1$	10
ads_oxic	124	11	4.4×10^3	1.4×10^1	9
ads_anoxic	34	3	2.3	9.6	3
des	1	1	irrelevant data		1
des_+120h	1	1	irrelevant data		1
des_pH+7-8	1	1	irrelevant data		1
des_mV+2000	1	1	irrelevant data		1
in situ	20	1	irrelevant data		1

TABLE 7. OPERATIONAL PDFS OF K_d VALUES (L kg^{-1}) FOR Mn

Parameters	N	NE	GM	GSD	#ref
ads	190	12	1.3×10^{5}	1.2×10^{1}	16
ads_-2h	33	5	1.4×10^{4}	6.0	6
ads_+2h-24h	66	11	1.5×10^{5}	6.8	14
ads_+24h-48h	43	3	1.3×10^{5}	2.0×10^{1}	5
ads_+48h-120h	24	5	4.5×10^{5}	8.6	6
ads_+120h	20	7	5.4×10^{5}	5.4	6
ads_pH-5	9	1			1
ads_pH+5-6	13	2	6.3×10^{2}	3.6	2
ads_pH+6-7	3	1			1
ads_pH+7-8	105	9	6.9×10^{4}	7.4	12
ads_pH+8-9	49	5	1.1×10^{6}	4.1	7
ads_pH+9					
ads_mV-50	89	8	2.2×10^{5}	1.2×10^{1}	11
ads_mV+50-500	79	6	6.0×10^{4}	1.1×10^{1}	7
ads_mV+500-2000	8	1			1

TABLE 8. OPERATIONAL PDFS OF K_d VALUES (L kg^{-1}) FOR Mn

Parameters	N	NE	GM	GSD	#ref
ads_mV+2000	10	1			1
des	46	3	6.9×10^{5}	6.6	3
des_-2h	17	2	5.4×10^{5}	6.7	2
des_+2h-24h	20	3	6.6×10^{5}	1.1×10^{1}	3
des_+24h-48h	5	1			1
des_+48h-120h	4	1			1
des_pH+7-8	22	3	2.0×10^{5}	5.0	3
des_pH+8-9	24	2	2.6×10^{6}	2.2	2
des_mV-50	32	3	1.3×10^{6}	5.6	3
des_mV+50-500	14	2	1.9×10^{5}	6.4	2
des_mV+500-2000	no data				
des_mV+2000	no data				
in situ	17	2	7.8×10^{4}	1.8	4

TABLE 9. OPERATIONAL PDFS OF K_d VALUES (L kg^{-1}) FOR Pu

Parameters	N	NE	GM	GSD	#ref
ads	37	7	7.9×10^{4}	2.2	3
ads_+2h-24h	1	1	irrelevant data		1
ads_+120h	15	2	irrelevant data		1
ads_mV-50	4	1	irrelevant data		1
ads_mV+50-500	27	6	8.7×10^{4}	2.3	2
ads_mV+500-2000	1	1	irrelevant data		1
des	41	6	3.0×10^{5}	4.2	4
des_-2h	1	1	irrelevant data		1
des_+2h-24h	5	2	irrelevant data		2
des_+24h-48h	3	2	irrelevant data		2
des_+48h-120h	1	1	irrelevant data		1
des_+120h	6	2	irrelevant data		2
des_mV+50-500	10	1	irrelevant data		1
des_mV+500-2000	30	4	3.4×10^{5}	2.5	2
in situ	79	9	2.4×10^{5}	6.6	6

TABLE 10. OPERATIONAL PDFS OF K_d VALUES (L kg^{-1}) FOR Sr

Parameters	N	NE	GM	GSD	#ref
ads	156	10	1.8×10^2	4.6	8
ads_+2h-24h	51	5	2.8×10^2	5.0	5
ads_+24h-48h	1	1	irrelevant data		1
ads_+48h-120h	48	3	8.5×10^1	4.0	3
ads_+120h	13	2	5.4×10^2	1.9	2
ads_pH+5-6	7	1	irrelevant data		1
ads_pH+6-7	3	1	irrelevant data		1
ads_pH+7-8	81	6	9.3×10^1	3.0	4
ads_pH+8-9	28	2	4.6×10^1	1.5	2
ads_pH+9	2	1	irrelevant data		1
ads_mV-50	2	1	irrelevant data		1
ads_mV+50-500	48	7	1.4×10^2	4.1	4
ads_mV+500-2000	7	1	irrelevant data		1
ads_mV+2000	84	3	4.5×10^1	4.1	3
des	34	5	6.2×10^2	2.1	3
des_+2h-24h	1	1	irrelevant data		1
des_+24h-48h	no data				
des_+48h-120h	no data				
des_+120h	6	1	irrelevant data		1
des_pH+6-7	6	1	irrelevant data		1
des_pH+7-8	18	3	4.4×10^2	1.4	2
des_pH+8-9	10	1	irrelevant data		1
des_mV+50-500	27	3	irrelevant data		1
des_mV+2000	7	2	irrelevant data		2
in situ	13	5	1.2×10^3	2.7	3

3.2 K_d values for radionuclides with moderate databases

For Be, Ba, Ce, Ra, Ru, Sb, and Th, the same methodology as previously detailed was used, i.e.: (i) construction of a database including information on experimental factors; (ii) application of quality criteria; (iii) determination of non-conditional *PDF*s by a bootstrap statistical procedure. Thus, the only difference with section 3.1. is that no conditional *PDF*s were defined for reducing the uncertainty over a given range of values for any parameter.

The literature sources are given in Appendix 1, and the various global *PDF*s that were thus generated were summarised in Table 11.

TABLE 11. OPERATIONAL *PDF*s OF K_d VALUES FOR Ba, Be, Ce, Ra, Ru, Sb AND Th, (L kg^{-1})

Element	N	NE	GM	GSD	#ref
Ba (Barium)	48	5	2.0×10^{3}	3.6	5
Be (Beryllium)	28	5	4.2×10^{4}	3.6	5
Ce (Cerium)	15	4	2.2×10^{5}	2.8	3
Ra (Radium)	75	8	7.4×10^{3}	3.1	7
Ru (Ruthenium)	74	5	3.2×10^{4}	1.9	3
Sb (Antimony)	23	5	5.0×10^{3}	3.8	5
Th (Thorium)	63	8	1.8×10^{5}	2.1×10^{1}	7

4.3.3 Generic K_ds to be used in risk assessment

Table 19 summarizes the generic *Kd* values, which can be used when conducting a study on a river for which no specific information is available. Information related to radionuclides others than given in the current document (i.e. Cr, Fe, Zn, Zr, Tc, Pm, Eu, U, Np, Cm) can be found in Technical Reports Series No. 364 (Table 14) [52]. Expected values for these radionuclides presented in Technical Reports Series No. 364 [52] are mainly product of expert estimates and based only on one review published in 1981 [53] while a majority of key references mentioned in the list of references to this paper were issued rather recently.

It is therefore advised to use these latter values with cautious because they were not reviewed and/or analysed as those represented in Table 13.

TABLE 12. K_d VALUES IN FRESHWATER ECOSYSTEMS FOR DIFFERENT ELEMENTS, (L kg^{-1})

Elements	N	GM	GSD	Min	Max	# ref.
Ag	81	8.5×10^{4}	2.3	2.2×10^{4}	3.3×10^{5}	7
Am	88	8.5×10^{5}	3.7	2.5×10^{4}	1.8×10^{6}	4
Ba	48	2×10^{3}	3.6	2.5×10^{2}	1.6×10^{4}	5
Be	28	4.2×10^{4}	3.6	5.1×10^{3}	3.4×10^{5}	5
Ce	15	2.2×10^{5}	2.8	4.2×10^{4}	1.2×10^{6}	3
Co	534	4.3×10^{4}	9.5	1.1×10^{3}	1.7×10^{6}	23
Cs	569	8.5×10^{3}	6.7	3.7×10^{2}	1.9×10^{5}	25
I	124	4.4×10^{3}	14	5.8×10^{1}	3.4×10^{5}	9
Mn	190	1.3×10^{5}	12	2.1×10^{3}	7.4×10^{6}	16
Pu	79	2.4×10^{5}	6.6	1.1×10^{4}	5.2×10^{6}	6
Ra	75	7.4×10^{3}	3.1	1.1×10^{3}	5.2×10^{4}	7
Ru	74	3.2×10^{4}	1.9	1.1×10^{4}	9.3×10^{4}	3
Sb	23	5×10^{3}	3.8	5.5×10^{2}	4.6×10^{4}	5
Sr	156	1.8×10^{2}	4.6	1.4×10^{1}	2.2×10^{3}	8
Th	63	1.8×10^{5}	21	1.2×10^{3}	2.7×10^{7}	7

TABLE 13. GROSS AVERAGE K_d VALUES WITH EMPHASIS ON OXIDIZING CONDITIONS IN AQUEOUS SYSTEMS, L kg^{-1}

Element	Expected value	Min.	Max.
Cm	5.0×10^3	1.0×10^1	7.0×10^4
Cr	Low		
Eu	5.0×10^2	2.0×10^2	8.0×10^2
Fe	5.0×10^3	1.0×10^3	1.0×10^4
Np	1.0×10^1	2.0×10^{-1}	1.0×10^2
Pm	5.0×10^3	1.0×10^3	1.0×10^4
Tc	$5.0\ 10^0$	-	1.0×10^2
U	5.0×10^1	2.0×10^1	1.0×10^3
Zn	5.0×10^2	1.0×10^2	1.0×10^3
Zr	1.0×10^3	1.0×10^3	1.0×10^4

5. CONCLUSIONS

The process of interaction of dissolved radionuclides with solids particles in suspension or deposited, is usually modelled according to the 'K_d concept', where K_d is the partition coefficient 'particulate form/dissolved form' based on the hypothesis of a reversible and rapid equilibration between the dissolved and the adsorbed radionuclides. However, this is not generally and rigorously true for every radionuclide. The equilibrium between the concentrations of the dissolved and the attached phases may be not instantaneously achieved and the adsorption and desorption processes are not always rapidly reversible.

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APPENDIX I

List of publication used for evaluation of freshwater K_d values

BARROS H., ABRIL J.M., Constraints in the construction and/or selection of kinetic box models for the uptake of radionuclides and heavy metals by suspended particulate matter, Ecological Modelling **185**(2-4) (2005) 371-385.

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APPENDIX II

DATA ON EXPERIMENTS USED FOR EVALUATION OF FRESHWATER K_d VALUES

TABLE.A2.1. DETAILS OF THE EXPERIMENTS USED FOR BUILDING PDFs FOR Ag, Am, Co, Cs, I, Mn, Pu AND Sr

References	Ag	Am	Co	Cs	I	Mn	Pu	Sr	Material	Environment	Experiment	Co-factors
Albrecht et al., 1995			X						SPM	Aare river, Switzerland	in situ	No
Albrecht, 2003			X						SPM	Aare river, Switzerland	in situ	No
Andreola et al., 1988			X			X			SPM	Pô river, Italy	in situ	No
Bale, 1987				X					SPM	Tamar river, UK	labo	pH
Benes et al., 1988			X	X					SPM/Sed	Dudvah river, Slovakia	labo	m/V ratio, contact time
Benes et al., 1989			X						Sed	Dudvah river, Slovakia	labo	contact time, pH, m/V ratio
Benes et al., 1989-a				X					SPM/Sed	Dudvah river, Slovakia ; Skryjsky and Jihlava rivers, Czechoslovakia	labo	pH, m/V ratio, material composition
Benes and Poliak, 1990								X	Sed	Dudvah river, Slovakia	labo	m/V ratio
Benes et al., 1992-I			X						SPM/Sed	Dudvah river, Slovakia ; Rhône and Loire river, France	labo	contact time, m/V ratio
Benes and Cernik, 1992-II			X						SPM/Sed	Dudvah river, Slovakia	labo	contact time, m/V ratio
Benes et al., 1992-b				X					SPM/Sed	Dudvah river, Slovakia	labo	m/V ratio
Bird et al., 1995-a					X				Sed	Lake 239 and Winnipeg river, Canada	labo	oxic/anoxic conditions
Bird et al., 1995-b					X				SPM/Sed	Lake 226 and 227, Canada	labo	contact time, particulate size
Bird and Schwartz, 1996					X				Sed	shield lake and Winnipeg river, Canada	labo	contact time, m/V ratio, oxic/anoxic conditions
Bunker et al., 2000			X	X				X	Sed	Botany pond and Esthwaite water lake, UK	labo	contact time
Cartier et al., 1990-a	X		X	X	X	X			SPM	Garonne river, France	labo	contact time, pH, m/V ratio, [K+]
Cartier et al., 1990-b	X		X	X	X	X			SPM	Meuse river, France	labo	contact time, m/V ratio
Cartier et al., 1990-c					X				SPM	Rhône, Loire and Seine rivers, France	labo	contact time, m/V ratio
Ciceri et al., 1988			X	X	X	X		X	SPM	Pô river, Italy	in situ/ labo	m/V ratio
Ciffroy et al., 1995	X		X			X			SPM	Garonne river, France	in situ	particle size
Ciffroy et al., 2001	X		X	X		X			SPM	Vienne river, France	labo	contact time, season
Ciffroy et al., 2003	X		X	X		X			SPM	Loire river, France	labo	contact time, season
Cochran et al., 2000		X		X	X				Sed	Ob river, Russian Federation	labo	m/V ratio, DOC

TABLE A2.1. DETAILS OF THE EXPERIMENTS USED FOR BUILDING FOR PDFs Ag, Am, Co, Cs, I, Mn, Pu AND Sr (Cont.)

References	Ag	Am	Co	Cs	I	Mn	Pu	Sr	Material	Environment	Experiment	Co-factors
Comans et al., 1989				X					Sed	Ketelmeer lake, Netherlands	in situ	[NH4+]
Comans et al., 1998				X					Sed	Devoke water lake, UK	in situ	[K+]
Conkic et al., 1988				X					Sed	Danube river, Yugoslavia	in situ	no
Drndarski and Lavi, 1996				X					Sed	Grliska artificial lake, Yugoslavia	in situ	no
Elprince, 1978				X					Sed	Savannah river Plant, USA	labo	pH
Garnier et al., 1996			X	X		X			SPM	Lena river, Russian Federation	labo	no
Garnier et al., 1997	X		X	X		X			SPM	Vienne river, France	labo	contact time, season
Hatje et al., 2003			X			X			SPM	Paramatta river, Australia	labo	contact time, pH,m/Vratio
Hemming et al., 1997								X	Sed	waste disposal pond and Snake river, USA	labo	m/V ratio
Ilus and Saxen, 2005				X					Sed	Some lakes, Finland	in situ	no
Joshi and Mc Crea, 1992			X	X				X	SPM	Ottawa river, Canada	in situ	m/V ratio
Konoplev et al., 2002				X					SPM	Constance lake and Rhine river, Germany	in situ	no
Li et al., 1984			X	X		X			SPM	Hudson and Mississippi rivers, USA	labo	contact time
Lima and Mazzilli, 1994			X	X					Sed	Pinheiros river, Brazil	labo	pH
Martin and Thomas, 1990	X	X	X	X		X	X		SPM	Rhône river, France	in situ	no
Matsunaga et al., 1991				X					SPM	Kuji river, Japan	in situ	pH, m/V ratio, [K+]
Matsunaga et al., 1998		X		X			X	X	SPM	Pripyat and Sahan rivers, Glubokoye lake, Ukraine	in situ	pH, [K+], [Ca2+], DOC
McCubbin et al., 2004							X		SPM/Sed	Ponds Water management system of AWE, UK	in situ	contact time, pH
Milton et al., 1992					X				SPM	Perch lake, Canada	labo	m/V ratio
Murdock et al., 1995		X		X			X		SPM	Drigg stream (waste disposal site), UK	in situ	m/V ratio
Nakayama and Nelson, 1988		X							Sed	Michigan lake, Saganashkee Slough, a pond, Volo Bog, USA	labo	COC
Nelson et al., 1985							X		SPM/Sed	Miami-Erie Canal and Michigan lake, USA	in situ/labo	contact time, m/V ratio, DOC
Onishi et al., 1981		X	X				X	X	SPM/Sed	Review	in situ/labo	pH, m/V ratio
Orlandini et al., 1990		X					X		SPM	Lake Trawsfynnydd, UK	in situ	m/V ratio

TABLE A2.1. DETAILS OF THE EXPERIMENTS USED FOR BUILDING PDFs FOR Ag, Am, Co, Cs, I, Mn, Pu and Sr (Cont.)

References	Ag	Am	Co	Cs	I	Mn	Pu	Sr	Material	Environment	Experiment	Co-factors
Pham and Garnier, 1998	X		X	X		X			SPM	Vienne river, France	labo	Colloidal concentration
Queirazza et al., 1987			X	X		X		X	SPM	Pô river, Italy	in situ/labo	no
Robbins et al., 1992				X					Sed	Constance lake, Germany	labo	contact time, m/V ratio
Robbins and Jasinski, 1995				X					Sed	lake Sniardwy, Poland	labo	contact time, m/V ratio
Santschi et al., 1990				X					SPM	Zurich lake, Switzerland	in situ	no
Schell et al., 1980		X	X	X			X	X	Sed	Michigan lake, Hudson and Clinch river, Cattaragus creek, USA	labo	pH, m/V ratio
Smith et al., 1997				X					SPM	Some Cumbrian lakes, UK	in situ	m/V ratio
Smith and Comans, 1996				X					Sed	Ketelmeer and Holland Diep lakes, Netherlands	in situ	[K+], [NH4+]
Standring et al., 2002-a						X			Sed	Flekke river, Sogn and Fjordane county, Norway	labo	contact time, sediment composition
Standring et al., 2002-b			X	X				X	Sed	Reservoir 10 of Mayak PA, Russian Federation	in situ	no
Stephens et al., 1998				X				X	Sed	Savannah river site, USA	labo	m/V ratio
Strand et al., 1999				X				X	Sed	Reservoir 10 and 11 of Mayak PA, Russian Federation	in situ	no
Svadlenkova et al., 1990					X				Sed	Vltava river, Czechoslovakia	labo	contact time
Thirion et al., 1983			X	X		X			SPM	Loire river, France	labo	contact time, m/V ratio
Turner et al., 1993				X		X			SPM	Ouse river, UK	labo	m/V ratio
Turner and Millward, 1994				X		X			SPM	Dee river, UK	labo	contact time, pH
Turner, 1996				X					SPM	Clyde, Conwy, Restronguet and Tweed rivers, UK	labo	no
Um et al., 2004					X				Sed	Hanford site, USA	labo	pH

TABLE A2.1. DETAILS OF THE EXPERIMENTS USED FOR BUILDING PDFs FOR Ag, Am, Co, Cs, I, Mn, Pu AND Sr (Cont.)

References	Ba	Be	Ce	Ra	Ru	Sb	Th	Material	Environment	Experiment	Co-factors
Barros et al, 2004	X							sediments	Odiel and Tinto estuaries	labo	
Barros et al, 2004b	X							sediments	Huelva estuary	labo	
Benes et al, 1986				X				sediments	Ploucnice river	labo	pH
Cartier et al., 1990-a						X		SPM	Garonne river, France	labo	contact time, pH, m/V ratio
Cartier et al., 1990-b						X		SPM	Meuse river, France	labo	contact time, m/V ratio
Carvalho, 1999				X				SPM	Tagus river, Spain	in situ	
Ciceri et al., 1988	X				X	X		SPM	Pô river, Italy	in situ/labo	m/V ratio
Ciffroy et al, 2003b		X						SPM	Loire river, France	in situ	
Dibb et al, 1989		X						SPM	Chesapeake Bay	in situ/labo	salinity
Drndarski et al, 1996				X			X	sediments	Grliska artificial lake, Yugoslavia	in situ	
Joshi et al, 1992		X	X				X	SPM	Ottawa river, Canada	in situ	
Kaplan et al, 2001							X	sediments	Savannah river, USA	in situ	
Li et al., 1984	X		X			X		SPM	Hudson and Mississippi rivers, USA	labo	contact time
Lima et al, 1994								Sed	Pinheiros river, Brazil	labo	pH
Maringer, 1996				X				SPM	Danube river, Austria	in situ	
Martin et al, 1990					X			SPM	Rhône river, France	in situ	
Mljac et al, 1996				X				Sed	Lake Velenje and Paka river	in situ	
Mundschenk, 1996	X							SPM	Moselle river	labo	M/V ratio, contact time
Olsen et al, 1986		X						SPM	James river, Hudson Bay, Susquehanna river	in situ	
Orlandini et al, 1996							X	SPM	Lake Trawsfynnydd, UK	in situ	m/V ratio
Perianez et al, 1997							X		Odiel river, Spain		
Sanchez et al, 1999				X			X	SPM	Jucar and Cabriel river, Spain	in situ	pH, temperature, conductivity
Schell et al., 1980					X			Sed	Michigan lake, Hudson and Clinch river, USA	labo	pH, m/V ratio
Steinmann et al, 1999		X						SPM	Lugano Lake, Switzerland	in situ	

APPENDIX III

GRAPHICAL PRESENTATION OF PROBABILITY DENSITY FUNCTIONS DERIVED FOR SOME RADIONUCLIDES

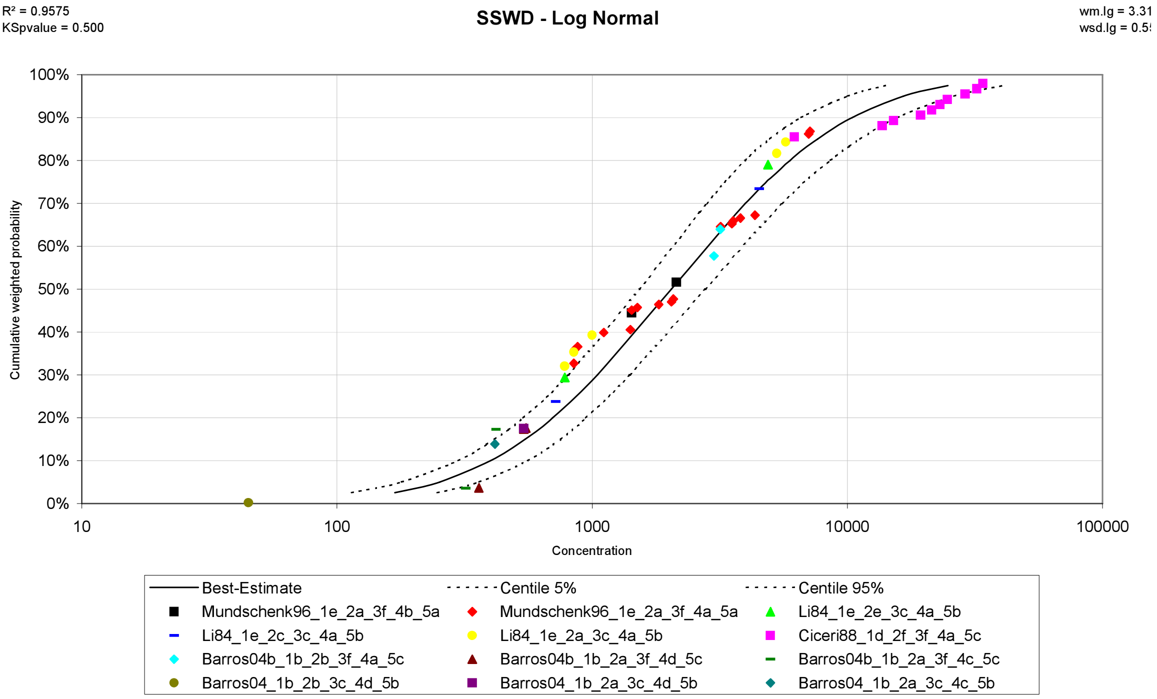

FIG.1. PDF for Ba.

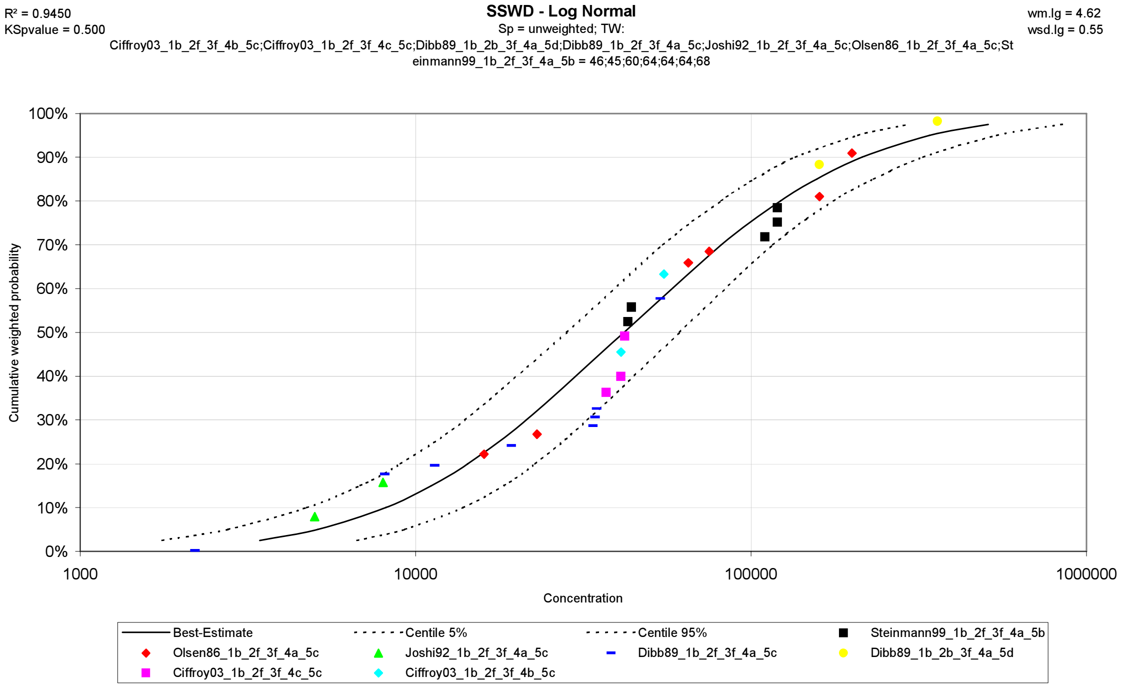

FIG. 2. PDF FOR Be.

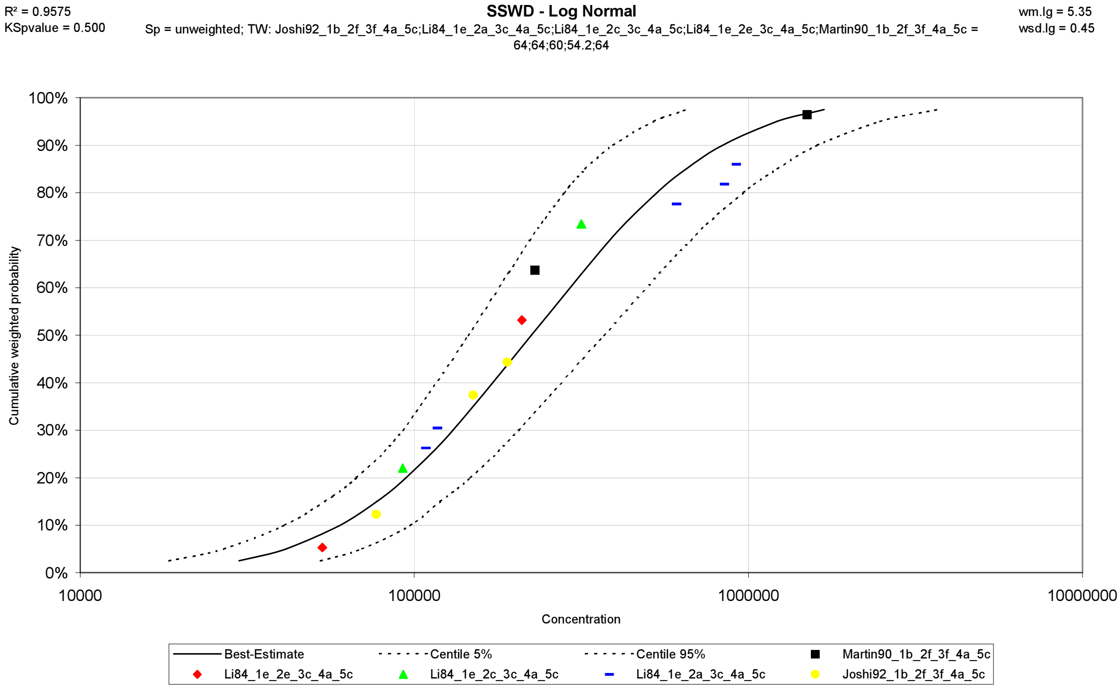

FIG. 3. PDF for Ce.

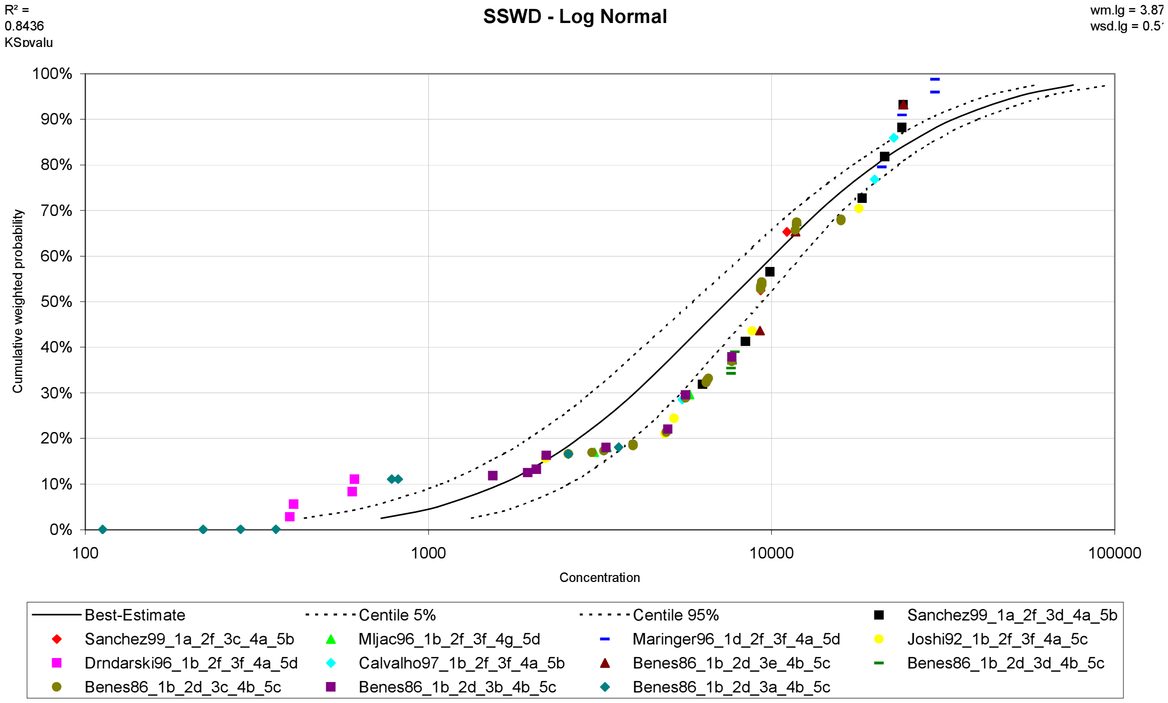

FIG. 4. PDF for Ra.

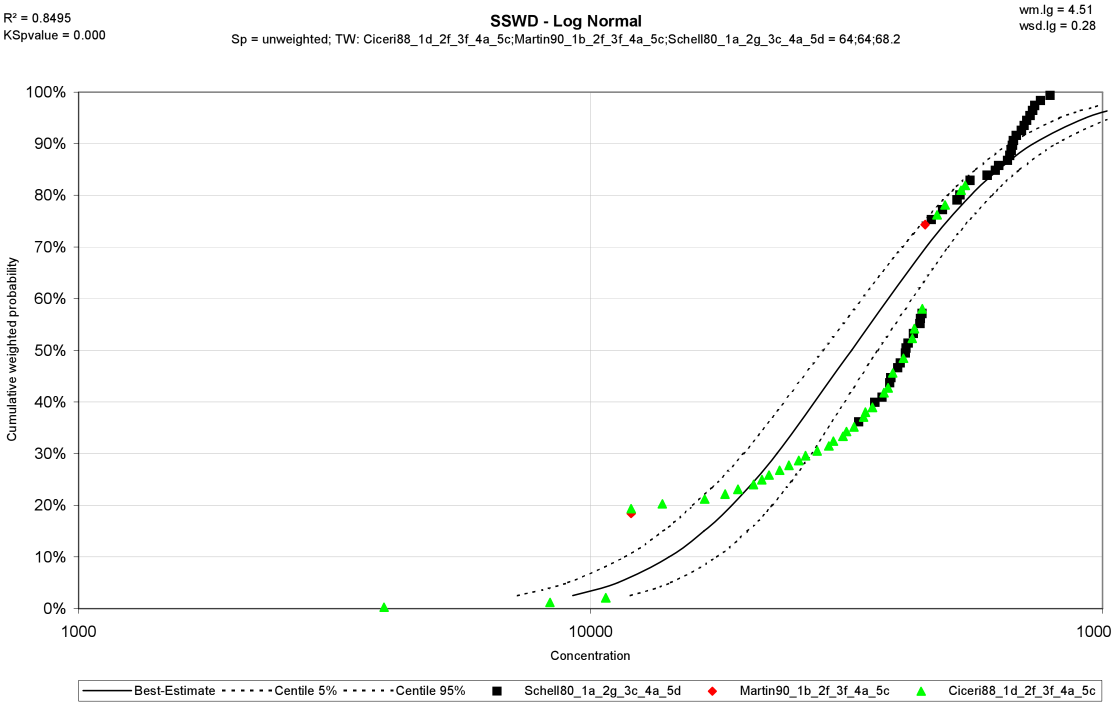

FIG. 5. PDF for Ru.

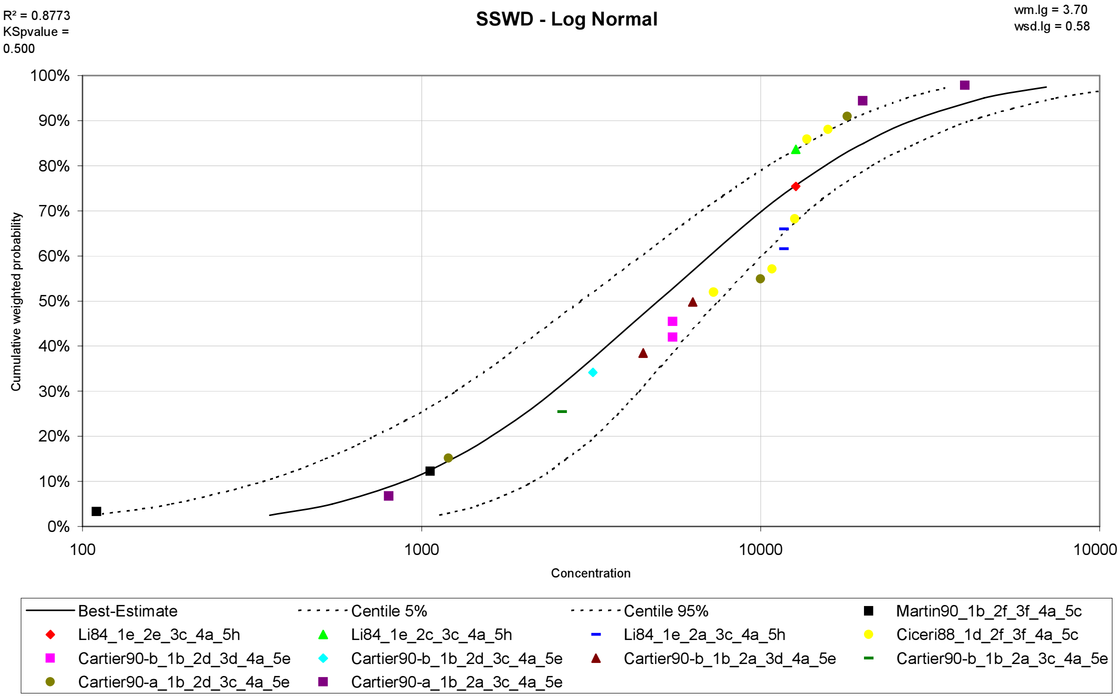

FIG. 6. PDF for Sb.

TRANSFERS TO FRESHWATER BIOTA

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Abstract

It remains important to monitor the fate of radionuclides, particularly in environmental compartments that comprise human pathways. Therefore, an extensive literature survey has been conducted to compile available data on the transfer of radionuclides and their analogues to edible freshwater biota. Focus was placed on compilation of steady state transfer parameters for two freshwater pathways, including water-to-biota and sediment-to-biota. In general, although in many cases, extensive data were available for fishes and invertebrates, relatively fewer data were available for freshwater primary producers, amphibians and reptiles. To fill in these gaps, data were also compiled on the internal partitioning of elements in the body with respect to tissue masses, which could be used to estimate radionuclide concentrations between compartments in the body.

1. MAIN CONCEPTS

Humans can consume a wide variety of freshwater biota, which broadly includes fish, freshwater primary producers, invertebrates, amphibians, reptiles, mammalian species and waterfowl. Within each type of freshwater foodstuff, humans may also consume a number of different tissue types (Tables 1 and 2) [1-13], which can contribute to the accumulation of radionuclides into human tissues, depending upon the propensity of a given tissue to accumulate radionuclides and the efficiency with which radionuclides in different types of food are incorporated into the human body [14-19].

Humans may also indirectly receive accumulated radionuclides through food-chain transfer processes whereby radionuclides are transferred from lower trophic levels that are not typically consumed by humans to higher, edible trophic levels (e.g. Figure 1). All such transfer processes can be mathematically described through the application of steady state or dynamic radionuclide transfer models for freshwater ecosystems [20-28].

Although it is generally recognized that accumulation of contaminants (in this case, radionuclides) by edible aquatic organisms is a dynamic process, many contaminant bioaccumulation models assume that the aquatic organisms are in equilibrium with reference media, such as water or sediments, in their surrounding environment. As a result, radionuclide accumulation into aquatic biota is often represented by simplified ratios that relate radionuclide concentrations in biotic tissues to concentrations in the reference media [25 and 29].

TABLE 1. SUMMARY OF EDIBLE TISSUES FOR EACH TYPE OF FRESHWATER PLANT SPECIES [13]

Type of Freshwater Biota	Whole	Leaves	Stems	Roots	Fruit
Cyanobacteria	√				
[a] Filamentous Algae	√				
Floating-leaved Macrophytes				√	√
Emergent Macrophytes		√	√	√	√

[a] e.g. epiphyton.

TABLE 2. SUMMARY OF EDIBLE TISSUES FOR EACH TYPE OF FRESHWATER ANIMAL SPECIES [1-13]

Type of Freshwater Animal	Whole	Soft Tissues	Muscle	Liver	Eggs	Comments
Freshwater Invertebrate:						
Freshwater Mussels		√				
Benthic Crustaceans		√			√	
Snails		√			√	
Macroinvertebrates		√				
Freshwater Fish:						
Forage Fish	√		√			
Benthic Fish			√	√	√	
Piscivorous Fish/Eels			√		√	
Reptiles:						
Snakes			√		√	ld[1]
Alligators			√		√	ld
Turtles			√		√	ld
Amphibians:						
Tadpoles	√					ld
Frogs	√		√			ld
Freshwater Birds:						
Waterfowl		√	√		√	
Aquatic Mammals:						
Muskrats			√			Not covered.
Otters			√			Not covered.
Beavers			√			Not covered.

[1] ld means that data are relatively limited.

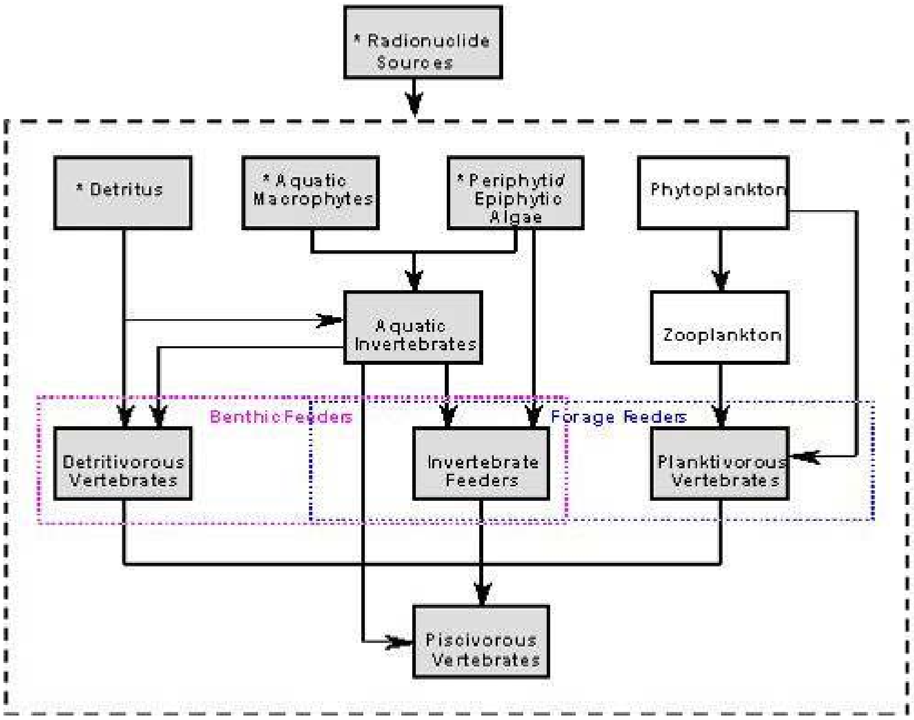

FIG. 1. Conceptual Model Depicting Trophic Interactions in Aquatic Ecosystems.

The steady state models can be sub-divided into two categories based on the chemical behaviour of a given radionuclide and its associated transfer processes to edible biotic tissues. These include: i) models that are based on simple radionuclide partitioning between organisms and reference phases (such as surface water or sediments), and ii) specific activity models, which assess partitioning of radionuclides relative to stable analogues in the body.

2. MODELS BASED ON BIOACCUMULATION FACTORS

2.1 Definitions

Depending upon the radionuclide uptake pathway being considered, a number of representations of partitioning can be defined. For example, the biota sediment accumulation factor (*BSAF*) is the concentration of a radionuclide in an organism (C_b) relative to the value measured in the sediment (C_{sed}) on a per unit fresh sediment basis, as described by:

$$BSAF = \frac{Cb}{Csed} \tag{1}$$

The *BSAF* is appropriately derived from studies in which only the sediment is contaminated.

The bioaccumulation factor (*BAF*) is the contaminant concentration in biota (C_b) from all exposure pathways (including water, sediment and ingestion/dietary pathways) relative to that of water (C_w), where:

$$BAF = \frac{Cb}{Cw} \tag{2}$$

The contribution of sediment-associated radionuclides can be of particular importance with respect to radionuclide uptake by benthic species.

2.2 Derived bioaccumulation factor values

Most contaminant transfer factors found in the literature do not distinguish between uptake pathways and therefore represent *BAF*s (also called concentration ratios, *CR*s). Water-to-biota *BAF*s from the literature have been compiled for radionuclides and their stable analogues in Tables 3-6, whereas sediment-to-biota accumulation factors (*BSAF*s) are provided in Tables from 7 to 8 (based on the data from [30-31]). The *BAF* data have been further sub-divided by type of edible tissue, as provided in Tables 9–21.All accumulation factors are given on a per unit fresh weight basis. In general, it is important to note that *BAF*s and *BSAF*s for stable elements are conservative when used to represent radionuclides with relatively short radiological half-lives and relatively long biological half-lives, since physical decay of short-lived radionuclides can significantly reduce their concentration in biota tissues [32]. To account for this, the *BAF* or *BSAF* can be multiplied by a factor, K, that accounts for the radionuclide-specific half-lives, as described by:

$$K = \frac{\lambda b}{\lambda r + \lambda b} \tag{3}$$

where λ_b is the biological decay constant = 0.693 t_b^{-1} (day^{-1}); λ_r = radioactive decay constant = 0.693 t_r^{-1} (day^{-1}); t_b = biological half-life (day); t_r = radiological half-life (day). For screening purposes, a t_b of 30 days (or a λ_b of 0.023 day^{-1}) can be assumed [32].

TABLE 3. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR ALL TYPES OF EDIBLE FRESHWATER PRIMARY PRODUCERS, L/kg FW

Element	N	GM/AM[1]	GSD/SD	Min	Max	# ref
Am	16	3.7×10^3	8.3×10^0	7.5×10^0	3.8×10^4	6
C	10	1.6×10^4	1.5×10^1	4.4×10^1	8.9×10^4	1
Cd	5	1.9×10^4	6.8×10^0	1.10×10^4	2.3×10^4	2
Cm	1	8.0×10^3	nd[2]	nd	nd	1
Co	18	7.1×10^2	5.1×10^0	5.0×10^1	2.0×10^4	11
Cs	26	8.7×10^1	1.6×10^1	1.8×10^0	3.3×10^3	2
Cu	5	3.0×10^3	3.19×10^2	2.40×10^3	3.6×10^3	3
Fe	5	9.1×10^3	1.8×10^0	5.20×10^3	1.5×10^4	1
I	3	1.3×10^2	3.7×10^0	7.8×10^1	2.7×10^2	1
Mn	6	1.2×10^4	7.22×10^2	3.10×10^{-1}	1.5×10^5	4
Ni	5	7.7×10^2	1.29×10^2	2.50×10^2	1.1×10^3	3
Np	2	7.2×10^3	1.1×10^3	6.50×10^3	8.0×10^3	3
Pb	5	1.9×10^3	7.6×10^1	1.30×10^3	2.2×10^3	1
Pu	40	2.6×10^4	1.4×10^1	1.20×10^2	4.9×10^7	4
Ra	8	2.8×10^3	4.1×10^0	6.40×10^2	1.1×10^4	2
Ru	8	2.9×10^2	2.0×10^0	7.4×10^1	6.7×10^2	1
Se	31	1.4×10^3	5.4×10^0	9.4×10^0	8.2×10^3	1
Sr	17	4.1×10^2	3.3×10^0	3.9×10^1	1.9×10^3	16
Tc	8	5.5×10^0	4.9×10^0	2.80×10^{-1}	8.8×10^1	1
U	4	2.10×10^2	1.9×10^0	8.1×10^1	5.2×10^2	1
Zn	5	2.10×10^4	1.3×10^1	1.40×10^4	2.7×10^4	1

[1] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2;

[2] nd means – no data.

TABLE 4. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR FRESHWATER INVERTEBRATES, L/kg FW

Element	N	GM/AM[1]	GSD/SD[1]	Min	Max	# ref
Ag	2	2.3×10^{2}	1.4×10^{3}	1.3×10^{2}	3.3×10^{2}	2
Al	2	3.4×10^{3}	4.0×10^{2}	3.1×10^{3}	3.7×10^{3}	2
Am	17	2.4×10^{3}	7.0×10^{0}	5.9×10^{1}	8.0×10^{4}	7
As	2	1.5×10^{3}	7.1×10^{2}	1.0×10^{3}	2.0×10^{3}	2
Au	2	1.4×10^{3}	3.5×10^{2}	1.0×10^{3}	1.5×10^{3}	2
Ba	2	1.4×10^{2}	3.5×10^{0}	1.1×10^{2}	1.6×10^{2}	2
Br	2	1.3×10^{3}	7.6×10^{2}	7.2×10^{2}	1.8×10^{3}	2
C	24	6.5×10^{4}	2.6×10^{0}	1.3×10^{4}	5.7×10^{5}	3
Ca	3	3.4×10^{1}	2.5×10^{0}	1.2×10^{1}	6.6×10^{1}	2
Cd	149	1.0×10^{2}	3.90×10^{1}	1.4×10^{-2}	3.1×10^{4}	8
Ce	2	4.3×10^{2}	1.9×10^{2}	2.9×10^{2}	5.6×10^{2}	2
Cl	2	1.6×10^{2}	3.5×10^{1}	1.3×10^{2}	1.8×10^{2}	2
Co	28	2.2×10^{1}	1.3×10^{2}	1.8×10^{-3}	4.1×10^{4}	9
Cr	2	3.0×10^{2}	1.2×10^{2}	2.1×10^{2}	3.8×10^{2}	2
Cs	29	2.3×10^{1}	7.5×10^{1}	5.4×10^{-3}	6.1×10^{3}	11
Cu	82	4.2×10^{1}	1.1×10^{1}	5.6×10^{1}	1.4×10^{3}	4
Cm	2	9.5×10^{3}	7.1×10^{2}	9.0×10^{3}	1.0×10^{4}	1
Eu	2	2.2×10^{2}	2.1×10^{1}	2.0×10^{2}	2.3×10^{2}	2
Fe	2	2.0×10^{3}	2.1×10^{2}	1.8×10^{3}	2.1×10^{3}	2
Hf	2	1.4×10^{3}	1.4×10^{2}	1.3×10^{3}	1.5×10^{3}	2
Hg	31	7.5×10^{2}	2.7×10^{0}	2.0×10^{2}	5.2×10^{3}	3
I	99	1.7×10^{1}	1.1×10^{1}	4.0×10^{-1}	1.3×10^{3}	5
K	2	5.8×10^{2}	5.0×10^{1}	5.4×10^{2}	6.1×10^{2}	2
La	2	3.5×10^{2}	2.8×10^{1}	3.3×10^{2}	3.7×10^{2}	2
Lu	1	1.1×10^{3}	-	-	-	2
Mg	2	3.2×10^{1}	1.6×10^{1}	2.1×10^{1}	4.3×10^{1}	2
Mn	4	2.1×10^{1}	3.90×10^{2}	1.1×10^{-1}	3.7×10^{3}	3
Mo	33	4.5×10^{-1}	1.30×10^{1}	2.9×10^{-2}	3.0×10^{3}	3
Na	4	3.4×10^{0}	3.60×10^{1}	1.4×10^{-1}	1.1×10^{2}	3
Np	2	9.5×10^{3}	1.1×10^{0}	9.0×10^{3}	1.0×10^{4}	1
Pb	79	2.2×10^{1}	2.00×10^{1}	4.5×10^{-2}	7.0×10^{2}	3
Pu	100	7.4×10^{3}	2.90×10^{1}	3.6×10^{-1}	5.5×10^{6}	13
Ra	5	1.0×10^{2}	3.00×10^{1}	1.8×10^{0}	1.8×10^{3}	1
Rb	2	2.0×10^{3}	2.8×10^{2}	1.8×10^{3}	2.2×10^{3}	2
Ru	9	3.8×10^{-2}	2.10×10^{1}	1.8×10^{-3}	8.3×10^{1}	2
Sb	2	2.1×10^{2}	1.9×10^{2}	7.4×10^{1}	3.5×10^{2}	2
Sc	2	3.5×10^{3}	2.8×10^{2}	3.3×10^{3}	3.7×10^{3}	2
Se	16	5.7×10^{2}	1.50×10^{1}	1.2×10^{1}	6.9×10^{4}	5
Sm	2	1.6×10^{3}	1.6×10^{3}	5.0×10^{2}	2.7×10^{3}	2
Sr	5	2.7×10^{2}	3.2×10^{0}	7.7×10^{1}	1.3×10^{3}	5
Tc	10	2.6×10^{1}	9.8×10^{0}	1.8×10^{0}	4.0×10^{2}	1
Th	2	2.9×10^{3}	1.4×10^{1}	2.9×10^{3}	2.92×10^{3}	2
U	8	1.7×10^{2}	1.9×10^{1}	3.6×10^{0}	6.0×10^{4}	1
V	2	3.8×10^{2}	2.8×10^{1}	3.6×10^{2}	4.0×10^{2}	2
Zn	82	9.2×10^{1}	2.9×10^{1}	6.3×10^{-2}	1.5×10^{3}	4

[1] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2, and the single observation is presented when number of data was equal to 1.

TABLE 5. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR FRESHWATER FISH TISSUES, L/kg FW

Element	Whole					Muscle					# ref
	N	GM/AM	GSD/SD	Min	Max	N	GM/AM	GSD/SD	Min	Max	
Ag	23	1.1×10^2	1.3×10^0	5.7×10^1	1.8×10^2	27	1.1×10^2	1.5×10^0	4.0×10^1	2.1×10^2	4
Al	93	6.6×10^1	7.1×10^0	4.5×10^0	5.2×10^3	31	5.1×10^1	3.8×10^0	5.9×10^0	3.0×10^2	3
Am	nd	nd	nd	nd	nd	2	2.4×10^2	2.3×10^2	7.2×10^1	4.0×10^2	11
As	33	3.8×10^2	2.3×10^0	8.1×10^1	1.0×10^3	15	3.3×10^2	2.1×10^0	5.0×10^1	8.5×10^2	4
Au	13	2.9×10^2	2.3×10^0	5.0×10^1	1.0×10^3	17	2.4×10^2	2.1×10^0	5.0×10^1	9.0×10^2	4
Ba	92	4.7×10^1	1.7×10^0	5.0×10^0	2.2×10^2	111	1.2×10^0	3.3×10^0	5.3×10^{-2}	3.2×10^1	4
Br	37	1.6×10^2	2.3×10^0	1.5×10^1	7.9×10^2	15	9.1×10^1	2.3×10^0	1.8×10^1	3.7×10^2	4
C	nd	nd	nd	nd	nd	6	4.0×10^5	2.9×10^0	1.9×10^5	3.2×10^6	3
Ca	118	1.0×10^3	3.4×10^0	8.4×10^1	5.6×10^3	104	1.2×10^1	2.5×10^0	2.0×10^0	9.7×10^1	5
Ce	90	1.2×10^1	2.7×10^0	3.0×10^0	1.1×10^2	71	2.5×10^1	8.5×10^0	9.0e-01	1.2×10^3	5
Cl	37	8.5×10^1	1.6×10^0	2.5×10^1	2.3×10^2	16	4.7×10^1	2.2×10^0	8.9×10^0	1.2×10^2	4
Co	118	4.0×10^2	1.6×10^0	2.3×10^1	2.4×10^3	65	7.6×10^1	2.4×10^0	9.0×10^0	5.6×10^2	14
Cr	51	2.1×10^2	2×10^0	3.5×10^1	7.6×10^2	57	4.0×10^1	2×10^0	1.3×10^1	1.2×10^2	5
Cs	145	3.0×10^3	2.6×10^0	7.5×10^1	2.4×10^4	106	2.5×10^3	2.4×10^0	1.4×10^2	1.5×10^4	52
Cu	102	2.7×10^2	1.5×10^0	8.6×10^1	1.2×10^3	86	2.3×10^2	1.7×10^0	8.9×10^1	7.2×10^2	5
Dy	1	3.0×10^2	-	-	-	2	6.5×10^2	6.4×10^2	2.0×10^2	1.1×10^3	3
Eu	53	1.5×10^2	3.2×10^0	7.6×10^0	2.2×10^3	24	1.3×10^2	4.9×10^0	1.1×10^1	7.2×10^2	4
Fe	114	1.4×10^2	5.7×10^0	1.6×10^1	5.3×10^3	86	1.7×10^2	6.9×10^0	6.6×10^0	2.0×10^3	6
Hf	20	2.1×10^3	3.2×10^0	3.0×10^2	2.8×10^4	10	1.1×10^3	1.8×10^0	3.3×10^2	2.0×10^3	3
Hg	20	4.5×10^3	2.2×10^0	1.1×10^3	2.2×10^4	14	6.1×10^3	1.8×10^0	1.9×10^3	1.7×10^4	3
I	84	6.5×10^2	2.1×10^0	1.0×10^2	4.5×10^4	50	$3.0x\ 10^1$	2.5×10^0	1.1e-01	4.0×10^2	8
K	120	4.0×10^3	2.0×10^0	5.7×10^2	1.5×10^4	87	3.2×10^3	1.6×10^0	1.2×10^3	9.0×10^3	5
La	102	1.6×10^1	3.2×10^0	3.6×10^0	3.4×10^2	74	3.7×10^1	4.9×10^0	1.1×10^0	6.6×10^2	4
Mg	111	1.1×10^2	3.0×10^0	1.4×10^1	4.3×10^2	86	3.7×10^1	2.2×10^0	7.9×10^0	1.8×10^2	4
Mn	110	4.5×10^2	4.0×10^0	4.8×10^1	7.0×10^3	87	2.4×102	6.7×10^0	1.3×10^1	1.4×10^5	6
Mo	81	2.7×10^1	1.9×10^0	2.1×10^0	1.9×10^2	64	1.9×10^0	2.1×10^0	4.0×10^{-3}	2.0×10^1	5
Na	42	1.4×10^2	2.1×10^0	3.4×10^1	6.0×10^2	87	7.6×10^1	3.0×10^0	1.7×10^1	6.1×10^2	5
Ni	24	7.1×10^1	2.1×10^0	1.9×10^1	6.6×10^2	5	2.1×10^1	1.9×10^0	1.1×10^1	4.4×10^1	1
P	nd	nd	nd	nd	nd	39	1.4×10^5	1.1×10^0	1.2×10^5	1.7×10^5	1
Pb	82	3.7×10^2	3.0×10^0	5.8×10^1	5.7×10^3	39	2.5×10^1	2.9×10^0	1.0×10^{-1}	2.7×10^2	1
Po	nd	nd	nd	nd	nd	5	3.6×10^1	4.3×10^0	6.0×10^0	1.7×10^2	2

TABLE 5. SUMMARY OF BIOACCUMULATION FACTORS (BAF) FOR FRESHWATER FISH TISSUES, L/kg FW (Cont.)

Element	Whole					Muscle					
	N	GM/AM	GSD/SC	Min	Max	N	GM/AM	GSD/SD	Min	Max	# ref
Pu	nd	nd	nd	nd	nd	3	2.1×10^4	2.6×10^0	7.7×10^3	5.0×10^4	15
Ra	2	2.1×10^2	6.0×10^1	1.6×10^2	2.5×10^2	21	4.0×10^0	6.8×10^0	6.0×10^{-2}	1.5×10^2	3
Rb	113	6.1×10^3	1.6×10^0	1.2×10^3	1.6×10^3	82	4.9×10^3	1.7×10^0	1.0×10^3	1.4×10^4	4
Ru	nd	nd	Nd	nd	nd	2	5.5×10^1	6.3×10^1	1.0×10^1	1.0×10^2	3
Sb	37	7.1×10^1	8.8×10^0	4.7×10^0	9.3×10^6	20	3.7×10^1	4.5×10^0	1.8×10^0	3.6×10^2	4
Sc	30	9.3×10^2	3.6×10^0	6.7×10^1	3.7×10^4	14	1.8×10^2	2.1×10^0	3.3 × 101	7.3×10^2	3
Se	28	6.8×10^3	1.3×10^0	3.6×10^3	1.2×10^4	14	6.0×10^3	1.3×10^0	3.5×10^3	9.4×10^3	6
Sr	116	1.9×10^2	2.2×10^0	2.2×10^1	7.1×10^2	88	2.8×10^0	3.8×10^0	1.4×10^{-1}	6.9×10^1	26
Tb	18	7.5×10^2	2.6×10^0	8.0×10^1	2.4×10^3	11	4.1×10^2	1.9×10^0	2.0×10^2	1.7×10^3	3
Te	9	4.2×10^2	1.5×10^0	2.2×10^2	8.9×10^2	3	1.5×10^2	1.5×10^0	9.6×10^1	2.1×10^2	1
Th	2	1.9×10^2	2.6×10^2	3.8×10^1	3.8×10^3	3	6.0×10^0	-	6.0×10^0	6.0×10^0	5
Ti	30	3.7×10^2	1.9×10^0	1.2×10^2	1.3×10^3	13	1.8×10^2	1.4×10^0	1.1×10^2	3.5×10^2	3
Tl	81	5.8×10^2	1.9×10^0	6.4×10^1	3.1×10^3	58	9.0×10^2	2.6×10^0	6.6×10^1	1.0×10^4	3
U	2	2.4×10^0	1.3×10^0	1.5×10^0	3.3×10^0	9	8.6×10^{-1}	12×10^0	2.0×10^{-2}	2.0×10^1	3
V	103	2.8×10^2	2.0×10^0	3.0×10^1	1.1×10^3	81	9.7×10^1	1.9×10^0	1.0×10^1	2.4×10^2	4
Y	12	3.1×10^1	1.6×10^0	1.1×10^1	6.2×10^1	18	4.0×10^1	2.5×10^0	4.5×10^0	1.2×10^2	1
Zn	114	4.7×10^3	1.8×10^0	1.2×10^3	1.8×10^4	86	3.4×10^3	2.8×10^0	3.3×10^2	1.6×10^4	5
Zr	9	9.5×10^1	1.5×10^0	5.7×10^1	2.4×10^2	10	2.2×10^1	2.4×10^0	9.2×10^0	1.2×10^2	1

[1] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2; [2]nd means – no data.

TABLE 6. SUMMARY OF BIOACCUMULATION FACTORS (BAF) FOR EDIBLE HERPETOFAUNA [30], L/kg FW

Element	Biota Type (tissue)	N	GM/AM	GSD/SD	Min	Max
Al	Tadpole (whole)	3	1.0×10^{4}	1.3×10^{0}	7.5×10^{3}	1.3×10^{4}
Al	Frog (muscle)	2	1.3×10^{2}	7.0×10^{0}	1.2×10^{2}	1.3×10^{2}
Al	Frog (carcass)	2	1.3×10^{2}	2.8×10^{1}	1.1×10^{2}	1.5×10^{2}
As	Tadpole (whole)	3	1.4×10^{2}	1.3×10^{0}	1.1×10^{2}	1.8×10^{2}
As	Frog (muscle)	2	5.2×10^{1}	4.0×10^{1}	2.4×10^{1}	8.0×10^{1}
As	Frog (carcass)	2	1.2×10^{2}	6.0×10^{1}	7.4×10^{1}	1.6×10^{2}
Ca	Tadpole (whole)	3	4.5×10^{1}	1.6×10^{0}	2.6×10^{1}	6.3×10^{1}
Ca	Frog (muscle)	2	3.5×10^{0}	7.0×10^{-2}	3.4×10^{0}	3.5×10^{0}
Ca	Frog (carcass)	2	2.9×10^{2}	7.0×10^{-2}	2.8×10^{2}	2.9×10^{2}
Ca	Reptile (carcass)	9	1.6×10^{2}	1.1×10^{1}	5.2×10^{1}	3.4×10^{2}
Cd	Tadpole (whole)	3	2.1×10^{2}	1.4×10^{0}	1.4×10^{2}	2.8×10^{2}
Cd	Frog (muscle)	2	1.2×10^{2}	7.0×10^{0}	1.1×10^{2}	1.2×10^{2}
Cd	Frog (carcass)	2	2.4×10^{2}	2.1×10^{1}	2.2×10^{2}	2.5×10^{2}
Co	Tadpole (whole)	3	8.3×10^{3}	1.1×10^{0}	7.3×10^{3}	9.5×10^{3}
Co	Frog (muscle)	2	5.5×10^{2}	5.0×10^{2}	1.9×10^{2}	9.0×10^{2}
Co	Frog (carcass)	2	2.4×10^{3}	8.5×10^{2}	1.8×10^{3}	3.0×10^{3}
Co	Reptile (carcass)	9	2.6×10^{3}	1.8×10^{0}	1.6×10^{3}	4.2×10^{3}
Cr	Tadpole (whole)	3	2.9×10^{2}	1.5×10^{0}	2.1×10^{2}	4.4×10^{2}
Cr	Frog (muscle)	2	8.2×10^{1}	7.0×10^{-1}	8.2×10^{1}	8.3×10^{1}
Cr	Frog (carcass)	2	2.6×10^{3}	3.3×10^{3}	1.9×10^{2}	4.9×10^{3}
Cs	Tadpole (whole)	3	3.0×10^{3}	1.3×10^{0}	2.5×10^{3}	4.0×10^{3}
Cs	Frog (muscle)	2	2.6×10^{2}	1.2×10^{0}	1.7×10^{2}	3.4×10^{2}
Cs	Frog (carcass)	2	2.1×10^{2}	6.4×10^{1}	1.6×10^{2}	2.5×10^{2}
Cs	Reptile (carcass)	9	2.8×10^{2}	1.3×10^{0}	1.3×10^{2}	5.0×10^{2}
Cu	Tadpole (whole)	3	2.2×10^{2}	1.3×10^{0}	1.7×10^{2}	2.6×10^{2}
Cu	Frog (muscle)	2	1.1×10^{2}	0	1.1×10^{2}	1.1×10^{2}
Cu	Frog (carcass)	2	4.4×10^{2}	2.3×10^{0}	2.8×10^{2}	6.0×10^{2}

[1] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is

[2] nd means – no data.

TABLE 6. SUMMARY OF BIOACCUMULATION FACTORS (BAF) FOR EDIBLE HERPETOFAUNA [30], L/kg FW (Cont.)

Element	Biota Type (tissue)	N	GM/AM	GSD/SD	Min	Max
Fe	Tadpole (whole)	3	2.4×10^{3}	1.2×10^{0}	1.9×10^{3}	2.8×10^{3}
Fe	Frog (muscle)	2	3.5×10^{1}	2.2×10^{1}	1.9×10^{1}	5.0×10^{1}
Fe	Frog (carcass)	2	1.0×10^{3}	1.0×10^{3}	3.0×10^{2}	1.7×10^{3}
K	Tadpole (whole)	3	4.7×10^{2}	1.5×10^{0}	3.1×10^{2}	7.0×10^{2}
K	Frog (muscle)	2	1.5×10^{3}	7.1×10^{1}	1.4×10^{3}	1.5×10^{3}
K	Frog (carcass)	2	1.6×10^{3}	7.1×10^{1}	1.5×10^{3}	1.6×10^{3}
K	Reptile (carcass)	9	1.4×10^{3}	7.0×10^{0}	8.8×10^{2}	2.0×10^{3}
Mg	Tadpole (whole)	3	2.7×10^{1}	1.2×10^{0}	2.3×10^{1}	3.4×10^{1}
Mg	Frog (muscle)	2	1.5×10^{1}	8.9×10^{0}	9.4×10^{0}	2.1×10^{1}
Mg	Frog (carcass)	2	2.4×10^{1}	7.0×10^{-1}	2.3×10^{1}	2.4×10^{1}
Mg	Reptile (carcass)	9	3.4×10^{1}	5.8×10^{0}	1.5×10^{1}	7.2×10^{1}
Mn	Tadpole (whole)	3	5.6×10^{2}	2.8×10^{0}	1.7×10^{2}	1.1×10^{3}
Mn	Frog (muscle)	2	2.0×10^{0}	1.5×10^{0}	9.3×10^{-1}	3.1×10^{0}
Mn	Frog (carcass)	2	3.0×10^{2}	5.0×10^{1}	2.6×10^{2}	3.3×10^{2}
Na	Tadpole (whole)	3	1.1×10^{2}	2.5×10^{0}	4.0×10^{1}	2.5×10^{2}
Na	Frog (muscle)	2	1.4×10^{2}	6.0×10^{1}	9.8×10^{1}	1.8×10^{2}
Na	Frog (carcass)	2	7.7×10^{1}	2.0×10^{0}	7.5×10^{1}	7.8×10^{1}
Na	Reptile (carcass)	9	7.3×10^{1}	1.1×10^{0}	5.6×10^{1}	1.3×10^{2}
Ni	Tadpole (whole)	3	3.8×10^{2}	1.9×10^{0}	1.9×10^{2}	7.2×10^{2}
Ni	Frog (muscle)	2	2.4×10^{1}	0.4×10^{0}	2.1×10^{1}	2.7×10^{1}
Ni	Frog (carcass)	2	2.0×10^{4}	2.8×10^{4}	6.6×10^{2}	1.0×10^{4}
Pb	Tadpole (whole)	3	6.4×10^{1}	1.2×10^{0}	5.3×10^{1}	7.6×10^{1}
Pb	Frog (muscle)	2	5.5×10^{0}	4.8×10^{0}	2.1×10^{0}	8.9×10^{0}
Pb	Frog (carcass)	2	1.7×10^{1}	6.0×10^{0}	1.2×10^{1}	2.1×10^{1}
Zn	Tadpole (whole)	3	5.7×10^{2}	2.7×10^{0}	2.7×10^{2}	1.8×10^{3}
Zn	Frog (muscle)	2	9.0×10^{2}	9.0×10^{2}	2.0×10^{2}	1.5×10^{3}
Zn	Frog (carcass)	2	1.0×10^{4}	1.1×10^{3}	9.5×10^{3}	1.1×10^{4}

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[1] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is

[2] nd means – no data.

TABLE 7. SUMMARY OF SEDIMENT-TO-BIOTA BIOACCUMULATION FACTORS (*BSAF*) FOR WHOLE FRESHWATER INVERTEBRATES [30, 31]

Element	N	GM[1]	GSD	Min	Max
Ag	40	7.3×10^{-1}	2.6×10^{0}	4.0×10^{-2}	7.1×10^{0}
Al	136	3.8×10^{-4}	3.8×10^{0}	8.5×10^{-6}	1.7×10^{-1}
As	138	1.5×10^{-1}	2.2×10^{0}	4.3×10^{-3}	5.7×10^{-1}
B	1	1.8×10^{-2}	-	-	-
Ba	137	1.6×10^{-2}	3.6×10^{0}	1.7×10^{-3}	1.7×10^{0}
Be	1	4.0×10^{-2}	-	-	-
Cd	115	7.8×10^{-1}	4.4×10^{0}	1.9×10^{-4}	8.7×10^{0}
Co	136	3.1×10^{-2}	2.0×10^{0}	3.7×10^{-3}	2.0×10^{-1}
Cu	149	4.7×10^{-1}	3.3×10^{0}	8.6×10^{-4}	3.3×10^{1}
Fe	140	3.4×10^{-3}	2.5×10^{0}	2.2×10^{-4}	3.5×10^{-1}
Hg	109	8.4×10^{-1}	3.5×10^{0}	5.6×10^{-2}	1.1×10^{1}
Mo	15	7.4×10^{-2}	1.7×10^{0}	3.6×10^{-2}	1.9×10^{-1}
Ni	131	2.1×10^{-2}	3.9×10^{0}	2.1×10^{-3}	1.6×10^{3}
Pb	75	8.6×10^{-3}	4.6×10^{0}	4.8×10^{-4}	4.5×10^{0}
Sb	1	1.5×10^{-1}	-	-	-
Se	103	1.3	2.2×10^{0}	6.5×10^{-2}	6.0×10^{0}
Sr	135	4.4×10^{-2}	3.0×10^{0}	1.9×10^{-3}	6.2×10^{-1}
Tl	1	2.3×10^{3}	-	-	-
U	6	1.7×10^{-2}	2.9×10^{0}	2.8×10^{-3}	6.4×10^{-2}
V	66	2.4×10^{-3}	2.1×10^{0}	5.3×10^{-4}	1.4×10^{-2}
Zn	151	5.2×10^{-1}	3.0×10^{0}	9.3×10^{-4}	2.3×10^{1}

[1] The individual value is given if the number of observations (N) is 1.

TABLE 8. SUMMARY OF SEDIMENT-TO-BIOTA BIOACCUMULATION FACTORS (*BSAF*) FOR EDIBLE TISSUES OF FRESHWATER FISH [30, 31]

	Whole fish					Fish muscle					Fish liver					Fish
Element	N	GM/AM[1]	GSD/SD[1]	Min	Max	N	GM/AM	GSD/SD	Min	Max	N	GM/AM	GSD/SD	Min	Max	eggs
Ag	8	6.8×10^{-1}	3.5×10^{0}	1.2×10^{-1}	4.9×10^{0}	nd	nd	nd	nd	nd	43	3.5×10^{-1}	3.2×10^{0}	2.4×10^{-2}	1.5×10^{1}	nd
Al	113	8.0×10^{-3}	2.0×10^{1}	3.6×10^{-6}	4.3×10^{1}	15	6.0×10^{-3}	4.1×10^{0}	1.2×10^{-3}	8.3×10^{-2}	41	1.3×10^{-5}	2.6×10^{0}	2.2×10^{-6}	1.0×10^{-4}	nd
As	226	1.4×10^{-1}	5.9×10^{0}	7.8×10^{-4}	6.6×10^{0}	13	2.7×10^{-1}	8.5×10^{0}	7.1×10^{-3}	13	56	1.8×10^{-2}	4.6×10^{0}	2.7×10^{-3}	4.4×10^{0}	nd
Ba	103	4.8×10^{-2}	9.3×10^{0}	4.7×10^{-5}	1.9×10^{0}	3	1.2×10^{-1}	2.3×10^{0}	4.5×10^{-2}	2.0×10^{-1}	39	6.3×10^{-5}	2.4×10^{0}	1.9×10^{-5}	1.9×10^{-3}	nd
Be	2	1.6×10^{-1}	4.0×10^{-2}	1.3×10^{-1}	1.8×10^{-1}	nd	nd	nd	nd	nd	1	1.4×10^{-1}	nd	nd	nd	nd
Ca	2	2.6×10^{2}	2.4×10^{2}	8.7×10^{1}	4.3×10^{2}	2	1.8×10^{1}	2.2×10^{1}	1.9	3.4×10^{1}	nd	nd	nd	nd	nd	nd
Cd	134	1.3×10^{-1}	5.9×10^{0}	2.1×10^{-6}	3.2×10^{3}	20	6.1×10^{-1}	3.7×10^{0}	2.2×10^{-2}	3.3×10^{0}	72	4.7×10^{-1}	4.1×10^{0}	1.7×10^{-2}	1.4×10^{1}	nd
Co	751	2.8×10^{-1}	4.3×10^{0}	7.1×10^{-4}	2.9×10^{1}	206	2.0×10^{-1}	4.4×10^{0}	3.0×10^{-4}	5.9×10^{0}	133	6.1×10^{-2}	1.1×10^{1}	1.3×10^{-3}	9.7×10^{1}	nd
Fe	71	4.3×10^{-3}	9.7×10^{0}	9.3×10^{-7}	2.3×10^{-1}	3	3.8×10^{-3}	5.2×10^{0}	7.5×10^{-4}	2.0×10^{-2}	78	2.0×10^{-3}	3.4×10^{0}	3.3×10^{-4}	3.2×10^{0}	nd
Hg	353	5.3×10^{0}	5.9×10^{0}	3.1×10^{-4}	1.3×10^{2}	128	7.3×10^{0}	6.5×10^{0}	2.3×10^{-2}	1.9×10^{2}	64	7.0×10^{-1}	4.4×10^{0}	3.0×10^{-2}	6.4×10^{1}	0.17[2]
Mg	7	3.7×10^{0}	2.9×10^{0}	3.6×10^{-1}	8.0×10^{0}	2	7.6×10^{-1}	1.1×10^{0}	7.3×10^{-1}	7.9×10^{-1}	nd	nd	nd	nd	nd	nd
Mo	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	6	2.0×10^{-2}	2.3×10^{0}	7.1×10^{-3}	6.0×10^{-2}	nd
Na	2	4.6×10^{-2}	5.7×10^{-2}	6.0×10^{-3}	8.6×10^{-2}	1	1.0×10^{1}	nd	nd	nd	nd	nd	nd	nd	nd	nd
Ni	138	2.1×10^{-1}	5.4×10^{0}	2.9×10^{-3}	2.9	14	4.6×10^{-1}	4.0×10^{0}	4.1×10^{-2}	2.8×10^{0}	45	2.9×10^{-3}	2.7×10^{0}	2.6×10^{-4}	7.3×10^{-2}	nd
Pb	365	2.8×10^{-1}	5.2×10^{0}	8.4×10^{-6}	6.3×10^{1}	20	1.1×10^{-1}	8.5×10^{0}	8.2×10^{-4}	1.2×10^{0}	21	2.2×10^{-3}	7.4×10^{0}	1.8×10^{-4}	4.4×10^{-1}	nd
Sb	9	6.6×10^{-1}	5.3×10^{0}	4.8×10^{-2}	8.8	nd	nd	nd	nd	nd	2	2.4×10^{-1}	8.0×10^{-2}	1.8×10^{-1}	2.9×10^{-1}	nd
Se	61	1.6×10^{0}	3.4×10^{0}	8.4×10^{-2}	4.0×10^{1}	16	4.8×10^{0}	2.9×10^{0}	2.6×10^{-1}	1.6×10^{1}	73	1.0×10^{0}	2.3×10^{0}	2.1×10^{-1}	8.6×10^{0}	nd
Sn	2	7.8×10^{-1}	3.4×10^{-2}	7.5×10^{-1}	8.0×10^{-1}	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Sr	35	1.4×10^{-2}	1.5×10^{1}	1.8×10^{-4}	9.5×10^{-1}	nd	nd	nd	nd	nd	71	4.7×10^{-4}	2.2×10^{0}	8.8×10^{-5}	2.6×10^{-3}	nd
Ti	1	1.6×10^{-2}				nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Tl	13	2.9×10^{2}	5.6×10^{0}	6.8×10^{0}	2.9×10^{3}	1	7.5×10^{1}	nd	nd	nd	nd	nd	nd	nd	nd	nd
V	28	2.1×10^{-3}	2.4×10^{0}	4.2×10^{-4}	1.3×10^{-2}	nd	nd	nd	nd	nd	51	1.3×10^{-3}	2.1×10^{0}	2.1×10^{-4}	5.4×10^{-3}	nd
Zn	703	2.1×10^{0}	4.0×10^{0}	8.1×10^{-3}	1.3×10^{2}	178	1.1×10^{0}	4.1×10^{0}	1.1×10^{-3}	4.4×10^{1}	80	2.2×10^{-1}	4.9×10^{0}	5.3×10^{-3}	1.2×10^{2}	nd

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[1] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1; [2] N=1.

TABLE 9. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER ALGAE AND PLANTS[1]

Element	Group of Plants	N	GM/AM[2]	GSD/SD[2]	Min	Max	# ref
Am	Algae	7	9.4×10^{3}	3.0×10^{0}	1.5×10^{3}	3.8×10^{4}	5
	Macrophytes (generic)[1]	1	6.5×10^{3}				1
	Floating-leafed Macrophytes	1	1.5×10^{4}				1
	Emergent Macrophytes	7	1.1×10^{3}	8.9×10^{0}	7.5×10^{1}	2.0×10^{4}	5
C	Algae	3	4.5×10^{4}	1.4×10^{0}	3.1×10^{4}	5.6×10^{4}	1
	Floating-leafed Macrophytes	1	5.4×10^{3}				1
	Emergent Macrophytes	6	1.1×10^{4}	1.7×10^{1}	4.4×10^{1}	8.9×10^{4}	2
Cd	Algae	4	1.8×10^{4}	1.4×10^{0}	1.1×10^{4}	2.3×10^{4}	2
	Floating-leafed Macrophytes	1	2.3×10^{4}	nd	nd	nd	1
Cm	Algae	1	8.0×10^{3}	nd	nd	nd	1
Co	Algae	8	1.5×10^{3}	2.6×10^{0}	6.5×10^{2}	9.0×10^{3}	6
	Macrophytes (generic)	3	1.1×10^{3}	1.2×10^{1}	2.6×10^{2}	2.0×10^{4}	3
	Floating-leafed Macrophytes	4	3.5×10^{2}	2.1×10^{0}	2.3×10^{2}	1.0×10^{3}	2
	Emergent Macrophytes	3	1.6×10^{2}	3.8×10^{0}	5.0×10^{1}	6.8×10^{2}	3
Cs	Algae	5	9.6×10^{2}	2.4×10^{0}	4.3×10^{2}	3.3×10^{3}	5
	Macrophytes (generic)	3	3.9×10^{2}	4.0×10^{0}	1.7×10^{2}	2.0×10^{3}	3
	Floating-leafed Macrophytes	2	3.6×10^{2}	3.0×10^{2}	1.5×10^{2}	5.7×10^{2}	2
	Emergent Macrophytes	16	2.7×10^{1}	8.8×10^{0}	1.8×10^{0}	1.4×10^{3}	4
Cu	Algae	4	3.0×10^{3}	1.2×10^{0}	2.4×10^{3}	3.1×10^{3}	4
	Floating-leafed Macrophytes	1	2.7×10^{3}				1
Fe	Algae	4	8.4×10^{3}	1.6×10^{0}	5.2×10^{3}	9.2×10^{3}	4
	Floating-leafed Macrophytes	1	1.2×10^{4}				1
I	[a] Macrophytes (generic)	3	1.3×10^{2}	2.0×10^{0}	7.8×10^{1}	2.7×10^{2}	2
Mn	Algae	4	9.2×10^{4}	1.5×10^{0}	6.1×10^{4}	9.8×10^{4}	4
	Macrophytes (generic)	1	3.1×10^{-1}				1
	Floating-leafed Macrophytes	1	1.1×10^{5}				1

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[1] Although four groups of aquatic plants are considered: algae, macrophytes (generic), floating-leafed macrophytes, emergent macrophytes, only those species are mentioned for individual radionuclides, which information was available for. Generic' macrophyte data are typically based on a compilation of very few data on accumulation factors for which little information is available. These values should not be used preferably for critical parameters.

[2] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1

TABLE 9. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER ALGAE AND PLANTS[1] (Cont.)

Element	Group of Plants	N	GM/AM	GSD/SD[2]	Min.	Max.	# ref
Ni	Algae	4	7.0×10^2	2.0×10^0	2.5×10^2	8.2×10^2	2
	Floating-leafed Macrophytes	1	1.1×10^3	nd	nd	1.1×10^3	1
Np	Algae	1	8.0×10^3	nd	nd	8.0×10^3	1
	Macrophytes (generic)	1	6.5×10^3	nd	nd	6.5×10^3	1
Pb	Algae	4	1.9×10^3	1.3×10^0	1.3×10^3	2.0×10^3	2
	Floating-leafed Macrophytes	1	1.8×10^3	nd	nd	1.8×10^3	1
Pu	Algae	16	4.1×10^4	4.4×10^1	8.0×10^2	4.9×10^7	4
	Macrophytes (generic)	8	6.4×10^2	2.4×10^0	1.2×10^2	2.1×10^3	4
	Floating-leafed Macrophytes	2	9.9×10^6	5.8×10^6	5.8×10^6	1.4×10^7	2
	Emergent Macrophytes	14	5.8×10^4	1.4×10^1	4.6×10^2	3.3×10^6	3
Ra	Algae	4	3.9×10^3	3.7×10^0	6.4×10^2	1.1×10^4	2
	Macrophytes (generic)	4	2.0×10^3	2.7×10^0	9.8×10^2	9.0×10^3	2
Ru	Algae	7	3.6×10^2	1.6×10^0	1.8×10^2	6.7×10^2	3
	Macrophytes (generic)	1	7.4×10^1	nd	nd	nd	1
Se	Algae	5	2.9×10^2	6.1×10^0	2.3×10^1	2.7×10^3	4
	Macrophytes (generic)	3	5.7×10^1	6.8×10^0	9.4	4.3×10^2	2
	Floating-leafed Macrophytes	11	2.6×10^3	1.6×10^0	1.2×10^3	5.9×10^3	3
	Emergent Macrophytes	12	3.1×10^3	1.7×10^0	1.3×10^3	8.2×10^3	2
Sr	Algae	3	6.6×10^2	2.5×10^0	3.8×10^2	1.9×10^3	3
	Macrophytes (generic)	2	1.7×10^2	1.8×10^2	3.9×10^0	2.9×10^1	2
	Floating-leafed Macrophytes	3	5.1×10^2	2.6×10^0	4.6×10	1.7×10^3	2
	Emergent Macrophytes	9	1.1×10^2	4.1×10^0	7.5×10	1.8×10^3	3
Tc	Algae	6	1.1×10^1	4.9×10^0	2.1×10^0	8.8×10^1	3
U	Algae	1	1.6×10^2	Nd	nd	nd	1
	[a] Macrophytes (generic)	3	2.3×10^2	2.6×10^0	8.1×10^1	5.2×10^2	2
Zn	Algae	4	2.0×10^4	1.3×10^0	1.4×10^4	2.7×10^4	3
	Floating-leafed Macrophytes	1	2.4×10^4	Nd	nd	nd	1

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[1] Although four groups of aquatic plants are considered: algae, macrophytes (generic), floating-leafed macrophytes, emergent macrophytes, only those species are mentioned for individual radionuclides, which information was available for. Generic' macrophyte data are typically based on a compilation of very few data on accumulation factors for which little information is available. These values should not be used preferably for critical parameters.

[2] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1.

TABLE 10. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER INVERTEBRATES AND THEIR TISSUES

Element	[a]Type of Invertebrate (and tissue)	N	GM/AM	GSD/SD	Min.	Max.	# ref
Ag	Freshwater Invertebrates	1	3.3×10^{2}				1
Al	Freshwater Invertebrates	1	3.1×10^{3}				1
Am	Freshwater Invertebrates	9	1.0×10^{3}	6.6×10^{0}	5.9×10^{1}	1.6×10^{4}	3
As	Freshwater Invertebrates	1	2.0×10^{3}				1
Au	Freshwater Invertebrates	1	1.5×10^{3}				1
Ba	Freshwater Invertebrates	1	1.6×10^{2}				1
Br	Freshwater Invertebrates	1	7.2×10^{2}				1
C	Freshwater Invertebrates	15	6.4×10^{4}	3.0×10^{0}	1.3×10^{4}	5.7×10^{5}	2
C	Molluscs (soft tissue)	1	1.5×10^{4}				1
Ca	Freshwater Invertebrates	1	6.6×10^{1}				1
Cd	Freshwater Invertebrates	121	1.6×10^{2}	4.0×10^{1}	1.4×10^{2}	1.6×10^{4}	7
Ce	Freshwater Invertebrates	1	5.6×10^{2}				1
Cl	Freshwater Invertebrates	1	1.8×10^{2}				1
Cm	Freshwater Invertebrates	1	1.0×10^{4}				1
Co	Freshwater Invertebrates	20	9.7×10^{0}	1.7×10^{2}	1.8×10^{3}	4.1×10^{4}	10
Co	Molluscs (soft tissue)	2	6.9×10^{2}	4.2×10^{1}	6.6×10^{2}	7.2×10^{2}	2
Cr	Freshwater Invertebrates	1	3.8×10^{2}				1
Cs	Freshwater Invertebrates	20	1.5×10^{1}	1.2×10^{2}	5.4×10^{3}	6.1×10^{3}	8
Cs	Molluscs (soft tissue)	1	3.0×10^{2}				1
Cu	Freshwater Invertebrates	81	4.1×10^{1}	1.1×10^{1}	5.6×10^{-1}	1.4×10^{3}	3
Eu	Freshwater Invertebrates	1	2.0×10^{2}				1
Fe	Freshwater Invertebrates	1	1.8×10^{3}				1
Hf	Freshwater Invertebrates	1	1.5×10^{3}				1
Hg	Freshwater Invertebrates	25	7.1×10^{2}	2.8×10^{0}	2.0×10^{2}	5.2×10^{3}	5
I	Freshwater Invertebrates	5	2.9×10^{2}	5.1×10^{0}	1.8×10^{1}	1.0×10^{3}	5
I	Crustaceans (muscle)	16	4.1×10^{1}	1.1×10^{1}	5.6×10^{-1}	1.4×10^{3}	3
I	Crustaceans (soft tissue)	19	4.5×10^{0}	3.4×10^{0}	6.1×10^{-1}	1.1×10^{2}	3
I	Molluscs (muscle)	24	1.6×10^{0}	6.8×10^{0}	3.6×10^{2}	1.8×10^{1}	2
I	Molluscs (soft tissue)	20	1.4×10^{1}	4.2×10^{0}	7.4×10^{1}	9.5×10^{1}	2

The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1.

TABLE 10. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER INVERTEBRATES AND THEIR TISSUES (Cont.)

Element	Type of Invertebrate[1] (and tissue)	N	GM	GSD	Min.	Max.	# ref
K	Freshwater Invertebrates	1	5.4×10^{2}				1
La	Freshwater Invertebrates	1	3.7×10^{2}				1
Lu	Freshwater Invertebrates	1	1.1×10^{3}				1
Mg	Freshwater Invertebrates	1	4.4×10^{1}				1
Mn	Freshwater Invertebrates	1	3.7×10^{3}				1
Mo	Freshwater Invertebrates	1	2.0×10^{3}				1
Mo	Crustaceans (muscle)	8	1.3×10^{1}	6.5×10^{0}	2.9×10^{2}	1.0×10^{1}	2
Mo	Crustaceans (soft tissue)	8	4.2×10^{1}	3.5×10^{0}	5.0×10^{2}	2.5×10^{0}	2
Mo	Molluscs (muscle)	8	1.8×10^{1}	2.3×10^{0}	3.6×10^{-2}	4.9×10^{-1}	2
Mo	Molluscs (soft tissue)	7	5.0×10^{1}	3.3×10^{0}	4.5×10^{-2}	1.60	2
Na	Freshwater Invertebrates	1	1.1×10^{2}				1
Np	Freshwater Invertebrates	1	1.0×10^{4}				1
Pb	Freshwater Invertebrates	74	1.9×10^{1}	2.1×10^{1}	4.5×10^{-2}	7.0×10^{2}	4
Pu	Freshwater Invertebrates	53	5.3×10^{3}	3.8×10^{1}	3.6×10^{1}	5.5×10^{6}	11
Ra	Freshwater Invertebrates	5	1.0×10^{2}	3.0×10^{1}	1.8	1.8×10^{3}	2
Rb	Freshwater Invertebrates	1	2.2×10^{3}				1
Ru	Freshwater Invertebrates	8	3.9×10^{2}	2.6×10^{1}	1.8×10^{-3}	8.3×10^{1}	3
Sb	Freshwater Invertebrates	1	7.4×10^{1}				1
Sc	Freshwater Invertebrates	1	3.7×10^{3}				1
Se	Freshwater Invertebrates	4	7.8×10^{2}	1.0×10^{1}	6.3×10^{1}	5.8×10^{3}	2
Sm	Freshwater Invertebrates	1	2.7×10^{3}				1
Sr	Freshwater Invertebrates	4	3.3×10^{2}	3.4×10^{0}	7.7×10^{1}	1.3×10^{3}	3
Tc	Freshwater Invertebrates	3	3.0×10^{1}	8.6×10^{0}	5.4×10^{0}	3.3×10^{2}	2
Tc	Molluscs (soft tissue)	2	1.9×10^{0}	1.4×10^{-1}	1.8×10^{0}	2.0×10^{0}	2
Th	Freshwater Invertebrates	1	2.9×10^{3}				1
U	Freshwater Invertebrates	8	1.7×10^{2}	1.9×10^{1}	3.6	6.0×10^{4}	2
V	Freshwater Invertebrates	1	3.3×10^{2}				1
Zn	Freshwater Invertebrates	81	8.9×10^{1}	2.9×10^{1}	6.3×10^{-2}	1.5×10^{3}	5

[1] A review of available data for whole freshwater invertebrates, crustacean muscle, crustacean soft tissue (including organs), mollusc muscle and mollusc soft tissue was conducted, but only records for which information was available is provided in the above table.

[2] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1.

TABLE 11. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH[1]

Element		Fish Muscle (L/kg FW)					Whole Fish (L/kg FW)					# ref
	Fish Type	N	GM/AM[2]	GSD/SD[2]	Min.	Max.	N	GM/AM	GSD/SD	Min.	Max.	(muscle, whole)
Ag	Benthic Fish	16	9.4×10^{1}	1.4×10^{0}	5.3×10^{1}	$1.7 \times 10^{2}0$	1	1.8×10^{2}				2, 1
	Forage Fish	11	1.2×10^{2}	1.6×10^{0}	4.0×10^{1}	2.1×10^{2}	37	1.1×10^{2}	1.3×10^{0}	5.7×10^{1}	1.7×10^{2}	2, 2
	Piscivorous Fish	4	1.2×10^{2}	1.2×10^{0}	9.3×10^{1}	1.5×10^{2}	nd	nd	nd	nd	nd	2, -
Al	Benthic Fish	20	3.4×10^{1}	4.8×10^{0}	5.9×10^{0}	1.9×10^{2}	4	1.4×10^{2}	1.2×10^{1}	1.7×10^{1}	5.2×10^{3}	3, 2
	Forage Fish	16	1.5×10^{2}	1.6×10^{0}	6.7×10^{1}	3.0×10^{2}	38	5.1×10^{2}	2.3	1.5×10^{2}	2.6×10^{3}	2, 3
	Piscivorous Fish	13	2.8×10^{1}	3.2×10^{0}	6.4×10^{0}	1.4×10^{2}	53	1.6×10^{1}	2.7	4.5	1.9×10^{2}	3, 3
Am	Forage Fish	2	2.3×10^{2}	2.3×10^{2}	7.2×10^{1}	4.0×10^{2}	nd	nd	nd	nd	nd	2, -
As	Benthic Fish	4	4.5×10^{2}	1.4×10^{0}	3.1×10^{2}	6.2×10^{2}	1	9.2×10^{2}	nd	nd	nd	2, 1
	Forage Fish	11	2.8×10^{2}	2.6×10^{0}	5.0×10^{1}	8.5×10^{2}	38	3.7×10^{2}	2.3×10^{0}	8.1×10^{1}	1.0×10^{3}	2, 3
	Piscivorous Fish	3	3.4×10^{2}	1.9×10^{0}	1.8×10^{2}	6.4×10^{2}	nd	nd	nd	nd	nd	1, -
Au	Benthic Fish	4	2.3×10^{2}	2.0×10^{0}	1.0×10^{2}	5.0×10^{2}	1	3.5×10^{2}	nd	nd	nd	2, 1
	Forage Fish	11	3.3×10^{2}	1.9×10^{0}	1.0×10^{2}	9.0×10^{2}	30	2.9×10^{2}	2.4×10^{0}	5.0×10^{1}	1.0×10^{3}	2, 3
	Piscivorous Fish	4	1.2×10^{2}	2.0×10^{0}	5.0×10^{1}	2.5×10^{2}	nd	nd	nd	nd	nd	2, -
Ba	Benthic Fish	27	1.5	2.8×10^{0}	2.5×10^{-1}	3.2×10^{1}	4	1.4×10^{1}	2.5×10^{0}	5.0×10^{0}	4.1×10^{1}	3, 3
	Forage Fish	9	4.8	3.8×10^{0}	1.1×10^{0}	2.7×10^{1}	14	5.7×10^{1}	2.3×10^{0}	1.4×10^{1}	2.2×10^{2}	3, 3
	Piscivorous Fish	79	9.7×10^{-1}	3.0×10^{0}	5.3×10^{-2}	1.4×10^{1}	78	4.9×10^{1}	1.5×10^{0}	1.3×10^{1}	1.5×10^{2}	4, 4

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[1] Although four groups of fish are considered: Fish (Generic), Benthic Fish, Forage Fish and Piscivorous Fish, only those species are mentioned for individual radionuclides, which information was available for

[2] The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1.

TABLE 11.11. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH (Cont.)

Element		Fish Muscle (L/kg FW)					Whole Fish (L/kg FW)					# ref
	Tissue Type	N	GM/AM	GSD/SD	Min.	Max.	N	GM/AM	GSD/SD	Min.	Max.	(muscle, whole)
Br	Benthic Fish	1	1.4×10^2				1	9.8×10^1				1, 1
	Forage Fish	11	8.1×10^1	2.8×10^0	1.8×10^1	3.7×10^2	38	1.6×10^2	2.3×10^0	1.5×10^1	7.9×10^2	2, 3
	Piscivorous Fish	4	1.1×10^2	1.4×10^0	7.4×10^1	1.5×10^2	nd	nd	nd	nd	nd	2, -
Ca	Benthic Fish	26	7.0×10^0	1.5×10^0	2.6×10^0	1.6×10^1	4	1.6×10^2	1.8×10^0	9.3×101	3.3×10^2	3, 3
	Forage Fish	16	1.3×101	1.9×10^0	4.1×10^0	$3.1x\ 10^1$	37	2.1×10^2	1.4×10^0	8.4×101	4.5×10^2	3, 2
	Piscivorous Fish	62	1.4×101	2.9×10^0	2.0×10^0	9.7×10^1	78	2.4×10^3	1.5×10^0	1.5×102	5.6×10^3	6, 5
Ce	Benthic Fish	14	8.6×10^1	1.2×10^1	1.7×10^0	1.2×10^3	4	4.8×10^1	2.2×10^0	1.6×10^1	9.3×10^1	3, 3
	Forage Fish	1	3.0×10^1	nd	nd	nd	9	3.2×10^1	2.6×10^0	8.6×10^0	1.1×10^2	1, 3
	Piscivorous Fish	56	1.8×10^1	7.2×10^0	9.0×10^{-1}	6.7×10^2	77	1.0×10^1	2.4×10^0	3.0×10^0	1.1×10^2	3, 5
Cl	Benthic Fish	1	4.4×10^1				1	1.2×10^2				1, 1
	Forage Fish	11	4.4×10^1	2.6×10^0	8.9	1.2×10^2	38	8.5×10^1	1.6×10^0	2.5×10^1	2.3×10^2	2, 2
	Piscivorous Fish	4	5.9×10^1	1.1	5.4×10^1	6.4×10^1	nd	nd	nd	nd	nd	2, -
Co	Benthic Fish	16	8.1×10^1	2.6×10^0	1.9×10^1	2.7×10^2	6	1.6×10^2	2.4×10^0	6.3×10^1	7.2×10^2	14, 6
	Forage Fish	11	3.0×10^2	1.5×10^0	1.9×10^2	5.6×10^2	32	5.3×10^2	1.9×10^0	2.4×10^2	2.4×10^3	11, 22
	Piscivorous Fish	40	5.5×10^1	2.0×10^0	1.7×10^1	2.7×10^2	80	3.9×10^2	1.5×10^0	2.3×10^1	7.5×10^2	23, 20
Cr	Benthic Fish	9	2.9×10^1	1.4×10^0	2.0×10^1	4.7×10^1	3	5.4×10^1	1.1×10^0	5.2×10^1	5.8×10^1	3, 3
	Forage Fish	1	3.9×10^1				5	5.9×10^1	1.5×10^0	3.5×10^1	8.8×10^1	1, 3
	Piscivorous Fish	47	4.2×10^1	2.1×10^0	1.3×10^1	1.2×10^2	43	2.6×10^2	1.6×10^0	5.9×10^1	7.6×10^2	5, 5
Cs	Benthic Fish	17	1.4×10^3	2.2×10^0	1.6×10^2	3.1×10^3	11	2.0×10^3	1.9×10^0	4.8×10^2	4.4×10^3	17, 11
	Forage Fish	12	1.4×10^3	2.2×10^0	1.4×10^2	2.2×10^3	43	9.8×10^2	2.6×10^0	7.5×10^1	5.2×10^3	12, 36
	Piscivorous Fish	78	3.1×10^3	2.5×10^0	2.6×10^2	1.5×10^4	91	5.3×10^3	2.2×10^0	2.4×10^2	2.4×10^4	34, 49

TABLE 11. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH (Cont.)

Element	Tissue Type	Fish Muscle (L/kg FW)					Whole Fish (L/kg FW)					# ref (muscle, whole)
		N	GM/AM	GSD/SD	Min.	Max.	N	GM/AM	GSD/SD	Min.	Max.	
Cu	Benthic Fish	15	2.6×10^{2}	1.5×10^{0}	1.3×10^{2}	4.1×10^{2}	4	2.9×10^{2}	2.7×10^{0}	1.4×10^{2}	1.2×10^{3}	3, 3
	Forage Fish	10	3.7×10^{2}	1.5×10^{0}	2.0×10^{2}	7.2×10^{2}	31	3.5×10^{2}	1.7×10^{0}	8.6×10^{1}	8.8×10^{2}	3, 3
	Piscivorous Fish	61	2.0×10^{2}	1.7×10^{0}	8.9×10^{1}	6.5×10^{2}	77	2.5×10^{2}	1.3×10^{0}	1.5×10^{2}	7.2×10^{2}	3, 4
[a] Dy	Forage Fish	2	6.5×10^{2}	6.4×10^{0}	2.0×10^{2}	1.1×10^{3}	2	3.0×10^{2}		3.0×10^{2}	3.0×10^{2}	2, 2
Eu	Benthic Fish	7	6.7×10^{1}	6.3×10^{0}	1.8×10^{1}	2.4×10^{2}	1	5.9×10^{2}	nd	nd	nd	3, 1
	Forage Fish	11	3.8×10^{2}	1.6×10^{0}	1.1×10^{2}	6.5×10^{2}	34	3.3×10^{2}	3.1×10^{0}	1.2×10^{1}	2.2×10^{3}	3, 3
	Piscivorous Fish	11	5.3×10^{1}	5.6×10^{0}	1.1×10^{1}	7.2×10^{2}	30	7.4×10^{1}	1.8×10^{0}	7.6×10^{0}	1.6×10^{2}	3, 3
Fe	Benthic Fish	15	3.6×10^{2}	6.0×10^{0}	2.6×10^{1}	2.0×10^{3}	4	3.1×10^{2}	6.6×10^{0}	9.3×10^{1}	5.2×10^{3}	3, 3
	Forage Fish	10	5.1×10^{2}	1.8×10^{0}	2.0×10^{2}	9.5×10^{2}	38	1.8×10^{3}	1.9×10^{0}	4.8×10^{2}	5.3×10^{3}	3, 3
	Piscivorous Fish	61	1.2×10^{2}	7.5×10^{0}	6.6	1.8×10^{3}	78	4.7×10^{1}	1.6×10^{0}	1.6×10^{1}	2.4×10^{2}	4, 4
Hf	Benthic Fish	1	1.9×10^{3}				1	1.2×10^{4}	nd	nd	nd	1, 1
	Forage Fish	10	8.6×10^{2}	1.9×10^{0}	3.3×10^{2}	1.8×10^{3}	37	2.0×10^{3}	3.1×10^{0}	3.0×10^{2}	2.8×10^{4}	3, 10
	Piscivorous Fish	4	1.5×10^{3}	1.3×10^{0}	1.1×10^{3}	1.9×10^{3}	nd	nd	nd	nd	nd	2, -
Hg	Benthic Fish	1	1.9×10^{3}				1	2.2×10^{4}	nd	nd	nd	1, 1
	Forage Fish	11	7.4×10^{3}	1.8×10^{0}	3.7×10^{3}	1.7×10^{4}	36	4.1×10^{3}	2.1×10^{0}	1.1×10^{3}	1.9×10^{4}	3, 3
	Piscivorous Fish	4	5.3×10^{3}	1.4×10^{0}	3.3×10^{3}	7.0×10^{3}	nd	nd	nd	nd	nd	2, -
I	Benthic Fish	10	5.5×10^{1}	1.9×10^{0}	3.3×10^{1}	1.6×10^{2}	1	6.6×10^{2}	nd	nd	nd	4, 1
	Forage Fish	3	2.9×10^{2}	1.4×10^{0}	2.0×10^{2}	4.0×10^{2}	35	5.9×10^{2}	3.1×10^{0}	1.0×10^{2}	4.5×10^{4}	2, 7
	Piscivorous Fish	42	2.4×10^{1}	9.1×10^{0}	1.1×10^{1}	3.2×10^{2}	59	6.8×10^{2}	1.6×10^{0}	1.5×10^{2}	1.3×10^{3}	8, 8
K	Benthic Fish	15	2.8×10^{3}	1.1×10^{0}	2.4×10^{3}	3.1×10^{3}	4	1.8×10^{3}	1.3×10^{0}	1.4×10^{3}	2.3×10^{3}	4, 4
	Forage Fish	11	2.4×10^{3}	1.5×10^{0}	1.2×10^{3}	3.4×10^{3}	38	1.8×10^{3}	1.6×10^{0}	5.7×10^{2}	3.9×10^{3}	4, 4
	Piscivorous Fish	61	3.4×10^{3}	1.7×10^{0}	1.4 × 103	$9. \times 10^{3}$	78	6.3×10^{3}	1.3×10^{0}	2.0×10^{3}	1.5×10^{4}	6, 6

TABLE 11. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH (Cont.)

Element		Fish Muscle (L/kg FW)					Whole Fish (L/kg FW)					# ref (muscle, whole)
	Tissue Type	N	GM	GSD	Min.	Max.	N	GM	GSD	Min.	Max.	
La	Benthic Fish	15	6.2×10^{1}	8.4×10^{0}	2.0×10^{0}	6.6×10^{2}	4	4.5×10^{1}	2.1×10^{0}	1.6×10^{1}	8.3×10^{1}	3, 3
	Forage Fish	14	4.0×10^{1}	1.7×10^{0}	1.5×10^{1}	1.1×10^{2}	30	9.7×10^{1}	2.2×10^{0}	2.5×10^{1}	3.4×10^{2}	3, 3
	Piscivorous Fish	46	3.1×10^{1}	4.8×10^{0}	1.1×10^{0}	2.3×10^{2}	78	9.7×10^{0}	2.0×10^{0}	3.6×10^{0}	7.7×10^{1}	4, 4
Mg	Benthic Fish	15	3.1×10^{1}	1.3×10^{0}	1.4×10^{1}	3.7×10^{1}	4	2.8×10^{1}	1.3×10^{0}	2.2×10^{1}	3.9×10^{1}	3, 3
	Forage Fish	11	1.3×10^{1}	1.2×10^{0}	7.9×10^{0}	1.8×10^{1}	36	2.1×10^{1}	1.2×10^{0}	1.4×10^{1}	3.4 × 101	3, 3
	Piscivorous Fish	61	4.6×10^{1}	2.2×10^{0}	1.4×10^{1}	1.8×10^{2}	78	2.1×10^{2}	1.4×10^{0}	3.5×10^{1}	4.3×10^{2}	4, 4
Mn	Benthic Fish	15	4.0×10^{2}	8.5×10^{0}	1.8×10^{1}	2.6×10^{3}	4	3.7×10^{2}	6.1×10^{0}	1.0×10^{2}	5.3×10^{3}	4, 3
	Forage Fish	11	4.5×10^{2}	2.5×10^{0}	6.0×10^{1}	1.1×10^{3}	38	4.0×10^{3}	1.4×10^{0}	1.9×10^{3}	7.0×10^{3}	3, 3
	Piscivorous Fish	61	1.9×10^{2}	7.0×10^{0}	1.3×10^{1}	1.4×10^{5}	78	2.1×10^{2}	1.6×10^{0}	4.8×10^{1}	6.9×10^{2}	5, 5
Mo	Benthic Fish	14	2.6×10^{0}	1.7×10^{0}	1.8×10^{0}	1.1×10^{1}	3	1.5×10^{1}	6.9×10^{0}	2.1×10^{0}	9.8×10^{1}	3, 3
	Piscivorous Fish	50	1.7×10^{0}	8.3×10^{0}	4.0×10^{-3}	2.0×10^{1}	78	2.8×10^{1}	1.8×10^{0}	3.0×10^{0}	1.9×10^{2}	5, 5
Na	Benthic Fish	15	4.3×10^{1}	1.4×10^{0}	1.9×10^{1}	7.3×10^{1}	3	5.7×10^{1}	1.7×10^{0}	3.9×10^{1}	1.0×10^{2}	5, 3
	Forage Fish	11	1.7×10^{2}	1.6×10^{0}	7.9×10^{1}	4.0×10^{2}	38	1.4×10^{2}	2.0×10^{0}	3.4×10^{1}	4.3×10^{2}	3, 4
	Piscivorous Fish	61	7.6×10^{1}	3.3×10^{0}	1.7×10^{1}	6.1×10^{2}	3	6.0×10^{2}	1.0×10^{0}	5.8×10^{2}	6.2×10^{2}	6, 3
Ni	Benthic Fish	1	4.4×10^{1}				1	1.9×10^{1}				1, 1
	Piscivorous Fish	4	1.7×10^{1}	1.7×10^{0}	1.1×10^{1}	3.4×10^{1}	24	7.5×10^{1}	2.0×10^{0}	3.0×10^{1}	6.6×10^{2}	
P	Benthic Fish	14	1.5×10^{5}	1.0×10^{0}	1.5×10^{5}	1.6×10^{5}	nd	nd	nd	nd	nd	1, -
	Piscivorous Fish	30	1.4×10^{5}	1.1×10^{0}	1.2×10^{5}	1.7×10^{5}	nd	nd	nd	nd	nd	1, -

TABLE 11. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH (Cont.)

Element	Fish Muscle (L/kg FW)						Whole Fish (L/kg FW)					# ref
	Tissue Type	N	GM/AM	GSD/SD	Min.	Max.	N	GM/AM	GSD/SD	Min.	Max.	(muscle, whole)
Pb	Benthic Fish	5	2.7×10^{1}	2.9×10^{0}	7.0×10^{0}	1.2×10^{2}	3	1.5×10^{2}	1.3×10^{0}	1.2×10^{2}	2.0×10^{2}	1, 1
	Piscivorous Fish	26	4.9×10^{1}	2.8×10^{0}	8.3×10^{0}	2.7×10^{2}	77	3.9×10^{2}	3.0×10^{0}	5.8×10^{1}	5.7×10^{3}	3, 3
	Fish (Generic)[1]	9	2.8×10^{0}	5.9×10^{0}	1.0×10^{-1}	3.0×10^{1}	2	1.2×10^{2}	1.8×10^{0}	7.5×10^{1}	1.8×10^{2}	1, 1
Po	Fish (Generic)	6	3.6×10^{1}	4.3×10^{0}	6.0×10^{0}	1.7×10^{2}	nd	nd	nd	nd	nd	2, -
Pu	Forage Fish	3	2.1×10^{4}	2.6×10^{0}	7.7×10^{3}	5.0×10^{4}	nd	nd	nd	nd	nd	2, -
Ra	Fish (Generic)	21	4.0×10^{0}	7.4×10^{0}	6.0×10^{-2}	1.5×10^{2}	nd	nd	nd	nd	nd	3, -
Rb	Benthic Fish	15	2.9×10^{3}	1.4×10^{0}	1.0×10^{3}	5.0×10^{3}	4	3.3×10^{3}	1.3	2.5×10^{3}	4.0×10^{3}	3, 3
	Forage Fish	11	3.3×10^{3}	1.3×10^{0}	2.3×10^{3}	5.3×10^{3}	38	3.9×10^{3}	1.5	1.2×10^{3}	1.0×10^{4}	3, 3
	Piscivorous Fish	60	5.8×10^{3}	1.6×10^{0}	2.1×10^{3}	1.4×10^{4}	78	7.6×10^{3}	1.4	3.8×10^{3}	1.6×10^{4}	4, 4
Ru	Fish (Generic)	2	3.2×10^{1}	5.1×10^{0}	1.0×10^{1}	1.0×10^{2}	nd	nd	nd	nd	nd	2, -
Sb	Benthic Fish	6	8.20	8.2×10^{0}	1.9×10^{0}	9.1×10^{1}	4	1.2×10^{1}	2.5×10^{0}	4.8×10^{0}	4.4×10^{1}	3, 3
	Forage Fish	10	8.9×10^{1}	2.0×10^{0}	3.7×10^{1}	3.6×10^{2}	38	1.3×10^{2}	1.2×10^{1}	2.6×10^{1}	9.3×10^{6}	4, 4
	Piscivorous Fish	8	2.5×10^{1}	4.4×10^{0}	3.6×10^{0}	1.1×10^{2}	10	3.7×10^{1}	2.3×10^{0}	4.7×10^{0}	9.4×10^{1}	3, 3
Sc	Benthic Fish	1	1.7×10^{2}				1	8.3×10^{3}				1, 1
	Forage Fish	10	2.0×10^{2}	2.5×10^{0}	3.3×10^{1}	7.3×10^{2}	37	8.7×10^{2}	3.4×10^{0}	6.7×10^{1}	3.7×10^{4}	2, 2
	Piscivorous Fish	4	1.4×10^{2}	1.3×10^{0}	1.0×10^{2}	2.0×10^{2}	nd	nd	nd	nd	nd	2, -
Se	Benthic Fish	1	9.4×10^{3}				1	6.1×10^{3}	nd	nd	nd	1, 1
	Forage Fish	10	5.7×10^{3}	1.4×10^{0}	3.5×10^{3}	8.2×10^{3}	38	6.9×10^{3}	1.4×10^{0}	3.6×10^{3}	1.2×10^{4}	2, 6
	Piscivorous Fish	4	6.1×10^{3}	1.1×10^{0}	5.5×10^{3}	6.9×10^{3}	nd	nd	nd	nd	nd	2, -

[1] Generic' fish data are typically based on very few data on accumulation factors for which little supplementary information is available. These values should not be adopted as preferred reference values unless no other, more specific data are available.

TABLE 11. SUMMARY OF BIOACCUMULATION FACTORS (BAF) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH AND THEIR TISSUES (Cont.)

Element	Tissue Type	Fish Muscle (L/kg FW)					Whole Fish (L/kg FW)					# ref (muscle, whole)
		N	GM/AM[1]	GSD/SD	Min.	Max.	N	GM/AM	GSD/SD	Min.	Max.	
Sr	Benthic Fish	15	2.2×10^{0}	1.8×10^{0}	1.10	1.1×10^{1}	4	4.0×10^{1}	2.0×10^{0}	2.2×10^{1}	1.0×10^{2}	5, 3
	Forage Fish	11	2.2×10^{1}	1.9×10^{0}	8.30	6.9×10^{1}	38	7.4×10^{1}	1.7×10^{0}	2.5×10^{1}	2.4×10^{2}	7, 11
	Piscivorous Fish	62	2.1×10^{0}	3.4×10^{0}	1.4×10^{-1}	2.3×10^{1}	78	3.0×10^{2}	1.4×10^{0}	3.3×10^{1}	7.1×10^{2}	27, 33
Tb	Benthic Fish	1	3.0×10^{2}				1	2.4×10^{3}	nd	nd	nd	1, 1
	Forage Fish	9	4.0×10^{2}	2.4×10^{0}	2.0×10^{2}	1.7×10^{3}	35	7.0×10^{2}	2.6×10^{0}	8.0×10^{1}	2.2×10^{3}	2, 3
	Piscivorous Fish	4	4.5×10^{2}	1.4×10^{0}	3.4×10^{2}	7.0×10^{2}	nd	nd	nd	nd	nd	2, -
Te	Piscivorous Fish	3	1.5×10^{2}	1.5×10^{0}	9.6×10^{1}	2.1×10^{2}	10	4.2×10^{2}	1.5×10^{0}	2.2×10^{2}	8.9×10^{2}	1, 1
Th	Fish (Generic)	1	6.00				1	3.8×10^{1}	nd	nd	nd	1, 1
	Benthic Fish	nd	nd	nd	nd	nd	1	3.8×10^{3}	nd	nd	nd	-, 1
Ti	Benthic Fish	1	1.5×10^{2}				1	1.3×10^{3}	nd	nd	nd	1, 1
	Forage Fish	9	1.8×10^{2}	1.6×10^{0}	1.1×10^{2}	3.5×10^{2}	34	3.6×10^{2}	1.8×10^{0}	1.2×10^{2}	1.3×10^{3}	2, 3
	Piscivorous Fish	4	1.8×10^{2}	1.2×10^{0}	1.4×10^{2}	2.1×10^{2}	nd	nd	nd	nd	nd	2, -
Tl	Benthic Fish	14	2.4×10^{3}	3.8×10^{0}	1.3×10^{2}	1.0×10^{4}	3	7.6×10^{1}	1.3×10^{0}	6.4×10^{1}	1.0×10^{2}	1, 1
	Piscivorous Fish	48	7.4×10^{2}	2.1×10^{0}	6.6×10^{1}	2.8×10^{3}	78	6.3×10^{2}	1.6×10^{0}	6.5×10^{1}	3.1×10^{3}	1, 1
U	Fish (Generic)	12	8.6×10^{-1}	1.2×10^{1}	2.0×10^{-2}	2.0×10^{1}	2	2.2×10^{0}	1.7×10^{0}	1.5×10^{0}	3.3×10^{0}	3, 2
V	Benthic Fish	15	8.0×10^{1}	1.4×10^{0}	2.5×10^{1}	1.1×10^{2}	4	2.8×10^{2}	2.4×10^{0}	1.5×10^{2}	1.1×10^{3}	3, 3
	Forage Fish	11	3.3×10^{1}	2.5×10^{0}	1.0×10^{1}	1.6×10^{2}	33	1.4×10^{2}	2.6×10^{0}	3.0×10^{1}	1.0×10^{3}	3, 3
	Piscivorous Fish	61	1.2×10^{2}	1.6×10^{0}	1.5×10^{1}	2.4×10^{2}	78	3.3×10^{2}	1.5×10^{0}	8.3×10^{1}	9.9×10^{2}	4, 4

1The arithmetical mean AM and the standard deviation (SD) are given if the number of observations (N) is 2 and the individual value is given if the number of observations (N) is 1.

TABLE 11. SUMMARY OF BIOACCUMULATION FACTORS (*BAF*) FOR VARIOUS TYPES OF EDIBLE FRESHWATER FISH (Cont.)

Element	Tissue Type	Fish Muscle (L/kg FW)					Whole Fish (L/kg FW)					# ref (muscle, whole)
		N	GM	GSD	Min.	Max.	N	GM	GSD	Min.	Max.	
Y	Benthic Fish	11	8.2×10^1	1.4×10^0	5.8×10^1	1.2×10^2	3	4.7×10^1	1.3×10^0	4.0×10^1	6.2×10^1	1, 1
	Piscivorous Fish	12	2.8×10^1	2.5×10^0	4.5×10^0	9.8×10^1	10	2.6×10^1	1.6×10^0	1.1×10^1	4.6×10^1	1, 1
Zn	Benthic Fish	15	6.3×10^3	2.3×10^0	3.8×10^2	1.6×10^4	4	6.3×10^3	2.2×10^0	1.9×10^3	9.8×10^3	5, 4
	Forage Fish	11	1.0×10^3	1.5×10^0	6.4×10^2	1.9×10^3	38	2.3×10^3	1.4×10^0	1.2×10^3	4.3×10^3	3, 3
	Piscivorous Fish	61	3.6×10^3	2.6×10^0	3.3×10^2	1.1×10^4	78	6.3×10^3	1.3×10^0	2.7×10^3	1.8×10^4	5, 5
Zr	Benthic Fish	5	1.4×10^1	1.5×10^0	9.2×10^0	2.1×10^1	nd	nd	nd	nd	nd	1, -
	Piscivorous Fish	7	2.8×10^1	2.7×10^0	1.2×10^1	1.2×10^2	10	9.5×10^1	1.5×10^0	5.7×10^1	2.4×10^2	1, 1

[a] Dy–dysprosium

Where possible, comparisons of geometric mean values (as opposed to arithmetic mean values) were made, since contaminant bioaccumulation data are typically log-normally distributed and consequently, the geometric mean provides a more reasonable estimate of the typical values than the arithmetic mean. A summary of available literature sources on *BAF* and *BSAF* values is provided in Appendix I. In addition, percent water contents of freshwater plants and animals have been compiled, since literature values may be provided either on a per unit fresh weight or per unit dry weight basis (Appendix II, Table AII.1) while data on carbon and hydrogen contents in freshwater dietary items and tissues consumed by humans (on a per unit dry weight basis) are presented in the same Appendix (Table AII.2).

3. RADIONUCLIDE PARTITIONING INTO EDIBLE BIOTIC TISSUES

3.1. Application of specific activity model approach for aquatic ecosystems

Although AFs are utilized to estimate the transfer of many radionuclides from environmental media to edible non-human biota, this approach is not applicable for all types of radionuclides. For example, in some cases, radionuclides have stable, non-decaying analogues that represent a relatively large proportion of the chemical composition of biotic tissues. As a result, in such situations, stable isotopes can essentially ‘dilute’ radioisotopes. To account for this effect, a specific activity model can be applied, which assesses concentrations of radioactive isotopes relative to all isotopes of that element found in biotic tissues. This relationship is described using the following general equation:

$$SA_{m,r} = \frac{C_{m,r}}{C_{m,a}} \cdot C_{b,a} \qquad (4)$$

where $SA_{m,r}$ is the specific activity of a given radioisotope, *r*, in a given environmental medium, m; $C_{m,r}$ is the concentration of a given radioisotope, *r*, in a given environmental medium, *m*; $C_{m,a}$ is the concentration of all isotopes of a given element, a, in that same environmental medium, *m*; and $C_{b,a}$ is the concentration of all isotopes of a given element, *a*, in a given type of biota or tissue.

Notable examples of radionuclides for which a specific activity models should be used include tritium, ^{14}C and ^{36}Cl, which are discussed in the next paper. In addition, in some cases, this approach can also be utilized for radionuclides that are analogues to stable elements that have high concentrations in tissues or whole organisms. For example, this is the case for ^{90}Sr and other bivalent cations, which exchange for calcium in bones and other hard tissues [e.g. 24, 33 and 34]. Carbon contents and hydrogen contents for various types of edible biota are provided in Table 6 of Appendix 1. This information can be used to assess the amount of radioactive forms of carbon and hydrogen in edible biota tissues from the total amounts of these elements present in biota tissues.

Fish-to-water *BAF* for radiocesium and radiostrontium can be tabulated accounting for the inverse relationship between the *BAF* and the analogous potassium and calcium concentration, respectively, in the surrounding water [35-37].

For radiocesium in predatory/omnivorous fishes, the BAF can be estimated using the following equation [37]:

$$BAF(predatory/omnivorous) = \frac{4880}{[K^+]} \qquad (5)$$

where [K$^+$] is concentration of potassium (K$^+$) concentration in lake water (in mg/L).

For non-predatory fishes, the following relationship can be applied:

$$BAF(non-predatory) = \frac{2390}{[K^+]} \qquad (6)$$

Similarly, strontium concentrations can be tabulated based on calcium concentrations in water ([Ca] in mg L^{-1}), since both elements behave in a similar manner, primarily partitioning in the bony parts of aquatic biota (e.g. skeleton, head, fins, bone, fish scales), as follows [25, 35]:

$$BAF(\text{muscle}) = exp(5.2 - 1.2\ ln[Ca]) \qquad (7)$$

$$BAF(\text{bone}) = exp(9.7 - 1.2\ ln[Ca]) \qquad (8)$$

Assuming that 20% of the wet weight of a fish is composed of bony parts, the whole fish *BAF* can be estimated using the following equation [37]:

$$BAF(\text{whole fish}) = exp(8.13 - 1.2\ ln[Ca]) \qquad (9)$$

An important consideration in the application of specific activity models is the choice of environmental medium, *m*, in Equation 1. Indeed, in some instances, organisms may obtain their supplies of an element from multiple sources, e.g. both water and sediments. In such cases, the specific activity in the organism will be some weighted average of the specific activities in the source media and this weighted average may vary in time and space depending on the relative availability of the different sources.

3.2. Derived parameters for radionuclide partitioning into edible biotic tissues

As discussed above, humans can consume a number of different types of biotic tissues, which may have varying propensities to accumulate radionuclides of interest. As a result, depending upon the species, radionuclide and tissue under consideration, it may be necessary to estimate the the percent radionuclide loading in specific edible tissues and/or in cases where whole organisms are consumed, to estimate the radionuclide load in the whole body of an organism based on data that have been collected for individual tissues [38-39]. Therefore, where possible, literature data relating tissue biomass to organism fresh weight were compiled for edible biota (Tables 12 and 13), along with data reflecting radionuclide partitioning patterns in edible biotic tissues (Tables 14-22). In the case of teleost fish and amphibians, predictive relationships were developed to estimate changes in tissue or organ mass with body size (Table 12).

Biomass estimates for biota and their internal components, and concentration measurements for each tissue, were then utilized to estimate the expected percent loadings of a given radionuclide in biotic tissues [40], as follows:

$$\text{Percent Loading in Tissue} = \frac{C_{tissue} \cdot m_{tissue}}{C_{whole} \cdot m_{whole}} \cdot 100\% = \frac{CR_{tissue} \cdot C_{\text{Reference Tissue}} \cdot m_{tissue}}{C_{whole} \cdot m_{whole}} \cdot 100\%$$

where C_{tissue} is the element concentration in a given tissue (in mg/kg fresh weight or Bq/kg fresh weight); m_{tissue} is the mass of that tissue (in kg fresh weight); C_{whole} is the element concentration in the whole organism (in mg/kg fresh weight or Bq/kg fresh weight); m_{whole} is fresh weight (in kg); CR_{tissue} is the concentration ratio of the tissue of interest relative to the reference tissue for a given type of biota (based on literature data); and $C_{Reference\ Tissue}$ is the concentration of the element of interest measured in the reference tissue (i.e. muscle).

It is important to note that percent loads can be tabulated using actual data on concentrations or by using concentration ratios for the tissue of interest in a given type of biota. A concentration ratio is defined as the ratio of the concentration of a given element in the tissue of interest relative to its concentration in the Reference Tissue (in this case, muscle) for a given type of biota (e.g. freshwater fish) [38]. For example, measurements of elements have been made for fish muscle tissue, but not for brain tissue. However, element concentrations in brain can be estimated by taking the product of the concentration in muscle (i.e. the Reference Tissue) and the brain-to-muscle concentration ratio for freshwater fish. With such information, it becomes possible to estimate the concentration of a given radionuclide in whole fish based on measurements taken for fish muscle tissue, for example, which could be relevant for fish species that humans eat whole. Available of the information for various types of freshwater biota are summarized in Appendix III [38].

Due to differences in the methodologies and objectives of various studies, several selection criteria were developed to choose which data would be included in the review. Therefore, of the more than 10,000 titles identified as being of potential relevance, about 1,000 reports and articles were selected for review on the basis of the following criteria:

- Tissue concentrations had been measured once steady-state conditions were achieved;
- Long-term exposure to the element of interest had occurred;
- Well-defined, realistic environmental conditions were predominant during the study; and
- Information was available on elemental concentrations in reference organs or tissues for each type of organism under consideration within a given reference paper.

Data that did not meet these criteria were excluded from the analysis. For example, some studies focused on the rate of change of element concentrations in non-human biota following changes in exposure through their environments [41]. However, when stable element concentrations in the environment are changing, corresponding tissue concentrations in non-human biota do not necessarily change at the same rate [41]. Therefore, comparison of tissue concentrations in a changing system could potentially result in misinterpretation of the relative concentrations of elements in various tissues. To overcome this, data were only included from reports in which element concentrations in non-human biota were constant with respect to their surrounding environment (i.e. at steady-state). This ensured that the relative concentrations in tissues and biota were stable and could be compared between ecosystems without misinterpretation.

In some cases, data were taken from laboratory studies or controlled field experiments. In these cases, it was necessary to ensure that the environmental and physicochemical conditions under which these studies were carried out were well-defined and realistic. For example, appropriate controls were required, as well as realistic pH, temperature, and concentrations of the elements or compounds of interest.

The final selection criterion accounted for the fact that it was often difficult to compare stable element concentrations in tissues from different studies. These difficulties can be attributed to homeostatic mechanisms in some types of organisms, which could cause elimination systems to be enhanced when a given element is in excess; regulation of some elements by the body, whereby the element may enter the organism's anabolic/catabolic cycles and concentrate in predictable 'biological compartments' (e.g. iron in hemoglobin, myoglobin and liver); differences in uptake thresholds between environmental conditions with varying contaminant loads; and other factors that can affect elemental partitioning between biota and their environments [16-18]. As a result, the relationships between environmental concentrations and tissue concentrations were often non-linear in nature. To address this inherent complexity, reference tissues were chosen for each type of biota and relative tissue concentrations were standardized between studies by dividing the concentration in a given tissue by that in the reference tissue. Reference tissues were chosen on the basis of data availability in the literature and relative importance of the tissue to human exposure and/or biotic health (Table 14) [40].

TABLE 12. PREDICTIVE RELATIONSHIPS BETWEEN WHOLE BODY WEIGHT (in kg), X, AND TISSUE WEIGHT (in g), Y, FOR TELEOST FISH AND AMPHIBIANS

Tissue	Equation	R^2
Teleost Fishes:		
Bone	$Y = 40.68\ X^{1.03}$	0.992
Brain	$Y = 0.960\ X^{0.504}$	0.7474
Eyes	$Y = 5.36\ X^{0.76}$	0.7267
Gonads (female)	$Y = 3.67\ X^{0.729}$	0.3398
Gonads (male)	$Y = 2.03\ X^{1.13}$	0.4205
Heart	$Y = 1.92\ X^{1.00}$	0.9146
Kidney	$Y = 5.16\ X^{1.03}$	0.8908
Liver	$Y = 13.42\ X^{1.08}$	0.8985
Spleen	$Y = 1.12\ X^{0.98}$	0.8562
Stomach & Intestine	$Y = 39.61X + 36.76$	0.8944
Thyroid	$Y = 0.0131X + 8 \times 10^{-5}$	0.6275
Amphibians:		
Eyes	$Y = 4.124\ X^{0.693}$	0.9868
Heart	$Y = 0.464\ \ln(X) + 1.96$	0.9857
Lung	$Y = 3.68\ X^{0.471}$	0.9775
Spleen	$Y = 0.0877\ \ln(X) + 0.443$	0.8142

TABLE 13. MEAN BIOMASSES OF TELEOST FISH AND AMPHIBIAN TISSUES RELATIVE TO WHOLE BODY WEIGHT

Tissue Type	N	Tissue-to-Body Weight (%)		Min	Max
		GM/AM	GSD/SD		
Teleost Fishes:					
Bone	17	4.7	1.2	2.3	9.1
Brain	183	0.087	1.0	7.0×10^{-5}	2.29
Eyes	174	0.50	1.3	0.034	1.65
Gill	4	1.3	1.2	0.7	1.8
Gizzard	2	1.8	1.0	1.8	1.8
Gizzard Contents	5	0.24	1.6	0.03	0.7
Gonads (F)	39	1.53	1.8	0.040	6.41
Gonads (M)	35	0.86	2.1	0.034	1.8
Heart	180	0.19	1.2	0.077	2.71
Kidney	137	0.52	1.2	0.16	1.44
Liver	216	1.4	1.2	0.22	6.23
Muscle	5	64.3	1.1	55.3	76.7
Scales	1	7.0	nd	nd	nd
Skin	1	7.1	nd	nd	nd
Skin & Scales	5	12.0	1.1	9.3	14.1
Spleen	77	0.11	1.1	0.031	0.413
Thyroid	170	1.4×10^{-3}	1.0	2.0×10^{-6}	0.162
Viscera	3	10.4	1.2	6.5	16.1
Stomach & Intestines	157	5.1	6.0	0.2	12.3
Amphibians:					
Adrenal Gland	12	0.029	1.1	0.018	0.037
Bone	1	10.0		nd	nd
Brain	11	0.20	1.3	0.089	0.55
Eyes	12	0.75	1.2	0.49	1.57
Gonads	estimated	0.02	nd	nd	
Heart	41	0.87	1.2	0.32	1.3
Kidney	41	0.98	1.2	0.27	2.5
Liver	41	5.7	1.2	1.9	7.8
Lung	41	1.7	1.2	0.53	2.5
Spleen	40	0.28	1.3	0.042	0.51
Stomach & Intestines	35	8.7	1.1	47.2	99.0
Thyroid	12	0.014	1.2	0.0065	0.0281
Muscle[1]	estimated	69.0	nd	nd	

[1] Muscle biomass for amphibians was estimated as the difference between whole body weight and the masses of the remaining tissues, nd= not determined.

TABLE 13. MEAN BIOMASSES OF TELEOST FISH AND AMPHIBIAN TISSUES RELATIVE TO WHOLE BODY WEIGHT (Cont.)

Tissue Type	N	Tissue-to-Body Weight (%)		Min	Max
		GM/AM	GSD/SD		
Teleost Fishes:					
Bone	17	4.71	1.1	2.34	9.1
Brain	183	0.087	1.0	7.0×10^{-5}	2.29
Eyes	174	0.504	1.3	0.034	1.65
Gill	4	1.3	1.2	0.7	1.8
Gizzard	2	1.80	1.0	1.8	1.8
Gizzard Contents	5	0.242	1.6	0.03	0.7
Gonads (F)	39	1.53	1.7	0.040	6.41
Gonads (M)	35	0.860	2.1	0.034	1.8
Heart	180	0.192	1.2	0.077	2.71
Kidney	137	0.518	1.2	0.155	1.44
Liver	216	1.43	1.2	0.222	6.23
Muscle	5	64.3	1.1	55.3	76.7
Scales	1	7.0	nd	nd	nd
Skin	1	7.1	nd	nd	nd
Skin & Scales	5	12.0	1.1	9.3	14.1
Spleen	77	0.112	1.1	0.031	0.413
Thyroid	170	1.42×10^{-3}	1.0	2.03×10^{-6}	0.162
Viscera	3	10.4	1.2	6.5	16.1
Stomach & Intestines	157	5.06	6.2	0.200	12.3
Amphibians:					
Adrenal Gland	12	0.0287	1.1	0.0177	0.0372
Bone	1	10		nd	nd
Brain	11	0.200	1.3	0.089	0.553
Eyes	12	0.751	1.2	0.494	1.57
Gonads	estimated	0.02	nd	nd	
Heart	41	0.871	1.2	0.317	1.26
Kidney	41	0.978	1.2	0.273	2.53
Liver	41	5.70	1.2	1.89	7.77
Lung	41	1.70	1.2	0.531	2.48
Spleen	40	0.275	1.3	0.042	0.512
Stomach & Intestines	35	8.67	1.1	47.200	99.0
Thyroid	12	0.014	1.2	0.00646	0.0281
Muscle[1]	estimated	69	nd	nd	nd

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[1] Muscle biomass for amphibians was estimated as the difference between whole body weight and the masses of the remaining tissues, nd= not determined.

TABLE 14. REFERENCE TISSUES CHOSEN FOR EACH TYPE OF FRESHWATER BIOTA.

Type of Organism	Reference Tissue
Amphibians	[1] Muscle or Carcass
Reptiles	Muscle
Fish	Muscle
Birds	Bone
Invertebrates	[1]Soft Tissue or Muscle
Aquatic Plants (or Macrophytes)	[1]Leaves or Shoots

[1] The Reference Tissue chosen depends on the data available in the paper. For example, Shoots represent Leaves + Stems, since in some papers, plant tissues are distinguished as ‘above-ground’ vs. ‘below-ground’. Note that in some cases (e.g. in the case of birds), a non-edible tissue was chosen as the Reference Tissue, since most of the available literature included data on this tissue.

TABLE 15. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN ELEMENT *CRs* FOR FRESHWATER AQUATIC MACROPHYTES. REFERENCE ORGANS ARE IN BOLD

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Cd	**Leaves**	**4**	**1.0**	**0.0**	**1.0**	**1.0**
Cd	Root	4	2.6	1.0	2.0	4.0
Cd	Stem	4	2.0×10^{2}	7.5×10^{1}	1.1×10^{2}	2.9×10^{2}
Co	Epiphyton	1.4×10^{1}	1.1×10^{1}	1.8×10^{1}	6×10^{-1}	7.0×10^{1}
Co	Root	1.4×10^{1}	1.0×10^{1}	1.2×10^{1}	2×10^{-2}	4.2×10^{1}
Co	**Shoot**	**1.4** $\times 10^{1}$	**1.0**	**0.0**	**1.0**	**1.0**
Cr	Root	2	5.0×10^{2}	3.2×10^{2}	2.7×10^{2}	7.3×10^{2}
Cr	**Shoot**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Cs	Epiphyton	7	1.9×10^{1}	2.1×10^{1}	8.8×10^{-1}	5.4×10^{1}
Cs	Fruit	4	2.8	2.7	4.0	6.0
Cs	**Leaves**	**3**	**1.0**	**0.0**	**1.0**	**1.0**
Cs	Root	9	3.2	2.3	0.3	7.1
Cs	**Shoot**	**7**	**1.0**	**0.0**	**1.0**	**1.0**
Cs	Stem	2	9.5×10^{-1}	7.1×10^{-2}	9.0×10^{-1}	1.0
Cu	Root	2	1.8×10^{1}	8.3	1.2×10^{1}	24
Cu	**Shoot**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Eu	Epiphyton	7	1.4×10^{1}	1.4×10^{1}	4.6	4.2×10^{1}
Eu	Root	7	7.4	3.4	3.2	1.2×10^{1}
Eu	**Shoot**	**7**	**1.0**	**0.0**	**1.0**	**1.0**
Mn	Epiphyton	1.7×10^{1}	1.3×10^{1}	4.4×10^{1}	4×10^{-1}	1.8×10^{2}
Mn	Root	1.4×10^{1}	7.3	5.8	2.2	19
Mn	**Shoot**	**1.4** $\times 10^{1}$	**1.0**	**0.0**	**1.0**	**1.0**
Mn	Whole plant	3	1.0	0.0	1.0	1.0
Pb	**Leaves**	**4**	**1.0**	**0.0**	**1.0**	**1.0**
Pb	Root	6	3.8×10^{1}	6.1×10^{1}	2.6	1.5×10^{2}
Pb	**Shoot**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Pb	Stem	4	4.1×10^{-1}	1.1×10^{-1}	2.7×10^{-1}	5.3×10^{-1}
Po	Root	1	4.8	[a] nd	[a] nd	[a] nd
Po	**Shoot**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**

[a] Not applicable since n = 1, nd=not detemined

TABLE 16. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN ELEMENT *CRs* FOR FRESHWATER MOLLUSCS. REFERENCE ORGANS ARE IN BOLD.

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Cd	Foot	3	4.4×10^{-1}	2.4×10^{-1}	2.4×10^{-1}	7.0×10^{-1}
Cd	Gills	3	2.2	1.2	1.5	3.6
Cd	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Cd	Mantle	3	8.6×10^{-1}	5.9×10^{-1}	3.6×10^{-1}	1.5
Cd	Viscera	1	7.2×10^{-1}	[a] nd	[a] nd	[a] nd
Cd	Whole body	1	1.0	[a] nd	[a] nd	[a] nd
Cu	Foot	2	1.5	2.7×10^{-1}	1.3	1.7
Cu	Gills	2	2.7	2.2×10^{-1}	2.6	2.9
Cu	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Cu	Mantle	2	2.0	1.1	1.2	2.8
Fe	Foot	2	1.6	8.7×10^{-1}	1.0	2.3
Fe	Gills	2	4.2	1.9	2.9	5.6
Fe	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Fe	Mantle	2	2.5	4.8×10^{-1}	2.2	2.8
Hg	Foot	3	1.1	4.5×10^{-1}	6.6×10^{-1}	1.6
Hg	Gills	3	2.6	2.1	9.7×10^{-1}	4.9
Hg	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Hg	Mantle	3	1.3	1.0	5.7×10^{-1}	2.5
Hg	Viscera	1	1.2	[a] nd	[a] nd	[a] nd
Hg	Whole body	1	1.0	[a] nd	[a] nd	[a] nd
I	Shell	1	8.0	[a] nd	[a] nd	[a] nd
I	**Soft tissue**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Mn	Foot	2	2.2	2.0	7.8×10^{-1}	3.7
Mn	Gills	2	2.3×10^{1}	2.0×10^{1}	8.4	3.7×10^{1}
Mn	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Mn	Mantle	2	6.1	2.0	4.7	7.5
Mn	Shell	2	2.7	3.3	3.0×10^{-1}	5.0
Mn	Soft tissue	2	1.0	0.0	1.0	1.0
Mn	Whole body	1	5.0	[a] nd	[a] nd	[a] nd
Pb	Foot	2	8.2×10^{-1}	5.2×10^{-1}	4.6×10^{-1}	1.2
Pb	Gills	2	3.3	2.5	1.5	5.1
Pb	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Pb	Mantle	2	1.5	1.1	7.2×10^{-1}	2.3
Po	Shell	2	1.4×10^{-2}	1.2×10^{-2}	5.5×10^{-3}	2.3×10^{-2}
Po	**Soft tissue**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Sr	Shell	1	2.3×10^{2}	[a] nd	[a] nd	[a] nd
Sr	**Soft tissue**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Zn	Foot	2	1.1	3.2×10^{-1}	8.4×10^{-1}	1.3
Zn	Gills	2	3.5	6.5×10^{-1}	3.0	3.9
Zn	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Zn	Mantle	2	1.4	1.1×10^{-1}	1.3	1.5

[a] Not applicable since n = 1, nd=not detemined.

TABLE 17. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER DECAPODS. REFERENCE ORGANS ARE IN BOLD

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Cd	Exoskeleton	2	1.4	4.2×10^{-1}	1.1	1.7
Cd	Hepatopancreas	2	7.1	3.7×10^{-1}	6.9	7.4
Cd	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Cd	Viscera	2	1.2	1.1×10^{-1}	1.1	1.3
Cu	Exoskeleton	2	6.5×10^{-1}	2.7×10^{-1}	4.6×10^{-1}	8.4×10^{-1}
Cu	Hepatopancreas	2	9.5	1.1	8.7	1.0×10^{1}
Cu	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Cu	Viscera	2	7.2×10^{-1}	4.4×10^{-1}	4.1×10^{-1}	1.0
Mg	Exoskeleton	2	1.7	1.3×10^{-1}	1.6	1.8
Mg	Hepatopancreas	2	5.7×10^{-1}	4.3×10^{-2}	5.3×10^{-1}	6.0×10^{-1}
Mg	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Mg	Viscera	2	2.5	6.3×10^{-1}	2.0	2.9
Mn	Exoskeleton	2	6.7	2.6×10^{-1}	6.5	6.9
Mn	Hepatopancreas	2	7.5	3.0×10^{-2}	7.5	7.5
Mn	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Mn	Viscera	2	5.8	1.0	5.1	6.5
Ni	Exoskeleton	1	9.7	[a] nd	[a] nd	[a] nd
Ni	Hepatopancreas	1	1.5	[a] nd	[a] nd	[a] nd
Ni	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Ni	Viscera	1	5.3×10^{-1}	[a] nd	[a] nd	[a] nd
Po	Exoskeleton	2	8.5×10^{-1}	0.0	8.1×10^{-1}	8.8×10^{-1}
Po	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Zn	Exoskeleton	2	2.5×10^{-1}	1.1×10^{-1}	1.8×10^{-1}	3.3×10^{-1}
Zn	Hepatopancreas	2	1.6	1.1×10^{-1}	1.6	1.7
Zn	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Zn	Viscera	2	8.9×10^{-1}	1.5×10^{-1}	7.8×10^{-1}	1.0

[a] Not applicable since n = 1.

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[a] Not applicable since n = 1, nd=not detemined.

TABLE 18. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR AMPHIBIANS. REFERENCE ORGANS ARE IN BOLD

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Al	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Al	Liver	1	2.1×10^{-1}	[a] nd	[a] nd	[a] nd
Ba	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Ba	Liver	1	4.3×10^{-1}	[a] nd	[a] nd	[a] nd
Cd	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cd	Liver	1	2.9	[a] nd	[a] nd	[a] nd
Co	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Co	Liver	1	2.0	[a] nd	[a] nd	[a] nd
Cr	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cr	Liver	1	2.0	[a] nd	[a] nd	[a] nd
Cs	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cs	Liver	1	2.0	[a] nd	[a] nd	[a] nd
Cs	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cs	Whole body	1	2.6	[a] nd	[a] nd	[a] nd
Cu	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cu	Liver	1	3.9×10^{1}	[a] nd	[a] nd	[a] nd
Mn	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Mn	Liver	1	3.9×10^{-1}	[a] nd	[a] nd	[a] nd
Mo	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Mo	Liver	1	30	[a] nd	[a] nd	[a] nd
Ni	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Ni	Liver	1	1.5×10^{-1}	[a] nd	[a] nd	[a] nd
Pb	Bone	5	7.0	6.9	1.2	1.8×10^{1}
Pb	Carcass	1	1.0	[a] nd	[a] nd	[a] nd
Pb	Kidney	5	7.5	6.7	1.0	1.5×10^{1}
Pb	Liver	4	3.3	1.9	1.5	5.4
Pb	**Muscle**	**3**	**1.0**	**0.0**	**1.0**	**1.0**
Pb	Skin	3	1.5	4.7×10^{-1}	1.1	2.0
Rb	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Rb	Liver	1	8.0×10^{-1}	[a] nd	[a] nd	[a] nd
Sr	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Sr	Liver	1	1.2×10^{-1}	[a] nd	[a] nd	[a] nd
Zn	**Carcass**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Zn	Liver	1	8.1×10^{-1}	[a] nd	[a] nd	[a] nd

[a] Not applicable since n = 1, nd=not detemined.

TABLE 19. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER REPTILES. REFERENCE ORGANS ARE IN BOLD.

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Ac	Liver	1	9.0	[a] nd	[a] nd	[a] nd
Ac	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Pb	Bone	1	1.2×10^{2}	[a] nd	[a] nd	[a] nd
Pb	Heart	1	1.1×10^{-1}	[a] nd	[a] nd	[a] nd
Pb	Liver	1	9.1	[a] nd	[a] nd	[a] nd
Pb	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Po	Bone	1	4.0	[a] nd	[a] nd	[a] nd
Po	Heart	1	1.6	[a] nd	[a] nd	[a] nd
Po	Liver	1	3.7×10^{1}	[a] nd	[a] nd	[a] nd
Po	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Ra	Bone	1	2.1×10^{2}	[a] nd	[a] nd	[a] nd
Ra	Heart	1	3.3	[a] nd	[a] nd	[a] nd
Ra	Liver	1	6.3	[a] nd	[a] nd	[a] nd
Ra	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Th	Bone	1	7.0×10^{1}	[a] nd	[a] nd	[a] nd
Th	Liver	3	1.6×10^{1}	1.3×10^{1}	7.3	3.0×10^{1}
Th	**Muscle**	**3**	**1.0**	**0.0**	**1.0**	**1.0**
U	Bone	2	3.3×10^{1}	3.8×10^{1}	6.3	6.0×10^{1}
U	Liver	4	2.1×10^{1}	2.0×10^{1}	5.0	5.0×10^{1}
U	**Muscle**	**4**	**1.0**	**0.0**	**1.0**	**1.0**

[a] Not applicable since n = 1, nd=not detemined.

TABLE 20. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER FISHES, REFERENCE ORGANS ARE IN BOLD

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Al	**Carcass**	**4**	**1.0**	**0.0**	**1.0**	**1.0**
Al	Gills	4	3.0	1.1	2.0	4.1
As	Bone	1	6.9	[a] nd	[a] nd	[a] nd
As	Liver	1	5.3×10^{-1}	[a] nd	[a] nd	[a] nd
As	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
As	Whole body	1	3.2	[a] nd	[a] nd	[a] nd
Ca	Bone	9	9.5×10^{-2}	7.1×10^{2}	91	2.0×10^{3}
Ca	Kidney	8	1.7×10^{-1}	7.6×10^{-2}	6.3×10^{-2}	2.9×10^{-1}
Ca	**Muscle**	**17**	**1.0**	**0.0**	**1.0**	**1.0**
Ca	Skin	8	2.3×10^{-1}	5.2×10^{-2}	1.4×10^{-1}	3.0×10^{-1}
Cd	Carcass	4	1.0	0.0	1.0	1.0
Cd	Fat	2	1.0	2.7×10^{-1}	7.8×10^{-1}	1.2
Cd	Gonad	22	2.8	4.1	1.2×10^{-1}	1.8×10^{1}
Cd	Gills	19	2.6	2.8	2.9×10^{-1}	1.2×10^{1}
Cd	Kidney	13	4.2	4.1	1.0	1.3×10^{1}
Cd	Liver	32	2.3×10^{1}	5.3×10^{1}	1.0×10^{-1}	2.3×10^{2}
Cd	**Muscle**	**33**	**1.0**	**0.0**	**1.0**	**1.0**
Cd	Skin	9	1.2	1.1	2.3×10^{-1}	3.2

TABLE 20. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER FISHES, REFERENCE ORGANS ARE IN BOLD (Cont.)

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Co	Bone	8	4.1	5.2	3.9×10^{-1}	1.7×10^{1}
Co	Blood	2	1.7	1.5	6.9×10^{-1}	2.8
Co	Gills	4	1.3×10^{2}	2.5×10^{2}	5.5×10^{-1}	5.0×10^{2}
Co	Heart	1	0.0	[a] nd	[a] nd	[a] nd
Co	Head	2	4.3	9.3×10^{-1}	3.7	5.0
Co	Kidney	5	1.7×10^{1}	2.3×10^{1}	1.6	5.7×10^{1}
Co	Liver	5	5.7	7.6	3.7×10^{-1}	1.9×10^{1}
Co	**Muscle**	**11**	**1.0**	**0.0**	**1.0**	**1.0**
Co	Skin	3	1.7	1.3	4.0×10^{-1}	3.1
Co	Spleen	1	2.2×10^{-3}	[a] nd	[a] nd	[a] nd
Co	Viscera	2	6.2	4.7×10^{-1}	5.8	6.5
Co	Whole body	4	1.2×10^{1}	2.2×10^{1}	1.4	4.5×10^{1}
Cr	Bone	1	2.7×10^{-1}	[a] nd	[a] nd	[a] nd
Cr	Gills	1	4.5×10^{2}	[a] nd	[a] nd	[a] nd
Cr	Kidney	1	7.3×10^{-1}	[a] nd	[a] nd	[a] nd
Cr	Liver	1	1.8	[a] nd	[a] nd	[a] nd
Cr	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cr	Whole body	1	3.8	[a] nd	[a] nd	[a] nd
Cu	Bone	1	4.1	[a] nd	[a] nd	[a] nd
Cu	Blood	2	6.5×10^{-1}	9.2×10^{-1}	$-^{3}$	1.3
Cu	Brain	2	3.1	3.1	9×10^{-1}	5.3
Cu	Eye	1	8.5×10^{-1}	[a] nd	[a] nd	[a] nd
Cu	Eye retina	1	1.0×10^{1}	[a] nd	[a] nd	[a] nd
Cu	Fat	3	1.8	1.5	5.8×10^{-1}	3.5
Cu	Gonad	14	2.3	1.4	6.3×10^{-1}	5.4
Cu	Gills	16	3.4	2.3	5×10^{-1}	8.7
Cu	Heart	1	1.5	[a] nd	[a] nd	[a] nd
Cu	Kidney	23	4.2	2.3	1.9	1.2×10^{1}
Cu	Liver	26	3.0×10^{1}	5.6×10^{1}	3.1	2.6×10^{2}
Cu	**Muscle**	**34**	**1.0**	**0.0**	**1.0**	**1.0**
Cu	Ovary	1	2.6	[a] nd	[a] nd	[a] nd
Cu	Pyloric caecae	1	4.9	[a] nd	[a] nd	[a] nd
Cu	Scales	1	1.3	[a] nd	[a] nd	[a] nd
Cu	Blood serum	1	6.3×10^{-1}	[a] nd	[a] nd	[a] nd
Cu	Skin	19	1.0	4.7×10^{-1}	4.2×10^{-1}	2.0
Cu	Skull	1	1.8	[a] nd	[a] nd	[a] nd
Cu	Spleen	2	6.2	1.8	4.9	7.5
Cu	Vertebra	1	1.8	[a] nd	[a] nd	[a] nd
Fe	Bone	5	1.5	9.1×10^{-1}	5.0×10^{-2}	2.2
Fe	Fat	2	2.1	8.5×10^{-1}	1.5	2.7
Fe	Gonad	9	2.4	2.4	4.1×10^{-1}	7.7
Fe	Gills	5	5.3×10^{2}	1.2×10^{3}	3.3	2.6×10^{3}
Fe	Kidney	15	1.6×10^{1}	7.2	4.5×10^{-1}	2.9×10^{1}
Fe	Liver	16	1.8×10^{1}	1.9×10^{1}	4.1×10^{-1}	7.3×10^{1}

TABLE 20. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER FISHES, REFERENCE ORGANS ARE IN BOLD (Cont.)

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Fe	**Muscle**	**24**	**1.0**	**0.0**	**1.0**	**1.0**
Fe	Scales	4	1.9	6.4×10^{-1}	9.3×10^{-1}	2.4
Fe	Skin	8	1.9	1.5	1.0	5.3
Fe	Whole body	1	5.0	[a] nd	[a] nd	[a] nd
Hg	Gills	4	1.1	5.9×10^{-1}	5.9×10^{-1}	1.9
Hg	Kidney	4	2.2	7.9×10^{-1}	1.1	2.9
Hg	Liver	11	1.4	8.0×10^{-1}	5.4×10^{-1}	3.2
Hg	**Muscle**	**11**	**1.0**	**0.0**	**1.0**	**1.0**
I	**Muscle**	**3**	**1.0**	**0.0**	**1.0**	**1.0**
I	Ovary	3	1.7×10^{1}	2.3	1.6×10^{1}	2.0×10^{1}
I	Thyroid	3	4.8×10^{3}	1.8×10^{3}	2.8×10^{3}	5.8×10^{3}
La	Bone	1	8.0	[a] nd	[a] nd	[a] nd
La	Gills	1	1.2×10^{2}	[a] nd	[a] nd	[a] nd
La	Kidney	1	2.5×10^{-1}	[a] nd	[a] nd	[a] nd
La	Liver	1	3.3	[a] nd	[a] nd	[a] nd
La	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
La	Whole body	1	13	[a] nd	[a] nd	[a] nd
Mg	Kidney	8	7.5×10^{-1}	1.0×10^{-1}	5.5×10^{-1}	8.6×10^{-1}
Mg	**Muscle**	**8**	**1.0**	**0.0**	**1.0**	**1.0**
Mg	Skin	8	3.0×10^{-1}	1.1×10^{-1}	1.7×10^{-1}	5.4×10^{-1}
Mn	Bone	4	8.9	2.3	6.9	1.1×10^{1}
Mn	Eye	2	1.1	0.0	1.1	1.1
Mn	Fat	2	1.3	8.6×10^{-1}	7.0×10^{-1}	1.9
Mn	Fin	4	1.1×10^{1}	1.2	1.0×10^{1}	1.2×10^{1}
Mn	Gonad	11	1.6	1.1	3.5×10^{-1}	3.7
Mn	Gills	19	9.2	7.0	2.1	2.6×10^{1}
Mn	Head	4	7.3	7.5×10^{-1}	6.6	7.9
Mn	Kidney	25	3.0	1.6	6.2×10^{-1}	6.0
Mn	Liver	19	4.6	4.0	6.2×10^{-1}	1.4×10^{1}
Mn	**Muscle**	**28**	**1.0**	**0.0**	**1.0**	**1.0**
Mn	Epidermal mucous	2	5.7	0.0	5.7	5.7
Mn	Rest of body	2	5.7	0.0	5.7	5.7
Mn	Skin	17	1.2	1.5	6.7×10^{-2}	6.2
Mn	Skin and scales	4	4.8	4.5×10^{-1}	4.5	5.2
Mn	Viscera	6	4.6	4.6	5.4×10^{-1}	1.0×10^{1}
Mn	Whole body	3	9.1	1.0×10^{1}	3.2	2.1×10^{1}
Na	Bone	1	8.0	[a] nd	[a] nd	[a] nd
Na	Blood	1	6.2	[a] nd	[a] nd	[a] nd
Na	Brain	1	4.1	[a] nd	[a] nd	[a] nd
Na	Eye retina	1	5.3	[a] nd	[a] nd	[a] nd
Na	Fat	1	1.0	[a] nd	[a] nd	[a] nd
Na	Heart	1	1.6	[a] nd	[a] nd	[a] nd
Na	Kidney	1	3.7	[a] nd	[a] nd	[a] nd
Na	Liver	1	1.1	[a] nd	[a] nd	[a] nd

[a] Not applicable since n = 1.

TABLE 20. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER FISHES, REFERENCE ORGANS ARE IN BOLD (Cont.)

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Na	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Na	Ovary	1	2.2	[a] na	[a] nd	[a] nd
Na	Scales	1	4.5	[a] nd	[a] nd	[a] nd
Na	Skin	1	1.3	[a] nd	[a] nd	[a] nd
Na	Spleen	1	4.8	[a] nd	[a] nd	[a] nd
Ni	Fat	2	6.2×10^{-1}	1.4×10^{-2}	6.1×10^{-1}	6.3×10^{-1}
Ni	Gonad	2	6.0×10^{-1}	1.1×10^{-2}	6.0×10^{-1}	6.1×10^{-1}
Ni	Gills	4	1.0	7.4×10^{-1}	4.3×10^{-1}	2.1
Ni	Kidney	2	1.3	4.4×10^{-1}	9.4×10^{-1}	1.6
Ni	Liver	11	2.1	1.1	3.7×10^{-1}	3.5
Ni	**Muscle**	**11**	**1.0**	**0.0**	**1.0**	**1.0**
P	Bone	1	1.6×10^{1}	[a] nd	[a] nd	[a] nd
P	Blood	1	2.0	[a] nd	[a] nd	[a] nd
P	Brain	1	1.9	[a] nd	[a] nd	[a] nd
P	Eye lens	1	6.3×10^{-1}	[a] nd	[a] nd	[a] nd
P	Eye retina	1	2.1	[a] nd	[a] nd	[a] nd
P	Fat	1	6.3×10^{-1}	[a] nd	[a] nd	[a] nd
P	Heart	1	2.5	[a] nd	[a] nd	[a] nd
P	Kidney	1	4.4	[a] nd	[a] nd	[a] nd
P	Liver	1	5.5	[a] nd	[a] nd	[a] nd
P	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
P	Ovary	1	2.4	[a] nd	[a] nd	[a] nd
P	Scales	1	9.9	[a] nd	[a] nd	[a] nd
P	Skin	1	1.0	[a] nd	[a] nd	[a] nd
P	Spleen	1	5.6	[a] nd	[a] nd	[a] nd
Pb	Bone	11	1.5×10^{1}	9.5	1.0	3.1×10^{1}
Pb	Brain	9	2.9	4.1	1.2×10^{-1}	1.3×10^{1}
Pb	Carcass	4	1.0	0.0	1.0	1.0
Pb	Eye	5	1.1	8.6×10^{-1}	1.7×10^{-1}	2.4
Pb	Fat	2	8.6×10^{-1}	1.1×10^{-2}	8.5×10^{-1}	8.7×10^{-1}
Pb	Gonad	19	1.9	1.9	4.8×10^{-1}	7.7
Pb	Gills	52	2.4	2.0	6.3×10^{-1}	1.1×10^{1}
Pb	Kidney	22	2.8×10^{1}	5.1×10^{1}	4.4×10^{-1}	2.0×10^{2}
Pb	Liver	67	3.9	1.1×10^{1}	0.0	8.1×10^{1}
Pb	**Muscle**	**74**	**1.0**	**0.0**	**1.0**	**1.0**
Pb	Ovary	9	6.3	9.6	4.2×10^{-1}	2.7×10^{1}
Pb	Skin	16	3.1	3.2	2.8×10^{-1}	1.1×10^{1}
Pb	Skin and muscle	3	2.3	1.5	1.3	4.0
Pb	Spleen	9	2.5×10^{1}	2.8×10^{1}	1.4	7.9×10^{1}
Pb	Testis	9	2.9	4.6	0.0	1.5×10^{1}
Pb	Whole body	14	9.0	1.5×10^{1}	1.0	6.0×10^{1}

[a] Not applicable since n = 1.

TABLE 20. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER FISHES, REFERENCE ORGANS ARE IN BOLD (Cont.)

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Po	Bone	6	2.3	2.6	1.7×10^{-1}	6.5
Po	**Muscle**	**6**	**1.0**	**0.0**	**1.0**	**1.0**
Ra	Bone	17	1.5×10^{1}	1.2×10^{1}	1.0	3.3×10^{1}
Ra	Liver	5	1.4×10^{1}	2.3×10^{1}	2.5	5.6×10^{1}
Ra	**Muscle**	**19**	**1.0**	**0.0**	**1.0**	**1.0**
Ra	Skin	7	2.3	2.8	8.7×10^{-1}	8.7
Ra	Skin and muscle	3	1.1	1.2×10^{-1}	1.0	1.2
Ra	Whole body	10	4.8×10^{1}	1.4×10^{2}	8×10^{-1}	4.4×10^{2}
Rb	Bone	1	2.4	[a]nd	[a]nd	[a]nd
Rb	**Soft tissue**	**1**	**1.0**	**[a]nd**	**[a]nd**	**[a]nd**
Ru	Bone	1	1.1	[a]nd	[a]nd	[a]nd
Ru	General	1	58	[a]nd	[a]nd	[a]nd
Ru	Gills	1	2.2×10^{2}	[a]nd	[a]nd	[a]nd
Ru	**Muscle**	**3**	**1.0**	**0.0**	**0.0**	**1.0**
Ru	Viscera	1	4.4×10^{2}	[a]nd	[a]nd	[a]nd
Ru	Whole body	3	8.3×10^{1}	2.9×10^{1}	5.0×10^{1}	1.0×10^{2}
Sc	Bone	2	1.1×10^{1}	1.5×10^{1}	0.0	2.1×10^{1}
Sc	Blood	1	7.7	[a]nd	[a]nd	[a]nd
Sc	Brain	1	0.0	[a]nd	[a]nd	[a]nd
Sc	Eye lens	1	0.0	[a]nd	[a]nd	[a]nd
Sc	Eye retina	1	0.0	[a]nd	[a]nd	[a]nd
Sc	Fat	1	0.43	[a]nd	[a]nd	[a]nd
Sc	Gills	1	460	[a]nd	[a]nd	[a]nd
Sc	Heart	1	0.0	[a]nd	[a]nd	[a]nd
Sc	Kidney	2	9.2×10^{-1}	1.3	0.0	1.8
Sc	Liver	2	3.2	4.6	0.0	6.4
Sc	**Muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
Sc	Ovary	1	0.0	[a]nd	[a]nd	[a]nd
Sc	Scales	1	0.0	[a]nd	[a]nd	[a]nd
Sc	Skin	1	1.0	[a]nd	[a]nd	[a]nd
Sc	Spleen	1	6.7	[a]nd	[a]nd	[a]nd
Sc	Whole body	1	3.0×10^{1}	[a]nd	[a]nd	[a]nd
Sr	air bladder	1	5.6	[a]nd	[a]nd	[a]nd
Sr	bone	7	2.9×10^{2}	3.9×10^{2}	7.2×10^{1}	1.1×10^{3}
Sr	blood	1	3.9	[a]nd	[a]nd	[a]nd
Sr	fin	1	1.2×10^{2}	[a]nd	[a]nd	[a]nd
Sr	gonad	1	1.1	[a]nd	[a]nd	[a]nd
Sr	gall bladder	1	2.1	[a]nd	[a]nd	[a]nd
Sr	gills	1	1.5×10^{2}	[a]nd	[a]nd	[a]nd
Sr	heart	1	3.6	[a]nd	[a]nd	[a]nd
Sr	kidney	1	3.3	[a]nd	[a]nd	[a]nd
Sr	liver	1	2.9	[a]nd	[a]nd	[a]nd

[a] Not applicable since n = 1, nd=not detemined.

TABLE 20. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR FRESHWATER FISHES, REFERENCE ORGANS ARE IN BOLD (Cont.)

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Sr	**muscle**	**10**	**1.0**	**0.0**	**1.0**	**1.0**
Sr	scales	1	4.6×10^1	[a] nd	[a] nd	[a] nd
Sr	skin	1	2.3	[a] nd	[a] nd	[a] nd
Sr	spleen	1	2.1	[a] nd	[a] nd	[a] nd
Sr	vertebra	1	2.0×10^1	[a] nd	[a] nd	[a] nd
Sr	whole body	2	2.3×10^1	2.7×10^1	3.7	4.2×10^1
Th	liver	2	5.5×10^1	7.7×10^1	0.0	1.1×10^2
Th	**muscle**	**2**	**1.0**	**0.0**	**1.0**	**1.0**
U	bone	12	5.6×10^1	7.2×10^1	2.0	2.1×10^2
U	liver	5	2.8	1.1	2.0	4.0
U	**muscle**	**12**	**1.0**	**0.0**	**1.0**	**1.0**
U	skin	7	8.1	9.8	2.0	3.0×10^1
U	skin and muscle	3	3.4	3.8×10^{-1}	3.0	3.8
U	whole body	7	1.4×10^1	2.2×10^1	5.1×10^{-1}	6.1×10^1
Zn	bone	6	7.3	2.1	5.0	1.1×10^1
Zn	blood	2	5.4	6.8	5.9×10^{-1}	1.0×10^1
Zn	brain	2	2.2	1.4	1.2	3.2
Zn	carcass	4	1.0	0.0	1.0	1.0
Zn	eye	1	5.8	[a] nd	[a] nd	[a] nd
Zn	eye lens	1	2.9	[a] nd	[a] nd	[a] nd
Zn	eye retina	1	1.0×10^2	[a] nd	[a] nd	[a] nd
Zn	fat	3	2.0	9.3×10^{-1}	1.2	3.0
Zn	gonad	21	7.3	5.1	5.4×10^{-1}	2.0×10^1
Zn	gills	21	1.6×10^1	6.3×10^1	1.0	2.9×10^2
Zn	heart	1	4.7	[a] nd	[a] nd	[a] nd
Zn	kidney	28	1.4×10^1	1.5×10^1	1.8	5.1×10^1
Zn	liver	39	5.8	6.6	8×10^{-1}	4.1×10^1
Zn	**muscle**	**48**	**1.0**	**0.0**	**1.0**	**1.0**
Zn	Ovary	1	1.6×10^1	[a] nd	[a] nd	[a] nd
Zn	Pyloric caecae	1	1.6	[a] nd	[a] nd	[a] nd
Zn	Scales	5	1.3×10^1	8.5	5.7	2.8×10^1
Zn	Blood serum	1	9.5×10^{-1}	[a] nd	[a] nd	[a] nd
Zn	Skin	19	8.1	5.0	1.6	2.3×10^1
Zn	Skull	1	1.3	[a] nd	[a] nd	[a] nd
Zn	Spleen	2	9.0	9.3	2.5	16
Zn	Vertebra	1	2.1	[a] nd	[a] nd	[a] nd
Zn	Whole body	1	9.9×10^1	[a] nd	[a] nd	[a] nd
Zr	Bone	1	1.1	[a] nd	[a] nd	[a] nd
Zr	Gills	1	2.0×10^2	[a] nd	[a] nd	[a] nd
Zr	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Zr	Viscera	1	2.2×10^2	[a] nd	[a] nd	[a] nd

[a] Not applicable since n = 1, nd=not detemined.

TABLE 21. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CRs* FOR EELS, REFERENCE ORGANS ARE IN BOLD

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Ca	Bone	1	26	[a] nd	[a] nd	[a] nd
Ca	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Cd	Brain	3	1.5	1.0	8.3×10^{-1}	2.7
Cd	Gill	4	4.7	3.8	1.2	10
Cd	Heart	3	8.1×10^{-1}	2.4×10^{-1}	6.5×10^{-1}	1.1
Cd	Kidney	3	5.0	2.1	2.8	6.9
Cd	Liver	4	8.1	1.1×10^{1}	1.1	2.5×10^{1}
Cd	**Muscle**	**4**	**1.0**	**0.0**	**1.0**	**1.0**
Cd	Skin	1	2.0	[a] nd	[a] nd	[a] nd
Cd	Whole body	1	2.7	[a] nd	[a] nd	[a] nd
Cu	Brain	1	2.0×10^{-1}	[a] nd	[a] nd	[a] nd
Cu	Gill	1	4.0	[a] nd	[a] nd	[a] nd
Cu	Heart	1	8.0	[a] nd	[a] nd	[a] nd
Cu	Kidney	1	4.0	[a] nd	[a] nd	[a] nd
Cu	Liver	1	1.6×10^{1}	[a] nd	[a] nd	[a] nd
Cu	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Hg	Gill	1	5.0	[a] nd	[a] nd	[a] nd
Hg	Liver	1	3.6	[a] nd	[a] nd	[a] nd
Hg	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Hg	Skin	1	1.4	[a] nd	[a] nd	[a] nd
Hg	Whole body	1	1.1	[a] nd	[a] nd	[a] nd
Sr	Bone	1	4.7×10^{1}	[a] nd	[a] nd	[a] nd
Sr	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**
Zn	Brain	1	1.7×10^{1}	[a] nd	[a] nd	[a] nd
Zn	Gill	1	1.3	[a] nd	[a] nd	[a] nd
Zn	Heart	1	1.8	[a] nd	[a] nd	[a] nd
Zn	Kidney	1	1.8	[a] nd	[a] nd	[a] nd
Zn	Liver	1	1.9	[a] nd	[a] nd	[a] nd
Zn	**Muscle**	**1**	**1.0**	**[a] nd**	**[a] nd**	**[a] nd**

[a] Not applicable since n = 1, nd=not detemined.

.TABLE 32. SUMMARY OF MEAN ORGAN-TO-REFERENCE ORGAN STABLE ELEMENT *CR*s, FOR MIGRATORY BIRDS. REFERENCE ORGANS ARE IN BOLD.

Element	Organ	N	AM, C_{org}/C_{ref}	SD	Min	Max
Cs	**Bone**	**10**	**1.0**	**0.0**	**1.0**	**1.0**
Cs	Liver	3	3.1	4.1×10^{-1}	2.8	3.6
Cs	Muscle	9	2.7	1.5	1.0	5.1
Pb	**Bone**	**11**	**1.0**	**0.0**	**1.0**	**1.0**
Pb	Liver	11	2.4×10^{-1}	1.7×10^{-1}	8.3×10^{-2}	6.7×10^{-1}

4. CONCLUSIONS

The paper that has been compiled on radionuclide transfer to freshwater edible biota is much more comprehensive than the information that had been provided in the 1994 version of Technical Reports Series No. 364. In addition to radionuclide water-to-muscle transfer to freshwater fishes, which was covered in the 1994 Technical Reports Series No. 364 report, the updated information also includes transfer from water to whole fishes, distinguishing benthivorous (sediment-dwelling) fish species, forage fishes and piscivorous fishes. In addition, new tables of water-to-edible aquatic primary producers (including algae, floating-leafed and emergent plant tissues); water-to-edible freshwater invertebrate tissues (including generic invertebrate, crustacean and mollusc tissues); and water-to-freshwater herpetofauna (including tadpoles, frogs and reptiles) have been added to provide a more comprehensive representation of transfer to edible freshwater plant and animal tissues. New tables of sediment-to-biota transfer factors have been compiled for fishes and invertebrates. Finally, data on the relative masses of tissues in a range of freshwater biota were compiled, along with a comparison in radionuclide and stable analogue concentrations in each tissue type were provided to facilitate estimation of radionuclide levels in edible tissues where data gaps existed.

Although, a number of additions have been included in the revised Technical Reports Series No. 364 chapter on transfer to edible freshwater biota, it is clear that the amount of data available is not uniform between radionuclides or types of edible plants and animals. For example, although a relatively large pool of data exists for such radionuclides as Cs, Sr or Co; much fewer data exist for many other radionuclides or their analogues. In addition, relatively few data are available for edible species of herpetofauna and invertebrates for many radionuclides.

The information that has been compiled in this paper is focused on steady state radionuclide transfer, although it would be useful to compile relevant data on dynamic parameters, as well as ecological half-lives of radionuclides in edible biota, in the future. Further work is also required to gain a better understanding of the factors that influence radionuclide transfer to edible biota, particularly for less well-studied species (such as invertebrates, herpetofauna, aquatic mammals and waterfowl), for less well-studied radionuclides, as well as for types of biota that show wide ranges of radionuclide transfer factors.

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APPENDIX I

List of publication used for evaluation of radionuclide transfer to aquatic biota species (BAFs), (Database on the BAFs for freshwater species)

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APPENDIX II

TABLE AII.1. COMPILATION OF DATA ON PERCENT WATER CONTENTS IN FRESHWATER DIETARY ITEMS AND TISSUES CONSUMED BY HUMANS

Food Type	AM	SD	.Min.	Max..	References
Aquatic Primary Producers:					
Algae	84	4.7	71	97	[1, 2]
Aquatic macrophytes	87	3.1			[1].
Emergent vegetation			45	93	[1, 2]
Aquatic macrophyte tubers	90	$3.0\text{x}10^{-2}$	86	92	[2]
Emergent vegetation tu6ers	90	$2.0\text{x}10^{-2}$	81	93	[2]
Aquatic Invertebrates:					
Bivalves (without shell)	82	4.5			[1].
Crabs (with shell)	74	6.1			[1].
Shrimp	78	3.3			[1].
Isopods			71	80	[1].
Cladocerans			79	87	[1].
Aquatic Vertebrates:					
Bony fishes	75	5.1	67	79	[1, 2]
Pacific herring	68	3.9			[1].
Reptiles and Amphibians:					
Snakes/Lizards	66 (n = 1)				[1].
Frogs/Toads	85	4.7			[1].
Mammals:					
Mice/Voles/Rabbits	68	1.6			[1].
Birds:					
Passerines (with typical fat reserves)	68 (n = 1)				[1].
Mallard duck (flesh only)	67 (n = 1)				[1].

TABLE AII.2. COMPILATION OF PERCENT CARBON AND PERCENT HYDROGEN CONTENTS IN FRESHWATER DIETARY ITEMS AND TISSUES CONSUMED BY HUMANS (ON A PER UNIT DRY WEIGHT BASIS) [3-7]

		% C (per unit DW)					% H (per unit DW)				
Type of Organism	Tissue Type	N	AM	SD	Min.	Max.	N	AM	SD	Min.	Max.
Algae	Whole	29	48	12	29	70	2	4.4	0.35	4.1	4.6
Aquatic Macrophytes	Not specified	19	31	3.1	26	38	nd	nd	nd	nd	nd
Animals	Not specified	2	47	2.4	45	411	2	6.6	0.07	6.5	6.6
Invertebrates	Whole	43	48	5.2	34	55	5	5.6	1.2	4.5	7.3
Molluscs	Soft tissue	1	40	nd	nd	nd	1	6.0	nd	nd	nd

– nd – no data.

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APPENDIX III: SUMMARY OF AVAILABLE BAF AND BSAF DATA SOURCES FOR EDIBLE FRESHWATER BIOTA

TABLE AIII.1. INVENTORY OF LITERATURE DATA COMPILED TO REPRESENT BIOACCUMULATION FACTORS (BAFS) FOR VARIOUS TYPES OF EDIBLE BIOTA AND THEIR TISSUES

	Primary Producers				*Invertebrates*					*Fishes*															*Herpetofauna*				
Element	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Fishes (Generic, muscle)	Fishes (Generic, whole)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
Ag					1										16	1			11	37			4						70
Al					1										20	4			16	38			13	53					145
Am	3	1		3	2														1										10
^{241}Am	4		1	4	7														1										17
As					1										4	1			11	38			3						58
Au					1										4	1			11	30			4						51
B																													0
Ba					1										27	4			9	14			79	78					212
Be																													0
Br					1										1	1			11	38			4						56
^{14}C	3		1	6	15				1																				26
Ca					1										26	4			16	37			62	78	3	2	2	9	240
Cd	4		1		121																								126
Ce					1										14	4			1	9			56	77					162
Cl					1										1	1			11	38			4						56
Cm	1				1																								2

	Primary Producers				*Invertebrates*					*Fishes*															*Herpetofauna*				
Element	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Fishes (Generic, muscle)	Fishes (Generic, whole)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
Co	6	2	1		4										15	4			11	38			39	78	3	2	2	9	214
^{60}Co	2	1	3	3	16				2						1	2							1	2					33
Cr					1										9	3			1	5			47	43					109
Cs	1	2			5										17	4			12	43			63	79	3	2	2	9	242
Cs Radioisotopes	1	1	1	1	1				1																				6
^{137}Cs	3		1	15	14											8				15			16	13					85
Cu	4		1		81										15	4			10	31			61	77					284
Dy																			2	2									4
Eu					1										7	1			11	34			11	30					95
Fe	4		1		1										15	4			10	38			61	78					212
Hf					1										1	1			10	37			4						54
Hg					25										1	1			11	36			4						78
I		2			3										10	1			3	35			32	59					145
^{131}I		1			2	16	19	16	20														10						84
K					1										15	4			11	38			61	78	3	2	2	9	224
La					1										15	4			14	30			46	78					188
Lu					1																								1
Mg															15	4			11	36			61	78	3	2	2	9	221
Mn	4	1	1		1										15	4			11	38			61	78					214

Element	Primary Producers				Invertebrates					Fishes															Herpetofauna				Total
	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Fishes (Generic, muscle)	Fishes (Generic, whole)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	
Mo					1										14	3							42	78					138
^{99}Mo						8	8	8	7														8						39
Na					1										15	3			11	38			61	3	3	2	2	9	148
Ni	4		1												1	1							4	24					35
Np	1	1			1																								3
P															14								30						44
Pb	4		1		74										5	3							26	77					190
^{210}Pb											9	2																	11
^{210}Po											6	2																	8
Pu	16	8	2	14	50														3										93
^{239}Pu					3																								3
Ra	4	4			5						2																		15
^{226}Ra											22	2																	24
Rb					1										15	4			11	38			60	78					207
Ru																													0
^{106}Ru	7	1			8																								16
Sb					1										6	4			10	38			8	10					77
Sc					1										1	1			10	37			4						54
Se	5	3	11	12	4										1	1			10	38			4						89
Sm					1																								1
Sn																													0

Element	*Primary Producers*				*Invertebrates*					*Fishes*																*Herpetofauna*				
	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Fishes (Generic, muscle)	Fishes (Generic, whole)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
Sr	3	2			4										15	4			11	38			62	78					217
90Sr			3	9																									12
Tb															1	1			9	35			4						50
Tc	3				3				2																				8
^{95m}Tc	3			2																									5
Te	3																						3	10					16
Th					1						1	1				1													4
Ti															1	1			9	34			4						49
Tl															14	3							48	78					143
U	1	3			8						12	2																	26
V					1										15	4			11	33			61	78					203
Y															11	3							12	10					36
Yb																													0
Zn	4		1		81										15	4			11	38			61	78					293
Zr															5								7	10					22
Total:	98	33	31	69	563	24	27	24	33	0	52	9	0	0	413	111	0	0	333	1102	0	0	1316	1669	18	12	12	54	6003

TABLE AIII.2. INVENTORY OF LITERATURE DATA COMPILED TO REPRESENT BIOTA-SEDIMENT ACCUMULATION FACTORS (BSAFS) FOR VARIOUS TYPES OF EDIBLE BIOTA AND THEIR TISSUES

	Primary Producers				*Invertebrates*					*Fishes*													*Herpetofauna*				
Element	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
Ag						1	7		32			35		6		1				7		2					91
Al							13		123			30	8	45		6		33		5	7	35					305
Am																											0
^{241}Am																											0
As						2	14	1	121			43	5	157		5	4	37		8	4	32					433
Au																											0
B									1																		1
Ba							12		125			27	1	95		9		2		3	2	6					282
Be									1					1				1		1							4
Br																											0
^{14}C																											0
Ca													1	1				1			1						40
Cd						1	7	1	106			52	15	55		10	2	28		10	3	51					341
Ce																											0
Cl																											0
Cm																											0
Co							12		124			29		24		9		3		11		4					216
^{60}Co																											0

	Primary Producers				Invertebrates					Fishes													Herpetofauna				
Element	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
Cr																											0
Cs																											0
Cs																											0
^{137}Cs																											0
Cu						2	16	2	129			62	84	370		10	44	181		12	78	169					1159
Dy																											0
Eu																											0
Fe							12		128			56	2	42		10		21		12	1	8					292
Hf																											0
Hg							12	2	95		1	44	79	211		7	11	48		13	38	94					655
I																											0
^{131}I																											0
K																											0
La																											0
Lu																											0
Mg													1	3				1			1	3					9
Mn																											0
Mo							1		14			1				1				4							21
^{99}Mo																											0
Na														1							1	1					3
Ni						1	15		115			28	4	60		7	4	31		10	6	47					328
Np																											0

	Primary Producers				Invertebrates					Fishes													Herpetofauna				
Element	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
P																											0
Pb						2	8	2	63			15	8	162		2	4	109		4	8	94					481
Pb-210																											0
Po-210																											0
Pu																											0
Pu-239																											0
Ra																											0
Ra-226																											0
Rb																											0
Ru																											0
Ru-106																											0
Sb							1							7				2		2							12
Sc																											0
Se							4		99			53	8	33		9	1	8		11	7	20					253
Sm																											0
Sn														1								1					2
Sr							12		123			52		29		8		1		11		5					241
Sr-90																											0

Element	Primary Producers				Invertebrates					Fishes													Herpetofauna				
	Algae	Macrophytes (Generic)	Macrophytes (Floating-leafed)	Macrophytes (Emergent)	Freshwater Invertebrates	Crustaceans (muscle)	Crustaceans (soft tissue)	Molluscs (muscle)	Molluscs (soft tissue)	Fishes (Generic)	Benthic Fishes (eggs)	Benthic Fishes (liver)	Benthic Fishes (muscle)	Benthic Fishes (whole)	Forage Fishes (eggs)	Forage Fishes (liver)	Forage Fishes (muscle)	Forage Fishes (whole)	Piscivorous Fishes (eggs)	Piscivorous Fishes (liver)	Piscivorous Fishes (muscle)	Piscivorous Fishes (whole)	Tadpoles	Frog (muscle)	Frog (whole)	Reptiles	Total
Tb																											0
Tc																											0
Tc-95m																											0
Te																											0
Th																											0
Ti																		1									1
Tl							1						1	6				5				2					15
U							3		3																		6
V							12		54			36		26		9				6		2					145
Y																											0
Yb																											0
Zn						2	16	2	131			57	57	342		10	15	177		13	106	184					1112
Zr																											0
Total:	0	0	0	0	0	11	178	10	1587	0	1	620	274	1677	0	113	85	690	0	143	263	760	0	0	0	0	6412

SUMMARY OF AVAILABLE ORGAN-TO-REFERENCE ORGAN CONCENTRATION RATIO DATA FOR EDIBLE FRESHWATER BIOTA TISSUES.

TABLE AIII.3. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF AQUATIC MACROPHYTES

Element	Whole	Shoots	Leaves	Stems	Roots	Fruits	Epiphytes
Al							
Ba							
Ca							
Cd			4	4	4		
Ce							
Co		14			14		14
Cr		2			2		
Cs		7	3	2	9	4	7
Cu		2			2		
Eu		7			7		7
Fe							
K							
La							
Mb							
Mg							
Mn	3	14			14		17
Na							
Ni							
P							
Pb		2	4	4	6		
Po		1			1		
Pu							
Rb							
Ru							
Sb							
Sr							
Te							
Th							
Tl							
U							
V							
Y							
Zn							
Zr							

TABLE AIII.4. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF FRESHWATER MOLLUSCS

Element	Whole	Shell	Soft Tissue	Viscera	Muscle	Foot	Gills	Mantle
Al								
Ba								
Ca								
Cd	1			1	2	3	3	3
Ce								
Co								
Cr								
Cs								
Cu					2	2	2	2
Eu								
Fe					2	2	2	2
Hg	1			1	2	3	3	3
I		1	1					
K								
La								
Mb								
Mg								
Mn	1	2	2		2	2	2	2
Mo								
Na								
Ni								
P								
Pb					2	2	2	2
Po		2	2					
Pu								
Rb								
Ru								
Sb								
Sr		1	1					
Te								
Th								
Tl								
U								
V								
Y								
Zn					2	2	2	2
Zr								

TABLE AIII.5. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF FRESHWATER DECAPODS

Element	Whole	Exoskeleton	Muscle	Hepatopancreas	Viscera
Al					
Ba					
Ca					
Cd		2	2	2	2
Ce					
Co					
Cr					
Cs					
Cu		2	2	2	2
Eu					
Fe					
K					
La					
Mb					
Mg		2	2	2	2
Mn		2	2	2	2
Mo					
Na					
Ni		1	1	1	1
P					
Pb					
Po		2	2		
Pu					
Rb					
Ru					
Sb					
Sr					
Te					
Th					
Tl					
U					
V					
Y					
Zn		2	2	2	2
Zr					

TABLE AIII.6. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF AMPHIBIANS

Element	[a] Carcass	Muscle	Liver	Kidney	Gonads	Blood	Skin	Bone
Al	1		1					
Ba	1		1					
Ca								
Cd	1		1					
Ce								
Co	1		1					
Cr	1		1					
Cs	2	1	1					
Cu	1		1					
Eu								
Fe								
K								
La								
Mb								
Mg								
Mn	1		1					
Mo	1		1					
Na								
Ni	1		1					
P								
Pb	1	3	4	5			3	5
Pu								
Rb	1		1					
Ru								
Sb								
Sr	1		1					
Te								
Th								
Tl								
U								
V								
Y								
Zn	1		1					
Zr								

[a] One of the Cs values for carcass represents a whole body measurement.

TABLE AIII.7. GAP ANALYSIS DEPICTING THE NUMBER CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF FRESHWATER REPTILES

Element	Muscle	Liver	Kidney	Gonads	Heart	Blood	Skin	Bone
Ac	1	1						
Al								
Ba								
Ca								
Cd								
Ce								
Co								
Cr								
Cs								
Cu								
Eu								
Fe								
K								
La								
Mb								
Mg								
Mn								
Mo								
Na								
Ni								
P								
Pb	2	1			1			1
Po	2	1			1			1
Pu								
Po								
Ra	2	1			1			1
Ru								
Sb								
Sr								
Te								
Th	3	3						1
Tl								
U	4	4						2
V								
Y								
Zn								
Zr								

TABLE AIII.8. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF FRESHWATER FISHES

Element	Whole	Muscle	Carcass	^a Bone	Gills	Skin	Scales	Blood	Blood Serum	Air Bladder	Liver	Kidney	Gonad	Fat	Eye	Pancreas	Brain	Spleen	Heart	Thyroid	Gall Bladder	Epidermal Mucous	Head	Fin	Viscera
Ag																									
Al			4		4																				
As	1	1		1							1														
Ba																									
Ca		17		9		8						8				8									
Cd		33	4		19	9					32	13	22	2											
Ce																									
Co	4	11		8	4	3		2			5	5						1	1				2		2
Cr	1	1		1	1						1	1													
Cs																									
Cu		34		3	16	19	1	2	1	26	23		15	3	3	8	2	2	1						
Eu																									
Fe	1	24		5	5	8	4				16	15	9	2		8									
Hg		11			4						11	4	3												
I		3											5							3					
K																									
La	1	1		1	1						1	1													
Mb																									
Mg		8				8						8				8									
Mn	3	28	2	4	19	17	4				19	25	11	2	2	8						2	4	4	6
Mo																									
Na		1		1		1	1	1			1	1	1	1	2		1	1	1						
Ni		11			4						11	2	2	2											
P		1		1		1	1	1			1	1	1	1	2		1	1	1						
Pb	14	77	4	11	52	16					67	22	37	2	5		9	9							

Element	Whole	Muscle	Carcass	[a] Bone	Gills	Skin	Scales	Blood	Blood Serum	Air Bladder	Liver	Kidney	Gonad	Fat	Eye	Pancreas	Brain	Spleen	Heart	Thyroid	Gall Bladder	Epidermal Mucous	Head	Fin	Viscera
Po		6		6																					
Ra	10	22		17		7					5														
Rb			1	1																					
Ru	3	3	1	1	1													1							
Sc	1	2		2	1	1	1	1			2	2	1	1	2		1	1	1						
Sr	2	10		8	1	1	1	1		1	1	1	1					1	1		1			1	
Te																									
Th		2									2														
Tl																									
U	7	15		12		7					5														
V																									
Y																									
Zn	1	48	4	8	21	19	5	2	1		39	28	22	3	3	8	2	2	1				1		
Zr		1		1	1																				1

[a] One bone value each for Cu and Zn represents skull and one of each represents vertebra measurements. One Sr value in bone represents a vertebra measurement.

TABLE AIII.9. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF EELS

Element	Whole	Muscle	Bone	Skin	Blood	Liver	Kidney	Gonad	Gill	Brain	Heart
Ag											
Al											
Ba											
Ca		1	1								
Cd	1	4		1		4	3		4	3	3
Ce											
Co											
Cr											
Cs											
Cu		1				1	1		1	1	1
Eu											
Fe											
Hg	1	1		1		1			1		
K											
La											
Mb											
Mg											
Mn											
Mo											
Na											
Ni											
P											
Pb											
Pu											
Rb											
Ru											
Sb											
Sr		1	1								
Te											
Th											
Tl											
U											
V											
Y											
Zn		1				1	1		1	1	1
Zr											

TABLE AIII.10. GAP ANALYSIS DEPICTING THE NUMBER OF CONCENTRATION RATIO VALUES AVAILABLE FOR STABLE ELEMENTS IN VARIOUS ORGANS OF MIGRATORY BIRDS

Element	Whole	Muscle	Bone	Feathers	Blood	Liver	Kidney	Gonad
Ag								
Al								
Ba								
Ca								
Cd								
Ce								
Co								
Cr								
Cs		9	10			3		
Cu								
Eu								
Fe								
Hg								
K								
La								
Mb								
Mg								
Mn								
Mo								
Na								
Ni								
P								
Pb			11			11		
Pu								
Rb								
Ru								
Sb								
Sr								
Te								
Th								
Tl								
U								
V								
Y								
Zn								
Zr								

MISCELLANEOUS TOPICS

SPECIFIC ACTIVITY MODELS AND PARAMETER VALUES FOR TRITIUM, ^{14}C AND ^{36}Cl

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Abstract

Tritium, ^{14}C and ^{36}Cl are long-lived isotopes of biologically-regulated, essential elements that are highly mobile in the environment. Under equilibrium conditions, specific activity (SA) models can be used to describe the environmental transfer of these isotopes based on the behaviour of the stable elements hydrogen, carbon and chlorine, respectively. SA models for the transfer of tritiated water and ^{14}C through the environment following release to air and water, and for transfer from soils contaminated via irrigation or releases from subsurface waste management areas, are described in this paper. Models are also presented for the environmental transfer of tritiated hydrogen gas following release to air, and for the transfer of ^{36}Cl to animal products. All of the tritium models account for the formation of organically bound tritium in plants and animals. Values of the parameters required by the models are listed in all cases. Areas in which further knowledge is needed to improve the confidence in the predictions of the models are identified.

1. INTRODUCTION

The models described in the previous papers are largely based on trace element partitioning and accumulation concepts, which are expressed quantitatively in terms of transfer factors that describe the transport of radionuclides between different environmental compartments. Under equilibrium conditions, the specific activity model provides an alternative approach for long-lived isotopes of biologically-regulated, essential elements that are highly mobile in the environment.

The behaviour of such an isotope mimics that of the stable form of the element in physical and biological processes. The flux of the isotope from one compartment to another is determined by the flux of the essential element (except in the case of diffusion, where the stable and active forms follow their respective vapour pressure gradients). Moreover, feedback mechanisms such as homeostasis act to control the flux of the essential element and hence, indirectly, the flux of the isotope. Models for these isotopes can be formulated in terms of specific activity concepts, where specific activity (SA) is defined as the radionuclide activity per mass of the stable element. SA models are used here for tritium, ^{14}C and ^{36}Cl (for plant to animal transfer only in the case of ^{36}Cl), based on the environmental behaviour of the stable elements hydrogen, carbon and chlorine, respectively. In principle, the SA model can be applied to ^{129}I as well, but this has not been common practice.

In the simplest application of the SA model, the radioisotope mixes physically and chemically with its corresponding stable element within some compartment of the environment, resulting in a certain specific activity. Any organism drawing the stable element from this compartment draws the radioisotope in proportion, and attains the same SA as the source compartment. The concentration of the isotope in the organism can be readily calculated from the concentration of the stable element, which is usually well known. In practice, the application of SA models requires a good understanding and precise definition of the environmental compartments within which the radionuclide of interest can be mixed, and of the fluxes that occur between those compartments. Mathematically, the specific activity (SA_J) in a compartment J that draws an element S from N other compartments is given by:

$$SA_J = \frac{A_J}{S_J} = \frac{\sum_{i=1}^{N} q_i \cdot D_{Fi} \cdot A_i}{\sum_{i=1}^{N} q_i \cdot S_i} \qquad (1)$$

Here A_i is the concentration of the active form of the element in compartment i (Bq kg^{-1}), S_i is the concentration of the stable form (g kg^{-1}), and q_i is the flux of the element from compartment i to compartment J (g d^{-1}). D_F is an isotopic discrimination factor that is introduced to allow for the possibility that the stable and active forms may have significantly different masses and therefore different transfer rates between compartments. The concentration of the radioisotope in compartment j is found by multiplying SA_J by the concentration of the stable element in the compartment. Isotopic exchange with relatively uncontaminated pools of the stable element results in progressive dilution of the isotope with distance from the source.

Most parameter values in specific activity models show relatively little variability and their distributions are best described as normal. However, for consistency with the rest of this document, the means and variations of the parameters discussed below are presented in terms of their geometric means (GM) and geometric standard deviations (GSD). This is not necessarily meant to imply that these parameters are lognormally distributed, or that the usual statistical properties of lognormal distributions apply to them. In reality though, the geometric and arithmetic means are almost identical for most of the parameters, and the spread in values implied by the geometric and standard deviations are similar.

The models and parameter values considered here are applicable only to steady-state conditions. Dynamic releases can be incorporated into the SA approach by considering the ratios of active-to-stable fluxes between compartments rather than the ratios of active-to-stable concentrations in the compartments. However, such an approach is complex and the uncertainties in the predictions of dynamic models are large. For tritium in particular, the model parameters depend upon meteorological and plant variables that are highly scenario-specific. It is not clear how to supply generic parameter values for dynamic models of tritium, ^{14}C or ^{36}Cl, and the discussion below is restricted to steady-state situations. This includes conditions near a continuous source, where concentrations at a given location fluctuate regularly with changes in meteorological conditions, particularly wind direction. SA models give reliable results in such situations provided the predictions of interest are long-term averages.

2. TRITIUM

Following traditional usage, the SA model for tritium is formulated in terms of the tritium concentration in water rather than the ratio of tritium activity to the mass of hydrogen in a given compartment. There is a one-to-one correspondence between these two quantities and it is the concentration in water (the HTO concentration) that is normally measured. Tritium can also be incorporated into the organic matter of plants and animals as non-exchangeable organically bound tritium (OBT). This is the form of tritium that remains in dry biomatter that has been repeatedly washed with tritium-free water. Its concentration is calculated as the activity in the water equivalent of the dry matter (the water produced by complete combustion of the dry material). Models and parameter values are also given for releases of tritiated hydrogen gas (HT).

2.1 Release of HTO to air

2.1.1 Transfer from air to soil

Tritium is transferred from air to soil through wet and dry deposition from the airborne plume. Concentrations in soil water are lower than those in air moisture, partly because precipitation is less contaminated than air moisture and partly because soil water concentrations are diluted by uncontaminated precipitation that falls when the plume is not present. Here, the soil water concentration (C_{sw}, Bq L^{-1}) is assumed to be proportional to the concentration in air moisture (C_{am}, Bq L^{-1}), with a proportionality constant CR_s:

$$C_{sw} = CR_s \cdot C_{am} \tag{2}$$

CR_s is difficult to estimate since HTO concentrations in both soil water and air moisture must be measured often enough to provide reliable averages over the growing season. The values depend upon a number of local factors, notably the frequency with which rain falls when the airborne plume is present at the site of interest. The few available data are listed in Table 1.

On balance, a default value of 0.3 is reasonable, consistent with the recommendation of IAEA [1]. A value of 0.5 is likely to be conservative, although values as high as 1.0 are possible. The data in Table 1 suggest that southern or wetter regions may have higher values of CR_s. Values based on local measurements should be used wherever possible.

Equation (2) should not be applied close to an elevated source where air concentrations are low or zero because the plume has not diffused down to the ground. Under such conditions, Equation (2) would predict low or zero concentrations in soil water, whereas in reality the concentration could be high due to wet deposition when precipitation falls through the elevated plume. This is not a serious restriction in practice because the model is usually applied to members of the public who are located far enough from the source that the plume has descended to the ground.

TABLE 1. RATIO OF SOIL FREE WATER TO AIR MOISTURE CONCENTRATION (CR_s)

Reference	Country of Measurement	Min.	Max	Conditions of Measurement
[2]	Russian Federation	1.3×10^{-1}	3.4×10^{-1}	3 values for sandy soils, each based on annual average air concentration and a single measurement of soil concentration in September. Annual precipitation: 560 mm
[3]	France	8×10^{-2}	2×10^{-1}	2 values, each based on air and soil concentrations averaged over a 20-month period. Air concentrations were measured continuously and soil concentrations monthly. Total precipitation over the study period: 1072 mm
[4]	Canada	1.5×10^{-1}	3.1×10^{-1}	7 values for sandy soils, each based on air and soil concentrations averaged over the growing season. Air concentrations were measured continuously and soil concentrations daily or weekly. Annual precipitation: 880 mm
P.M. Ravi, BARC[1]	India	4×10^{-1}		Mean of 9 measurements in sandy loam soils covering rainy and winter seasons
Y. Inoue, NIRS[1]	Japan	7.6×10^{-1}		Based on weekly air and soil concentrations measured between mid-April and June. Air concentrations were weekly averages and soil concentrations were measured once per week.
All data		GM = 2.3×10^{-1}		GSD = 1.7

[1] Personal communication.

The HTO concentration in air moisture is calculated from the HTO concentration in air (C_{air} Bq m^{-3}) and the absolute humidity (H_a, kg m^{-3}):

$$C_{am} = \frac{C_{air}}{H_a} \tag{3}$$

The value of C_{air} is assumed known through measurement or modelling. The absolute humidity is commonly measured by national weather services, and site-specific values are usually readily available and preferred. In the absence of local data, default values are given in Table 2 for different climates [5]. If the relative humidity (RH) is known, H_a can be found from:

$$H_a = \frac{2.17 \cdot 10^{-3} \cdot e_s \cdot RH}{T_a} \tag{4}$$

where e_s is the saturation vapour pressure (Pa) and T_a is air temperature (K). The saturated vapour pressure is a function of temperature only and is available in a number of lookup tables [6].

TABLE 2 SELECTED DEFAULT VALUES OF THE RELATIVE AND ABSOLUTE HUMIDITY FOR DIFFERENT CLIMATES AVERAGED OVER THE GROWING SEASON [5]

Climate	Latitude	Absolute humidity (H_a) (kg m^{-3})	Relative Humidity (RH)
Mediterranean (Cordoba)	34	0.0115	0.6
Continental (Munich)	48	0.0087	0.71
Maritime (London)	50	0.0078	0.795
Arctic (Northern Finland)	70	0.0067	0.73

2.1.2 Transfer to plants

HTO: Plants take up tritium from both air and soil. The HTO concentration in the free water of the leaf (commonly referred to as the tissue free-water tritium (TFWT) concentration) is calculated using a steady-state model [7] that explicitly considers the contributions from these two sources. The model is based on theoretical considerations and controlled experiments [8-10], and is written as:

$$C_{TFWT} = [RH \cdot C_{am} + (1 - RH) \cdot C_{sw}] / \gamma \,, \qquad (5)$$

where C_{TFWT} is the HTO concentration in the leaf free water (Bq L^{-1}) and γ (= 0.909) is the ratio of the HTO vapour pressure to that of H_2O.

When the relative humidity is 1, the transpiration stream shuts down and Equation (5) predicts that no soil tritium is carried to the leaves. In this case, the tritium in the leaves is due entirely to transfer from the air via diffusion through the stomates, and the leaf concentration comes into equilibrium with the concentration in air moisture. As the humidity drops, transpiration brings low concentration water up to the leaves, diluting the input from the air. In the limit RH = 0 (a physically impossible situation) there is no HTO in the air and no transfer from air to leaf, and the plant concentration equals the concentration in the transpiration stream.

The relative contributions of air moisture and soil water to the tritium concentration in the plant depend on the part of the plant under consideration. The partitioning in Equation (5) in terms of the relative humidity applies specifically to plant leaves that, under normal circumstances, draw the majority of their tritium from the air. The equation does not apply to fruit, tubers or root crops, which draw a larger fraction of their tritium from the soil [11]. The concentration in tubers and root crops is close to that in soil water. The concentration in fruit water is intermediate between the concentration in leaves and soil water. However, the available data do not permit the fractional contributions of air and soil water to the tritium concentration in fruit, tubers or root crops to be quantified with any confidence. Thus Equation (5) is used for all plant parts, with the recognition that it will be conservative for all parts other than leaves, since concentrations in soil water are lower than those in air moisture for an atmospheric release.

Equation (5) is a simplification of a more general formula [7, 12] that considers the effect of leaf/air temperature differences, which are ignored in the simplified version. Equation (5) can also be derived as the steady-state solution of a more complex dynamic equation [13] that considers tritium fluxes between air, soil and plants using a resistance model. Equation (5) can be obtained from this model if it is assumed that the temperature of the air, soil surface and leaf are the same; that the mass of plant water is constant; and that the HTO concentration at the soil surface is equal to the concentration in root water. Equation (5) is in common use in the tritium modelling community [1] and requires a minimal amount of input data. Various validation exercises have demonstrated that, when it is used to predict average concentrations close to harvest, it provides realistic or slightly conservative estimates of the amount of HTO in the plant.

The averaging time of the parameters appearing in Equation (5) depends on the application. To predict the tritium concentration in leaf water at harvest, the input values should reflect conditions that prevail close to the harvest time, since turnover of tritium in plant leaves occurs on time scales of minutes to hours. In contrast, if Equation (5) is used as the first step in calculating OBT concentrations, the input values should be averaged over the growing season of the organs to be harvested. This is because the OBT concentration at harvest is roughly proportional to the mean TFWT concentration in leaves over the growing phase, assuming that growth is nearly linear and that most of the OBT, once formed, remains in the plant until harvest.

The *HTO* concentration in the fresh weight (FW) plant (Bq kg^{-1} FW) is given by

$$C_{pfw}^{HTO} = WC_p \cdot C_{TFWT} \tag{6}$$

where WC_p is the fractional water content of the plant. Water contents for a number of broad plant categories are listed in Table 3. These are the same categories defined in this TECDOC earlier, except that some groups have been combined (leafy with non-leafy vegetables, cereals with rice, and grass with fodder and pasture), and the categories for herbs and "other' plants are not considered. The dry matter contents reported in Appendix 1 for individual species have been synthesized, converted to water contents and combined with data from other sources to produce the values in Table 3. These values apply to the edible part of the plant as harvested. Some grasses are dried before use as animal feed, in which case their water contents are better represented by the value for cereals (12%). A value for silage is also provided since this is a common form of animal feed.

TABLE 3. DEFAULT WATER CONTENTS (WC_p, %) FOR PLANT CATEGORIES [14 -24]

Plant category	N	GM	GSD	Min.	Max
Leafy and non-leafy vegetables	88	92	1.03	84	97
Leguminous vegetables–seed	11	12	1.19	9.3	17
–vegetative mass	16	81	1.10	69	91.4
Root crops	39	87	1.05	76.9	95
Tubers	10	75	1.08	62.2	82.2
Fruit	102	85	1.06	73.4	96
Grass, Fodder, Pasture	33	76	1.07	67.1	90
Cereals (including rice)	22	12	1.17	10	16
Maize–sweet corn	4	71	1.05	68.1	76.4
–feed corn	11	16	1.46	10	25.2
Silage	13	66	1.15	55	82

OBT: Tritium is incorporated into the organic matter of plants during photosynthesis in the presence of light and through metabolic processes in the dark. Tritiated organic material occurs in two main forms: exchangeable and non-exchangeable. The former is in equilibrium with plant tritiated water and behaves as TFWT in all respects. The latter is more stable and has a longer residence time in plants and animals, making it a greater radiological risk. Organically bound tritium (OBT) is usually considered to refer to the non-exchangeable form, and that will be the case here. Since most dry matter is formed in the presence of water, the plant OBT concentration is similar to the TFWT concentration. However, due to the larger mass of tritium compared to hydrogen, isotopic fractionation results in an OBT specific activity that is lower than that for HTO [25]. Accordingly, the OBT concentration in water produced from the complete oxidation of organically bound hydrogen in plant tissues (Bq L^{-1}) is equal to the TFWT concentration in the leaf water modified by a partition factor R_p :

$$C_{pcw}^{OBT} = R_p \cdot C_{TFWT} \tag{7}$$

The partition factor accounts for the reduction in dry weight (DW) concentration due to the presence of exchangeable hydrogen in combustion water, as well as for isotopic discrimination.

Values of R_p must be determined empirically for steady-state conditions. Because TFWT and OBT have very different formation and clearance times in plants (a few hours and several weeks, respectively), and because tritium concentrations in air vary rapidly in time due to fluctuations in wind direction, it is difficult to obtain reliable estimates from field data when both TFWT and OBT concentrations are measured at a point in time. Dependable values can be obtained if the TFWT concentration in the plant is monitored continuously over the period of OBT formation, or if the plant is grown in an environment where the TFWT concentration does not change. However, these conditions are rarely met in practice, even in areas far from local tritium sources. Thus the most reliable estimates of R_p come from controlled laboratory experiments, where the plant is exposed to an HTO concentration that is held constant or monitored continuously. Table 4 summarizes literature values of OBT/HTO ratios obtained in controlled conditions [25-27]. The plants were typically grown from seed over a period of a few weeks in an enclosure in which the HTO concentration in air was held constant. Although the number of data points is small, all values are less than one, with a GM of 0.54 for the crops considered (maize, barley and alfalfa). In the absence of other information, this value is assumed to apply to all plant types. Regardless of the plant in question, the TFWT concentration used in Equation (7) should be the concentration in the plant leaves, the primary location of dry matter production.

TABLE 4. EMPIRICAL VALUES OF THE PARTITION FACTOR FOR PLANTS (R_p) OBTAINED UNDER CONTROLLED LABORATORY CONDITIONS

Plant type	R_p		Reference
	AM	SD	
Maize	5.1×10^{-1} [a]	3×10^{-2}	[26]
	6.6×10^{-1}	1×10^{-2}	[27]
Barley	4.2×10^{-1} [a]	1×10^{-2}	[26]
	5.5×10^{-1}	1×10^{-2}	[27]
Alfalfa	5.8×10^{-1} [a]	2×10^{-2}	[25]
GM (GSD)	5.4×10^{-1} (1.16)		

[a] The concentrations measured in these experiments as 'OBT' consisted of both exchangeable and non-exchangeable organically bound tritium. To eliminate the contribution of exchangeable OBT, the OBT/HTO ratios reported in the publications were corrected by subtracting 0.22, which represents the proportion of exchangeable hydrogen in the total hydrogen of the samples [28].

The OBT concentration in the fresh weight plant is given by:

$$C_{pfw}^{OBT} = (1 - WC_p) \cdot WEQ_{p} \cdot R_p \cdot C_{TFWT}\ , \tag{8}$$

where WEQ_p is the water equivalent factor (kg of water produced per kg dry matter combusted). The water equivalent factor is difficult to measure but can be calculated reliably from the hydrogen content of protein, fat and carbohydrate (7%, 12% and 6.2%, respectively) and the fractions of protein, fat and carbohydrate in the dry matter of the plant in question. The calculated values, which are shown in Table 5, vary little among the various plant categories.

TABLE 5 DEFAULT WATER EQUIVALENT FACTORS (WEQ_p, g water g^{-1} DW) FOR THE VARIOUS PLANT CATEGORIES (calculated from data in [17, 20, 24])

Plant category	N	GM	GSD	Min.	Max
Leafy vegetables	10	5.1×10^{-1}	1.05	4.7×10^{-1}	5.5×10^{-1}
Non-leafy vegetables	12	5.3×10^{-1}	1.03	5.0×10^{-1}	5.5×10^{-1}
Root crops	11	5.2×10^{-1}	1.06	4.5×10^{-1}	5.5×10^{-1}
All others	91	5.6×10^{-1}	1.04	5.0×10^{-1}	6.0×10^{-1}

2.1.3. Transfer to terrestrial animal products

Animals can ingest tritium as HTO in feed and drinking water and as OBT in the organic fraction of feed. Inhalation and skin absorption are also possible routes of HTO intake. Exchangeable organic tritium and HTO rapidly equilibrate with body water. Most of the HTO taken in by an animal remains as HTO in the body, with a small fraction converted to OBT. In contrast, about half the OBT taken in is converted to HTO, with the other half remaining in organic form. Animal concentrations are generally lower than concentrations in air or plants because their tritium inventory is diluted by drinking water, which has a concentration much lower than that of air moisture for most atmospheric releases.

Here, concentrations in animal products are based on a metabolic model [29, 30] that relates the concentrations in animal products to the concentration in feed and drinking water. The model explicitly takes into account transfers from HTO in the diet to HTO and OBT in the product, and from OBT in the diet to HTO and OBT in the product. It is expressed in terms of transfer coefficients; the equilibrium activity concentrations of HTO and OBT in fresh weight animal products are given by

$$C_{afw}^{HTO} = F_{HH} \cdot I^{HTO} + F_{OH} \cdot I^{OBT} \tag{9}$$

$$C_{afw}^{OBT} = F_{HO} \cdot I^{HTO} + F_{OO} \cdot I^{OBT} \tag{10}$$

Here F_{HH} is the transfer coefficient from HTO in diet to HTO in animal product (d kg^{-1} FW); F_{HO} is the transfer coefficient from HTO in diet to OBT in animal product (d kg^{-1} FW); F_{OH} is the transfer coefficient from OBT in diet to HTO in animal product (d kg^{-1} FW); F_{OO} is the transfer coefficient from OBT in diet to OBT in animal product (d kg^{-1} FW); I^{HTO} is the daily intake of HTO (Bq d^{-1}); and I^{OBT} is the daily intake of OBT (Bq d^{-1}).

Expressions for the transfer coefficients were derived from a SA approach based on the metabolism of hydrogen in the body:

$$F_{HH} = \frac{V_{tw}}{I_w} \tag{11}$$

$$F_{OH} = \frac{v_{tw} \cdot F_D}{I_w} = F_{HH} \cdot F_D \tag{12}$$

$$F_{HO} = \frac{SAR \cdot m_{ot}}{0.111\ I_w} \tag{13}$$

$$F_{OO} = \frac{m_{ot} - 0.111\ F_{HO} \cdot I_w}{I_{dm} \cdot C_{oh}} \tag{14}$$

In these equations, v_{tw} is the fraction of tissue or organ, *t*, composed of water; F_D is the dry matter diet digestibility; m_{ot} is the mass of organically bound hydrogen in 1 kg of tissue (kg H kg^{-1} FW); I_w is the total water intake (L d^{-1}); I_{dm} is the total dry matter intake (kg DW d^{-1}); SAR is the ratio of the specific activity of OBT in the animal product to the specific activity of HTO in the body water; C_{oh} is the concentration of organic hydrogen in the animal diet (kg H kg^{-1} DW); and the constant 0.111 is the mass of hydrogen in water (kg H kg^{-1}).

The total water intake (I_w) includes drinking water, water from food, water from the metabolism of the dry matter in food, inhalation and skin absorption. Similarly, the total dry matter intake (I_{dm}) includes both contaminated and uncontaminated feed.

The output of the model is expressed as the ratio CR_a of the concentration in the animal product to the concentration in the feed, drinking water and inhaled air. Separate ratios are determined for HTO and OBT intakes. The total tritium concentrations (HTO plus OBT) in the animal product are given by:

$$C_{afw}^{T\_HTO} = CR_a^{HTO} \cdot C_f^{HTO} \tag{15}$$

and

$$C_{afw}^{T\_OBT} = CR_a^{OBT} \cdot C_f^{OBT} \tag{16}$$

where $C_{afw}^{T\_HTO}$ is the total tritium concentration in the animal product from HTO intake (Bq kg^{-1} FW), CR_a^{HTO} is the concentration ratio for HTO intake ((Bq kg^{-1} FW)/(Bq L^{-1})), C_f^{HTO} is the average HTO concentration in ingested water (Bq L^{-1}), $C_{afw}^{T\_OBT}$ is the total tritium concentration in the animal product from OBT intake (Bq kg^{-1} FW), CR_a^{OBT} is the concentration ratio for OBT intake ((Bq kg^{-1} FW)/(Bq kg^{-1} DW)), and C_f^{OBT} is the average OBT concentration in feed (Bq kg^{-1} DW).

C_f^{HTO} is the sum of the HTO concentrations in the water taken in with feed, drinking water and respiration (including skin absorption), weighted by the fractional contribution of each of these sources to the total water intake. Generally speaking, inhalation contributes about 2-5% of the total water intake of the animal and metabolic water about 10%. The fraction of water coming from the diet varies among practices and must be user defined. C_f^{OBT} is a weighted average that includes uncontaminated as well as contaminated feed since local sources supply only a fraction of the total animal feed in modern industrial farming.

By definition:

$$CR_a^{HTO} = (F_{HH} + F_{HO}) \cdot I_w \text{ and} \tag{17}$$

$$CR_a^{OBT} = (F_{OH} + F_{OO}) \cdot I_{dm} \tag{18}$$

Substituting Equations (11) to (14) into (17) and (18) yields the following expressions for the concentration ratios:

$$CR_a^{HTO} = \frac{v_{tw} + SAR \cdot m_{ot}}{0.111} \tag{19}$$

$$CR_a^{OBT} = \frac{v_{tw} \cdot FD \cdot I_{dm}}{I_w} + m_{ot} \cdot \frac{1 - SAR}{C_{oh}} \tag{20}$$

For the most part, values for all the parameters required by the model are readily available and can be selected to reflect a particular breed, mass, diet composition, production rate or agricultural practice. Different climates can be taken into account by adjusting water intakes, which are influenced by air temperature; similarly the activity level of an animal can be accounted for by adjusting feed intakes.

Representative results for a number of animal products for temperate climates are shown in Tables 6 and 7. For a given product, the central value of the concentration ratio pertains to the specific mass, production rate and intake rate shown in the table. The ranges were derived by considering the variability in animal mass, production level and diet under temperate climate conditions. The ratios show much less variability across a broad range of conditions than the corresponding transfer coefficients. This is a major advantage of the concentration ratio approach. Larger values are conservative and should be used for animals that are raised in cold climates or have high fat contents in their products.

To implement the model in practice, Equations (15) and (16) are used to find the total tritium concentration in the animal product by multiplying the average HTO or OBT concentration in the ingested water or feed by the appropriate value of CR_a from Table 6 or 7 (or from Equation (19) or (20)).

The OBT concentration in the animal product can be split out by multiplying the total concentration by f_{OBT} from Tables 6 and 7, where f_{OBT} is the fraction of the total tritium in the animal product in the form of OBT; the HTO concentration is found by multiplying the total concentration by (1-f_{OBT}). In the model, f_{OBT} was calculated as $F_{HO}/(F_{HH} + F_{HO})$ for HTO intakes and $F_{OO}/(F_{OH} + F_{OO})$ for OBT intakes.

2.2 Contaminated soils

2.2.1 Waste management areas

Agricultural soils can become contaminated with tritium due to the upward migration of water from subsurface waste management areas. In this case, the SA model for transfer to air, plants and animals is driven by the HTO concentration in soil water (C_{sw}, Bq L^{-1}), which is assumed to be known.

TABLE 6. CONCENTRATION RATIOS FOR HTO INTAKE (CR_a^{HTO})

Product	Animal mass (kg)	Intake rate (kg DW d^{-1})	Production rate (kg d^{-1} or L d^{-1})	Fraction OBT (f_{OBT})	CR_a^{HTO}, (Bq kg^{-1} FW product per Bq L^{-1} intake)		
					Value	Min.	Max.
Cow milk	550	14	15	4×10^{-2}	8.7×10^{-1}	8.1×10^{-1}	9.2×10^{-1}
Sheep milk	50	1.8	1.3	6×10^{-2}	7.8×10^{-1}	7.6×10^{-1}	8.9×10^{-1}
Goat milk	50	2.5	2.5	4×10^{-2}	8.3×10^{-1}	8.1×10^{-1}	9.3×10^{-1}
Beef meat	500	9.3	7×10^{-1}	9×10^{-2}	6.6×10^{-1}	6.4×10^{-1}	8.2×10^{-1}
Veal meat	160	4.9	8×10^{-1}	8×10^{-2}	6.9×10^{-1}	6.4×10^{-1}	8.2×10^{-1}
Sheep meat	50	1.2	8×10^{-2}	1.2×10^{-1}	7.4×10^{-1}	6.7×10^{-1}	7.8×10^{-1}
Lamb meat	20	1.0	2×10^{-1}	1.2×10^{-1}	7.8×10^{-1}	6.0×10^{-1}	8.1×10^{-1}
Goat meat	50	1.2	8×10^{-2}	1.0×10^{-1}	6.7×10^{-1}	6.4×10^{-1}	8.1×10^{-1}
Pork meat	100	2.7	8×10^{-1}	1.5×10^{-1}	6.7×10^{-1}	6.1×10^{-1}	7.7×10^{-1}
Hen meat	2.5	1.2×10^{-1}	7×10^{-3}	1.0×10^{-1}	7.6×10^{-1}	7.0×10^{-1}	8.0×10^{-1}
Broiler meat	1.7	1.1×10^{-1}	3×10^{-2}	1.0×10^{-1}	7.6×10^{-1}	7.0×10^{-1}	9.0×10^{-1}
Egg	2.5	1.5×10^{-1}	4.5×10^{-2}	8×10^{-2}	6.6×10^{-1}	6.4×10^{-1}	8.1×10^{-1}

TABLE 7. CONCENTRATION RATIOS FOR OBT INTAKE (CR_a^{OBT})

Product	Animal mass (kg)	Intake rate (kg DW d^{-1})	Production rate (kg d^{-1} or L d^{-1})	Fraction OBT	CR_a^{OBT}, (Bq kg^{-1} FW product per Bq kg^{-1} DW intake)		
					Value	Min	Max
Cow milk	550	14	15	4.7×10^{-1}	2.4×10^{-1}	1.7×10^{-1}	3.0×10^{-1}
Sheep milk	50	1.8	1.3	5.7×10^{-1}	3.2×10^{-1}	2.3×10^{-1}	3.9×10^{-1}
Goat milk	50	2.5	2.5	4.0×10^{-1}	3.2×10^{-1}	2.5×10^{-1}	3.8×10^{-1}
Beef meat	500	9.3	7×10^{-1}	8.0×10^{-1}	4.0×10^{-1}	3.5×10^{-1}	5.3×10^{-1}
Veal meat	160	4.9	8×10^{-1}	7.2×10^{-1}	3.5×10^{-1}	3.1×10^{-1}	4.5×10^{-1}
Sheep meat	50	1.2	8×10^{-2}	7.5×10^{-1}	4.0×10^{-1}	3.5×10^{-1}	5.6×10^{-1}
Lamb meat	20	1.0	2×10^{-1}	7.8×10^{-1}	5.5×10^{-1}	4.1×10^{-1}	6.7×10^{-1}
Goat meat	50	1.2	8×10^{-2}	6.0×10^{-1}	4.3×10^{-1}	3.6×10^{-1}	4.5×10^{-1}
Pork meat	100	2.7	8×10^{-1}	7.4×10^{-1}	6.4×10^{-1}	4.5×10^{-1}	7.5×10^{-1}
Hen meat	2.5	1.2×10^{-1}	7×10^{-3}	5.5×10^{-1}	5.0×10^{-1}	4.2×10^{-1}	6.0×10^{-1}
Broiler meat	1.7	1.1×10^{-1}	3×10^{-2}	5.5×10^{-1}	5.0×10^{-1}	4.2×10^{-1}	7.0×10^{-1}
Egg	2.5	1.5×10^{-1}	4.5×10^{-2}	7.8×10^{-1}	6.4×10^{-1}	5.3×10^{-1}	6.8×10^{-1}

The lower levels of the atmosphere will become contaminated due to evapotranspiration of tritium from soil and plants. Yim and Caron [31] have shown that the tritium profile above a contaminated, vegetated surface remains roughly constant with height in the canopy. This suggests that the specific activity of air in the plant canopy is the same as in the soil water:

$$C_{am} = C_{sw} \tag{21}$$

The HTO concentration in plant water is then calculated using Equation (5), with $C_{air} = H_a \cdot C_{am}$ The OBT concentration in plants and the HTO and OBT concentrations in terrestrial animals following soil contamination are then calculated using the models and parameter values for release to air (see Sections 2.1.2 and 2.1.3).

2.2.2 Irrigation

Soils can also become contaminated through irrigation. From specific activity concepts, tritium reaching the soil with irrigation water is mixed into the total amount of water reaching the soil. Thus the concentration in soil pore water is given by

$$C_{sw} = \frac{Q_{irr} \cdot C_{irr}}{P + Q_{irr}} \tag{22}$$

where Q_{irr} is the irrigation rate (L m^{-2} d^{-1} averaged over the growing season), C_{irr} is the tritium concentration in irrigation water (Bq L^{-1}) (assumed known), and P is the precipitation rate (L m^{-2} d^{-1} averaged over the growing season).

From this point forward, the model follows that for contamination by upward migration from waste management areas. Air moisture concentrations are calculated using Equation (21) and TFWT concentrations using Equation (5). The OBT concentration in plants and the HTO and OBT concentrations in terrestrial animal products are calculated using the models and parameter values for release to air (see Sections 1.1.2 and 1.1.3). When calculating animal concentrations, only those crops that are irrigated should be considered contaminated. Moreover, the drinking water of the animals should be considered contaminated if it comes from the same source as the irrigation water.

Precipitation rate is commonly measured and a site-specific value is usually available. The irrigation rate depends strongly on the crop in question and the local climate and practice. Nominal values are shown in Table 8. Forage crops are not usually irrigated.

2.3. Release of HTO to water bodies

Fish are the only aquatic organisms considered here since they are the only aquatic organisms that play a major role in the human diet.

HTO: The assumption of full SA equilibrium is a good approximation for HTO concentrations in most aquatic compartments [38, 39]. Concentrations in the top few centimetres of sediment water are likely to be the same as those in the water column itself. Submerged plants, and the submerged parts of emergent plants, have the same concentration as the local water, whether they are rooted in the sediments or not. The HTO concentrations in fish and benthic organisms are the same as the concentration in the water that they access. Only the emergent parts of plants such as cattails diverge from the SA assumption, but these do not play a major role in the diets of aquatic animals. Thus, the water to which freshwater fish are exposed, including lake water and water derived from foods at different trophic levels, has more or less the same HTO concentration.

TABLE 8 IRRIGATION RATES (Q_{irr}) *(averaged over the growing season for various crops and climates)*

Country	Crop type	Irrigation rate (L m^{-2} d^{-1})		Reference
		Min	Max	
France (Loire Valley)	Garden vegetables	2	6	[32]
	Fruit	2	6	[32]
	Maize	0.8	2.8	[32]
	Grain	0	2.4	[32]
Republic of Korea	Rice	10		[33]
India	Rice	0.9		[34]
	Improved jowar	2.0		[34]
	Gram	1.8		[34]
	Wheat	4.3		[34]
	Ground nut	6.3		[34]
	Cotton	1.6		[34]
	Pigeon Pea	1.6		[34]
	Taurmeric	2.1		[34]
	Sugar cane	4.1		[34]
	Banana	2.6		[34]
Canada	Garden vegetables	0	1.8	[35]
	Forage	0		[35]
California*	Barley	2.4	3.2	[36]
	Oats, rye	2.0	2.5	[36]
	Alfalfa	4.9	7.1	[36]
	Hay	3.2	5.4	[36]
	Garden vegetables	4.2	5.2	[36]
	Fruit orchards	3.9	4.9	[36]
	Pasture	3.7	4.4	[36]
	Grapes	1.7		[37]

* Annual rates scaled to an nominal growing season of 180 days.

The assumption of full *SA* equilibrium implies that the HTO concentration in the fish (C_{ffw}^{HTO}, Bq kg^{-1} FW) is given by:

$$C_{ffw}^{HTO} = WC_f \cdot C_w \tag{23}$$

WC_f is the fractional water content of the fish, which is roughly constant at 0.78 for most fish that form part of the human diet [17]. C_w is the HTO concentration in water (Bq L^{-1}), which is assumed to be known through measurement or modelling.

OBT: The *SA* in fish soft tissues is determined by water use in anabolic processes, and direct incorporation of labelled small organic molecules derived from food. Because fish are immersed in an environment of uniform HTO concentration, it is reasonable to assume that the OBT concentration in the combustion water of the fish is the same as the HTO concentration, apart from a partition factor that takes account of the presence of exchangeable hydrogen in the combustion water and isotopic discrimination arising both in the fish and in different components of its food and water intakes. The OBT concentration in the fresh weight fish is given by:

$$C_{ffw}^{OBT} = (1 - WC_f) \cdot WEQ_f \cdot R_f \cdot C_w, \tag{24}$$

where WEQ_f is the water equivalent factor of the fish and R_f is the partition factor.

As was the case for the partition factor for plants, values of R_f must be determined empirically for steady state conditions. Because HTO and OBT have very different biological half-lives in fish, and because fish can encounter different tritium concentrations as they move through their water body, it is difficult to obtain reliable estimates from fish collected in the wild. Dependable values can be obtained if the fish lives its entire life in an environment where the HTO concentration does not change, or if the fish is exposed over a long period of time to contaminated food and water in a controlled setting. Table 9 summarizes literature values of OBT/HTO ratios obtained under such conditions. The GM of the observations is 0.66, which is the recommended default value for R_f.

The water equivalent factor is difficult to measure but can be calculated reliably from the hydrogen content of protein, fat and carbohydrate (7%, 12% and 6.2%, respectively) and the fractions of protein, fat and carbohydrate in the fish in question. The calculated values for four fish species are shown in Table 10. The GM of 0.65 is recommended for generic assessments.

TABLE 9 VALUES OF THE PARTITION FACTOR FOR FISH (R_f) OBTAINED UNDER EQUILIBRIUM CONDITIONS

Fish type	R_f	Reference
Bullhead (catfish)	7.7×10^{-1}	[38]
Pike	8.4×10^{-1}	[38]
Bass	1.3^{a}	[39]
Bluegill	$9.7 \times 10^{-1\,a}$	[40]
	$4.9 \times 10^{-1\,a}$	[41]
Largemouth bass	$4.1 \times 10^{-1\,a}$	[41]
Mosquito fish	$3.4 \times 10^{-1\,a}$	[41]
	$6.7 \times 10^{-1\,a}$	[42]
Carp	$6.0 \times 10^{-1\,a}$	[43]
GM (GSD)	$6{,}6 \times 10^{-1}$ (1.5)	

[a] The concentrations measured in these experiments as ‘OBT’ consisted of both exchangeable and non-exchangeable organically bound tritium. To eliminate the contribution of exchangeable OBT, the OBT/HTO ratios reported in the publications were corrected by subtracting 0.2, which represents the proportion of exchangeable hydrogen in the total hydrogen of the samples [44].

TABLE 10 DEFAULT WATER EQUIVALENT FACTORS ($WEQ_{f,}$) FOR VARIOUS FISH SPECIES *(calculated from data in [17])*

Fish	Water equivalent factor (g water g^{-1} DW)
Carp	7.1×10^{-1}
Perch	6.1×10^{-1}
Pike	6.5×10^{-1}
Trout	6.4×10^{-1}
GM (GSD)	6.5×10^{-1} (1.06)

Equation (24) and the values of R_f and WEQ_f in Tables 9 and 10 will give accurate estimates of OBT concentrations in fish only for cases in which the OBT is formed through metabolic processes involving HTO in the water. The model does not apply to situations in which dissolved tritiated organics are released directly to the water, as has been observed on a number of recent occasions [45]. Many aquatic species can selectively take up such material, in which case they exhibit OBT concentrations much higher than the HTO concentrations in the water. Models for such situations are outside the scope of the present document.

2.4 Release of tritiated hydrogen gas

Tritium can be released in the form of tritiated hydrogen gas (HT) as well as HTO. HT imparts a very low radiological dose relative to HTO because it is only weakly absorbed by the body. However, HT released to the atmosphere diffuses into soil pore spaces, where it is oxidized to HTO by microorganisms [46]. Some of the HTO so formed is taken up by plants through their roots with transpiration water and some is emitted to the atmosphere, where it is diluted by uncontaminated air. This HTO is available for uptake by plants, and by animals through inhalation and ingestion. Tritium concentrations in plants and animal products following an atmospheric HT release are determined entirely by the behaviour of the HTO formed in the soil.

The transfer of HT from air to HTO in plants as described above is a complex process and a detailed treatment cannot be given here. Instead, the HT model is based on data collected during a chronic HT release experiment carried out at Chalk River Laboratories (CRL) in 1994 [47-49]. The HTO concentration in plant water (C_{TFWT}, Bq L^{-1}) is given by:

$$C_{TFWT} = R_{HT} \cdot C_{air}^{HT} \qquad (25)$$

where R_{HT} is the ratio of HTO concentration in plant water to HT concentration in air (Bq L^{-1} plant HTO per Bq m^{-3} air HT) and C_{air}^{HT} is the concentration of HT in air (Bq m^{-3} air) (assumed known through measurement or modelling). R_{HT} is an empirical bulk parameter that accounts for HT oxidation and the subsequent re-emission and uptake of the HTO. The value found in the CRL experiment was 6 Bq L^{-1} HTO per Bq m^{-3} HT. However, the value could be different at sites where the soil properties that control HT deposition and oxidation (water content, porosity and distribution of microorganisms) are significantly different from those at CRL. Based on expert judgment, a value of R_{HT} = 12 Bq L^{-1} HTO per Bq m^{-3} HT is believed to be conservative in most cases and is recommended here. This value applies to all plant types since the data do not show any significant difference among leafy vegetables, non-leafy vegetables and root crops.

Once the TFWT concentration in plants is known following an HT release, the remainder of the HT model (OBT formation, transfer to animals) is the same as that for HTO (Sections 1.1.2 and 1.1.3). There is no direct transfer of HT to plants, animals or water bodies. Because HT is insoluble in water, there is no need to model releases to soil or aquatic systems.

3. CARBON-14

3.1 Release to air

3.1.1 Transfer to terrestrial plants

The assumption of full SA equilibrium throughout the terrestrial environment is completely satisfactory for ^{14}C releases to the atmosphere if, as is usual, the ^{14}C is emitted as $^{14}CO_2$. This is the only form that is readily taken up by plants, so that active carbon is incorporated into the plant dry matter via photosynthesis at the same rate as stable carbon. Moreover, the organic components of the soil are made up of decayed plant matter and so will reflect the SA ratio of the plants. Accordingly, the ^{14}C concentration in Bq/g stable carbon is the same in the plant as it is in air:

$$C_{pfw} = \frac{C_{air} \cdot S_p}{S_{air}}, \tag{26}$$

where C_{pfw} is the ^{14}C concentration in the plant (Bq kg^{-1} FW); S_p is the concentration of stable carbon in the plant (gC kg^{-1} FW), C_{air} is the concentration of ^{14}C in air (Bq m^{-3}), and S_{air} is the concentration of stable carbon in air (gC m^{-3}).

The only parameters required for this model are the stable carbon concentrations in air and in the plants of interest. S_{air} is presently about 0.20 g m^{-3}. Measured values of the carbon contents for the various plant categories are shown in Table 11. The data are augmented by values calculated from the carbon content of protein, fat and carbohydrate (52%, 77% and 42%, respectively) and the fractions of protein, fat and carbohydrate in the plant in question [17]. The plant carbon content varies strongly with plant category due largely to differences in plant water content. When the carbon contents are expressed on a dry weight basis (the last column of Table 11), the variability in S_p is greatly reduced.

TABLE 11 CONCENTRATION OF STABLE CARBON IN TERRESTRIAL PLANTS (S_p) (from [17, 20, 24, 50-52])

Plant category	Stable carbon (gC kg^{-1} FW)					Stable carbon (gC kg^{-1} DW), GM
	N	GM	GSD	Min.	Max.	
Leafy/non-leafy vegetables	49	3.0×10^1	1.4	1.8×10^1	6.5×10^1	3.8×10^2
Leguminous vegetables						
–seed	7	4.1×10^2	1.1	3.8×10^2	4.7×10^2	4.7×10^2
–vegetative mass	5	5.9×10^1	1.5	4.1×10^1	1.1×10^2	3.1×10^2
Root crops	23	4.6×10^1	1.5	2.2×10^1	9.5×10^1	3.5×10^2
Tubers	6	1.0×10^2	1.2	8.6×10^1	1.3×10^2	4.1×10^2
Fruit	48	6.2×10^1	1.3	3.1×10^1	1.0×10^2	4.1×10^2
Grass, Fodder, Pasture	25	1.0×10^2	1.3	4.0×10^1	1.6×10^2	4.2×10^2
Cereals (including rice)	29	3.9×10^2	1.1	3.6×10^2	4.3×10^2	4.4×10^2
Maize–sweet corn	3	1.2×10^2	1.0	1.2×10^2	1.2×10^2	4.1×10^2
–feed corn	1	3.8×10^2	-	-		4.6×10^2
Silage	13	1.3×10^2	1.4	6.5×10^1	1.8×10^2	3.8×10^2

2.1.2 Transfer to terrestrial animal products

Animals take up carbon almost entirely through ingestion and the *SA* ratio in plants is maintained in the animal. Therefore the ^{14}C concentration in animal products is given by:

$$C_{apf} = \frac{f_c \cdot C_{pfw} \cdot S_a}{S_p} \tag{27}$$

where C_{apf} is the ^{14}C concentration in the animal product (Bq kg^{-1} FW), *f* is the fraction of animal feed that is contaminated, and S_a is the concentration of stable carbon in the animal product (gC kg^{-1} FW). The factor *f* is introduced to allow for the fact that animals may be fed supplementary concentrates or feed from remote sources that is uncontaminated. If an animal is fed stored hay or silage, the concentration of relevance is that in the stored feed. The value of *f* should be set from a consideration of local farming practices; if a site-specific value is not available, *f* should be conservatively set to 1.

Carbon contents of various animal products are shown in Table 12. A few of these were directly measured [50, 52] but most were calculated from the carbon content of protein, fat and carbohydrate and the fractions of protein, fat and carbohydrate in the product in question [17].

TABLE 12 CONCENTRATION OF STABLE CARBON CONTENT IN TERRESTRIAL ANIMAL PRODUCTS (S_a, gC kg^{-1} FW)

Animal product	N	GM	GSD	Min.	Max.
Cow milk	8	6.5×10^1	1.03	6.2×10^1	6.9×10^1
Sheep milk	1	1.1×10^2	-	-	
Goat milk	1	7.1×10^1	-	-	
Beef meat	14	2.0×10^2	1.19	1.6×10^2	2.9×10^2
Veal meat	3	1.6×10^2	1.21	1.3×10^2	1.9×10^2
Sheep meat	-	2.9×10^2	-	-	
Lamb meat	2	2.8×10^2	1.26	2.3×10^2	3.2×10^2
Goat meat	1	1.7×10^2	-	-	
Pork meat	12	3.0×10^2	1.39	1.7×10^2	5.5×10^2
Hen meat	1	2.4×10^2	-	-	
Broiler meat	5	1.5×10^2	1.23	1.1×10^2	2.0×10^2
Egg	2	1.6×10^2	1.01	$1.\ 6 \times 10^2$	1.6×10^2

3.2 Contaminated soils

3.2.1 Irrigation

Soils can become contaminated with ^{14}C, as they can with tritium, by irrigation with contaminated water. C-14 in irrigation water enters the soil compartment and part is reemitted to the air by volatilization, where it is taken up by plants. Sheppard et al. [12.53] have developed a dynamic model for this pathway. Assuming that the irrigation rate is controlled so that there are no losses by leaching, a conservative, steady-state solution to the equations is:

$$SA_p = SA_{air} = CD_C \cdot \frac{Q_{irr} \cdot C_{irr}}{F_c} \tag{28}$$

where SA_p is the specific activity in the plant (Bq/gC); SA_{air} is the specific activity in the canopy air (Bq/gC); CD_C is the canopy dilution factor for ^{14}C (unitless); Q_{irr} is the average irrigation rate over the growing season (L m^{-2} d^{-1}); C_{irr} is the ^{14}C concentration in irrigation water (Bq L^{-1}); and F_C is the average production rate of carbon by decomposition of crop residues (gC m^{-2} d^{-1}).

The term $\frac{Q_{irr}C_{irr}}{F_C}$ in Equation (28) is the specific activity of ^{14}C in the soil, calculated as the ^{14}C flux to the soil with irrigation water divided by the stable carbon flux generated by the decomposition of soil organic matter, which consists mainly of crop residues. The SA in the air in the plant canopy is assumed equal to the SA in the soil, reduced by the canopy dilution factor to account for dilution with uncontaminated air from the free atmosphere. The SA in the plant is then set equal to the SA in the canopy air. The concentration in the fresh weight plant is given by Equation (26) (with $\frac{C_{air}}{S_{air}} = SA_{air}$), and the ^{14}C concentration in animal products by Equation (27).

Values for F_C, the production of carbon by decomposition of crop residues, can be calculated as the total biomass minus the harvested biomass using site-specific data. However, total biomass is not a readily available parameter; it includes root biomass, which is not easy to measure under realistic conditions. Petersen et al. [54] have suggested a central value of 0.66 gC m^{-2} d^{-1}, with a range from 0.37 to 0.94.

The canopy dilution factor, CD_C, indicates the degree to which canopy air is diluted by uncontaminated air from the free atmosphere. Its value depends on the area of contaminated soil and the crop height and density. Plants represent a sink for atmospheric ^{14}C, suggesting that the ^{14}C profile decreases with height in the canopy when the ^{14}C source is in the soil. Sheppard et al. [55] have suggested a value for CD_C of 0.15 for crops with an open canopy (forage) and 0.3 for crops with a closed canopy (garden crops), although values up to 1 are possible.

Equation (28) is conservative, particularly for soils that have a high pH or other properties that result in low volatilization rates. Under such conditions, most of the carbon is fixed in soil and is unavailable to plants. Moreover, Equation (28) assumes equilibrium conditions where none may exist, as the ^{14}C soil concentration due to irrigation will build up gradually over time.

3.2.2 Waste management areas

Soils can also become contaminated with ^{14}C by the upward migration of water from subsurface waste management areas. The chemistry of such a situation is complex and the form in which the ^{14}C might occur is not clear. Microbial degradation of the wastes leads to the production of $^{14}CO_2$ and radiomethane, which may be oxidized to $^{14}CO_2$ in the unsaturated zone and released to the soil gas. In addition, ^{14}C may escape from a repository in solution in groundwater and may also be converted to $^{14}CO_2$. All of these scenarios depend strongly on site-specific conditions, and their quantitative analysis is beyond the scope of this document. Instead, the present model is driven by the concentration of $^{14}CO_2$ in soil gas, which is assumed to be known regardless of how it originates. Part of the ^{14}C volatilizes from the soil to the atmosphere, where it is diluted with uncontaminated air and incorporated into plants by photosynthesis. The ^{14}C concentration in air is calculated using a model identical to that for tritium (Section 1.2.1), assuming that the specific activity of air in the plant canopy is the same as in the soil gas, modified to account for dilution with uncontaminated air:

$$SA_{air} = CD_C \cdot SA_{sg} \tag{29}$$

where SA_{air} is the ^{14}C specific activity in air (Bq/gC); CD_C is the canopy dilution factor for ^{14}C (unitless); and SA_{sg} is the ^{14}C specific activity in soil gas (Bq/gC) (assumed known).

The SA of the plant is the same as the SA of the air (Section 2.2.1). The concentration in the fresh weight plant (C_{pfw}, Bq kg^{-1} FW) is given by

$$C_{pfw} = SA_{air} \cdot S_p, \tag{30}$$

where S_p is the concentration of stable carbon in the plant (gC kg^{-1} FW). The ^{14}C concentration in animal products is then given by Equation (27).

An alternative approach is available if the ^{14}C is associated with soil solids. In this case, the flux of ^{14}C from the soil to the atmosphere is given by Yu et al. [56]:

$$F = C_s \cdot E_c \cdot \rho_b \cdot d_s \tag{31}$$

where F is the flux of ^{14}C from soil to air (Bq m^{-2} d^{-1}); C_s is the ^{14}C concentration in soil (Bq kg^{-1} DW) (assumed known); E_c is the ^{14}C evasion loss rate (d^{-1}); ρ_b is the soil dry bulk density (kg DW m^{-3}); and d_s is the soil depth (m).

The specific activity of the carbon evading the soil (SA_e, Bq/gC) is given by

$$SA_e = \frac{F}{F_C} \tag{32}$$

where F_C is the average production of stable carbon by decomposition of crop residues (Section 2.2.1). Although this is a production rate, it should equal the evasion rate under steady-state conditions. The specific activity in air is then calculated from Equation (29) with SA_{sg} replaced with SA_e. Concentrations in plants and animal products are found using Equations (30) and (27) respectively.

The parameters E_c and ρ_b depend on soil type. Nominal values are shown in Table 13 [57].

TABLE 13 DEFAULT VALUES FOR SOIL BULK DENSITY (ρ_b) AND ^{14}C EVASION RATE (E_c) [57]

Soil type	Evasion rate E_c (d^{-1})	Bulk density ρ_b (kg DW m^{-3})
Sand	0.060	1500
Loam	0.033	1300
Clay	0.033	1400
Organic	0.060	400

3.3 Transfer to fish

Modelling ^{14}C in aquatic systems is complicated by the existence of several carbon pools: dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), particulate inorganic carbon (PIC) and particulate organic carbon (POC). DIC and DOC are found primarily in the water column and PIC and POC in the sediments. Different aquatic organisms are exposed to different carbon pools and care must be taken to base the SA model on the pools that are seen by the organism of interest. DIC is the dominant carbon pool accessed by most aquatic organisms. DIC is the form of carbon taken up by most aquatic plants and animals that are important for human consumption. Moreover, DIC is the form of carbon normally released to aquatic systems, the most common form in the water column and the most easily measurable form of carbon. Thus it is reasonable to use DIC as the basis for the SA model for fish. This approach may underestimate concentrations in rooted macrophytes, filter feeders and benthic fish, which see particulate organic carbon in bottom sediments, but these organisms, play a small role in the human diet.

The assumption of full SA equilibrium based on DIC for aquatic ^{14}C releases implies that the ^{14}C concentration in Bq/g stable carbon is the same in fish as it is in dissolved inorganic carbon in the water column. The ^{14}C concentration in fresh weight fish (C_{ffw}, Bq kg^{-1} FW) is given by

$$C_{ffw} = C_{DIC} \cdot S_f \tag{33}$$

where C_{DIC} is the ^{14}C concentration in DIC in the water column (Bq/gC) and S_f (gC kg^{-1} FW) is the concentration of stable carbon in the fish.

As was the case for terrestrial animals, the carbon contents of fish are most reliably determined from the carbon content of protein, fat and carbohydrate and the fractions of protein, fat and carbohydrate in the fish [17]. The calculated values are shown in Table 14. A conservative default value of 150 gC kg^{-1} FW can be used for fish types not listed in Table 14.

TABLE 14 CONCENTRATION OF STABLE CARBON IN FISH (S_f) *(calculated from data in [17])*

Fish type	Stable carbon content (gC kg^{-1} FW)
Carp	1.6×10^2
Perch	1.0×10^2
Pike	1.0×10^2
Rainbow Trout	1.2×10^2
GM (GSD)	1.2×10^2 (1.2)

4. CHLORINE-36

4.1 Characteristics and speciation of chlorine-36

Long-lived ^{36}Cl (half-life = 3 × 10^5 years) is an important potential dose contributor to humans from the disposal of solid radioactive wastes. The main exposure pathways are ingestion of drinking water, milk and milk products, meat and vegetables [58].

Special attributes of ^{36}Cl are its long half-life and high mobility. ^{36}Cl is not significantly absorbed by the mineral part of the soil and its degree of transfer from soil to plants is extremely high. On the other hand, chlorine can be bound by soil organic matter. It is also a biologically essential element and the chloride form is subject to homeostatic regulation in the body.

4.2 Behaviour in soils

A key characteristic of ^{36}Cl is its speciation in soils. The assumption is that the predominant chemical species is anionic chloride (Cl^-) with high mobility in soils governed by water fluxes (leaching and transpiration). However, Oberg [59] concluded that native chlorine in soils can be converted into various organic forms due to microbial activity and has established biochemical cycling of chlorides. Lee et al. [60] showed complexation with organic soil constituents. It is evident that soil organic matter could play a slight retardation role, reducing the rate and extent of ^{36}Cl migration within the biosphere. Nevertheless, the distribution coefficient between water and soil is very low; Sheppard et al. [61] recommend a geometric mean equal to 0.1 L kg^{-1}. Values of the distribution coefficient for different soil types are given in the previous paper.

Results of a batch experiment show that chlorine is linked in soil only by live biota and passes quickly (within hours) from organic material to soil solution even without decomposition of the dead plants. Thus chlorine can be involved in migration processes in soil [62].

4.3 Uptake by plants

The literature data on ^{36}Cl transfer from soil to plants are fragmentary, but the data that do exist suggest that uptake rates vary by three orders of magnitude.

Kashparov et al. [62] showed that concentration ratios are not time- or soil-dependent, and that the specific activity of chlorine in plants is not the same as that in soil. This is because of rapid changes in the stable chlorine content of the root zone during the vegetation period due to uptake by plants, migration with the advective flux and injection of mineral fertilizers. A change of chlorine content in soil results in an analogous change in its content in plants, within the limited range of contents available to specific plants. Thus, transfer factors are more appropriate than specific activity for modelling soil to plant transfer [62]. On the basis of all experimental data on ^{36}Cl and stable chlorine transfer into plants from various soils, average concentration ratio (C_r) values for ^{36}Cl were estimated for the most important types of agricultural production [63].

Sheppard and Evenden [64] and Hara et al. [65] have shown very marked decreases in C_r with increasing substrate concentrations of ^{36}Cl, implying weakness of the C_r model if substrate concentrations of stable chlorine are not taken into account; moreover, it is obvious that chlorine concentrations in the soil vary through the seasons and even perhaps day to day.

Where the stable chlorine concentration in soil varies in the same way as the ^{36}Cl concentration, the SA concept can be used. The uptake of ^{36}Cl (and of stable chlorine) depends on the plant group. For example, halophyte species can take up a large quantity of chlorine.

4.4 Transfer to animals

Chlorine is essential to animals; it controls osmotic pressure and internal solution electroneutrality. Recent work on transfer to cow meat and milk has shown that the isotopic ratio in animal products is the same as in their foodstuffs [63]. Due to homeostatic control, the stable chlorine concentrations in animal tissues were not related to the amount ingested daily in herbage at intake rates in the normal physiological range of up to 188 g d^{-1} for cows and up to 40 g d^{-1} for pigs. The metabolism of stable chlorine and ^{36}Cl in an animal's body is identical. Therefore, the average equilibrium ratios of ^{36}Cl to stable chlorine in the daily ration and in the animal body will be the same. Thus it is concluded that the average equilibrium chlorine isotopic ratio in the dietary daily intake should be used to predict the contamination of meat and milk with ^{36}Cl.

4.5 Modelling

For these several reasons, it is more appropriate to use a specific activity model to describe the transfer of ^{36}Cl to animal products than the classical transfer factor approach, as long as the different inputs are well defined. Following Equation (1), the specific activity in the animal product is given by:

$$\frac{C_{animal}}{S_{animal}} = \frac{q_{water} \cdot C_{water} + q_{foodstuff} \cdot C_{foodstuff}}{q_{water} \cdot S_{water} + q_{foodstuff} \cdot S_{foodstuff}} \qquad (34)$$

where q is the intake rate (L d^{-1} or kg FW d^{-1}), C is the ^{36}Cl concentration (Bq L^{-1} or Bq kg^{-1} FW), and S is the stable chlorine concentration (g L^{-1} or g kg^{-1} FW). The intake of stable chlorine should include the contribution from salt (NaCl) licks for cattle.

4.6 Stable chlorine contents in environmental media

The stable chlorine values required for Equation (34) are shown in Table 15. These values represent the means of literature data for soil [66], plants and animal products [67] and air [68-69]. They should be used with caution and only when site-specific data are not available. In fact, there exist only a few data in the literature for stable chlorine contents in environmental media, and those show some large variations.

5. CONCLUSIONS

The specific activity concepts upon which the tritium, ^{14}C and ^{36}Cl models are based are theoretically sound for long-term safety assessments with constant release rates. However, the models do not apply to short-term term (accidental) releases where concentrations are time-dependent. For example, the ^{36}Cl content of soils varies by more than an order of magnitude between winter and the growing season, and plant uptake depends on the growth stage. This contributes some uncertainty to the predictions of ^{36}Cl concentrations in animal products calculated using the specific activity approach.

TABLE 15 STABLE INORGANIC CHLORINE CONTENTS IN ENVIRONMENTAL MEDIA *([66-70] as summarized in [71]*

Environmental media	Content	Unit	Environmental media	Content	Unit
Air[1]			Root vegetable	5.0×10^{-1}	g/kg
Gaseous	6.0×10^{-2}	mg/m^3	Beet	1.3	g/kg
Aerosol	3.0×10^{-2}	mg/m^3	Sugar beet	3.5×10^{-1}	g/kg
Water[1]	1.0×10^{-2}	g/L	Potatoes	1.0	g/kg
Groundwater[2]	1.6×10^{-2}	g/L	Red beet	6.0×10^{-1}	g/kg
River[3]	7.0×10^{-3}	g/L	Carrot	5.0×10^{-1}	g/kg
Rain[4]	1.1×10^{-2}	g/L	Celery	5.0×10^{-1}	g/kg
Soil	2.0×10^{-1}	g/kg (DW)	Turnip	5.5×10^{-1}	g/kg
Terrestrial plants			Onion	2.5×10^{-1}	g/kg
Cereals (grains)	5.0×10^{-1}	g/kg (FW)	Radish	3.0×10^{-1}	g/kg
Oat	5.0×10^{-1}	g/kg	Horse radish	1.7×10^{-1}	g/kg
Wheat	5.0×10^{-1}	g/kg	Rutabaga	3.0×10^{-1}	g/kg
Maize	4.5×10^{-1}	g/kg	Salsify	3.1×10^{-1}	g/kg
Millet (bird seeds)	1.9×10^{-1}	g/kg	Leafy vegetable	5.0×10^{-1}	g/kg
Barley	1.0	g/kg	Artichoke	2.2×10^{-1}	g/kg
Rice	2.3×10^{-1}	g/kg	Celery	1.4	g/kg
Saracen	3.0×10^{-1}	g/kg	Cabbage	1.1	g/kg
Cereals (flour)			Brussels sprout	1.0×10^{-1}	g/kg
Oat	4.9×10^{-1}	g/kg	Cauliflower	2.9×10^{-1}	g/kg
Wheat	5.0×10^{-1}	g/kg	Red cabbage	4.5×10^{-1}	g/kg
Fruits and nuts	5.0×10^{-1}	g/kg	Chives	4.3×10^{-1}	g/kg
Apricot	2×10^{-2}	g/kg	Watercress	1.0	g/kg
Almond	2.0×10^{-1}	g/kg	Endive	7.1×10^{-1}	g/kg
Pineapple	3.0×10^{-1}	g/kg	Spinach	7.5×10^{-1}	g/kg
Peanut	1.7×10^{-1}	g/kg	Curled salad	2.5×10^{-1}	g/kg
Eggplant	5.0×10^{-1}	g/kg	Lettuce	5.0×10^{-1}	g/kg
Banana	1.0	g/kg	Corn salad	1.0×10^{-1}	g/kg
Nectarine	5.0×10^{-2}	g/kg	Sorrel	6.0×10^{-1}	g/kg
Cherry	3.0×10^{-2}	g/kg	Parsley	1.3	g/kg
Chestnut	1.0×10^{-1}	g/kg	Dandelion	1.0	g/kg
Lemon	3.0×10^{-2}	g/kg	Leek	4.0×10^{-1}	g/kg
Pumpkin	1.8×10^{-1}	g/kg	Animal Products		
Quince	2.0×10^{-2}	g/kg	Milk	1.0	g/L
Cucumber	2.7×10^{-1}	g/kg	Woman	4.0×10^{-1}	g/L
Pickle	2.7×10^{-1}	g/kg	Cow	1.0	g/L
Courgette	1.8×10^{-1}	g/kg	Ewe	1.0	g/L
Date	2.5	g/kg	Buffalo	6.2×10^{-1}	g/L
Fig	1.6×10^{-1}	g/kg	Camel	1.1	g/L
Strawberry	1.2×10^{-1}	g/kg	Goat	5.0×10^{-1}	g/L
Raspberry	2.2×10^{-1}	g/kg	Mare	3.0×10^{-1}	g/L
Guava	4.5×10^{-1}	g/kg	Egg	1.2	g/kg
Currant	1.0×10^{-1}	g/kg	Meat	7.5×10^{-1}	g/kg
Bean	2.3×10^{-1}	g/kg	Beef	7.0×10^{-1}	g/kg
Mandarin	2.0×10^{-2}	g/kg	Horse	9×10^{-2}	g/kg
Melon	4.3×10^{-1}	g/kg	Sheep	1.0	g/kg
Blackberry	2.0×10^{-1}	g/kg	Lamb	8.5×10^{-1}	g/kg
Medlar	3.0×10^{-2}	g/kg	Veal	7.5×10^{-1}	g/kg
Coconut	1.2	g/kg	Pork	6.0×10^{-1}	g/kg
Olive	4.0×10^{-2}	g/kg	Turkey	1.2	g/kg
Grapefruit	2.0×10^{-2}	g/kg	Chicken	6.0×10^{-1}	g/kg
Watermelon	8.0×10^{-2}	g/kg	Pig liver	9.0×10^{-1}	g/kg
Peach	3.0×10^{-2}	g/kg	Aquatic Plants	5.0×10^{-1}	g/kg
Pear	2.0×10^{-2}	g/kg	Aquatic Animals		
Pea	3.6×10^{-1}	g/kg	Freshwater fishes	1.0	g/kg
Bell pepper	1.9×10^{-1}	g/kg	Bream	1.2	g/kg
Apple	3.0×10^{-2}	g/kg	Pike	1.0	g/kg
Plum	5.0×10^{-2}	g/kg	Perch	8.5×10^{-1}	g/kg
Grape	3.0×10^{-2}	g/kg	Tench	9.5×10^{-1}	g/kg
Rhubarb	5.3×10^{-1}	g/kg	Trout	1.0	g/kg
Tomato	4.0×10^{-1}	g/kg	Fresh water invertebrates	1.0	g/kg

[1] Dependent on distance from the sea.

[2] variation 1×10^{-3}-7×10^{-2}.

[3] variation: 1×10^{-3}-3.5×10^{-2}.

[4] variation: 1×10^{-3}-2.0×10^{-2}.

In applying the models, all inputs of the stable and active forms of the isotope to each environmental compartment must be taken into account. For example, the stable chlorine taken up by animals from salt licks should be accounted for in calculating ^{36}Cl concentrations.

The models for tritium and ^{14}C transfer following contamination of soil (Sections 2.2 and 3.2) are generally quite simple and should be reconsidered as more information becomes available.

Many of the parameters required by the models are bulk parameters that describe the net effect of several processes, and that are empirically derived. If the models are to provide reliable predictions, the experimental data must cover the full range of environmental conditions of interest. This is not always the case, and the main gaps in the data are listed below:

For tritium:

- More data are required on the relative contributions of air moisture and soil water to the tritium content in plants, to allow Equation (5) to be applied more realistically to fruit and root crops.
- More data are required on the ratio CR_s of soil water concentration to air moisture concentration (Equation (2)). Values are needed for different soil types and for sites that exhibit different frequencies with which rain falls when the airborne plume is present. New values of CR_s must be based on HTO measurements made frequently enough to provide reliable averages of air and soil concentrations over the growing season. Alternatively, the model of transfer from air to soil could be amended to treat wet and dry deposition explicitly (e.g. [72]).
- More data collected under steady-state conditions are required to better define the isotopic discrimination factors in the formation of OBT in a variety of plants and fish (Equations (7) and (24)).
- More information is needed on the ratio R_{HT} of the HTO concentration in plant water to the HT concentration in air (Equation (25)). Given the difficulty in measuring R_{HT} experimentally, new values are more likely to arise through model simulations than through measurements.

For ^{14}C:

- More information is needed on the canopy dilution factor CD_C (Equations (28) and (29)) as a function of the size of the contaminated soil source and the height and density of the plant canopy. Given the difficulty in measuring CD_C experimentally, new values are most likely to come from models that account for the emission of ^{14}C from soil and plants and dispersion from ground-level area sources.
- More data are required for F_C, the average production rate of carbon by decomposition of crop residues (Equations (28) and (32)), for different crop types.
- More data are needed for E_C, the ^{14}C evasion rate (Equation. (31)), for different soil and meteorological conditions.

For ^{36}Cl:

More data on stable chlorine concentrations in the drinking water and foodstuffs of domestic animals, and in the animal products themselves, would be useful.

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FOOD PROCESSING

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Abstract

The paper discusses the processes involved in food preparation for consumption and technological processing that could be used to reduce contamination of foodstuffs. The food processing transfer parameters–food processing retention factor, processing efficiency and the food processing factor–are defined and their relationships discussed. Data are presented for vegetables, fruit, cereals, dairy products, meat (mammals, birds and fish–edible fraction), fungi, seafoods and drinks. Processing procedures considered includes: simple washing and boiling for fruit, vegetables and mushrooms, boiling and milling for cereals, cooking and curing for meats, butter and cheese productions for dairy products and washing and cooking for seafoods. Storage times for foodstuffs have also been considered. A wide range of chemical elements are considered, with the majority of data presented caesium, strontium and iodine, stable element databases have been reviewed and relevant data used to extend the data set. A discussion of application of the data is presented with consideration of areas where cautions is needed.

1. DEFINITIONS AND CONCEPTS

The concentration of radionuclides in food can be affected by industrial and domestic processes such as extraction during boiling, removal of certain parts of the raw food (eg. bran, peel, shell, bone) and drying or dilution [1]. Neglecting of radionuclide losses during food processing can lead to overestimation of the calculated dose. It should be noted that processing of raw materials of vegetable and animal origin is often the most effective countermeasure for reducing the radioactive contamination of the foodstuff to permissible levels or below. It can be applied both domestically and in industrial processing of food.

Experience gained after the Chernobyl accident has shown that many commonly used methods of domestic and industrial processing of food products results in significant decreases of contamination of those foodstuffs and hence of internal radiation doses to people [2, 3]. Food processing allows significant reduction in the radionuclide contamination of foodstuffs. It can be achieved by many of the normal practices used in the preparation, cooking and processing of food. The effects of processing on contaminated food depend on the radionuclide, the type of foodstuff and the method of processing. The effectiveness of radionuclide removal from raw material during processing can vary widely, but can be up to 99% (for instance in the production of ethyl alcohol and vegetable oil) [2].

In addition, it is relevant to note that standard food preparation techniques will be used irrespective of whether it is known that food is contaminated with radionuclides. Therefore, when evaluating the radiological impacts of routine releases, e.g. in the context of optimisation studies, consideration may need to be given to the degree to which those impacts are affected by food processing methods. Finally, it has to be recognised that the waste streams generated in food processing may be contaminated by radionuclides and the radiological impacts of disposal or recycling of this material, e.g. in animal feed, may need to be addressed.

Data on the behaviour of many radionuclides during food processing are scarce. The exceptions are radioisotopes of caesium, strontium and iodine. Some measurements were made in the 1960s at a time when there was concern over the consequences of radionuclide transfer from nuclear weapons testing into the human food chain. Following the accident at the Chernobyl NPP, new measurements have become available. Noordijk and Quinault [4] reviewed the existing literature with in the framework of the CEC and VAMP programmes. These results were mainly reported in Technical Reports Series No. 364 [1]. This updated account includes the more recent results and information from various reviews [5-9], as well as the experimental data from the database of the UK Food Standards Agency (for ^{137}Cs, ^{90}Sr and stable Na, K, Ca, Mg, P, Fe, Cu, Zn, Cl, Mn, Se, I, Cd and Pb) [10] and from the database created within the framework of the Franco-German Initiative FGI (includes 783 cases of technological processing of ^{137}Cs and ^{90}Sr contaminated raw materials after the Chernobyl accident: animal products – 384 and plant products–399 cases) [2, 3]. The main results obtained, focusing on the most effective methods, are shown in Tables 1-9.

In reporting the quantitative results of food processing, the following food processing transfer parameters are applied:

1. Food processing retention factor (F_r) is the fraction of activity of radionuclides that is retained in the food after processing. F_r is defined as the total amount of a radionuclide in processed food divided by the total amount of the radionuclide in the original raw food (Bq processed per Bq raw, i.e. F_r cannot exceed 1):

$$F_r = \frac{A_{pf}}{A_{rf}} \tag{1}$$

Where A_{pf} is the total activity of processed food, Bq and A_{rf} is the total activity of raw material.

2. Processing efficiency (P_e) which is the ratio of the fresh weight of processed food (M_{pf}) divided to weight of original raw material (M_{rf}) i.e.:

$$P_e = \frac{M_{pf}}{M_{rf}} \tag{2}$$

3. Processing factors (P_f)[1] for a foodstuff which is the ratio of the radionuclide activity concentrations (analogous to CR–concentration ratio) in the food after (SA_{pf}) and before processing (SA_{rf}) (Bq/kg processed per Bq/kg raw for fresh weight):

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[1] In some publications, in particular the ICRU report 65 this value is called the Food processing retention factor.

$$P_f = \frac{SA_{pf}}{SA_{rf}} \tag{3}$$

Processing factors in terms of dry mass are also sometimes reported, but, since consumption rates are generally expressed in terms of fresh mass of food [5], these are not included here. (Note that values of P_f can exceed 1).

There is a simple relationship among these three factors. F_r is the product of P_f and P_e**:**

$$F_r = P_f \cdot P_e \tag{4}$$

Food processing retention factor (F_r) is mainly applied for assessment of the total losses of radionuclides during processing (removal of a radionuclide from the food chain and/or estimation of discharges to waste streams) and calculations of collective dose [1, 5]. For calculation of individual doses, and for comparisons of concentrations in foods with permissible levels, the processing factor (P_f) concept is mainly applied [2, 9]. Also for some processes where the activity remains in the waste product rather than being removed from the foodstuff, notably the production of oil from olives, rapeseed and wine from grapes, the parameter P_f is more appropriate [5].

Application of these various factors is illustrated with reference to caesium and strontium. Thus, an F_r value of 0.4 for caesium in boiled meat indicates that only 40% of the caesium in raw meat is retained after boiling and that 60% is removed into the boiling liquid (Table 6). In the case of dairy products (Table 5), the yield of each product is important. For example, an F_r value of 0.61 for strontium in goat cheese indicates that 39% is removed by the conversion of goat milk to cheese, but, owing to the 12% yield of cheese, the concentration of strontium in goat cheese is 0.61/0.12 = 5 times the concentration in goat milk. Therefore, the processing factor (P_f) is 5. Thus, if the permissible levels for ^{90}Sr in milk and milk products are similar (for instance, in Ukraine they are 20 Bq/L(kg) [11]), and the ^{90}Sr specific activity in milk is somewhat lower than the permissible level, its specific activity in cheese will be 5 times higher than in milk and may well exceed the permissible level. Therefore, in Ukraine milk with ^{90}Sr specific activity more than 4 Bq·l^{-1} can not be used for cheese production, although the milk itself does not exceed the permissible level of 20 Bq L^{-1} and can be consumed.

F_r values for animal food products are all based on contamination *in vivo*. All data on plants refer to the contamination of the edible product, generally contaminated via root uptake followed by translocation. However, it should be noted that often the radionuclide transfer factors from soil to plant are experimentally determined and reported for the washed and peeled vegetables and fruits (for example, for potato). In this case, application of the radionuclide losses at washing and peeling to concentrations estimated using experimentally determined transfer factors will lead to underestimation of the predicted activity of radionuclides in foodstuffs. Therefore, it is important to know whether the transfer factor values were obtained for washed and peeled vegetables and fruits.

For vegetables, F_r values based on 'external contamination' are also presented. A product is said to be externally contaminated when the leaves are contaminated by spraying, painting, deposition, etc. and the time lag between contamination and processing is short enough to ensure that the majority of the radionuclides have not migrated from the surface into the plant.

It must be stressed that a thorough knowledge of the food consumption pattern is required. For example, all F_r values referring to procedures such as cooking and frying apply only when cooking liquid is removed from the food and not used for other culinary purposes or food preparation.

Milk products may require careful consideration, due to the variety of processes employed and products generated. It should be determined which coagulation process is used for cheese making – the acid or rennet process. Further, it should not be assumed beforehand that all whey will be discarded as waste or animal feed. The food industry has found ways to use whey as an additive to human food. If all the whey and the buttermilk is used for human consumption, it is more accurate to use for collective dose assessment an F_r value of 1.0 for all milk. However, such approach may not be appropriate for individual dose assessments, depending on the mix of milk products consumed by the individuals of interest, and it may be more appropriate to use the food processing factors for the different products and assess the doses to the population groups separately, using their specific consumption rates of the different products.

As for forest products such as mushrooms and berries, culinary treatment is one of the most effective and technically feasible options for reduction of radiocaesium uptake into the human body. Raw mushrooms are usually peeled, washed and boiled in water, which leads to more than one order of magnitude reduction of radiocaesium activity in the cooked mushrooms [12].

Long storage and processing times will reduce the activity contents of short-lived radionuclides in foodstuffs, with implications for assessments of doses from releases of radionuclides to the environment [5, 9]. The delay between harvest and consumption is important for short-lived radionuclides such as ^{131}I. For instance, processing of milk with a high concentration of ^{131}I during the acute phase of Chernobyl accident into long-stored foodstuffs (such as butter, cheese and dried milk) ensured significant decreases of ^{131}I concentrations in these foodstuffs due to the radioactive decay before their delayed consumption. For that reason, storage and processing times for the main foodstuffs are also reported here [5, 8, 9, 125].

2. PROCESSING OF PLANT PRODUCTS

Available data on food processing of plant products are given in Tables 1-6. Processing of fruits and vegetables includes surface cleansing or washing and other more vigorous or deeply penetrating measures. The efficiency of radionuclide removal through processing of plant products varies widely and can remove up to 99% of the initial activity in raw material (Table 1). However, the efficiency of surface cleansing or washing of fruit and vegetables is rather low and gives a reduction in the ^{137}Cs content of up to 10-30% of the initial activity. Some more vigorous processing can be more effective. Thus, the ^{137}Cs content is reduced by 30-80% after boiling, salting, pickling, and juice and wine production. Most of the initial radionuclide content remains in water wastes and filter-pressed precipitate. The technological processing of grain to flour, sugar-beet to sugar and potatoes to starch provides products with low ^{137}Cs and ^{90}Sr concentrations (Table 1).

Although most data are related to ^{90}Sr and ^{137}Cs, information on the behaviour of tritium and ^{14}C during food processing is available and summarized in Tables 5-6. The values for ^{14}C and tritium assume that the water used in cooking is uncontaminated, which may not always be the case. Moreover, it is the custom in some cultures to consume the cooking water, in which case any tritium lost to the water would still be ingested. For these reasons, it is recommended that, in the absence of specific information, the food-processing factor for these radionuclides should be 1 (i.e. concentrations in food products should not be reduced when the food is processed).

TABLE 1. FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR VEGETABLES AND FRUIT (DATA ARE BASED ON TOTAL CONTAMINATION OF THE PLANT) [1-3, 5, 8-10, 12-49]

Method of processing	Food processing retention factor F_r			P_e	
	Element	Min.	Max	Min.	Max.
Washing of vegetables, berry and fruits	Cs	6×10^{-1}	1.0	1.0	
	I	8×10^{-1}		1.0	
	Ru	7×10^{-1}	9×10^{-1}	1.0	
	Sr	4×10^{-1}	1.0	1.0	
Peeling of vegetables	Am, Pu	1×10^{-1}	1.0	7×10^{-1}	9×10^{-1}
	Cs	5×10^{-1}	9×10^{-1}	7×10^{-1}	9×10^{-1}
	Po	3×10^{-1}	5×10^{-1}	7×10^{-1}	9×10^{-1}
	Sr	5×10^{-1}	9×10^{-1}	7×10^{-1}	9×10^{-1}
Boiling in water of vegetables, berries and fruits	Am, Ca, Cu, Fe, K, Mg, Na, P, Po, Pu, Ru, S, Zn	3×10^{-1}	1	8×10^{-1}	1
	Cl, ^{3}H	3×10^{-1}	6×10^{-1}	8×10^{-1}	1
	Cs	4×10^{-1}	9×10^{-1}	8×10^{-1}	1
	Sr	6×10^{-1}	1.0	8×10^{-1}	1
Canning, blanching and pickling of vegetables	Cs	1×10^{-1}	1	5×10^{-1}	9×10^{-1}
	Sr	3×10^{-1}	1.0	5×10^{-1}	9×10^{-1}
Producing of sugar from beetroot	Cs	1×10^{-3}	1×10^{-2}	1.2×10^{-1}	
Producing of starch from potato	Cs	2×10^{-2}	3×10^{-2}	1.8×10^{-1}	
Olive press – oil;	Cs	1.3×10^{-1}		2×10^{-1}	
- cake		4.3×10^{-1}		5×10^{-1}	
Processing rapeseed to oil	Cs	$P_f = 4 \times 10^{-3}$			
	Sr	$P_f = 2 \times 10^{-3}$			

TABLE 2. FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR VEGETABLES AND FRUIT (DATA ARE BASED ON EXTERNAL CONTAMINATION ONLY) [1,22,23,50]

Method of processing	Food processing retention factor F_r			P_e	
	Element	Min.	Max.	Min.	Max.
Washing of vegetables, berries and fruits	Cs	1×10^{-1}	9×10^{-1}	1.0	
	I	1×10^{-1}	9×10^{-1}	1.0	
	Ru	2×10^{-1}	8×10^{-1}	1.0	
	Sr	1×10^{-1}	5×10^{-1}	1.0	
Boiling of vegetables and berries	Ba	6×10^{-1}	9×10^{-1}	8×10^{-1}	1
	Cs	1×10^{-1}	5×10^{-1}	8×10^{-1}	1
	I	1×10^{-1}	5×10^{-1}	8×10^{-1}	1
	Ru, Te	3×10^{-1}	7×10^{-1}	8×10^{-1}	1
	Sr	1×10^{-1}	2×10^{-1}	8×10^{-1}	1
	Zr:	1.0		8×10^{-1}	1

TABLE 3. FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR CEREALS [1-3, 9, 10, 14, 15, 17, 18, 22, 23, 28, 47, 51-62]

Raw material	Method of processing	Food processing retention factor, F_r			P_e	
		Element	Min.	Max.	Min.	Max.
Wheat, rye, barley, oats grain	Milling to white flour	Am, Pu	1×10^{-1}	2×10^{-1}	6×10^{-1}	8×10^{-1}
		Cd, Pb	5×10^{-1}	6×10^{-1}	6×10^{-1}	8×10^{-1}
		Cs	2×10^{-1}	6×10^{-1}	6×10^{-1}	8×10^{-1}
		Sr	1×10^{-1}	6×10^{-1}	6×10^{-1}	8×10^{-1}
	Milling to dark flour	Cs	5×10^{-2}	2×10^{-1}	5×10^{-2}	1×10^{-1}
		Sr	1×10^{-1}	2×10^{-1}	5×10^{-2}	1×10^{-1}
	Milling to semolina	Cs	1.5×10^{-1}	5×10^{-1}	1×10^{-1}	3×10^{-1}
	Milling to bran	Cs	4×10^{-1}	7×10^{-1}	1×10^{-1}	4×10^{-1}
		Sr	6×10^{-1}	9×10^{-1}	1×10^{-1}	4×10^{-1}
	Cooking wheat sprouts	Cs	8×10^{-1}	9×10^{-1}	1.8	2.4
	Shredding or puffing wheat	Cs	1×10^{-1}	1.5×10^{-1}	9×10^{-1}	9.5×10^{-1}
Rice grain	Polishing	Ca, P, Fe, K, Mg	1×10^{-1}	6×10^{-1}		
		Cs	2×10^{-1}	4×10^{-1}		
		Na, Cu, Zn	7×10^{-1}	9×10^{-1}		
		Sr	1×10^{-1}	4×10^{-1}		
Brown, savoury, easy cook white rice	Boiling	Ca, Cl, Cu, Fe, K, Mg, Na, P, Se, Zn	F_r =0.3-0.4			
Macaroni, spaghetti, pasta	Boiling	Cs	1×10^{-1}	4×10^{-1}		
		Ca, Cl, Cu, Fe, K, Mg, Na, P, Zn	P_f =0.1-0.4			

TABLE 4. FOOD PROCESSING RETENTION FACTOR (P_e) AND THE PROCESSING EFFICIENCY P_e FOR DRINKS [1, 5, 8, 10, 17, 22, 29, 63-68]

Raw material	Method of processing	Food processing retention factor, F_r			P_e	
		Element	Min.	Max	Min.	Max.
Surface waste water	Conventional treatment to tap water	Co	4×10^{-1}		1	
		Cs	7×10^{-1}		1	
		I:	8×10^{-1}		1	
		Ru	3×10^{-1}		1	
		Sr	1.0		1	
Tea	Brewing 2-8 minutes	Cs	4×10^{-1}	6×10^{-1}		
		Cs	9×10^{-1} for external contamination			
Herb tea	Brewing	Cs	4×10^{-1}	6×10^{-1}		
Berries and fruits	Juice	Am, Pu	5×10^{-1}		3×10^{-1}	9×10^{-1}
		Cs	2×10^{-1}	9×10^{-1}	3×10^{-1}	9×10^{-1}
		^{3}H	6×10^{-1}		3×10^{-1}	9×10^{-1}
		S	2×10^{-1}		3×10^{-1}	9×10^{-1}
Grapes	Wine	Cu, K, P, Zn	3×10^{-1}	8×10^{-1}	6×10^{-1}	8×10^{-1}
		Cs	3×10^{-1}	7×10^{-1}	6×10^{-1}	8×10^{-1}
		Sr	2×10^{-1}	6×10^{-1}	6×10^{-1}	8×10^{-1}

TABLE 5. FOOD PROCESSING FACTORS FOR TRITIUM [20-21]

Raw material	Method of processing	Food processing retention factor F_r		Processing efficiency P_e
		HTO	OBT	
Blackberries	Washed and stewed	5.5×10^{-1}	5.6×10^{-1}	5.9×10^{-1}
Broad beans	Boiled	2.8×10^{-1} [a]	6.9×10^{-1} [b]	9.1×10^{-1}
Cabbage	Washed and steamed	2.8×10^{-1} [a]	-	9.8×10^{-1}
Carrots	Washed and boiled	2.8×10^{-1} [a]	4.3×10^{-1} [b]	8.5×10^{-1}
	Scrubbed and boiled	5.5×10^{-1} [b]	-	9.2×10^{-1}
New potatoes	Peeled and roasted	2.2×10^{-1} [b]	-	6.2×10^{-1}
	Peeled and boiled	5.5×10^{-1}	7.4×10^{-1}	9.2×10^{-1}
Old potatoes	Peeled and roasted	2.1×10^{-1}	-	6.5×10^{-1}
Hulled rice	Boiled	8.4×10^{-1}	-	-
Soybean	Boiled	7.7×10^{-1}	-	-
Rice flour	Boiled	6.9×10^{-1}	-	-
Soybean flour	Boiled	7.4×10^{-1}	-	-

[a]Some data are below the detection limit, [b]not significant at the 5% level.

TABLE 6 FOOD PROCESSING FACTORS FOR ^{14}C [20, 21]

Raw material	Method of processing	F_r	P_e
New potatoes	Scrubbed and boiled	6.9×10^{-1}	9.2×10^{-1}
Hulled wheat	Boiled	8.2×10^{-1}	-
Hulled rice	Boiled	9.8×10^{-1}	-
Soybean	Boiled	8.6×10^{-1}	-
Wheat flour	Boiled	9.2×10^{-1}	-

3. PROCESSING OF ANIMAL PRODUCTS

3.1. Milk processing

Milk is one of the important foods that can contribute to internal irradiation doses and available data for this product are shown in Table 7.

Radiocaesium is concentrated in the water phase of milk, whereas radiostrontium is bound by casein and milk protein. Neither radionuclide is preferentially associated with the fat content of milk and hence neither tends to accumulate in high-fat products.

The ^{137}Cs contents of cream, sour cream, butter, natural cheese, and casein after processing are 1-30% of the initial radionuclide content in raw milk (Table 7). The ^{90}Sr contents in cream, sour cream and butter products with high contents of fats are reduced to 1-30% of those in raw milk. Contaminated milk should not be used for producing of dried milk because the drying process does not remove radionuclides.

TABLE 7. FOOD PROCESSING RETENTION FACTOR (F_r) AND THE PROCESSING EFFICIENCY (P_e) FOR DAIRY PRODUCTS (**bold** denotes recommended values) [1-3, 9, 10, 14, 18, 21, 22, 51, 69-103]

Product	Food processing retention factor F_r				P_e		
	Element	Mean	Min	Max	Mean	Min	Max
Cream	Ca, Cl, K, Mg, Na		3×10^{-2}		$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	Cd		6×10^{-2}	1×10^{-1}	$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	Cs	$\mathbf{5 \times 10^{-2}}$	3×10^{-2}	1.6×10^{-1}	$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	Fe		7×10^{-2}		$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	I	$\mathbf{6 \times 10^{-2}}$	6×10^{-3}	1.9×10^{-1}	$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	P		2×10^{-2}		$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	Pb, Zn		5×10^{-2}		$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
	Sr	$\mathbf{4 \times 10^{-2}}$	2×10^{-2}	2.5×10^{-1}	$\mathbf{8 \times 10^{-2}}$	3×10^{-2}	2.4×10^{-1}
Sour cream	Cs	$\mathbf{1 \times 10^{-1}}$	1×10^{-1}	2×10^{-1}	$\mathbf{1 \times 10^{-1}}$	1×10^{-1}	2×10^{-1}
	Sr	$\mathbf{1 \times 10^{-1}}$	1×10^{-1}	1.3×10^{-1}	$\mathbf{1 \times 10^{-1}}$	1×10^{-1}	2×10^{-1}
Skim milk	Cs	$\mathbf{9.5 \times 10^{-1}}$	8.5×10^{-1}	9.9×10^{-1}	$\mathbf{9.2 \times 10^{-1}}$	7.6×10^{-1}	9.7×10^{-1}
	I		8.1×10^{-1}	9.4×10^{-1}	$\mathbf{9.2 \times 10^{-1}}$	7.6×10^{-1}	9.7×10^{-1}
	Sr	$\mathbf{9.3 \times 10^{-1}}$	7.5×10^{-1}	9.6×10^{-1}	$\mathbf{9.2 \times 10^{-1}}$	7.6×10^{-1}	9.7×10^{-1}
Butter	Ca, Cl, K, Mg, Na		8×10^{-3}		$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	Cd		1×10^{-1}		$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	Cs	$\mathbf{1 \times 10^{-2}}$	3×10^{-3}	2×10^{-2}	$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	I	$\mathbf{2 \times 10^{-2}}$	1×10^{-2}	3.5×10^{-2}	$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	P		4×10^{-3}		$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	Pb		2×10^{-2}		$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	Sr	$\mathbf{6 \times 10^{-3}}$	2.5×10^{-3}	1.2×10^{-2}	$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
	Zn		1×10^{-2}		$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	5×10^{-2}
Buttermilk	Cs	$\mathbf{5 \times 10^{-2}}$	2×10^{-2}	1.3×10^{-1}	$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	0.14
	I		5×10^{-2}	1.3×10^{-1}	$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	0.14
	Sr	$\mathbf{6 \times 10^{-2}}$	3×10^{-2}	7×10^{-2}	$\mathbf{4 \times 10^{-2}}$	3×10^{-2}	0.14
Butterfat	I		2×10^{-2}		$\mathbf{4 \times 10^{-2}}$	4×10^{-2}	4×10^{-2}
	Sr		1×10^{-3}	2×10^{-3}			
Milk powder (dried)	Ca Cl K Mg Na Zn		1		$\mathbf{1.2 \times 10^{-1}}$	1.1×10^{-1}	1.2
	Cs	**1.0**	1.0		$\mathbf{1.2 \times 10^{-1}}$	1.1×10^{-1}	1.2
	I	**1.0**	1.0		$\mathbf{1.2 \times 10^{-1}}$	1.1×10^{-1}	1.2
	Sr	**1.0**	1.0		$\mathbf{1.2 \times 10^{-1}}$	1.1×10^{-1}	1.2
Condensed milk	Ca Cl Cu Fe K Mg Na, Zn		1		$\mathbf{4 \times 10^{-1}}$	3.7×10^{-1}	
	Cs	**1.0**	1.0		$\mathbf{4 \times 10^{-1}}$	3.7×10^{-1}	
	I	**1.0**	1.0		$\mathbf{4 \times 10^{-1}}$	3.7×10^{-1}	
	Sr	**1.0**	1.0		$\mathbf{4 \times 10^{-1}}$	3.7×10^{-1}	
Cheese[1]							
goat	Cs		7×10^{-2}	1.5×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.7×10^{-1}
	I		8×10^{-2}	1.4×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.7×10^{-1}
	Sr		6.1×10^{-1}		$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.7×10^{-1}
Cow rennet	Ca		5×10^{-1}	7×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	Cd Fe Mg P Pb		2×10^{-1}	4×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	Cl K		1×10^{-1}		$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	Cs	$\mathbf{7 \times 10^{-2}}$	5×10^{-2}	2.3×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	Cu		4×10^{-1}	6×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	I	$\mathbf{2.0 \times 10^{-1}}$	1.1×10^{-1}	5.3×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	Se, Zn		7×10^{-1}	1	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
	Sr	$\mathbf{7 \times 10^{-1}}$	2.5×10^{-2}	8.0×10^{-1}	$\mathbf{1.2 \times 10^{-1}}$	8×10^{-2}	1.8×10^{-1}
Cow acid	Cs	$\mathbf{6 \times 10^{-2}}$	1×10^{-2}	1.2×10^{-1}	$\mathbf{1.0 \times 10^{-1}}$	8×10^{-2}	1.2×10^{-1}
	I		2.2×10^{-1}	2.7×10^{-1}	$\mathbf{1.0 \times 10^{-1}}$	8×10^{-2}	1.2×10^{-1}
	Sr	$\mathbf{8 \times 10^{-2}}$	4×10^{-2}	8×10^{-2}	$\mathbf{1.0 \times 10^{-1}}$	8×10^{-2}	1.2×10^{-1}
Cottage cheese rennet	Cs		1×10^{-2}	5×10^{-2}			
	Sr		7×10^{-2}	1.7×10^{-1}			

Cottage cheese rennet	Cs		1×10^{-2}	5×10^{-2}			
	Sr		7×10^{-2}	1.7×10^{-1}			
Cottage cheese acid	Cs	**1×10^{-1}**	1×10^{-1}		**1.2×10^{-1}**	1×10^{-1}	1.4×10^{-1}
	Sr		2×10^{-1}	7×10^{-1}	**1.2×10^{-1}**	1×10^{-1}	1.4×10^{-1}
Whey [1]							
rennet	Cs		7.3×10^{-1}	9.6×10^{-1}	**9.0×10^{-1}**	7.0×10^{-1}	9.4×10^{-1}
	I		4.7×10^{-1}	8.9×10^{-1}	**9.0×10^{-1}**	7.0×10^{-1}	9.4×10^{-1}
	Sr		2.0×10^{-1}	8.0×10^{-1}	**9.0×10^{-1}**	7.0×10^{-1}	9.4×10^{-1}
acid	Cs		7.5×10^{-1}	9.0×10^{-1}		8.2×10^{-1}	
	I		6.0×10^{-1}	7.3×10^{-1}		8.2×10^{-1}	
	Sr		7.0×10^{-1}	9.0×10^{-1}		8.2×10^{-1}	
Casein[1]							
rennet	Cs		1×10^{-2}	8×10^{-2}		3×10^{-2}	6×10^{-2}
	I		2×10^{-2}	1.2×10^{-1}		3×10^{-2}	6×10^{-2}
	Sr		1.0×10^{-1}	8.5×10^{-1}		3×10^{-2}	6×10^{-2}
acid	Cs		1×10^{-2}	4×10^{-2}		1×10^{-2}	6×10^{-2}
	I		3×10^{-2}	4×10^{-2}		1×10^{-2}	6×10^{-2}
	Sr		5×10^{-2}	8×10^{-2}		1×10^{-2}	6×10^{-2}
Casein whey[1]							
rennet	Cs		7.7×10^{-1}	8.3×10^{-1}	**7.6×10^{-1}**	7.3×10^{-1}	7.9×10^{-1}
	I		6.9×10^{-1}	8.2×10^{-1}	**7.6×10^{-1}**	7.3×10^{-1}	7.9×10^{-1}
	Sr		8×10^{-2}	1.6×10^{-1}	**7.6×10^{-1}**	7.3×10^{-1}	7.9×10^{-1}
acid	Cs		8.3×10^{-1}	8.4×10^{-1}	**7.8×10^{-1}**	7.5×10^{-1}	7.9×10^{-1}
	I		7.8×10^{-1}	8.0×10^{-1}	**7.8×10^{-1}**	7.5×10^{-1}	7.9×10^{-1}
	Sr		6.7×10^{-1}	8.6×10^{-1}	**7.8×10^{-1}**	7.5×10^{-1}	7.9×10^{-1}
Milk[2]							
ion exchange	Cs	**5×10^{-2}**	4×10^{-2}	6×10^{-2}	**1.0**	1	
	I	**1×10^{-1}**			**1.0**	1	
	Sr	**1×10^{-1}**			**1.0**	1	

[1] Separate values are given for the rennet and acid coagulation procedures; [2] Decontamination of milk by ion exchange on a commercial scale.

3.2 Meat processing

Meat processing is an effective method for reducing the ^{137}Cs content in food (Table 8). Boiling and soaking in salt or acid solution are the most effective types of meat processing. Mechanical removal of bone removes ^{90}Sr contamination from meat.

4. PROCESSING OF FOREST PLANT PRODUCTS (MUSHROOMS, BERRIES, MEDICINAL PLANTS)

The most effective processing techniques for berries and mushrooms are soaking, boiling and salting. Use of these measures gives decreases of factors of 2 to 10 in the ^{137}Cs content of forest products. Washing of mushrooms alone gives a reduction in the radiocaesium content of the product by a factor of two. Significant reductions in ^{137}Cs activity concentrations can be achieved by soaking of dried mushrooms and berries for several hours and salting of mushrooms. Some general data are given in Table 9, while Table 10 gives some information for individual mushrooms species and types of mushroom processing.

TABLE 8 FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR MEAT (**bold** data denote recommended values) [1-3, 9, 10, 22, 69, 104-123]

Raw material	Method of processing	Food processing retention factor F_r				P_e	
		Elem.	Mean	Min.	Max	Min.	Max.
Tissues of animals (cow, pig, sheep, deer, rabbit)	Boiling meat	Cs	**4×10^{-1}**	2×10^{-1}	7×10^{-1}	5×10^{-1}	7×10^{-1}
		I		6×10^{-1}		5×10^{-1}	7×10^{-1}
		Ru		3×10^{-1}		5×10^{-1}	7×10^{-1}
		Sr	**5×10^{-1}**	4×10^{-1}	9×10^{-1}	5×10^{-1}	7×10^{-1}
	Boiling bone	Cs	**3×10^{-1}**	2×10^{-1}	3×10^{-1}	1.0	
		I		9.8×10^{-1}		1.0	
		Ru		7×10^{-1}		1.0	
		Sr		9.9×10^{-1}		1.0	
	Frying, roasting or grilling meat	Ca Cl Cu Fe K Mg Na P Se Zn		5×10^{-1}	1		
		Cs	**7×10^{-1}**	5×10^{-1}	8×10^{-1}	4×10^{-1}	7×10^{-1}
		I		2×10^{-1}	6×10^{-1}	4×10^{-1}	7×10^{-1}
		Sr		8×10^{-1}		4×10^{-1}	7×10^{-1}
	Microwave baking	Ca Cl Fe K Mg Na P Se Zn		5×10^{-1}	1	4×10^{-1}	7×10^{-1}
		Cs	**5×10^{-1}**	4×10^{-1}	5×10^{-1}	4×10^{-1}	7×10^{-1}
	Pickling wet (salting), marinating	Cs	**5×10^{-1}**	1×10^{-1}	7×10^{-1}	9×10^{-1}	1
	Sausage production	Cs		4×10^{-1}	1.0		
Birds	Boiling meat	Sr		5×10^{-1}			
	Baking meat	Cs		7×10^{-1}	8×10^{-1}		
	Roasting	Ca Cl Cu Fe I K Mg Mn Na P Se Zn		5×10^{-1}	1	4×10^{-1}	7×10^{-1}
Fish	Boiling flesh	Cs		2×10^{-1}	9×10^{-1}	5×10^{-1}	9×10^{-1}
		Sr		9×10^{-1}		5×10^{-1}	9×10^{-1}
	Frying flesh	Cs		8×10^{-1}	9×10^{-1}	7×10^{-1}	8×10^{-1}
	Grilling	Ca Cl Cu Fe I K Mg Mn Na P Se Zn		P_f=1.1-1.2			

TABLE 9 FOOD PROCESSING RETENTION FACTOR (F_r) AND THE PROCESSING EFFICIENCY (P_e) FOR ^{137}Cs and ^{90}Sr FOR FOREST PLANT PRODUCTS (MUSHROOMS AND BERRIES) (*Data are based on total contamination of the plant*) [1-3, 12, 13]

Raw material	Method of processing	F_r		P_e	
		Min.	Max	Min.	Max.
Berries (bilberry, blackberry)	Washing	8×10^{-1}	1	1	
	Boiling	5×10^{-1}	6×10^{-1}	1	
	Drying of berries	1		1×10^{-1}	
	Soaking in water of dried berries	8×10^{-1}		1×10^{-1}	
Mushrooms	Washing	4×10^{-1}		1	
	Drying of mushrooms	1		1×10^{-1}	1.2×10^{-1}
		(Sr: 1)		1×10^{-1}	1.2×10^{-1}
	Washing of dried mushrooms	5×10^{-1}		1×10^{-1}	
	Soaking of dried mushrooms in water	1×10^{-1}	2×10^{-1}	1×10^{-1}	
	Salting	7×10^{-2}	1×10^{-1}	6×10^{-1}	9×10^{-1}
	Boiling (30-60 min)	1×10^{-1}	3×10^{-1}	6×10^{-1}	8×10^{-1}
		Sr: 2×10^{-1}	9×10^{-1}	6×10^{-1}	8×10^{-1}
	Boiling of dried mushrooms	1×10^{-1}		1.5×10^{-1}	
	Pickling	6×10^{-2}	1×10^{-1}	6×10^{-1}	
		(Sr: 5×10^{-1})		6×10^{-1}	

TABLE 10. FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR ^{137}Cs in SOME EDIBLE MUSHROOM SPECIES [126, 127]

Mushroom species	Type of culinary processing	F_r		P_e	
		Min.	Max.	Min.	Max.
Boletus edulis (*dry weight*)	*Consecutive processing*				
	Washing by flowing water for 10 min.	9.0×10^{-1}	9.5×10^{-1}	1.1	
	Soaking in 0.85 % salt solution for10 h followed by washing in flowing water	1.5 x 10^{-1}	2.0×10^{-1}	1.5	1.7
	Boiling for 5 min with extract removal	8×10^{-2}	1.0×10^{-1}	8×10^{-1}	9×10^{-1}
	Boiling for 20 min with extract removal	3×10^{-2}	5×10^{-2}	7×10^{-1}	
Suillus variegatus, (*fresh weight*)	*Consecutive processing*				
	Cleaning of mushroom cap	8.0×10^{-1}	8.5×10^{-1}	1.0	
	Washing by flowing water for10 min.	5.0×10^{-1}	5.5×10^{-1}	1.3	
	Boiling for 20 min. and washing by flowing water for 10 min.	1.5×10^{-1}	2.0×10^{-1}	8×10^{-1}	
	Pickling	5×10^{-2}	1.0×10^{-1}	5×10^{-1}	
Xerocomus badius (*fresh weight*)	Boiling for 5 min.	2.5×10^{-1}	3.0×10^{-1}	9×10^{-1}	
	Boiling for 10 min.	1.5×10^{-1}	2.0×10^{-1}	8×10^{-1}	
	Boiling for 20 min.	5×10^{-2}	7×10^{-2}	8×10^{-1}	
	Soaking for 20 min.	8.0×10^{-1}	8.5×10^{-1}	1.3	
	Soaking for 40 min.	6.0×10^{-1}	7.0×10^{-1}	1.3	
	Soaking for 60 min.	3.0×10^{-1}	4.0×10^{-1}	1.3	
Lactarius deliciosus, *L. necator,* *Russula delica,* (*fresh weight*)	*Consecutive processing:*				
	Cleaning of mushroom cap	7.0×10^{-1}	7.5×10^{-1}	1.0	
	Washing by flowing water for10 min.	6.5×10^{-1}	7.0×10^{-1}	1.0	
	Soaking for 24 h	2.5×10^{-1}	3.0×10^{-1}	1.2	
	Soaking for 48 h	1.0×10^{-1}	1.2×10^{-1}	1.2	
	Soaking for 72 h	2×10^{-2}	3×10^{-2}	1.2	
	Salting in 2-3 % salt solution 72 h	3×10^{-3}	5×10^{-3}	1.0	

It can be seen from the data presented that culinary processing of forest products can considerably decrease ^{137}Cs concentrations in mushrooms consumed by man. Consecutive processing – cap cleaning and washing of fresh fruit bodies leads to decreases of ^{137}Cs activity concentration to about 10 % of the initial value in *Boletus edulis* to 40-45 % in *Suillus variegatus*. Soaking and boiling of fresh fruit bodies are characterized by values from 0.85 to 0.03 and salting – 0.003-0.005. Soaking and boiling of dried fruit bodies are characterized by values from 0.20 to 0.03.

After the Chernobyl accident some information was also obtained on processing retention factors for wild (mainly forest) medicinal plants (Table 11). Normally, wild medicinal plants are used after drying to an air-dry state. Liquid medicinal forms are then prepared from dried plant material.

There are five main groups of plant medicinal raw material according to parts of plants used: fruits (mainly of arboreal species) – *Alnus glutinosa*, *Rosa canina*, *Crataegus sp.*, *Vaccinium myrtillus*, etc.); flowers (both grass and arboreal species) – *Tilia cordata*, *Crataegus sp.*, *Helichrysum arenarium*, etc.; buds (mainly of arboreal species) – *Pinus sylvestris*, *Betula pendula*; leaves, shoots (more than 50 grass and dwarf-shrub species) – *Leonurus quinquelobatus*, *Hypericum perforatum*, *Vaccinium vitis-idaea*, etc.; rhizomes and roots (mainly of grass species) – *Potentilla erecta*, *Acorus calamus*, *Valeriana officinalis, Ononis arvensis* etc.; bark (mainly of arboreal species) – *Quercus robur, Frangula alnus, Viburnum opulus* etc.

TABLE 11. Cs-137 AND ^{90}Sr FOOD PROCESSING RETENTION FACTORS F_r FOR PREPARATION OF LIQUID WATER MEDICINAL FORMS (INFUSIONS AND BROTHS) FROM AIR DRY MEDICINAL PLANT RAW MATERIAL [128-134]

Group of medicinal plant raw material	N	Food processing retention factor F_r							
		^{137}Cs				^{90}Sr			
		AM	SD	Min	Max	AM	SD	Min	Max
Fruits	25	4.9×10^{-1}	2.7×10^{-1}	1.1×10^{-1}	8.7×10^{-1}	4.3×10^{-1}	9×10^{-2}	2.9×10^{-1}	5.9×10^{-1}
Flowers	20	6.0×10^{-1}	2.9×10^{-1}	1.5×10^{-1}	9.3×10^{-1}	4.7×10^{-1}	2.1×10^{-1}	1.6×10^{-1}	7.3×10^{-1}
Buds	20	5.5×10^{-1}	8×10^{-2}	4.4×10^{-1}	5.8×10^{-1}	5.0×10^{-1}	1.0×10^{-1}	4.0×10^{-1}	5.5×10^{-1}
Grass, leaves, shoots	115	5.7×10^{-1}	1.5×10^{-1}	2.0×10^{-1}	9.2×10^{-1}	4.6×10^{-1}	1.2×10^{-1}	2.2×10^{-1}	7.5×10^{-1}
Rhizomes and roots	20	4.8×10^{-1}	2.0×10^{-1}	1.9×10^{-1}	8.9×10^{-1}	2.3×10^{-1}	6×10^{-2}	1.4×10^{-1}	3.1×10^{-1}
Bark	15	2.9×10^{-1}	6×10^{-2}	1.8×10^{-1}	3.8×10^{-1}	1.6×10^{-1}	5×10^{-2}	1.2×10^{-1}	2.8×10^{-1}

In wide use now are ethanol and water-ethanol liquid medicinal forms made from medicinal plants. For some medicinal plants, these forms are the main, and sometimes the only possible, way to produce medicines. There is a close relationship between ^{137}Cs concentrations in raw plants and in water-ethanol medicinal forms that depends on the alcohol concentration in the solvent [135, 136]. It has been found that for concentrations of ethanol in solvent of 96.5 % the mean value is around 0.12 (range 0.05-0.19), for 70 % it is 0.58 (range 0.51-0.69), and for 40 % it is 0.70 (range 0.64-0.86) [135, 136].

Targeted chemical processing of medicinal plants with extraction of clean effective medicinal substances (flavonoids, cortical glycozids etc.) leads to medicines of very low contamination and values are in the range $1\text{-}3\cdot10^{-5}$ [135].

5. PROCESSING OF LOWER SEA ORGANISMS

Data on the behaviour of radionuclides during food processing of lower sea organisms are rather scarce (Table 12) and more research is necessary to provide robust data on sea food processing factors to be implemented for environmental assessments.

6. EFFECT OF STORAGE AND PROCESSING TIMES

As mentioned earlier, taking into account the delay between the harvest (production of raw product) and consumption of processed product is important for realistic dose estimates from short-lived radionuclides such as ^{131}I and the necessary data which allow such effects into to be taken into account are given in Table 13.

TABLE 12 FOOD PROCESSING RETENTION FACTOR F_r AND THE PROCESSING EFFICIENCY P_e FOR LOWER SEA ORGANISMS [17, 123, 124]

Raw mat.	Method of processing	Element	F_r values		P_e
			Min.	Max	
Crevette (Shrimp)	Wash with tap water	Ca	9×10^{-1}		1
		^{90}Sr	7×10^{-1}		1
	Wash with 1-3% solution of NaCl	Ca	9×10^{-1}		1
		^{90}Sr	3×10^{-1}	4×10^{-1}	1
	Cooking	Pb	0.0 [a]	4×10^{-1}	3.5×10^{-1}
		Po:	4×10^{-1}	8×10^{-1}	3.5×10^{-1}
		Ra:	4×10^{-1}	5×10^{-1}	3.5×10^{-1}
Oyster	Wash with 1-3% solution of NaCl	Ca	8×10^{-1}		1
		^{90}Sr	7×10^{-1}	8×10^{-1}	1
Mussels	Washing and removal of flesh	Pb	5×10^{-1}		2.5×10^{-1}
		Po	2×10^{-2}		2.5×10^{-1}
		Ra	1×10^{-2}		2.5×10^{-1}
Clam	Wash with tap water	Ca	8×10^{-1}		1
		^{90}Sr	7×10^{-1}		1
	Wash with 1-3% solution of NaCl	Ca	7×10^{-1}	5×10^{-1}	1
		^{90}Sr	3×10^{-1}	6×10^{-1}	1
Algae	Alginate production	Ru, Rh	7×10^{-2}		4×10^{-2}
		Sr	6×10^{-1}		4×10^{-2}
		Te	2×10^{-2}		4×10^{-2}
	Satiagum production	Co	4×10^{-2}		8×10^{-2}
		Ru, Rh	4×10^{-2}		8×10^{-2}

[a] Lower than detection limit.

TABLE 13 DELAY TIMES (STORAGE AND PROCESSING TIMES) BETWEEN HARVESTING AND CONSUMPTION OF FOOD PRODUCTS [5, 8, 9, 125]

Raw material	Typical value	Minimum	Maximum
Cereals and cereal products	6 months	45 days	1 year
Potatoes and beet	3 months	7 days	6 months
Leafy vegetables	4 days	1 day	7 days
Root vegetables	10 days	7 days	14 days
Fruit vegetables	7 days	2 days	14 days
Fresh apples and pears	3.5 months	0 day	8 months
Fresh drupe fruits, soft fruit, rhubarb	4 days	0 day	8 days
Canned fruit	1 year	14 days	2 years
Frozen fruit	6 months	7 days	1 year
Jams and jellies	1 year	1 day	2 years
Milk	2 days	1 day	6 days
Butter	1 month	3 days	3 months
Cream	5 days	2 days	10 days
Condensed milk	6 months	7 days	1 year
Pasteurized skim milk	2 days	1 day	6 days
Cheese (rennet coagulation)	1.5 months	30 days	3 months
Cheese (acid coagulation)	1 month	7 days	2 months
Fresh* beef	20 days	14 days	28 days
Fresh* pork, veal	4 days	2 days	7 days
Fresh* chicken	4 days	2 days	7 days
Fresh* lamb	10days	7 days	14 days
Fresh* game	10 days	2 days	20 days
Eggs	14 days	2 days	28 days

* Refers to fresh meat, frozen meats would have longer delay times of up to 6 months.

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APPENDIX

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USE OF ANALOGUES

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Abstract

The use of analogues is necessary in the case where directly applicable data are not available or are of dubious quality. Three main types of analogue are considered in this document: the same parameter obtained for another isotope of the same element (e.g. using data obtained for a stable isotope of a radioactive element) – analogue isotopes; the same parameter obtained for another element – analogue elements; and the differening parameter obtained for the same element (e.g. applying the soil–plant transfer factor for one crop to another crop) – analogue parameters. An analogue could only be proven to be valid by comparing its behaviour in the conditions of interest with that of the thing for which it is an analogue. Hence, while confidence in the validity of an analogue will increase as the quality of the justification increases, there will always be some residual uncertainty.

1. INTRODUCTION

The bioavailability of hazardous materials depends on their physical and chemical properties, as well as the nature of release and the biogeochemistry of the environment at their location of deposition. The self-cleaning capacity of the receptor medium also has a determining role. The movement of radionuclides between different physical and chemical phases in different parts of various ecosystems has been studied, but only for a few radionuclides (mainly of caesium and strontium) and is still not well understood in details. Finding a proper analogue either for a process or for an isotope often leads to a better, more reliable description of environmental behaviour. The use of analogues is not an accurate way of modelling, but may be used if no data are available in the frame of screening models. Analogues can be used to provide input values for model parameters as default values, though care has to be exercised in not pushing such analogies beyond their legitimate domain of applicability. Thorough knowledge is required of the system for which an analogue is sought to derive parameter values. Relevant processes and features have to be known, such as time scales of these processes, physical, chemical and biological properties of the environment and relevant media.

Three main types of analogue that can be used for derivation of values if measured or recommended values given in this document are not available:

- The same parameter obtained for another isotope of the same element (e.g. using data obtained for a stable isotope of a radioactive element) – analogue isotopes;
- The same parameter obtained for another element – analogue elements; and
- A different parameter obtained for the same element (e.g. applying the soil–plant transfer factor for one crop to another crop) – analogue parameters.

2. ANALOGUE ISOTOPES

Application of analogue isotopes is the most common form and is often used without any specific justification or even recognition that data for an analogue are being used. Short-lived fission products whose environmental behaviour has been extensively studied in the context of reactor accidents or routine discharges used as analogues for long-lived isotopes of relevance for solid waste disposal, like data of ^{131}I for long lived ^{129}I and data on well studied ^{134}Cs or ^{137}Cs for the long lived ^{135}Cs; similarly, short-lived and readily available tracer radionuclides are used in experiments as analogues for the isotopes found in radioactive discharges or disposed as wastes.

Stable isotopes whose environmental behaviour has been extensively studied in the context of chemical toxicity can be used as analogues for less common and less studied radioactive isotopes found in radioactive discharges or disposed as wastes.

Since radionuclides remain in the environment for a long time, obtaining transfer factors values under equilibrium conditions is important because a freshly added radionuclide may not be at equilibrium in the environment. Many long-lived radionuclides generated in nuclear power plants do not exist in the natural environment and there is a general lack of information on their environmental behaviour over decades. To fill these gaps, measurement of F_v values of naturally existing elements rather than radionuclides can be a powerful tool to obtain F_v values under equilibrium conditions.

It has been also reported that the F_v values of fallout ^{137}Cs are 3-6 times higher than those of native Cs [1, 2, 3] suggesting that the physico-chemical forms of global fallout ^{137}Cs differ from those of naturally occurring stable Cs. Fig 1. compares F_v values for ^{137}Cs and stable Cs for white rice collected in Akita, Japan over 1987 to 1999 [4, 5]. F_v data for the period 1959-1986 were excluded because they were affected by global fallout [6]. F_v data for fallout ^{137}Cs and stable Cs for crops collected from the 2002 to 2005 are also plotted for comparison in Fig 2. The values recorded were rather similar but the F_v values for ^{137}Cs were usually slightly higher than the F_v values for stable caesium ($p<0.01$).

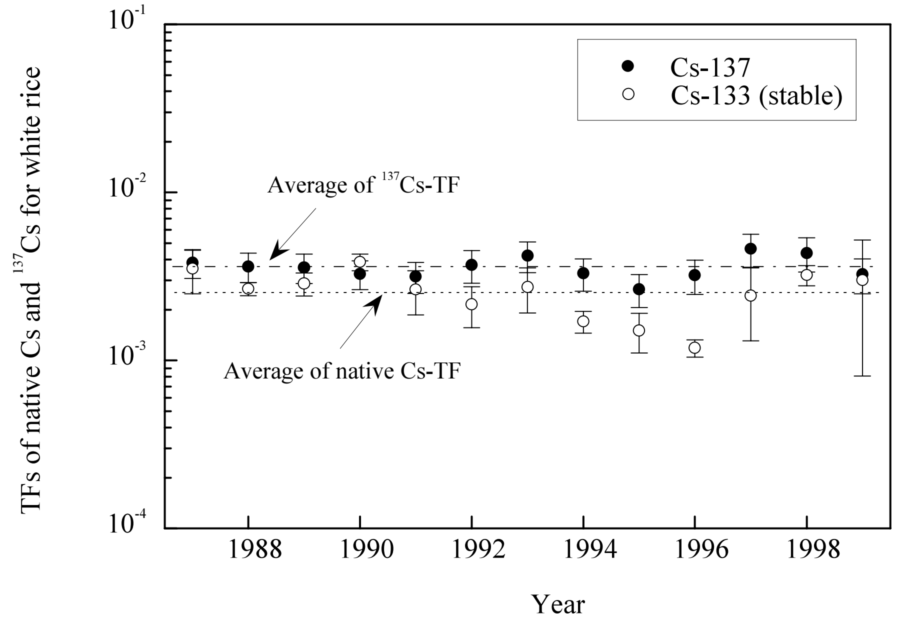

FIG. 1. Transfer factors of ^{137}Cs and stable native ^{133}Cs for white rice collected in Akita, Japan [4, 5].

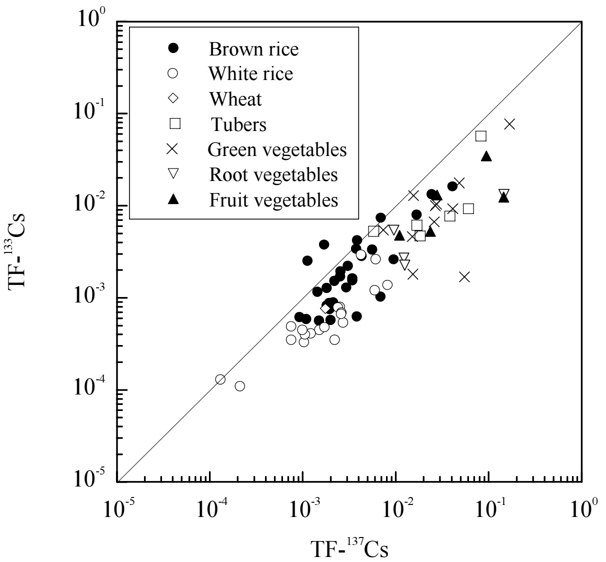

FIG. 2. Relationship between F_v ^{137}Cs and F_v stable native ^{133}Cs for various crops [1-3].

This phenomenon of initial higher bioavailability of ^{137}Cs than stable Cs can be explained as follows: fresh, artificial ^{137}Cs is more mobile and more easily adsorbed by plants than stable Cs in the soil. Stable Cs is present in the interlayer lattice of clays, where it is relatively unavailable compared with ^{137}Cs.

However, ^{137}Cs slowly becomes incorporated in the interlayer lattice, so that eventually it is well-mixed with the whole stable Cs pool in soil; therefore, in the long term, the F_v value for stable Cs can be used to evaluate transfer of ^{137}Cs in the environment. The same reasoning can be adopted for other elements. For several radioisotopes, there is/are stable isotope(s), such as ^{88}Sr for ^{90}Sr, ^{127}I for ^{129}I and ^{133}Cs for ^{135}Cs or ^{137}Cs, so that the stable isotopes can serve as analogues. Table 1 provides, for some crops, F_v values of elements that might serve as analogues for radionuclides [3, 9-11].

The uptake of ^{79}Se by plants will be strongly determined by the stable selenium status of the soil. Similarly, in upland areas, soils can be low in stable iodine and chlorine, so affecting transfers of radioisotopes of these elements. In particular, for iodine the fraction of a radioisotope entering the blood that is taken up by the thyroid is enhanced in iodine-deficient conditions. For chlorine, homeostatic requirements mean that the half life of retention in the bodies of animals varies in inverse proportion to the level of the stable element in the diet. This may be a significant factor in selecting data values, but should not affect the selection of analogues.

For radioisotopes that have no stable isotopes, it might be possible to use other elements with similar biogeochemical characteristics as a way to add information. For instance, Ba is considered a good analogue for Ra [7, 26] and Re for Tc [8].

In general, the behaviour of isotopes of the same element is identical, except for light elements especially in case of hydrogen. An important limitation and consideration is whether the timescale over which behaviour of a short-lived radionuclide can be studied is sufficient to reveal the significance of long-term processes that may influence the behaviour of a long-lived radioisotope of the same element. In particular, equilibration of a short-lived isotope in environmental media may be strongly influenced by its physical decay, whereas equilibration of a long-lived isotope may be almost entirely determined by biogeochemical transfer processes.

TABLE 1. SOIL TO PLANT TRANSFER FACTORS OF ELEMENTS TO CROPS

Element	Green vegetables[1]		Fruit vegetables[1]		Tubers[1]		Root vegetables[1]		White rice[2]		Brown rice[2]	
	N	GM	N	GM	N	GM	N	GM	N	GM	N	GM
Al	19	3.1×10^{-4}	12	1.0×10^{-4}	10	4.1×10^{-4}	8	2.1×10^{-4}	37	2.3×10^{-5}	50	2.9×10^{-5}
As	20	1.7×10^{-3}	14	1.6×10^{-3}	11	1.2×10^{-3}	10	1.6×10^{-3}	37	7.1×10^{-3}	50	9.3×10^{-3}
Ba	21	2.8×10^{-2}	14	7.7×10^{-3}	11	4.3×10^{-3}	10	3.3×10^{-2}	37	3.9×10^{-4}	50	1.8×10^{-3}
Ca	21	6.6×10^{-1}	14	1.4×10^{-1}	11	3.8×10^{-2}	10	2.5×10^{-1}	37	4.1×10^{-3}	50	8.8×10^{-3}
Cd	20	3.4×10^{-1}	14	1.7×10^{-1}	11	2.1×10^{-1}	8	2.0×10^{-1}	37	9.1×10^{-2}	50	9.4×10^{-2}
Ce	20	1.2×10^{-3}	14	5.1×10^{-4}	11	6.9×10^{-4}	9	1.3×10^{-3}	27	3.3×10^{-5}	33	3.3×10^{-5}
Co	21	5.8×10^{-3}	14	3.7×10^{-3}	11	3.5×10^{-3}	10	3.5×10^{-3}	37	4.6×10^{-4}	49	9.3×10^{-4}
Cr	21	4.8×10^{-3}	14	1.6×10^{-3}	11	1.3×10^{-3}	10	2.3×10^{-3}	37	1.5×10^{-3}	50	2.0×10^{-3}
Cs	21	6.1×10^{-3}	14	6.1×10^{-3}	11	4.5×10^{-3}	10	4.0×10^{-3}	37	5.9×10^{-4}	50	9.5×10^{-4}
Cu	19	9.8×10^{-2}	14	1.7×10^{-1}	11	1.3×10^{-1}	8	3.6×10^{-2}	37	8.8×10^{-2}	50	1.0×10^{-1}
Fe	21	1.3×10^{-3}	14	9.3×10^{-4}	11	7.5×10^{-4}	10	7.3×10^{-4}	37	9.8×10^{-5}	50	2.9×10^{-4}
K	21	3.2×10^{0}	14	3.2×10^{0}	11	1.9×10^{0}	10	3.1×10^{0}	37	6.2×10^{-2}	50	2.2×10^{-1}
La	21	2.7×10^{-3}	14	7.9×10^{-4}	11	7.1×10^{-4}	10	3.8×10^{-3}	34	3.5×10^{-5}	45	4.9×10^{-5}
Mg	19	2.8×10^{-1}	12	2.4×10^{-1}	10	2.4×10^{-1}	8	1.6×10^{-1}	37	5.6×10^{-2}	50	2.4×10^{-1}
Mn	21	2.5×10^{-2}	14	1.5×10^{-2}	11	8.4×10^{-3}	10	9.8×10^{-3}	37	1.6×10^{-2}	50	4.6×10^{-2}
Mo	20	3.6×10^{-1}	14	3.5×10^{-1}	11	1.7×10^{-1}	9	1.8×10^{-1}	37	7.1×10^{-1}	50	7.6×10^{-1}
Na	21	6.7×10^{-2}	14	1.1×10^{-2}	11	5.0×10^{-3}	10	1.1×10^{-1}	37	6.9×10^{-4}	50	9.8×10^{-4}
Ni	21	3.5×10^{-2}	14	2.3×10^{-2}	11	1.4×10^{-2}	10	1.4×10^{-2}	37	1.4×10^{-2}	50	1.3×10^{-2}
Rb	19	4.6×10^{-1}	12	6.0×10^{-1}	10	2.2×10^{-1}	8	5.2×10^{-1}	37	5.6×10^{-2}	50	1.2×10^{-1}
Sb	21	7.0×10^{-3}	14	3.9×10^{-3}	10	2.1×10^{-3}	9	4.2×10^{-3}	34	1.4×10^{-3}	34	1.6×10^{-3}
Se	14	3.5×10^{-2}	11	1.9×10^{-2}	9	1.9×10^{-2}	8	2.3×10^{-2}	26	5.4×10^{-2}	41	6.7×10^{-2}
Sr	21	2.6×10^{-1}	14	4.5×10^{-2}	11	2.3×10^{-2}	10	1.3×10^{-1}	37	8.6×10^{-4}	48	3.2×10^{-3}
Th	18	9.0×10^{-4}	14	2.3×10^{-4}	11	6.4×10^{-4}	10	7.6×10^{-4}	37	1.3×10^{-4}	40	1.0×10^{-4}
Ti	16	1.0×10^{-3}	14	6.4×10^{-4}	10	8.2×10^{-4}	9	8.0×10^{-4}	26	1.8×10^{-4}	50	3.1×10^{-4}
U	16	1.1×10^{-3}	12	3.8×10^{-4}	11	8.9×10^{-4}	10	1.2×10^{-3}	29	1.1×10^{-4}	32	5.6×10^{-5}
V	19	4.4×10^{-4}	12	1.6×10^{-4}	11	4.5×10^{-4}	9	3.8×10^{-4}	23	3.5×10^{-5}	50	4.6×10^{-5}
Zn	21	2.7×10^{-1}	14	2.2×10^{-1}	11	1.1×10^{-1}	10	1.9×10^{-1}	37	2.0×10^{-1}	50	2.4×10^{-1}

[1] Data are from [10, 16-17.], [2] data are from [11].

3. ANALOGUE ELEMENTS

The purely chemical properties of elements follow fairly well established patterns that can be used as a basis for identifying potential analogues. Elements in the same group (column) of the periodic table usually exhibit similar chemical behaviour, because they have the same number of outer electrons available to form chemical bonds (i.e. they form compounds in the same valence state).

In case of essential macro-elements for plants occurring in soil, the uptake and transfer of a chemically similar element will be affected depending upon any lack or excess of the essential one. However, generally similar chemistry does not necessarily imply similar metabolic characteristics in plants and animals, because of the high specificity of biochemical pathways. Thus, although chlorine and iodine have many chemical similarities, their behaviour in mammals is very different because of the role of iodine in the production of thyroid hormones.

The most commonly used analogue element- pairs are K and Cs, Ca and Sr, Ba and Ra. Ba, Ca and K are regarded as biogenic elements indicating the influence of biogenic processes. Ca is often used as analogue element for Sr, for example the correlation coefficient between Ca and Sr in suspended particulate matter was found 0.9986 according to the work of Varnavas et al. [12] and Table 2 shows another example for good correlation of the dissolved ^{90}Sr and stable Ca content of the fraction of molecular weight less than 10000 Daltons after two-step ultrafiltration of surface water [13].

TABLE 2. FRACTIONAL DISTRIBUTION OF DISSOLVED ^{90}Sr AND STABLE Ca IN THE FRACTION LESS THAN 10000 Daltons AFTER TWO-STEP ULTRAFILTRATION OF THE SURFACE WATER [13]

Sampling place	^{90}Sr, %	Ca, %
Sahan River, March 1996	80	97
Sahan River, August 1996	98	97
Glubokoye Lake, August 1996	97	98
Kiev Reservoir, April 1998	97	99

The counter-example is that an increase in soil strontium concentration of over 4 orders of magnitude range was accompanied by increase in tissue strontium burdens of earthworms according to the work of Morgan et al. [14], whereas tissue calcium burdens appeared to be maintained at fairly uniform levels, irrespective of soil calcium concentrations. The contrasting behaviour suggest that differences exist in the ability of earthworms to physiologically control tissue burdens of these two analogue elements and this may relate to differences in the solubility of sequestration products of strontium and calcium.

The availability and distribution of radiocaesium in soil-plant systems is affected by the soil exchangeable K status, clay content and time. Cs is a chemical analogue of K, therefore, they behave as competitors for plant and soil sorption sites [15]. Transfer factors can be reduced by increasing K supply at low K status of a soil, whereas the transfer factor will be increased by increasing K supply at high K status of the soil. At K concentrations up to 1mM, the plant and soil solution concentration ratio can be decreased by more than 2 orders of magnitude with increasing K concentrations. However, this dependence is not linear and above 1mM K, the concentration ratio is almost unaffected by K supply.

These results show that K fertilisation can be a successful countermeasure for reduction of ^{137}Cs transfer, as has been demonstrated for cereals [16]. Transport patterns in bean plants following the foliar uptake for periods longer than 1 day are similar for K and Cs [17]. It is likely Cs is taken up by the K transport systems of the plasma membranes of root cells.

Uptake of K (and therefore also probably of Cs) at low external K^+ concentrations (<10-12 mg/l) is dominated by a carrier-mediated high-affinity transport system, which can operate against an electrochemical gradient requiring energy. Conversely, at higher external concentrations of K^+ (>12-40 mg/l) the electrochemical gradient is diminished or reversed, and uptake has dominantly channel-like properties [18].

Unfortunately, no stable Ra isotope exists, but several studies have demonstrated that Ba behaves in the environment similarly to ^{226}Ra. For example, the oceanic ^{226}Ra/Ba ratio is constant nearly everywhere [19]. This similarity of the two elements means that ^{133}Ba can be used as a tracer in laboratory experiments instead of ^{226}Ra having high radiotoxicity for example to quantify the uptake kinetics by sediments in aqueous suspensions [20].

Transition elements in the same period (row) of the periodic table also tend to be chemically similar to each another.

There are strong similarities among all the lanthanides, though there are some subtle differences due to variations in the dominant and subsidiary oxidation states along the series. Also, there are analogies between the lanthanides and higher actinides. Lanthanides are oxidation state analogues for actinides, so their distribution can give an indication of the long-term behaviour of the radioactive transuranic elements. Exceptions to the typical lanthanide pattern are cerium and europium with their 4+ and 2+ oxidation states, contrasting with the 3+ oxidation state common to all lanthanides. This property gives the possibility of using cerium as an analogue of isotopes of various higher actinides. Their ionic radii are close enough to justify this analogy, for example 0.114 nm – Ce^{3+}, 0.111 nm – Ce^{4+}, 0.105 nm – Th^{4+}, 0.096 nm – Pu^{4+}, 0.106 nm – Am^{3+}. There are several geological studies where thorium serves as an analogue for Pu^{4+} which is a most common oxidation state of plutonium in the environment, and La^{3+} as an analogue for Cm^{3+} and Am^{3+}. In the migration studies Th serves as usually as an analogue for Pu^{4+}, and La^{3+} as an analogue for Cm^{3+} and Am^{3+}. Moreover the precipitation of lanthanides is closely linked to the behaviour of the transition metals iron and manganese. Nd^{3+}, which occurs only in this chemical form, gives more adequate analogy for Pu^{3+} and Am^{3+}, which have solubilities at least one order higher than that of Pu^{4+} in the environmental conditions, in different kind of natural water. Regarding the bioconcentration factors for lanthanides in river biota are in a narrow range [21].

Chemical similarity does not necessarily translate into similar behaviour in the environment; sometimes the size of the ionic form of a radionuclide can cause differences particularly in the chemical association processes in environmental conditions. Subtle differences in chemistry can result major distinctions in biochemical processes.

4. ANALOGUE PARAMETER

Soil K_d values can vary significantly with the soil type[19]. Where knowledge of soil characteristics is not available, a generic soil K_d value can be adopted. This may be an average over soil types, a value for the soil type expected to maximise doses, or simply a value for the soil type for which data are most extensive. If data are limited, the K_d for a soil can be used for a sediment with similar characteristics (pH, Eh *etc.*).

In case of plants, some analogues may seem relatively obvious, e.g. between pasture grass and forage. However, closer inspection may show that the analogy is not close and may be misleading, e.g. if information is required for ^{60}Co in a non-leguminous pasture and the available data are for ^{60}Co in clover. Similarly, generic data for 'grain' might be expected to provide a good analogue for rice, but the growing conditions for rice are so different from those for cereals that the analogue is not, in general, a good one[20]. For example the PATHWAY [22] model uses a single transfer factor value for all crops and soil types, whereas ECOSYS has crop-specific transfer factor values that are independent of soil type [23]. In the case of PATHWAY the higher uncertainty of the predicted values should be taken into account. During the parameterisation of the model of ECOSYS, to the results observed for ryegrass appeared satisfactory for a variety of crops for example for grass and cereals [15], applying experimental values of ryegrass as an analogue plant to cereals. If plant analogues are used, it is normally assumed that the soil–plant transfer factor is the same for different plant types on a dry weight (of plant) basis, allowing for the consideration that the fresh weight values differ according to the water content of the different plant types.

Phylogenetic effects at the level of plant family and order have been observed for C, Ca, Cs, Cu, K, Mg, Ni and Zn, from this comes a hypothesis, by analogy with the behaviour of Ca and other minerals, that phylogenetic attributes of a species can be used to estimate the soil to

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[19] See paper by Vidal et al 'Soil-Radionuclide Interactions' in this publication.

[20] See paper by Uchida et al 'Transfer to rice' in this publication.

plant transfer of many radionuclides. Phylogenetic effects are less strong for Cd, Cr, and Na, and hardly occur for N, P and Pb [24].

When making comparisons between animals of different types, consideration also has to be given to the mass of the animal. Conventionally, transfers to animal products have been expressed through the use of transfer factors that are the ratio of the concentration in the product to the rate of intake of the radionuclide. For unit rate of intake, the concentration in a particular product tends to be higher for animals of smaller mass, though this effect may be counteracted by more rapid metabolic turnover in smaller animals.

In the case of different products from the same animal the assumption of similar transfer factors, e.g. to chicken meat and eggs, might seem tempting, but needs to be approached with caution. In the most common of these cases, one product (milk or eggs) is collected during the life of the animal whereas another (meat) arises only when the animal is slaughtered, and the two products are very different in nature. Other examples are different parts of the slaughtered animal (flesh, liver, etc.) and, as with humans, many elements concentrate preferentially in certain tissues or organs. In particular, because of the major role of the liver in detoxification, many transition metals, heavy metals, lanthanides and actinides are concentrated in it, giving rise to concentrations that are an order of magnitude or more larger than concentrations in meat.

In the other case of comparable products from different animals, e.g. meat from different animals, or cow’s milk and goat’s milk, analogies are also of limited use, because for most assessments the animals considered are quite different from each other. Poultry have quite different physiology and metabolism from mammals, and there are important differences between ruminants (such as cattle, sheep and goats) and other mammals (such as pigs). Another obvious difference concerns the nature of the food eaten by the animals. Cattle and sheep may be assumed to eat primarily grass, whereas goats and pigs often have much more varied diets and poultry primarily eat grain.

Therefore, although analogies may be drawn between different animal types, this is likely to involve the use of allometric relationships or more properly biological scaling, which is the effect of size on biological variables. Many biological phenomena scale as quarter powers of the mass, and the direct application of transfer factors for one animal type to another is almost never appropriate.

In the context of estimating exposures to non-human biota, reference organisms are used as analogues for broad classes of organisms. Such reference organisms have been selected on the basis of their radioecological and radiological sensitivities, ecological niche and sustainability for future monitoring or research. However this still leaves a requirement for a potentially large number of radionuclide-reference organism transfer values. The reference organism concept provides a series of organism types, which can be considered representative of different trophic levels. In each case, it is possible to determine relevant ecological parameters for a real species that is taken to correspond to the reference organism in the particular context under consideration. The ecological parameters provide information on prey, predators and the time spent in different compartments of the ecosystem; it is also generally possible to define a simplified geometry, usually ellipsoid for the dosimetric calculations. For each reference organism a concentration factor has been derived relative to soil, water or air depending on the type of assessment and radionuclide [25].

5. CONCLUSIONS

Although care is needed to consider the characteristics of each individual case, a general order of preferences for data sources is as follows:

- Data for the specific parameter of interest for the specific radionuclide of interest.
- Data for the specific parameter of interest for another isotope of the same element (preferably not a short-lived isotope for a long-lived isotope, as it may not persist for long enough in the environment to reveal the characteristic behaviour).
- Data for the specific parameter of interest for an analogue element.
- Data for a related parameter (e.g. different plant type or animal product) for the specific radionuclide/element of interest. In general, plant type analogues tend to be more reliable than animal product analogues.
- Data for a related parameter for an analogue element.

The ordering of options 3 and 4 in particular will depend on the specific case, and judgement will be necessary. For example, the order shown above would be valid if the choice were between a well-recognised element analogue and a cross-species animal product analogue; on the other hand, the order would be reversed if choosing between data for a similar plant type for an element with high plant uptake and a speculative element analogue.

The use of analogues is not the preferred approach to modelling, but it is necessary in those contexts in which directly applicable data are not available or are of dubious quality.

One can never be sure exactly how good any specific analogue is. An analogue could only be proven to be valid by comparing its behaviour in the conditions of interest with that of the thing for which it is an analogue. Hence, while confidence in the validity of an analogue will increase as the quality of the justification increases, there will always be some residual uncertainty.

As with any other choice of parameter values for modelling, decisions on using analogues must take account of the assessment context and particularly the level of realism or conservatism of the assessment. The best analogue for a realistic assessment might not be the best for a conservative assessment.

It is preferable to use elemental analogues that lie close to each other in a chemical series, for example, amongst the lanthanides it could be samarium and europium. However, in practice, by far the most extensive data amongst the lanthanides are for cerium, so it is often most appropriate to use this as the analogue when information is lacking for other lanthanide elements.

There are two main issues that could affect the validity of using isotope analogue. The timescales for experiments or observations on short-lived isotopes may be limited by radioactive decay and so might not reflect all aspects of environmental behaviour in the long-term. An important example is that of iodine isotopes. The majority of experimental data relate to ^{131}I, which is of great importance in the context of accidental releases from nuclear power stations, and has a half-life of about 8 days, whereas the isotope of interest for solid waste disposal is ^{129}I, with a half life of 17 million years. Observations of ^{131}I are limited by radioactive decay to a period of a few months at most, and so could be of little value for identifying and characterising long-term behaviour, because the time scale of the relevant processes in the environment is much longer than the half life of ^{131}I. In the opposite case, data for an analogue isotope that is long-lived or stable should exhibit the same short-term behaviour as a short-lived isotope (with the exception of radioactive decay, which is generally modelled explicitly), provided observations of the long-lived species have been made on short enough time scales. However, although some care is needed in cases where there are large differences in half-life – and especially when the analogue isotope is short-lived – isotopic analogues can normally be assumed to be more reliable than element or media analogues.

In addition to consideration of the effects of radioactive decay, it should be recognised that although isotopes of an element have the similar chemical behaviour, this chemical similarity is not exact and the differences will translate into subtle differences of behaviour in the environment. This effect is demonstrated by the absorption of common elements from the atmosphere – most plants show higher ratios of $^{12}C/^{13}C$, $^{14}N/^{15}N$ and $^{16}O/^{18}O$ than are found in the atmosphere, due to the difference in their chemical behaviour. These differences tend to be more important for lighter elements because the relative mass differences are larger (e.g. the relative difference between ^{7}Be and ^{10}Be nucleus is higher, than the difference between ^{226}Ra and ^{228}Ra which is less than 1% regarding the mass). Except for hydrogen and several light elements biochemical differences caused these isotopic differences will in general be much smaller than most other uncertainties in the system. The environmental behaviour of different isotopes may differ even simply because their modes of release or more general entry into the biosphere and consequently their distribution in the biosphere are different. The simplest example is the different chemical behaviour of released CH_4 and CO_2 regarding carbon and from the point of view of hydrogen isotopes released as CH_4 and H_2O.

For element analogues, chemical similarity does not necessarily translate into similar behaviour in the environment. For chemical group analogues, e. alkali earths, these differences will normally be large. For period analogues, such as the lanthanides, the differences may be much smaller. Key considerations include variations in valence and ionic radius. Thus, in the case of the lanthanides, there is a consistent trend across the series from predominantly 2+ through to 4+, and this trend can be reflected in trends in environmental behaviour. A problem of element analogue is that the initial distribution of elements in the environment can affect the behaviour of the radionuclides being modelled. If the soil is naturally (or as a result of past activities) poor or rich in a particular element that is (or behaves like) and important plant or animal nutrient, then the uptake and transfer of chemically similar radionuclides released to the environment will be affected. This may be a significant factor in selecting data values, but should not affect the selection of analogues because, by definition, if the analogue is good then it will behave in the same way as the radionuclide of interest would.

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