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Modelling the migration and accumulation of radionuclides in forest ecosystems

Final Report of the Forest Working Group
activities 1998–2000
BIOMASS Theme 3

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FOREWORD

The BIOMASS Programme on BIOsphere Modelling and ASSessment was launched in Vienna in October 1996, and is sponsored by the International Atomic Energy Agency and a number of other organisations, as shown below. The Programme addresses radiological issues associated with accidental and routine releases of radionuclides to the environment, and solid waste management. Three important themes involving environmental assessment modelling are covered:

Theme 1: Radioactive Waste Disposal. The objective is to develop the concept of "Reference Biospheres" into a practical system for application to the assessment of the long-term safety of repositories for radioactive waste. The following Task Groups have been set up to achieve this:

TG1: Principles for the Definition of Critical and Other Exposure Groups.

TG2: Principles for the Application of Data to Assessment Models.

TG3: Consideration of Alternative Assessment Contexts.

TG4: Biosphere System Identification and Justification.

TG5: Biosphere System Descriptions.

TG6: Model Development.

Theme 2: Environmental Releases. BIOMASS provides an international forum for activities aimed at increasing the confidence in methods and models for the assessment of radiation exposure related to environmental releases. Two Working Groups have been set up, concerned with Dose Reconstruction for previous releases; and Remediation Assessment to evaluate the efficacy of remedial measures.

Theme 3: Biosphere Processes. The aim of this Theme is to improve capabilities for modelling the transfer of radionuclides in particular parts of the biosphere which have been identified as being of potential radiological significance. This topic is being explored using a range of methods including reviews of the literature, model inter-comparison exercises and where possible, model testing against independent sources of data. Three Working Groups have been set up, to examine modelling of: 1) long-term tritium dispersion in the environment; 2) radionuclide uptake by fruits; and 3) radionuclide migration and accumulation in forest ecosystems.

This report describes results of the studies undertaken by the Forest Working Group under Theme 3. The support provided to this work by organisations listed below is also gratefully acknowledged.

Statens Stralskyddinstitut (SSI), Sweden

BIOMASS Theme 1

EXECUTIVE SUMMARY

This document describes the work carried out during a two year period from 1998 to 2000 by the IAEA's BIOMASS Forest Working Group (FWG). The primary objective of the FWG's activities was to bring together modellers working in the field of radionuclide behaviour in forest ecosystems and to facilitate their interaction in the following areas:

- Review and exchange of existing information and peer review.
- Development of a list of Features, Events and Processes (FEPs) for forests using an interaction matrix approach.
- Inter-comparison of existing models to identify and investigate significant areas of uncertainty and differences in modelling approach.
- Testing and validation of existing or new models against independent data sets, where available.
- Recommendation for possible future directions in the modelling of radionuclide behaviour in forests.

The work of the FWG began with a review of the biogeochemical cycling of radiocaesium in forest ecosystems, based on the best available knowledge obtained both before and after the Chernobyl accident of 1986. It was concluded from this review there is still a need for a standardisation of current approaches to data acquisition in a format which can be used to facilitate the comparison of radionuclide cycling in forests at different geographical locations and, preferably, on a whole-ecosystem basis.

The application of an interaction matrix method to forest model design was investigated as a means of both improving the design of models based on best available knowledge and achieving a greater degree of consensus between modelers on the objectivity and fitness for purpose of forest models. The definitions and application of transfer factors to forests was also reviewed in some detail and a new method of determining radiocaesium uptake within the woody tissues of trees (the Wood Immobilisation Potential, or WIP) was proposed.

The major part of the FWG's work was devoted to conducting three model inter-comparison studies based on three different scenarios. It was considered that an early emphasis on such practical modelling exercises was desirable so an inter-comparison of the predictions of ten models for a hypothetical Chernobyl-type scenario was conducted within the first six months of the FWG's activities. The results indicated a generally high level of consistency between model predictions for 'structural components' of the forest such as soils and trees. Predictions for biological endpoints such as edible mushrooms, however, were more variable. A detailed statistical analysis of these results was carried out.

A second model inter-comparison exercise was carried out using previously unseen data obtained from the Zhitomir region of Ukraine, which became contaminated with ^{137}Cs in 1986. Blind predictions were made by nine modellers and the results revealed a similarly high level of consistency between model predictions as in the first scenario. Many of the model predictions proved rather accurate, although the available data only covered a period of some 5 to 12 years after initial contamination when the early dynamics of ^{137}Cs within the forest ecosystem are likely to have been superseded by slower, long-term rates of redistribution. Particularly accurate and consistent predictions were made for the tree-related compartments and for certain soil compartments. As in the case of the first scenario, a detailed statistical analysis of these results was carried out.

The third and final modelling exercise was again hypothetical and involved comparison of model outputs against each other. Whereas the first two scenarios had involved Chernobyl-type scenarios, however, this scenario involved a shallow landfill type repository as a source term. Even though none of the FWG models had been constructed with this type of scenario in mind, seven models were used to provide predictions in this exercise, although some degree of model re-coding was required by at least some of these models. The degree of agreement between predictions for tree components and for biological endpoints was striking, although the reasons for this are not immediately evident. A key question arising from the results obtained is whether physical or biological transport of ^{137}Cs through the soil is likely to dominate when the source is below the ground surface and when deep-rooted plants, such as trees, are allowed to access such subterranean sources.

The conclusions of the FWG's activities are finally summarised and recommendations for future experimental and modelling studies within the broad field of forest radioecology are made. These include the issue of time-dependency in forest processes (including tree growth), high versus low deposition scenarios, process-orientated models, integration of forest radioecology with other ecological models, and the question of whether it is possible to construct a 'generic' model of radionuclide behaviour in forests.

Appended to this document are summary descriptions of the individual models which participated in the FWG modelling studies, as well as detailed descriptions of each of the modelling scenarios addressed by the Forest WG.

CONTENTS

1. INTRODUCTION

1.1. BACKGROUND

Forests are extensive natural resources that provide economic, nutritional, recreational and social benefits to people in many countries. Following the Chernobyl accident, radioactive contamination of forests occurred in Ukraine, Belarus and Russia, and also in many countries beyond the borders of the former Soviet Union, notably Finland and Sweden. The degree of contamination with ^{137}Cs in these countries ranged from >10 MBq m^{-2} in some locations to between 10 and 50 kBq m^{-2}, the latter range being typical of ^{137}Cs deposition to several countries of western Europe. In each of these countries, not only do forests provide an economic resource of major importance, but they are at the heart of many social and cultural activities which, in some cases, have been curtailed by the deposition of ^{137}Cs in 1986. Despite such problems, when compared with the research efforts committed to understanding the radiological impact of ^{137}Cs contamination of agriculture, contaminated forest ecosystems have remained on the periphery of interest of many national radiological protection organisations.

In the fourteen years since the Chernobyl accident it has become apparent in countries across Europe and the former Soviet Union that natural decontamination of contaminated forests is proceeding extremely slowly. Since net export of ^{137}Cs from forest ecosystems has been determined to be less than 1% per year [Tikhomirov et al, 1993, Nylen, 1996], it is likely that, without artificial intervention, it is the physical decay rate of ^{137}Cs that will determine the duration over which forests continue to be affected by the Chernobyl legacy. Despite the fact that the absolute natural losses of ^{137}Cs from the forest are minimal, recycling of ^{137}Cs within the forest is a dynamic process in which reciprocal transfers occur on a seasonal, or longer-term, basis between biotic and abiotic components of the ecosystem. In order to facilitate long-term management of forests which may, potentially, represent a radiological hazard to both human and non-human populations, a reliable understanding of these exchange processes is required. Primary information on such processes is obtained from experiments and field measurements, but for highly diverse systems such as forests it is inevitable that this information can be better assimilated in a holistic sense through the application of mathematical models.

Radioecological models can be used to simulate system responses to radionuclide inputs and system manipulations of various kinds and, thereby, to assist in the assessment of different contamination scenarios and post-contamination management options. For any reliability to be placed on system simulations, however, it is essential that models are correctly scrutinised by appropriate validation studies. Validation exercises involving more than one model and more than one group of model authors/users are likely to yield the most satisfactory results since this approach should ensure that an individual model is tested in scenarios which are not restricted to the original situations or cases for which that model was originally developed. The activities of the Forest Working Group (Forest WG) within the IAEA's BIOMASS programme were designed to provide an international platform on which forest radioecology models could be tested.

The Forest WG activities within the IAEA BIOMASS programme have their roots in the preceding VAMP (Validation of Environmental Model Predictions) programme, which ran from 1988 to 1993. One of the themes within the Terrestrial Working Group of VAMP was 'Food Chain Transfer in Natural and Semi-Natural Ecosystems'. Forests belong to this group of ecosystems and a small Forest Group began to address the problem of appropriate model design and data sources as part of the working group's activities. Unfortunately, at that time, there were no suitable radioecological models available for forests which could be put forward for a validation study. However, by the time the BIOMASS programme began this situation had changed radically. It was evident in 1997/98 that several research groups across Europe and North America had developed models which could be used to simulate radiocaesium behaviour in forest ecosystems (e.g. in EU Projects such as Seminat and Landscape).

1.2. OBJECTIVES

The main purpose of the BIOMASS Forest WG was to provide a forum for inter-comparison and validation of models designed to simulate the ecosystem behaviour of radionuclides, especially ^{137}Cs, within forest ecosystems of the temperate and boreal latitudes.

Five specific objectives were drawn up for the Forest WG, as follows:

(1) To bring together modellers in the field of radionuclide transfer within forests to facilitate exchange of information and peer review.

(2) To undertake model inter-comparisons to identify and investigate significant areas of uncertainty and differences in approach.

(3) Where possible and practicable, to undertake testing and validation of existing or new models against independent data sets.

(4) To take account of the wider implications of the specific results of the Forest WG by developing a list of Features, Events and Processes (FEPs) for forests.

(5) Based on the above, to make recommendations for the future direction of the modelling of radionuclides in forests.

1.3. SCOPE

It is important to recognise that the model testing activities listed in Section 1.2 are closely associated with model building and data gathering. The activities of the Forest WG have also provided an opportunity to consolidate existing and new information on such key issues as biogeochemical cycling within forests and the use and abuse of transfer parameters. Over and above these activities, however, the Forest WG has attempted to bring together forest modellers and experimentalists from 12 countries to share their models, their data and their opinions on the status of forest radioecological modelling at present and where it should be heading in the future.

1.4. STRUCTURE

This report provides a summary of the activities of the Forest WG, based on the above objectives, between 1998 and 2000. Sections 2, 3 and 4 are devoted to reviews of the biogeochemical cycling of radiocaesium in forest ecosystems, the application of the interaction matrix method to forest model design, the definition and application of transfer factors to forests and a new method of determining radiocaesium uptake within the woody tissues of trees. Sections 5, 6 and 7 describe the three model inter-comparison exercises (Scenarios 1, 2 and 3, respectively) carried out with the participation of 11 modelling groups. Finally, Sections 8 and 9 summarise the main conclusions from the activities of the Forest WG and make recommendations for future experimental and modelling studies within the broad field of forest radioecology.

Appended to the report are summary descriptions of the individual models that participated in the Forest WG modelling studies, as well as detailed descriptions of each of the modelling scenarios addressed by the Forest WG.

2. RADIOCAESIUM CYCLING IN FOREST ECOSYSTEMS

2.1. BIOGEOCHEMICAL CYCLING OF RADIOCAESIUM IN FORESTS

The study of elemental cycling within environmental systems allows us to determine the dominant processes controlling the transport of elements in the system and is usually a prerequisite to modelling the system. Normally, a conceptual biogeochemical cycle for a forest ecosystem is represented as the combination of two interconnected cycles, namely the geochemical and biological cycles. The geochemical cycle is an open cycle comprising two main components – the input and output of an element into or out of the forest system. The geochemical cycle is, in fact, intimately connected to the biological cycle which comprises the fluxes of elements mobilised during biomass growth and degradation. The schematic figure below illustrates a generally accepted forest biogeochemical cycle, applicable to radiocaesium, in terms of the major compartments and fluxes.

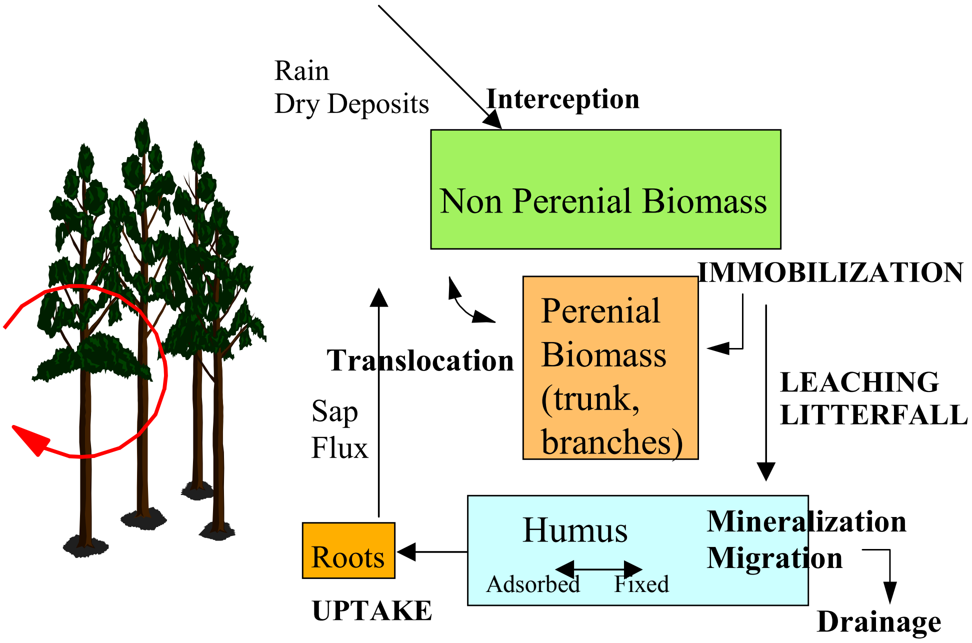

FIG. 2.1. A general scheme of the biogeochemical of radiocaesium within a forest ecosystem.

Most of the recent modelling developments in forest radioecology deal with the cycling of ^{137}Cs deposited in 1986 as a result of the Chernobyl accident. Recent reviews and analyses of information available on processes and models of radiocaesium cycling in forests have been provided through peer-reviewed publications [Thiry, and Myttenaere 1993; Myttenaere et al 1993; Nimis 1996; Mamikhin et al 1997; Avila et al 1998; Ipatyev et al 1999 and Thiry et al 2000], within conference proceedings [Linkov and Schell, 1999; Riesen et al 1999; and Delvaux et al 1999] or in PhD theses [Linkov 1995; Thiry 1997 and Avila 1998].

Following deposition of ^{137}Cs from the Chernobyl plume, the primary source of tree contamination (60-90%) was direct interception of aerosol-associated radiocaesium 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 was due to two main processes. The first of these was a dominant and rapid self-decontamination process of the tree canopy, effected by precipitation wash-off (throughfall) and litterfall, and this was followed by root uptake which has prevailed over the longer term as radiocaesium has migrated into the soil profile. Just as in the case of its nutrient analogue, potassium, the rate of radiocaesium cycling within forests is relatively rapid and quasi-equilibrium of its distribution is probably reached a few years after atmospheric fallout. The upper, organic-rich, soil layers act as a long term sink but also as a general source of radiocaesium for contamination of forest vegetation, although individual plant taxa differ greatly in their ability to accumulate radiocaesium from this organic soil. Output from the system via the drainage water is generally limited due to radiocaesium fixation on micaceous clay minerals [Nylen 1996]. An important role of the forest vegetation in the recycling of radiocaesium in the forest is the partial and transient storage of radiocaesium, particularly in perennial woody components such as tree trunks and branches which can have a large biomass. The major portion of radiocaesium accumulated by vegetation from the soil, however, is recycled annually through leaching and litterfall, resulting in the long-lasting biological availability of radiocaesium in surface soil. Internal translocation of radiocaesium within vegetation also occurs, but involves generally low radiocaesium activities compared with exchange (uptake/return) between the soil and the forest vegetation.

2.2. TOWARDS A GENERAL, CONCEPTUAL MODEL OF RADIOCAESIUM CYCLING WITHIN FORESTS

As implied by the preceding section, biogeochemical cycling integrates most of the driving functions of the ecosystem and its study thus involves the ecosystem as a whole. Ecosystem functioning involves, *inter alia*, a continual exchange of elements between various biotic and abiotic components. These exchanges also apply to trace contaminants like radiocaesium, the rate of recycling of which varies with factors such as climate, soil type, vegetation species and with the stage of tree development.

Different interpretations of the spatial and temporal complexities of forest ecosystems can result in different conceptual models of biogeochemical functioning of a forest. A simplified model was proposed by Ulrich [1973], developed by Cole and Rapp [1981] and refined by Ranger and Bonneau [1984]. This model stratifies the ecosystem into compartments delineated by the specificity and homogeneity of their respective fate. The main components are diverse compartments of the vegetation, the soil organic and mineral horizons and their soil solution, and certain key fluxes which are directly measurable (*i.e.* atmospheric inputs, leaching and water drainage, litterfall). A mass inventory of elements can be defined for each compartment and fluxes are deduced from input-output measurements. Certain fluxes which are not directly measurable, such as root uptake, can be calculated. This approach allows us to quantify the dominant elemental fluxes within the ecosystem and to establish an overall elemental balance. Furthermore, when this approach is applied to a suitable forest chronosequence [Cole and Van Miegroet 1989], the effect of forest age and stage of development on elemental fluxes can be estimated.

During the last decade, numerous studies of radiocaesium transfer have been conducted at a wide range of forest sites located across Western Europe and the countries of the CIS. In such studies, the extent of radiocaesium exchange between forest compartments such as soil and vegetation is most often characterised through the determination of a transfer coefficient or factor (TF) even though the use of the TF to quantify the dynamics of radiocaesium exchanges in forest ecosystems presents obvious limitations. Even at those sites where the dynamics of radiocaesium cycling have been studied, the bulk of research has often been limited to single biogeochemical pathways rather than examining the system *in toto*. In recently published reviews, therefore, reasonably complete data for radiocaesium cycling within forests have not been widely available. Thus, there is still a need for a standardization of current approaches to data acquisition in a format which can be used to facilitate the comparison of radiocaesium cycling at different geographical locations and, preferably, on a whole-ecosystem basis. An alternative way to improve the development of conceptual models is the application of a systematic method of identifying dominant features, events and processes (FEPs) using an 'interaction matrix' approach [Avila and Moberg 1999]. The reliability of simulation models depends largely on our knowledge of individual processes represented within the model and the way in which these interact. Objectivity in identifying and characterising these processes can be increased by involving a number of experts in creating and coding the matrices.

The advantages of the 'interaction matrix' approach for designing and testing mathematical simulation models, and the adequacy of transfer factors to assess radiocaesium redistribution in specific biological components of forest ecosystems, are discussed below in Section 2.3 and Section 3, respectively.

2.3. A SYSTEMATIC APPROACH TO THE MIGRATION OF ^{137}CS IN FOREST ECOSYSTEMS USING INTERACTION MATRICES

As described above, the migration of radionuclides in a forest ecosystem is the result of multiple biotic and abiotic interactions between many components. When developing conceptual and mathematical models of such complex systems, there is always a risk that important components and/or interactions will be omitted or underestimated. This risk can be reduced if a systematic approach to model development is applied, for example by using interaction matrices [Avila and Moberg 1999]. This method (the matrix method) was used in the BIOMASS Forest WG to develop a general conceptual model of radiocaesium migration in the forest, which was afterwards used as a reference for identifying differences and similarities between the compared models.

The essence of the matrix method is to study processes occurring in a system using an interaction matrix. In such a matrix the components of the system in question are elements in the leading diagonal (top left to bottom right) and the interactions between these are the off-diagonal elements. The number of diagonal elements will be a measure of the resolution of the matrix (*i.e.* the degree of complexity, or simplicity, of the interpretation of the system by the modeller). The larger the number of leading diagonal terms, the higher the number of possible interaction terms and the higher the resolution. For a matrix with N diagonal terms, there are N (N-1) interaction terms. A key question determining the complexity of the model developed as a result of the matrix development is the selection of the diagonal elements and the optimal resolution based on the modeller's knowledge about the system. There is no universal objective method for this and, thus, the selection of diagonal elements strongly depends on the expert judgement of the person building the matrix. One way to increase the objectivity of this process is to involve several experts, for instance a multidisciplinary group such as the Forest WG. To make the matrix more useful for studies of cause-effect relationships, pathway analysis, *etc.*, the diagonal elements should be selected in such a way that as many binary interactions as possible are placed in off-diagonal elements. The resulting matrix can be checked for completeness by assuming that each binary interaction is, in principle, possible and by a comparison with information on these interactions to be found in the literature.

The matrix shown in Figure 2.2 was developed by the BIOMASS Forest WG. This matrix can be regarded as the general consensus within the group of the principal transfer processes that are relevant and necessary to describe the migration of ^{137}Cs in a forest ecosystem. It also shows a conceptualisation that was considered suitable to represent the interactions prevailing in the system. Hence, the matrix in Figure 2.2 is also a conceptual representation of a compartmental model of radiocaesium cycling in a forest ecosystem, in which the diagonal elements represent the compartments and the off-diagonal elements represent the transfer pathways between the compartments.

The derivation of a mathematical model from an original conceptual model is not a straightforward task and different modellers will have their own approaches to this problem. When the modelled system is complicated it is difficult to foresee which level of aggregation is optimal and which interactions or pathways should be included. A rather common approach is to start with a very detailed model (a ‘research’ model), which is afterwards simplified by sensitivity analyses and screening procedures. However, this method can not always be applied due to a lack of knowledge of parameter values for the many exchanges and interactions which a research model usually involves. Such models also require large calculation efforts.

Atmosphere	intercept. rainfall snowfall	intercept.rainfall snowfall			intercept. rainfall, snowfall			intercept. rainfall snowfall	intercept. rainfall snowfall	intercept. inhalation
Transpir.	Tree leaves	weathe-ring	translocation	translocation	leaf fall, weathe-ring			weather. intercept.	weather. intercept.	ingestion
		External bark	translocation		weather. intercept.			weather. intercept.	weather. intercept.	ingestion
	translocation	translocation	Living wood	translocation		fertilisation	fertilisation	mycorrhizae transfer		ingestion
			translocation	Dead wood						
Resuspension		rain splash	root uptake		Litter	Decomp. Percolation soil biota		uptake	rain splash, root uptake	ingestion
			root uptake			Soil organic	percolat. diffusion Advect., soil biota	uptake	root uptake	
			root uptake			diffusion, capillary rise, soil biota	Soil Mineral	uptake	root uptake	
			root upt. (mycorrhizae)		fertilisation	fertilisation	fertilisation	Fungi	Root upt. (mycorrhizae)	ingestion
transpir.					leaf fall, weather. intercept.	fertilisation	fertilisation	mycorrhizae transfer	Under storey	ingestion
					fertilisation					Game

FIG. 2.2. An interaction matrix which describes the migration of ^{137}Cs in a forest ecosystem. The diagonal elements are components of the system (i.e. model compartments) and the off-diagonal elements are the intections between them (transfer processes between compartments). In order to identify the transfer processes the matrix should be read clockwise.

An alternative method is to start the process by building a simple compartment model, which includes only the major pathways. These can be identified after ranking the pathways with the help of an interaction matrix and an initial version of the model would include only those pathways with the highest rank. Thereafter, other pathways can be systematically added to this model according to their rank. After each addition the effect on a predefined optimisation function is evaluated. The uncertainty of the estimations or the differences between the estimations and a set of experimental data are examples of possible optimisation functions. The addition of pathways ends when the desired level of the optimisation is reached. So, the matrix in Figure 2.2 can be used by modellers as a starting point for the derivation of models with more specific endpoints. Other practical uses of this matrix are discussed in Avila and Moberg [1999].

2.4. COMPARISON OF FOREST WORKING GROUP MODELS WITH MATRIX

The models which participated in the three inter-comparison studies undertaken by the BIOMASS Forest WG are listed in Table 2.1. A brief description of each model can be found in Annex A. The forest ecosystem compartments (and intercompartmental transfers) depicted in the matrix in Figure 2.2 are represented to a varying degree in these models. Some of the 11 compartments and 75 transfers depicted in Figure 2.2 are represented explicitly in the models, but others are only represented implicitly. Explicit/implicit representation of compartments and transfers in each model as a percentage of the total is shown graphically in Figures 2.3 and 2.4.

The degree of representation by each of the participating models of compartments and transfer processes identified in the matrix is of interest since, as stated in Section 2.3, the matrix represents a consensus view of what an idealised forest model should represent. Deviations from this idealised model could lead to enlargement of Type B uncertainty in individual models, i.e. those uncertainties due to incorrect representation of compartments or processes within in a model [Hoffman and Hofer, 1988]. Explicit representation of the 11 compartments identified in the matrix varied from 90% to 45% in the models included in the Forest WG inter-comparison exercises. This discrepancy, which indicates the variability in the level of complexity of each of the models, can be explained by the fact that the models identified in Table 2.1 were either conceived as general forest radioecology models, which attempt to model the behaviour of radiocaesium in the forest system as a whole, or as more focussed models intended to address specific transport pathways, such as soil migration and uptake of radiocaesium by understorey vegetation. Hence, models such as FORESTLAND, FAO and FORM, each designed to simulate radiocaesium behaviour in the forest system as a whole, represent explicitly 90% of the 11 possible compartments. In the case of the FAO model, the remaining 10% of compartments is represented implicitly. In contrast, FORSUN represents explicitly only 45% of the 11 compartments identified in the matrix. This model was more narrowly focussed on the problem of soil-plant transfer of radiocaesium to understorey vegetation in the forest ecosystem.

While it is perhaps straightforward for modellers to agree on the notional compartments of the forest ecosystem which should be represented within their idealised model, it is far less certain which of the processes effecting transfer between these compartments should be represented. Indeed, it is often the case that the operation and significance of specific transfer processes are in doubt, due either to a lack of fundamental knowledge concerning the process or a lack of quantitative data on the rates of individual processes. A very good example is the role of mycorrhiza in the transfer of radiocaesium from forest soils to trees and understorey vegetation.

TABLE 2.1. MODELLERS AND MODELS PARTICIPATING IN THE BIOMASS FOREST WG INTER-COMPARISON STUDIES

Modeller(s)	Model	Institute
R. Avila and L. Moberg	FORESTLAND	SSI, Stockholm, Sweden
G. Shaw	RIFE	Imperial College, UK
S. Fesenko and S. Spiridonov	FORESTLAND	RIARAE, Russia
R. Bergman	FOA	NDRE, Umea, Sweden
P. Calmon	RODOS	IPSN, France
A. Dvornik and T. Zhuchenko	FORESTLIFE	BFI, Gomel, Belorus
M. Frissel	FORM	Consultant (IAEA)
I. Linkov	FORESTPATH	Harvard University, USA
S. Mamikhin	ECORAD	MSU, Moscow, Russia
A. Rantavaara	FINNFOOD	STUK, Helsinki, Finland
A. Konoplev and A. Bulgakov	FORSUN	Typhoon, Obninsk, Russia
A. Konoplev and A. Bulgakov	FORWASTE	Typhoon, Obninsk, Russia
A. Rantavaara and J. Wendt	S-RODOS[1]	STUK, Helsinki, Finland
M. Scimone	LOGNAT	Trieste, Italy

The general uncertainty surrounding the significance of many of the possible transfer processes identified in Figure 2.2 is reflected in the rather low degree of representation of the 75 processes shown in the matrix in the models themselves. Explicit representation of these processes ranged from 50% in the FAO model to only 10% in the RIFE model. Figure 2.4 also indicates that in some models (RIFE and ECORAD-C) the implicit representation of transfer processes accounted for 50% or more of the total number of transfers included in the model. This indicates that lumping together of transfer processes into single effective transfer rates or transfer coefficients is a common and perhaps inevitable approach to modelling transfer of radionuclides in complex ecosystems such as forests for which there is still incomplete information on transfer rates and processes (see Section 3).

This comparison of the models used within the Forest WG with the matrix shown in Figure 2.2 illustrates that, while it is certainly possible to obtain agreement from a diverse group of modellers on which components and transfer processes should be represented in an idealised forest radioecology model, the working models which individual modellers have built are inevitably designed and modified to suit the individual needs of the modeller. The ‘ideal’ forest radioecology model has yet to be constructed.

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[1] S-RODOS is a dynamic submodel for tree and understorey calculations in the RODOS model.

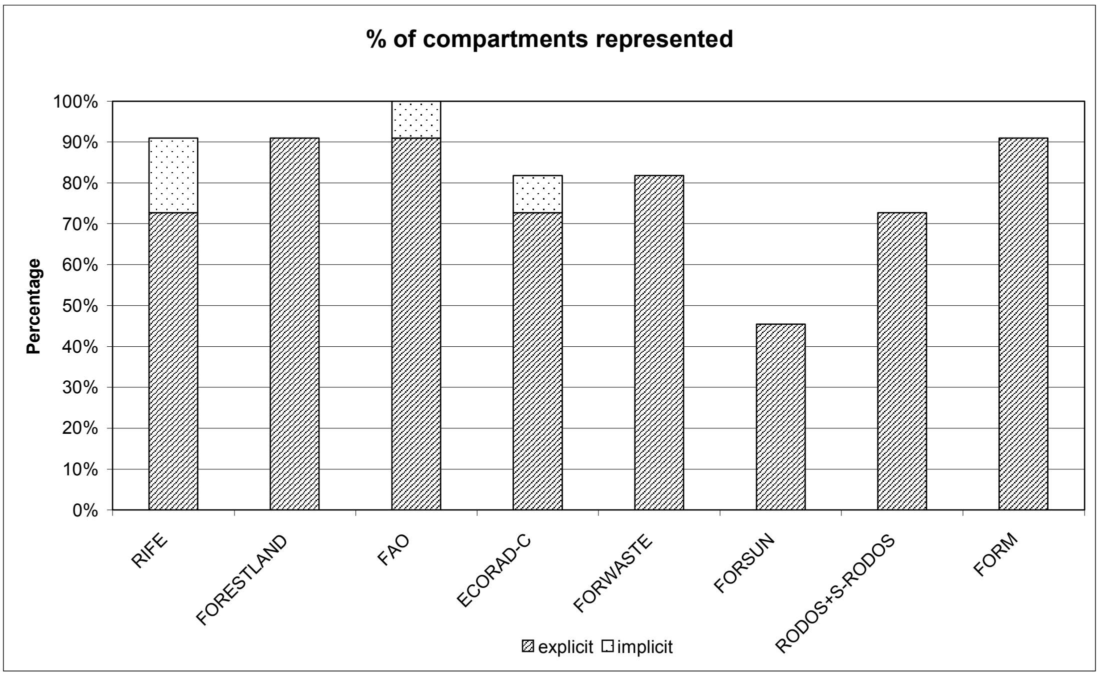

FIG. 2.3. Explicit/implicit model representation of Forest Ecosystem compartments listed in the Forest Matrix (Figure 2.2).

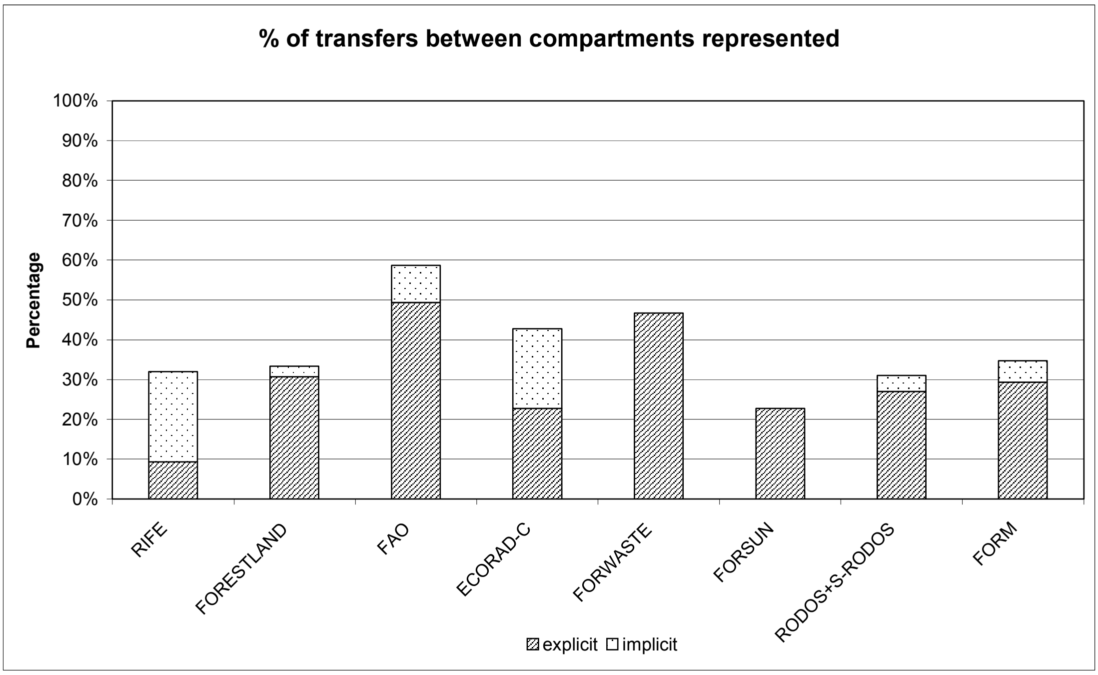

FIG. 2.4. Explicit/implicit model representation of transfers between forest ecosystem compartments listed in the Forest Matrix (Figure 2.2)[2].

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[2] Models such as FORSUN and RODOS were still under development during this study.

3. DEFINITIONS OF TRANSFER PARAMETERS FOR UNDERSTOREY VEGETATION AND FUNGAL FRUIT BODIES

3.1. INTRODUCTION

The quantitative evaluation of radioactivity levels in fungal fruit bodies and understorey vegetation is a challenging task. The concept of transfer factors and concentration ratios, commonly used in agricultural radioecology, has also been applied extensively to quantify the transfer of radionuclides from soil to fungi and understorey vegetation in non-agricultural situations. This concept has been recommended by several agencies: IAEA [1994], IUR [1992] and ICRP [1979]. In general, transfer factors are expressed as parameters representing the contamination (activity concentration) of green plants or fungal fruit bodies divided by parameters representing the contamination (activity concentration or inventory) of soil. It is implicitly assumed that the radionuclide activity concentration in plants or fungal fruit bodies can be described by a linear function of soil contamination passing through the origin. The contamination of plants and fruit bodies is usually expressed as the amount of radioactivity per unit weight, either on a dry weight (Bq kg^{-1} dry weight) or a fresh weight basis (Bq kg^{-1} fresh weight). The contamination of soil is usually expressed as the amount of radioactivity per surface area (*i.e.* an inventory, Bq m^{-2}) or per unit dry weight (Bq kg^{-1} dry weight), where the latter definition commonly refers to standardized soil depths.

Until now, there has been no universal agreement as to whether transfer parameters should refer to the total radionuclide inventory in soil, the activity concentration within a certain soil depth or specifically the activity concentration of the soil layers exploited by fungal mycelia or fine roots. In general, there is no ‘best definition’ for transfer parameters, since these empirical parameters are often intended to be used in different ways in particular radioecological models. Consequently, the suitability of a certain definition of transfer parameters depends on the purpose of the radioecological model and the knowledge about the ecosystem to be modelled. If, for example, activity levels have to be estimated very quickly after an accidental release of radionuclides, aggregated transfer factors, which refer to the total deposition on the ground surface, might be a good choice to assess the order of magnitude of contamination levels to be expected in fungal fruit bodies and green plants. For longer-term predictions of activity levels, however, more sophisticated definitions of transfer parameters might be more suitable. It is commonly found that concentration ratios explicitly referring to the soil horizons within which fungal mycelium or fine roots are located, work well, notably for sampling sites with a significant vertical migration of radionuclides in the soil.

Despite the conceptual simplicity of most transfer parameters, their inherent limitations should always be acknowledged and taken into account to avoid erroneous applications of the parameters. It is generally accepted that the large observed variations of radionuclide levels in fungi and green plants cannot be entirely explained by varying concentrations in the soil alone. The radionuclide activity concentrations in fungi and green plants depend also on other ecological parameters, such as the type of soil, its content and type of clay minerals, *etc.* In addition it is well known that green plants have the ability to control the uptake of ions, thus regulating their content of essential elements. Wirth et al [1985] argue that there are only slight variations in the concentrations of essential elements in green plants, and therefore the variances of the corresponding transfer factors mainly represent the concentration variance of the essential elements in soil. In this case, the naive application of transfer factors or concentration ratios would lead to values that decrease with increasing soil content. McGee et al [1996] conclude that such a decreasing concentration ratio value is a mathematical artefact that arises as a result of dividing a denominator (soil concentration) which shows considerable variation, into a numerator (plant concentration) which is relatively constant. A critical review of the concept of ratios in the field of radioecology was published by McGee et al [1996].

A considerable lack of knowledge exists concerning the mechanisms and processes involved in radionuclide uptake and retention by fungi and green plants which, in forest soils, often live in close symbiotic relationships. It is therefore not surprising that transfer factors and concentration ratios are a popular empirical parameter to describe quantitatively the uptake of radionuclides from soil to fungi and green plants, notably in forest ecosystems.

3.2. DEFINITIONS OF TRANSFER FACTORS

This section presents common definitions of transfer factors for soil – green plant transfer and soil – fungi transfer of radionuclides. The specific advantages and limitations of different definitions are briefly discussed.

3.2.1. Aggregated transfer factors

Aggregated transfer factors (often referred to as T_{agg} values) are defined as the ratio of the activity in plant (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the total deposition on the soil (Bq m^{-2}). Sometimes, other area-related definitions of transfer factors, *e.g.* area-related transfer factors referring to a certain soil depth instead of the total deposition, are also called aggregated transfer factors, a fact that might lead to confusion.

The concept of aggregated transfer factors was developed, *inter alia*, to avoid difficulties in defining a suitable average radionuclide activity concentration in soils with a multi-layered structure, such as podzols, which are common in many natural and semi-natural environments. Aggregated transfer factors are a useful tool to estimate quickly, but only approximately, the uptake of radionuclides by vegetation, often during the short-term following accidental release of radionuclides but also over the long term (ie. decades). However, aggregated transfer factors suffer from some disadvantages in connection with radioecological models. Since they refer to the total deposition on soil, they usually exhibit a time-dependence: this effect is particularly pronounced for plants and fungal species which exploit distinct soil horizons, especially when significant vertical migration of radionuclides occurs within the soil. A wide range of aggregated transfer factors (more than four orders of magnitude) has been reported for fungi. As will be discussed later, this large variation is, at least partially, attributable to non-uniform vertical distributions of radionuclides in the soil and the fact that fungal mycelia often colonise distinct soil horizons.

3.2.2. Transfer factors (concentration ratios) referring to standardized soil depths

Transfer factors referring to standardised soil depths are defined as the ratio of the activity concentration in plant (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration in soil (Bq kg^{-1} dry weight) within the uppermost layer of a standardised soil thickness. This definition was designed especially for agricultural ecosystems, where radionuclides are often distributed homogeneously within the rooting depth of agricultural plants due to ploughing.

As in the case of aggregated transfer factors, transfer factors defined on the basis of standardised soil depths are of limited usefulness in the case of soils with a multi-layered structure and a pronounced vertical profile of activity concentration. Averaging the radionuclide concentration over a standardised soil depth, irrespective of the location of the mycelium or the fine roots, might lead to a large variation and a time dependence of transfer factors. Details will be discussed in Section 3.2.4.

3.2.3. Transfer factors soil solution – plant

Several studies [Horrell et al 1990; Desmet et al 1991; Schell et al 1996, Mytenaere et al 1993 and Tikhomirov et al 1993] suggested that the bioavailability of a given radionuclide in soil, not its bulk concentration, is important for uptake by fungi and green plants. Expressing transfer parameters on a dry soil basis has been criticised as being inadequate. Desmet et al [1991] argue that the metal ion concentration and its chemical form in soil solution are the most important factors in determining the root uptake and transfer to plant tissues. Consequently, the possibility to calculate transfer factors as the ratio of the activity concentration in plant (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration in soil solution (Bq l^{-1}) has been discussed. Nevertheless, Desmet et al [1991] state that long-term bioavailability relates to the more strongly binding organic complexes and minerals and to the solid phase of the soil. In this context, the available fraction has been defined qualitatively as the fraction of an element that has the *potential* to move into the soil solution.

Experimental results for forest soils have revealed that the fraction of easily exchangeable radiocaesium in organic horizons is low but, nevertheless, radiocaesium is highly available for uptake by fungi. This effect can be very pronounced for organic horizons of forest soils. At present, there is no experimental method available to quantify the bioavailable fraction of radionuclides in soil under semi-natural conditions. The results of field studies indicate that most radiocaesium in the organic horizons may be available for uptake by fungi. Rühm et al [1999] reported that the bioavailabilities of stable ^{133}Cs and radioactive ^{137}Cs and ^{134}Cs are approximately the same in the organic horizons of a German forest site.

3.2.4. Transfer factors relating to specific soil horizons

Transfer factors (concentration ratios), defined as the ratio of the activity concentration in fungal fruit bodies or green plants (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration of the specific soil layer exploited by the mycelium or the root system (Bq kg^{-1} dry weight) have proved to be useful, especially for application in dynamic radioecological models. This definition of the transfer factor was proposed in the late 1980s.

It is very difficult to determine the precise vertical location of fungal mycelium within soils on a species-by-species basis. In the case of radiocaesium, Byrne [1988] and Guillitte et al [1990] suggested that the isotopic ratio of ^{134}Cs/^{137}Cs could be used for this purpose. The approach is based on the idea that the isotopic ratio in fungal fruit bodies should reflect the isotopic ratio of the soil horizon from which radiocaesium is predominantly taken up. At several sampling sites the time-dependent isotopic ratio ^{137}Cs/^{134}Cs has turned out to be a ‘fingerprint’ of the different layers of forest soil, a consequence of the mixing of the residual ^{137}Cs from global fallout from atmospheric nuclear tests with ^{134}Cs and ^{137}Cs from the Chernobyl fallout. Hence, the location of fungal mycelia in forest soils can be determined by comparing the isotopic ratios of ^{137}Cs/^{134}Cs in fruit bodies with the corresponding values of different soil horizons. This basic idea was developed in an operational tool by Rühm et al [1997] who determined the vertical location of the mycelia of 14 fungal species in German forests.

As an illustration, Figure 3.1 presents the isotopic ratios of ^{137}Cs/^{134}Cs as functions of time measured in samples of *Clitocybe nebularis* and *Russula cyanoxantha,* together with 95 percent confidence bands. The ^{137}Cs/^{134}Cs ratios for different soil horizons are also shown for comparison. Obviously, *Clitocybe nebularis* has a superficial mycelium located in the L and/or Of horizon. The ^{137}Cs/^{134}Cs ratios in samples of *Russula cyanoxantha* are significantly higher and indicate that this symbiotic species obtains radiocaesium from both Oh and Ah horizons.

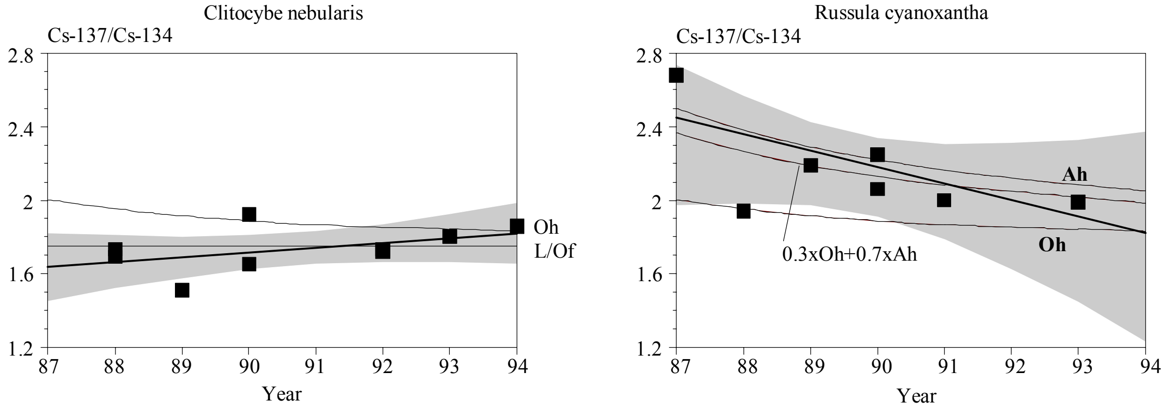


FIG. 3.1. ^{137}Cs/^{134}Cs ratios as functions of time in two mushroom species at a German sampling site, decay-corrected to May 1, 1986. The black boxes denote measurements. The thick full line represents linear regression curves and the shaded areas are the corresponding 95 percent confidence bands. The predictions of a compartment model for the isotopic ratios of different horizons are shown for comparison.

The advantage of transfer factors referring to specific soil horizons, from which nutrients and radionuclides are mainly taken up, is three-fold:

(1) The variation of transfer factors for specific horizons is significantly lower compared with other definitions, notably in the case of a non-uniform vertical distribution of radionuclides in soil. Guillitte et al [1990] even conclude that, in the case of fungi, “soil sampling at a constant depth has no practical interest when the element is unequally distributed across the soil profile”.

(2) Transfer factors for specific horizons offer a conceptual advantage for predictive modelling. Transfer factors defined in this way have not changed significantly over a period of several years after the Chernobyl accident [Rühm et al 1998] and there is evidence that they will stay fairly constant in future [Rühm et al 1999].

(3) Transfer factors for specific horizons are a direct measure of the availability of radionuclides for uptake by fungi or green plants.

The methodology described above is very difficult to apply now, since ^{134}Cs is a short-lived radionuclide. Currently, research projects are dealing with the question of whether the ratio ^{137}Cs/stable caesium can alternatively be used to localise fungal mycelia or fine roots of green plants *in situ*, an approach which is expected to work well for organic horizons of forest soils. In organic soil layers, where the fraction of caesium fixed within mineral particles is likely to be very small, the ratios of radiocaesium/stable caesium in fruit bodies were close to those of the soil layers, from which certain species of fungi take up radiocaesium [Rühm et al 1999]. If mycelia colonise deeper soil horizons, where the concentration of mineral particles is usually increased, the ratios of radiocaesium/stable caesium in fruit bodies can be higher compared with the ratio in the corresponding soil layer [Tsukada et al 1998; Yoshida et al 1998]. This trend is to be expected if a significant fraction of stable caesium is enclosed in mineral particles and thus not available for uptake by fungi.

3.2.5. Rhizosphere – plant transfer factors

Based on the results of a recent study, Delvaux et al [2000] proposed transfer factors that refer to the rhizosphere, *i.e.* the volume of soil influenced by plant root activity. Plant roots are "dynamic weathering agents" which strongly modify their soil environment while taking up their nutrients. The uptake of potassium, for example, induces potassium depletion around plant roots and hence a rapid weathering of mica through the release of interlayer potassium. Delvaux et al [2000] argue that the mobilisation of otherwise unavailable radiocaesium might be directly linked to this process.

The concept of rhizosphere – plant transfer factors under extreme potassium deficiency has been tested with pot experiments, which were specifically designed to investigate the rhizospheric effects of radiocaesium uptake [Delvaux et al 2000]. A root mat of ryegrass was brought into close contact with a soil-agar mixture spiked with carrier-free ^{137}Cs. The plants were supplied with a nutrient solution free of potassium to simulate the potassium depletion in the rhizosphere of unfertilised soil. The rhizosphere – plant transfer factor could not be related to any common physical or chemical soil property but was significantly correlated with the radiocaesium interception potential (RIP). Thus, the RIP quantitatively relates the rhizospheric mobilisation of radiocaesium with an intrinsic caesium binding property of soils.

Up until now, the concept of rhizospheric transfer factors has not been applied to field investigations. Moreover, biological interactions, such as the effects of mycorrhizal fungi, which occur particularly in the upper horizons of forest soil, have not yet been considered.

3.3. CONCLUSIONS

The concepts of transfer factors and concentration ratios provide a popular approach to quantify the transfer of radionuclides from soils to plants and fungal fruit bodies. This section has reviewed common definitions of transfer factors and highlighted their advantages and limitations. It has been shown that there is no 'best choice' of any particular type of transfer parameters since, in most cases, these empirical parameters do not stand alone but belong to a specific radioecological model. Consequently, the suitability of a particular definition of a transfer parameter depends on the purpose of the radioecological model, the manner in which the data to calculate the transfer parameter have been obtained and the level of understanding of the ecosystem to be modelled (usually limited by field data, either obtained directly by measurement or from the literature).

In the past, numerous values for transfer factors have been published in the literature. In many cases, however, the authors did not specify the precise definition of the transfer factors they used. This might lead to confusion and erroneous application of such transfer factors. Area-related transfer factors, for example, are often called 'aggregated transfer factors', even if they do not refer to the total inventory of radionuclides within the soil. It is strongly recommended that any published transfer factor values are accompanied by a detailed description of the way in which the values were obtained and how they should be used to calculate soil-to-plant or soil-to-fruit body transfer.

This section has been exclusively concerned with the definition and application of transfer factors for herbaceous vegetation and fungal fruiting bodies. Quantification of the uptake of radiocaesium by trees for the purposes of predictive modelling has been given very little consideration in comparison with these vegetation types. This problem is addressed in Section 4.

4. THE ADEQUACY OF TRANSFER FACTORS TO ESTIMATE LONG TERM ACCUMULATION OF RADIOCAESIUM IN WOOD

4.1. INTRODUCTION

In forests affected by Chernobyl fallout, long-term management of ^{137}Cs-contaminated wood stocks requires a reliable tool to estimate the evolution of the ^{137}Cs content in timber wood in the future [Ipatyev et al 1999]. In the long term, root uptake is the dominant source of ^{137}Cs contamination in growing wood. As for other radionuclides, ^{137}Cs contamination in forest vegetation is generally assessed with Transfer Factor (TF) or aggregated Transfer Factor (T_{agg}) coefficients that express the ratio of the average radionuclide concentration in a plant compartment (Bq kg^{-1}) to that in soil (Bq kg^{-1} for TF or Bq m^{-2} for T_{agg}). Depending on the time elapsed after an atmospheric pulse, it is not possible to distinguish whether the measured radiocaesium contained in timber wood compartment originated from initial atmospheric deposits or from root uptake [Ertel and Ziegler 1991; Bonnett and Anderson 1993; Fawaris and Johanson, 1994; Barci-Funel et al 1995; Haas et al 1995]. TF's refer to the total content of ^{137}Cs in wood and can therefore not be used to distinguish between the respective contributions of each process. The adequacy of TF's to predict future ^{137}Cs accumulation in wood due to root uptake is therefore questionable, as clearly illustrated by comparing TF values with the real ^{137}Cs stock in wood in two neighbouring contaminated Pine (*Pinus sylvestris L.*) stands (17 and 58 years) located near Gomel in Belarus and affected with similar ^{137}Cs deposits (Table 4.1).

With increasing age, measured TF values decrease while ^{137}Cs total content in trunk wood is greatly enhanced. Moreover, the increase in standing wood biomass can only partly explain the discrepancy in ^{137}Cs amounts accumulated in timber wood between young and old stands. This indicates that historical accumulation of ^{137}Cs in wood does not simply relate to root uptake as the main vector of wood contamination. Initial interception and subsequent incorporation of ^{137}Cs were likely very important in older trees.

Radiocaesium is indeed highly mobile and rapidly recycled in trees. Stemwood acts as an important reservoir of ^{137}Cs but its radial distribution between rings is not conservative. However, the observed radial gradient of ^{137}Cs reflects a particular diffusion process in trunk biomass. The radial pattern of ^{137}Cs thus poses a problem when using the average level of ^{137}Cs in wood and derived TF coefficients for a prospective estimation of further ^{137}Cs accumulation in wood. A new estimate of the radiocaesium immobilisation, i.e. the wood immobilisation potential (WIP), was therefore introduced and is described below.

TABLE 4.1. COMPARISON OF TF WITH ^{137}CS CONTENT IN WOOD IN PINE PLANTATIONS (VETKA, BELARUS)

	17 years	58 years
Trunk biomass (t MS/ha)	37.03	152.27
Transfer factor (TF) (m^2/kg)	0.0047 (0.0013)	0.0035 (0.0008)
^{137}Cs content in soil (kBq/m^2)	1462.1 (116.7)	1103.9 (228.3)
^{137}Cs content in trunk wood		
calculated (% of soil activity)	1.74 (0.14)	5.33 (1.10)
measured (% of soil activity)	1.52 (0.35)	8.58 (0.31)

4.2. THE WOOD IMMOBILISATION POTENTIAL

The definition of WIP is based on the shape of the cumulated curve schematised in Figure 4.1 for radiocaesium (plain curve). This curve was plotted by adding, for each annual wood increment from the pith to the last formed ring, respectively the volume of trunk wood (cm³) and its radiocaesium content (Bq) as x- and y-axes (the point 0 corresponds to the present situation). The shape of this curve, based on ^{137}Cs levels measured more than ten years after the initial contamination pulse, will probably not evolve drastically in the near future, and the newly incorporated ^{137}Cs (which only originates from root absorption at this time) will be redistributed accordingly. The WIP, defined as the limit of the mathematical slope of the curve in point 0 (curve 3), can consequently be considered as a suitable estimate of the future net ^{137}Cs accumulation in wood due to root uptake. The WIP is therefore a compromise between two extreme approaches: a conservative approach (each ring has the memory of the annual uptake episode, i.e. the ^{137}Cs is not redistributed in the trunk – curve 1) and the TF approach (there is no radial gradient of ^{137}Cs which is homogeneously redistributed in the trunk – curve 2).

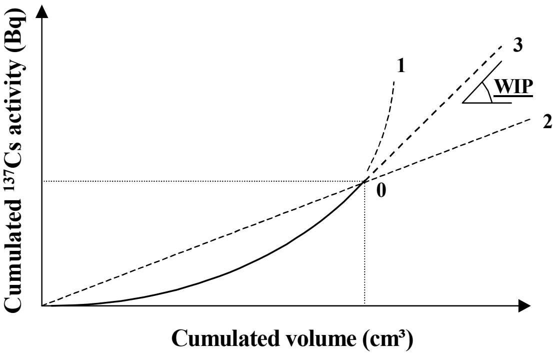


FIG. 4.1. Conceptual definition of the Wood Immobilisation Potential (WIP).

The comparison of WIP with Transfer Factors (Table 4.2) shows that WIP values increase as tree ages, contrary to what is suggested by Transfer Factors.

TABLE 4.2. COMPARISON OF WIP WITH OTHER WAYS TO CALCULATE ^{137}CS ACCUMULATION IN WOOD

Immobilisation	17 years	58 years
Measured		
Transfer factor (m^2/kg)	0,0047 ± 0,0013	0,0035 ± 0,0008
Total Cs content (wood) (% soil activity)	1,52 ± 0,35	8,58 ± 0,31
Calculated		
Cs WIP (Bq/cm^3·y)	1,99 ± 0,30	3,12 ± 0,23
Annual Cs flux to wood (% of soil activity)	0,15 ± 0,04	0,34 ± 0,07
Cs content in wood due to root uptake (% of soil activity)	1,80 ± 0,48	4,08 ± 0,84

Contrary to what is shown for TF values, ^{137}Cs WIP's increase as tree ages. This means that, at present, old trees immobilise about 1.5-fold more ^{137}Cs per volume unit of newly formed wood than the younger tree via root uptake. The WIP approach allows to estimate the current annual ^{137}Cs flux to wood by multiplying WIP values with the current annual wood increment of the stand, reported to soil activity. The annual ^{137}Cs flux to wood, multiplied by the time spent from the Chernobyl accident (12 years), gives then an idea of the current average ^{137}Cs content due to root uptake. This calculated value is in good agreement with the measured total ^{137}Cs content in trunk wood of the young stand that was, in fact, few affected by initial interception. For the old stand, the same calculation points out that soil-to-wood transfer explains only 48 % of the measured total ^{137}Cs content in wood which is consistent with the previous assumption. Finally, the root uptake of ^{137}Cs in a forest soil depends on the contamination level, the ^{137}Cs bio-availability and the distribution of roots in the different layers. In the older stand (58 years), the higher WIP was connected with a higher accessibility of the ^{137}Cs located in the organic layers (data not shown).

5. MODEL-MODEL INTER-COMPARISON STUDY – 'SCENARIO 1'

5.1. INTRODUCTION

This section describes results from the first model inter-comparison study undertaken by the BIOMASS Forest Working Group. The study involved comparison of outputs from 10 different models (see Section 5.3). The contamination scenario ('Scenario 1') that was posed for this exercise was hypothetical, but based on data sets from a real forest. It was deliberately made simple in order to allow all potential contributors of model predictions to provide results with the minimum of difficulty, thus ensuring a high rate of return of results from modellers. The scenario, summarised in Section 5.2, was based on a Chernobyl-type pulse input to a coniferous forest ecosystem of which all modellers were likely to have experience and appropriate model calibrations. Since Scenario 1 was hypothetical, the aim of the inter-comparison was not to validate model predictions against actual data but to compare the results produced by each modeller and his/her respective model. This model-model comparison was intended to serve as a baseline against which future scenarios and model-data validations could be gauged. Thus, one of the primary aims of the exercise was to allow modellers to assess the need to modify their model approach before undertaking further exercises within the Forest WG.

5.2. SUMMARY OF SCENARIO 1

The complete scenario is listed in Annex B. The scenario was based on a hypothetical Chernobyl-type pulse input to a coniferous forest ecosystem although the description of the forest ecosystem was based on real data from a coniferous forest typical of Scandinavia. The main soil type is a soddy-podzolic loamy sand. The dominant tree species is Scots pine (*Pinus sylvestris*), with some birch (*Betula pendula*). The understorey includes red raspberry (*Rubus idaeus*) and blackberry (*Rubus trivialis*), as well as mushroom species such as *Boletus edulis*, *Leccinum scabrum*, *Cantharellus cibarius* and *Russula* species. Grasses are rather sparse, but mosses cover 90% of the ground area. The main game species are moose and roe deer.

Participants were requested to predict activity concentrations on a fresh weight basis. Preferred endpoints could be chosen from the following list:

- total tree;
- total wood (i.e. trunk plus branches);
- needles (annual average);
- other parts of tree, especially bark;
- soil profile, including litter;
- animals (annual average for moose and roe deer); and
- vegetation ('mushrooms', berries, shrubs and grass).

Scots pine (*Pinus sylvestris*) was the main tree of interest, but participants could report on both pine and birch (*Betula pendula*) if they wished. Similarly, participants could report generically on mushrooms, berries, shrubs and grasses, or on particular species of these.

Each chosen endpoint was to be considered as a function of time and results were requested at 1 year intervals from 1 to 20 years after the date of initial deposition. It was requested that calculations be based on best estimates of parameter input values and that results be reported as both 'best estimates' and/or 95% confidence intervals, if possible.

5.3. PARTICIPANTS AND MODELS

Individual modellers and their respective models are listed in Table 2.1. Brief descriptions of each of these models are provided in Annex A

5.4. RESULTS

The results are presented graphically as 20 year time courses following initial deposition. As requested in the scenario sent out to respondents, the resolution of these time courses is 1 year.

All the modellers reported results as 'best estimates'. The results obtained with the FORESTPATH model were also reported as 95% confidence intervals. To simplify the graphs, normally only the median value calculated with the FORESTPATH model is plotted. The 95% confidence intervals are only plotted when it differs significantly from the median value.

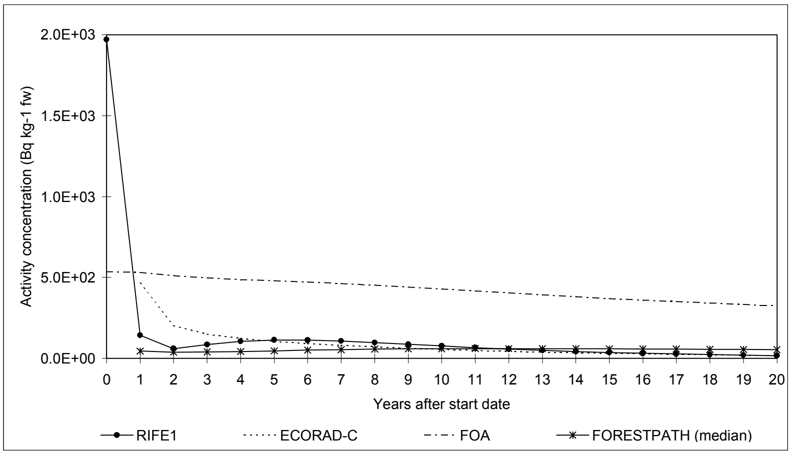


FIG. 5.1. ^{137}Cs activity concentration in total tree (fresh weight, Bq kg^{-1}).

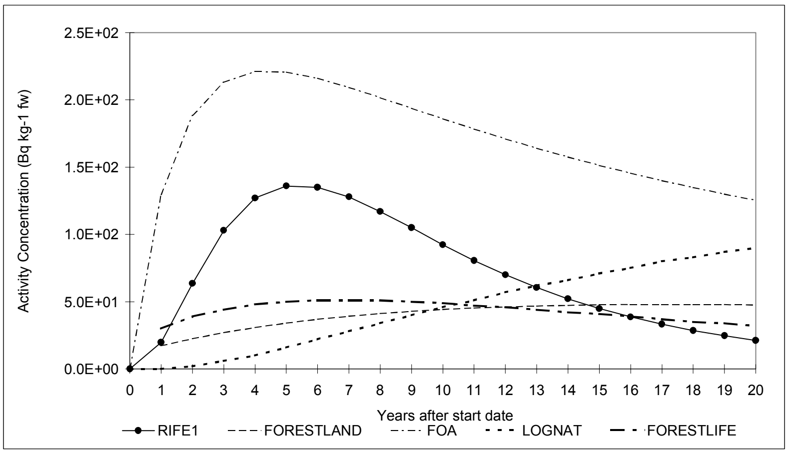


FIG. 5.2. ^{137}*Cs activity concentration in total wood (trunk + branches) (fresh weight, Bq kg*$^{-1}$*).*

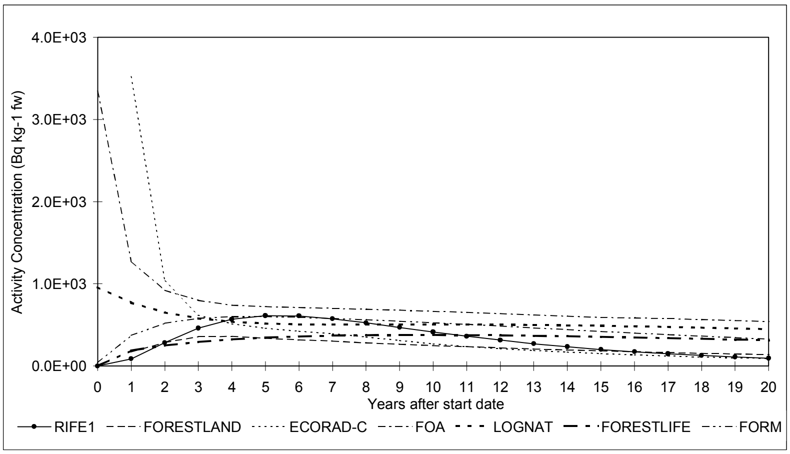


FIG. 5.3. ^{137}*Cs activity concentration in needles (fresh weight, Bq kg*$^{-1}$*).*

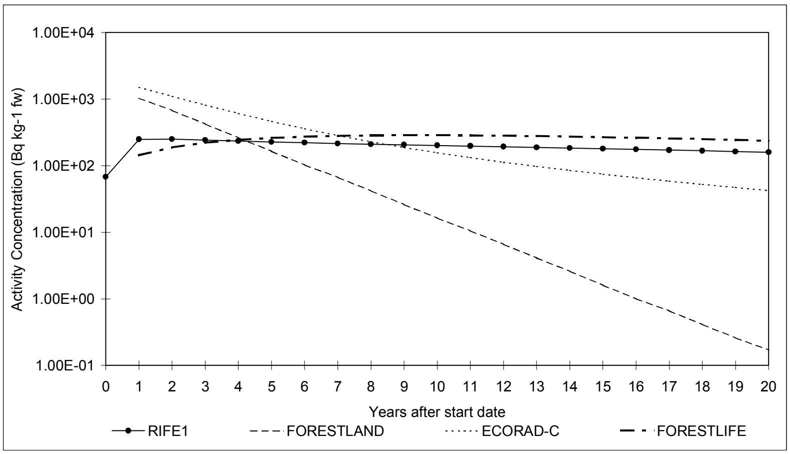


FIG. 5.4. ^{137}*Cs activity concentration in bark (fresh weight, Bq kg*$^{-1}$*).*

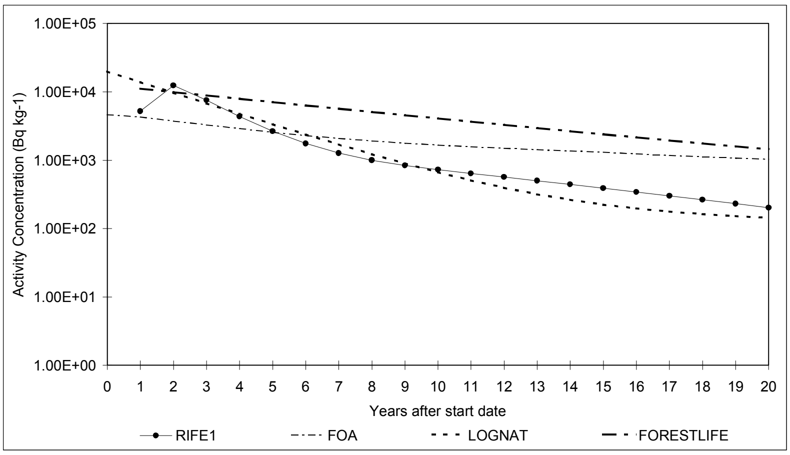


FIG. 5.5. ^{137}Cs activity concentration in litter layer (Bq kg^{-1}).

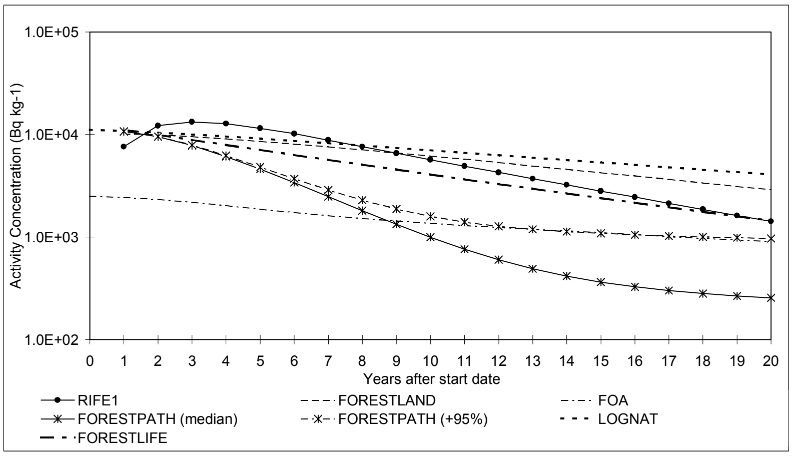


FIG. 5.6. ^{137}Cs activity concentration in organic soil layer (Bq kg^{-1}).

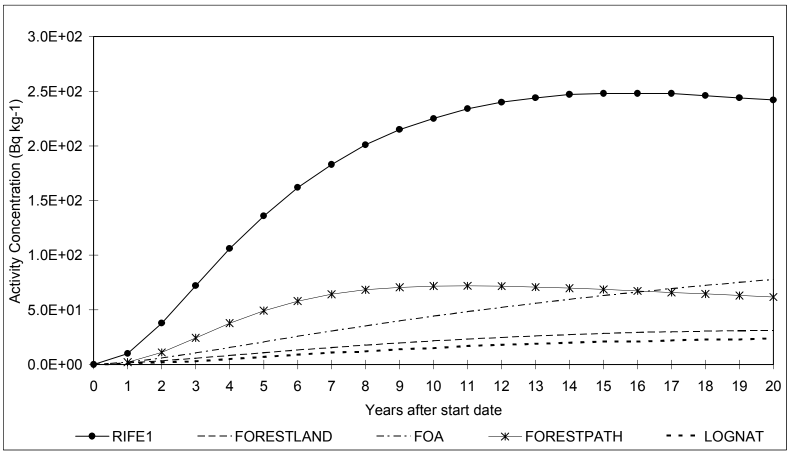


FIG. 5.7. ^{137}Cs activity concentration in mineral soil layer (Bq kg^{-1}).

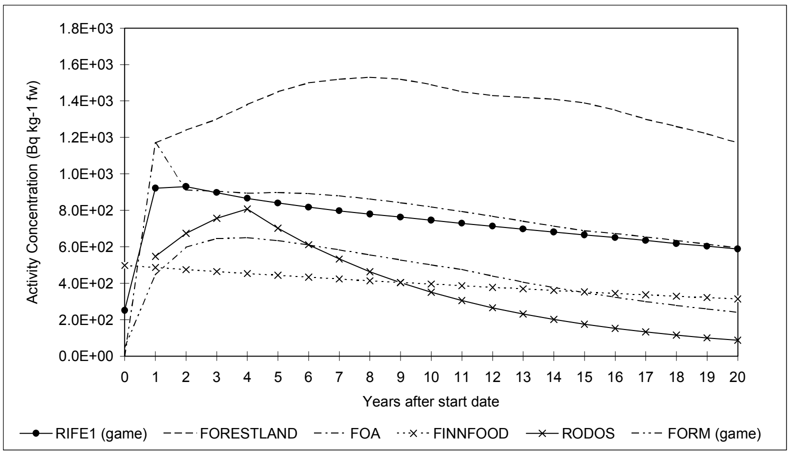


FIG. 5.8. ^{137}Cs activity concentration in moose (fresh weight, Bq kg^{-1}).

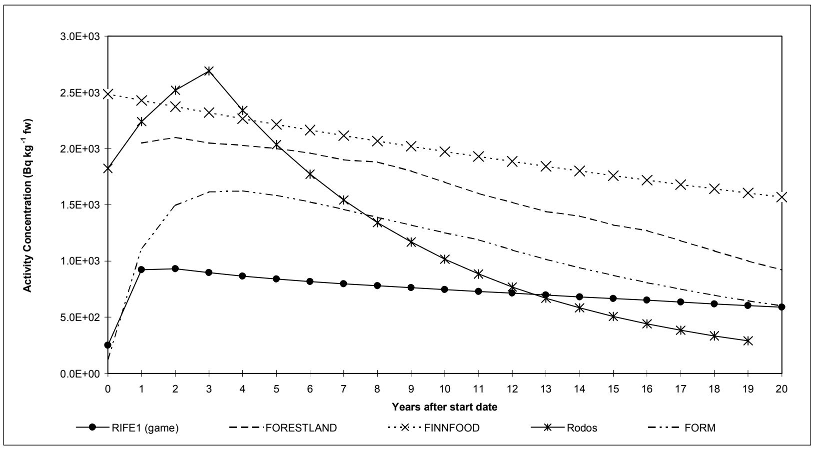


FIG. 5.9. ^{137}Cs activity concentration in deer (fresh weight, Bq kg^{-1}).

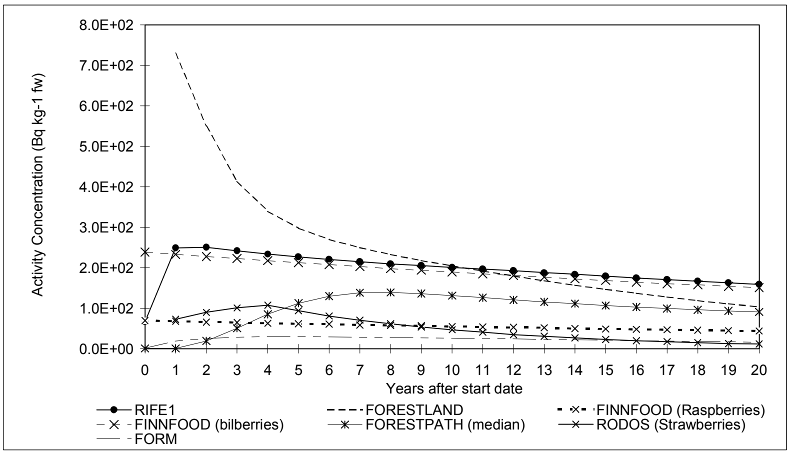


FIG. 5.10. ^{137}Cs activity concentration in berries (fresh weight, Bq kg^{-1}).

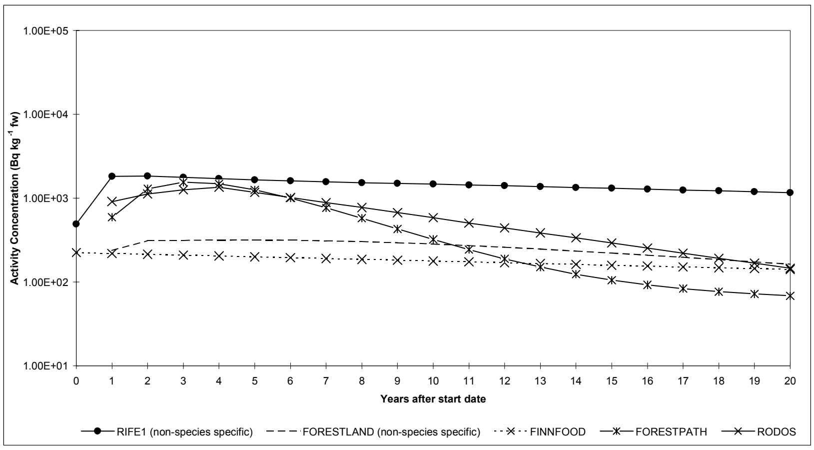


FIG. 5.11. ^{137}Cs activity concentration in mushrooms(Boletus, fresh weight, Bq kg^{-1}).

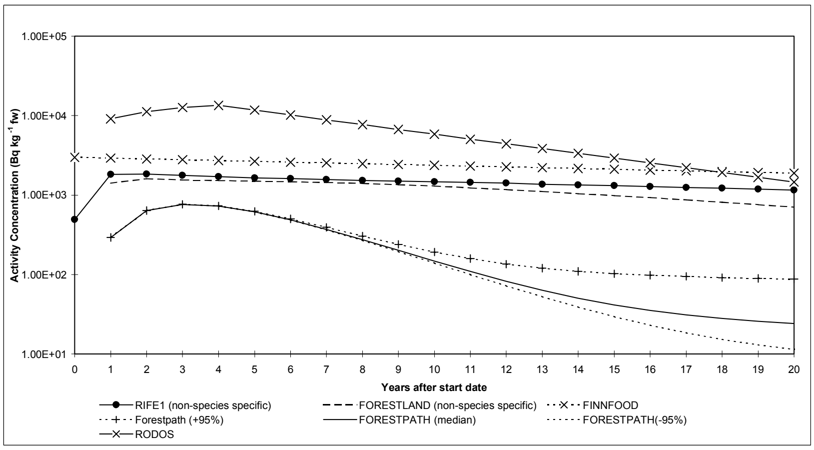


FIG. 5.12. ^{137}Cs activity concentration in mushrooms (Russula, fresh weight, Bq kg^{-1}).

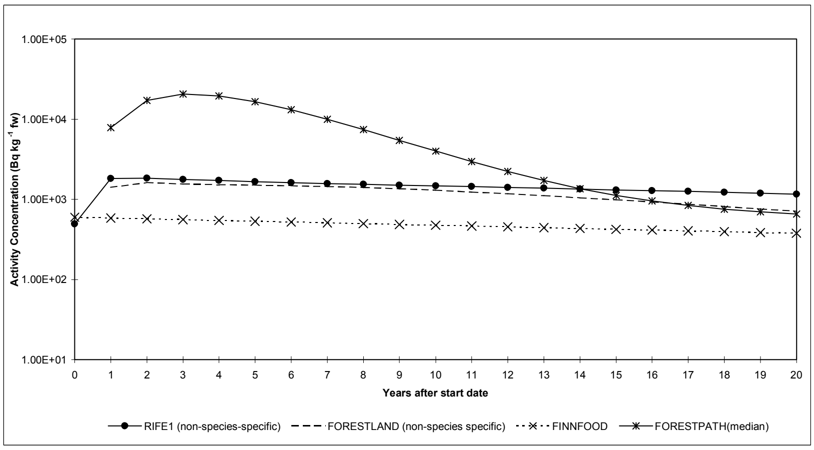


FIG. 5.13 ^{137}Cs activity concentration in mushrooms (Cantharellus, fresh weight, Bq kg^{-1}).

5.4.1. Trees and associated components

Seven respondents provided predictions for either total tree activity concentrations and/or activity concentrations of specific components of trees. Figures 5.1, 5.2, 5.3 and 5.4 show plotted results for the specific tree-related endpoints requested, namely total tree (all tissues averaged), wood (trunk plus branches), needles and bark, respectively.

In the case of total tree, bark and needle predictions the relative uniformity of predicted time courses after approximately 5 years was striking. The FOA model consistently predicted the highest activity concentration within the total tree (Figure 5.1), a result which is accounted for by the relatively high activity concentration in wood predicted by this model (Figure 5.2). The range of predicted activity concentrations in total tree tissues was greatest at 0 years and 20 years (*i.e.* at the very beginning and end of the simulation period) although this maximum range was just greater than one order of magnitude. A similar maximum range of values for activity concentrations in wood was observed. Five modellers returned predictions for wood and within this group of predictions two distinct types of predicted time course emerged. In the case of FOA, RIFE and FORESTLIFE the predicted time courses were characterised by an initial increase in wood activity concentration to a maximum (at times ranging from approximately 4 to 8 years following deposition) after which the activity concentration declined. Somewhat different time courses were predicted by FORESTLAND and LOGNAT, which showed a general increase in wood activity concentration over the 20-year simulation period.

As with predictions for wood, the predicted time courses of activity concentrations in bark could be separated into two sub-groups. Of the four models that provided bark predictions the FORESTLAND and ECORAD models predicted approximately exponentially declining time courses while RIFE and FORESTLIFE predicted rather lower initial bark activity concentrations and rather higher activity concentrations in bark after approximately nine years (Figure 5.4). This is probably indicative of different conceptual approaches in each of these two pairs of models. For instance, in FORESTLAND bark is defined as the outer surface of the tree, whereas the other models do consider the internal bark to a certain extent.

The maximum range of results for bark was large with a range of approximately three orders of magnitude being spanned by predictions at 20 years – this was largely accounted for by the exponentially decaying time course predicted by FORESTLAND.

Predicted time courses of activity concentrations in needles (Figure 5.3) show a remarkable degree of consistency after three years with a maximum range of about one order of magnitude at 20 years. In the initial three years of the predictions, however, both ECORAD and FOA predicted time courses of needle activity concentrations rapidly declining from initially high values. LOGNAT predicted a similar though less pronounced time course. FORESTLAND starts simulating the activity in needles one year after deposition only, as it is assumed that most of the intercepted activity is released during the first year (this assumption is made for long-term calculations only). The other models (RIFE and FORESTLIFE) predict initially zero needle activity concentrations followed by a steady increase over a period of three to ten years. The discrepancy between the models points unequivocally to a difference in interpretation of the desired endpoint by the modellers. In the case of ECORAD, FOA and LOGNAT it is clear that initial external contamination of needles, due to direct deposition from the atmosphere, is taken into account whereas in RIFE, and FORESTLIFE only contamination *via* root uptake and subsequent internal translocation is considered.

5.4.2. Soils

Soils are subdivided into three major layers for the purposes of the analysis of results from the model-model inter-comparison, namely litter (Figure 5.5), organic soil (Figure 5.6) and mineral soil (Figure 5.7). Interpretation of the actual soil horizons which constitute each of these broad layers can vary between modellers but, broadly, the litter layer is taken to be the AoL horizon, the organic layer is taken to be the AoF plus AoH horizons and the mineral soil is taken to be anything below the organic horizons (on the basis of this definition the mineral soil may, in practice, include hemi-organic horizons marking the transition between organic and mineral layers).

As for the prediction of contamination of trees, seven modellers returned predictions of contamination time courses for soils. There was generally good quantitative consistency between predicted time courses for all three major soil layers with maximum variation between predictions never exceeding slightly more than one order of magnitude. Qualitatively, too, the predicted time courses were very consistent with each other, especially in the case of the mineral soil predictions (Figure 5.7). This probably represents a high degree of consistency in the way in which soil migration is represented conceptually within each model. In most of the models a classical compartmental leaching approach is taken in which downwards migration of radiocaesium occurs from one discrete compartment to the next. The FORESTLIFE model represents this process by means of a physical advection/diffusion approach, yet the predictions of this model are very similar to those of the other models. The FOA model distinguishes between two soil compartments “available soil” and “unavailable soil”, principally not related to position in any soil layers. However, “available soil” is expected to correspond mainly with the organic layers below the AoL horizon, and “unavailable soil” to the mineral soil, allowing for exchange of caesium between the “unavailable” condition to the “available”.

The particularly high degree of consistency between predictions of contamination of soil layers doubtless reflects the strong effort which has been put into understanding soil migration, not only in forests but particularly in agricultural ecosystems.

5.4.3. Other biological endpoints

The biological endpoints (other than trees) specified in Scenario 1 are of concern because their consumption by man represents a radiological exposure route *via* which internal doses may be incurred. Thus, wild animals, understorey (berries) and mushrooms are the main biological endpoints of interest in this study – model predictions of contamination time courses for each of these components of the forest ecosystem are shown in Figures 5.8, 5.9, 5.10, 5.11, 5.12 and 5.13 respectively.

5.4.3.1. Wild animals

Five modellers provided predictions for time courses of contamination in wild animals (game). Of the five models, one (RIFE) interprets game in a generic sense while the remaining four specify the species of interest, namely deer and/or moose. The spread of predictions produced by these models appears large in Figure 5.8 (moose) and Figure 5.9 (deer) but is generally within one order of magnitude. The time trends predicted for moose, deer and generic ‘game’ are all qualitatively similar, with similar predictions of gently declining activity concentration over the 20 year simulation period. There are striking quantitative differences between predictions for moose and deer contamination, with the greatest variability being evident in predicted moose time courses. The differences between the uppermost and lowermost moose contamination predictions (FORESTLAND and FINNFOOD, respectively) was approximately a factor of three, although there was very good agreement between these two models for predictions of deer contamination. Finally, the agreement between the RIFE model (using a generic, IAEA recommended, T_{agg} value) and the FOA model (which provided a prediction for moose only) was striking, particularly so since the FOA moose model is based on seasonally dependent intake rates from vegetation compartments, and does not use the T_{agg} principle.

5.4.3.2. Understorey

Predictions of time courses of contamination of the understorey focussed on berries (Figure 5.10). Results were provided by four of the modellers. Three of the models did not attempt to make predictions for specific species of berries but instead provided predictions on a generic basis. In the case of the FINNFOOD model, which had not been used to provide predictions for any of the previous endpoints, two particular species of berry bearing plant were modelled – bilberry (*Vaccinium myrtillus*) and raspberry (*Rubus idaeus*).

As for the tree predictions, the degree of qualitative agreement between all the predicted time courses was good after the first few years of the simulation. The greatest divergence in predictions during these first five years was provided by FORESTLAND which, as for needles of trees, predicted that an initially maximal degree of contamination immediately after deposition would be followed by a slow, approximately exponential decline in activity concentration over the 20-year simulation period. This is most probably due to an assumption of considerable external contamination of plant tissues immediately after deposition. None of the other models made this assumption and consequently predicted only a slow decline in activity concentration of berries following a (sometimes rapid) increase in activity concentration of berries. After five to seven years the rate of decline in activity concentrations of berries predicted by each model was very similar and probably reflects the physical decay of ^{137}Cs. The maximum range of predicted activity concentrations during the initial phase following deposition was approximately three orders of magnitude, but less than one order of magnitude during the 10 to 20 year period of the simulation.

5.4.3.3. Fungal fruiting bodies ('Mushrooms')

One of the confounding factors in the provision of results for Scenario 1 was that different modellers provided results for different mushroom species and it is well known that major species differences occur in terms of ^{137}Cs uptake by mushrooms. In an attempt to clarify the range of predictions received, the mushroom predictions were plotted species-by-species (Figures 5.11, 5.12 and 5.13). Nevertheless, the variability of predictions for contamination was high, with a range of results spanning about two orders of magnitude for each mushroom species over the entire simulation period (Figures 5.11, 5.12 and 5.13).

Of the four models from which predictions of mushroom contamination time courses are available most predict qualitatively similar curves over the 20-year period. On closer inspection, however, the models can be divided into different groups with respect to the mushroom contamination dynamics that they predict. FINNFOOD predicts that, having reached peak contamination rapidly (immediately in the case of FINNFOOD) after ^{137}Cs deposition, a steady reduction in mushroom contamination occurs at the same rate as the physical decay of ^{137}Cs. This implies that removal by leaching of ^{137}Cs from the region of the soil exploited by fungal mycelia does not occur during the 20-year simulation period. On the other hand, both the FORESTPATH and RIFE models predict that reduction of mushroom contamination after deposition will occur more rapidly than the physical decay of ^{137}Cs, thereby implying that leaching of radiocaesium from the exploited soil will occur over a 20 year period. In the case of FORESTLAND there is a weak decrease of radiocaesium levels in mushrooms with time because the radiocaesium inventory in available form does not decrease substantially with time. This is despite the fact that a decrease in total radiocaesium levels in the soil exploited by fungal mycelia does occur.

The high degree of variability between the predictions confirms that, conceptually, the modelling of mushroom contamination remains probably the most contentious aspect of forest modelling. This is an important problem both to recognise and solve since, as far as the ingestion dose pathway for forest food products is concerned, consumption of contaminated mushroom is probably the most important single component.

5.5. GENERAL SUMMARY

In brief, the results obtained for Scenario 1 and reviewed in this report have shown that, for the soil and tree compartments of forests, there is generally a high level of consistency between predictions made by the models tested. This is particularly true for the soil compartment and it doubtless reflects the research efforts that have been made over the last 10 years in understanding processes controlling radiocaesium migration in soils.

Predictions of all the biological endpoints proved more variable, especially predictions for mushroom contamination. This raises the question of whether our current (deterministic) modelling approaches are suitable to predict the behaviour of biological entities that will inevitably exhibit a high degree of variability. One or two specific mushroom species will be selected for modelling in future scenarios and this may allow a better understanding of any shortcomings of current modelling approaches (although the Scenario 1 exercise has already demonstrated that predictions for individual mushroom species are currently highly variable).

In addition to these general observations the following summary points were compiled during a meeting of the Forest Working Group to discuss the results of Scenario 1:

— Accurate model descriptions are needed which give accurate definitions of compartments (and interpretation of endpoints) as well as a description of solution algorithms.

— A description of why a particular model structure was used would be helpful in understanding the individual modeller’s conceptual approach as well as an account of how parameter values were obtained.

— As well as reporting results as activity concentrations it is felt that the Forest Working Group modellers should also report results as % distributions (or fractional distributions) of ^{137}Cs within the forest ecosystem after deposition. Results may also be more informative if reported without physical decay.

— Tagg values (or other appropriate transfer coefficients) for compartments such as trees should be calculated and reported.

— In the model-data inter-comparison (see Section 6) the variability between model predictions should be compared with the variability between data (this may be facilitated by comparison of variability between predicted Tagg values).

- The question has been raised as to whether the available models consider tree age and growth adequately; this may form a specific part of a future inter-comparison scenario.
- Compilation of a table of fresh to dry weight ratios for forest products and components is seen as being necessary and desirable. Considerable difficulty was encountered in the preparation of figures of results from Scenario 1 due to the fact that some modellers provided results on a fresh weight basis while others preferred to use a dry weight basis.
- Finally, during discussions within the Forest Working Group it was established that certain processes are currently not (or very poorly) represented in models. One potentially important process is that of foliar absorption, especially by trees, of which our current understanding is very weak.

5.6. STATISTICAL ANALYSIS OF THE RESULTS OF THE MODEL-MODEL INTER-COMPARISON (SCENARIO 1)

5.6.1. Introduction

The objective of the analysis presented here was to analyse the results of the Scenario 1 exercise using an original method developed for evaluating predictive models. The analysis was designed to address the following questions:

- What is the degree of agreement between model predictions?
- Do the models predict similar time dynamics, and does the agreement between the model predictions decrease (do predictions converge) or increase (do predictions diverge) with time?
- For which endpoint (wood, bark, needles, *etc.*) is the degree of agreement best (or worst)

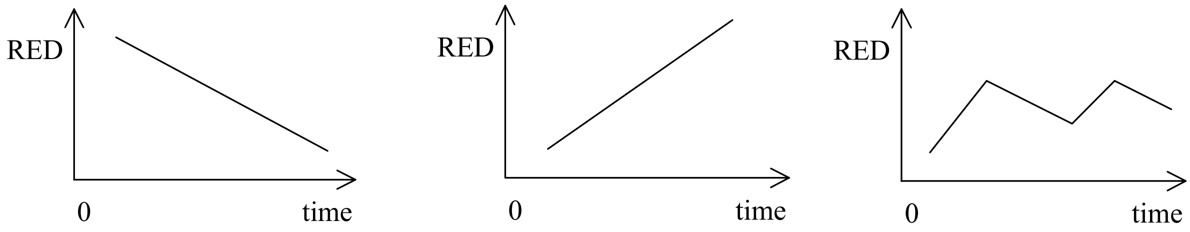
5.6.2. Statistical method of model comparison

The methodology applied here is derived from an original statistical method developed initially to compare predictive models with experimental data (Williams and Leggett, 1983). If only models outputs are compared (with no reference to experimental values), the Relative Euclidean Difference (RED), which is an expression of the sum of differences between every couple of model outputs, is an appropriate tool to evaluate different compartments of a forest ecosystem:

$$RED = \sqrt{\sum_i \left(\frac{x_i - y_i}{x_i + y_i} \right)^2}$$

x_i and y_i refer to the predictions made by each pair of models for the same time "i". The sum contains as much members as the number of possible pair combinations between the "n" models compared (C_n^2).

The RED is a qualitative measure of the average agreement between model predictions. Decreasing (respectively increasing) time series of RED values mean that the predictions of the different models globally converge (respectively diverge) with time. If the RED curve is non-monotonous, the inflexion points correspond to changes in the modelling dynamic of some of the models. If all the models predict the same value (perfect agreement), the RED equals zero (Figure 5.14).



The models converge with time

The models diverge with time

The models are based on different time dynamics

FIG. 5.14. Schematised shapes of the evolution of RED with time.

As the RED is generally not constant with time, its maximum value (max(RED), worst case) in a given time series of model predictions has been used to define the reliability index "k" ($0 < k < 1$):

$$k = \frac{1 - \max(RED)}{1 + \max(RED)}$$

5.6.3. Models clustering

The RED analysis highlights the forest compartments for which no clear consensus exists between models predictions. For these compartments, it is interesting to go a step forward and refine the analysis through models clustering. For a given model *m*, this analysis consists in comparing the differences between all the pairs of models (*global* value) with the differences calculated without considering the pairs of models in which the model *m* appears (*limited* value). As clustering is carried out independently for each compartment, the absolute differences between models predictions (Absolute Euclidean Difference – AED) is generally used:

$$AED = \sqrt{\sum_i \left(x_i - y_i\right)^2}$$

An individual AED value which significantly diverges from the global AED value means that model *m* significantly differs from the other models for the compartment studied.

5.6.4. Results and discussion

Some models involved in our comparison exercise are generic, others are specific to one or several compartments. Table 5.2 synthesises the ecosystem compartments which are respectively considered by the different models.

The RED values have been calculated for the different compartments according to Table 5.2. The results, grouped in a logical way (tree, soil, understorey), are presented in Figures 5.15 to 5.17. In these figures, the real value of RED has no clear significance, only the comparison of the relative position of the curves and of their tendencies is meaningful.

TABLE 5.2. MODEL-MODEL COMPARISON: SYNTHESIS

	Total tree	Wood	Needles	Bark	Litter	Organic soil	Mineral soil	Berries	Moose	Roe deer	Mush-rooms
Ecorad-C	×	×	×	×							
Forestland		×	×	×		×	×	×	×	×	×
FOA	×	×	×		×	×	×		×		
Forestpath	×					×	×	×			×
Rodos								×	×	×	
Rifel	×	×	×	×	×	×	×	×	×		×
Form								×		×	
Lognat		×	×		×	×	×				
Forestlife		×	×	×	×	×					
Finfood								×	×	×	×

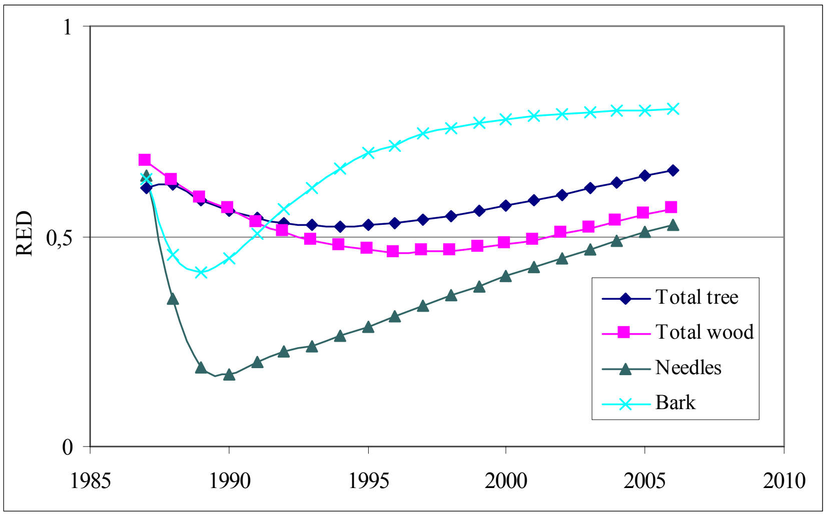


FIG. 5.15. Time variation of the RED for tree compartments.

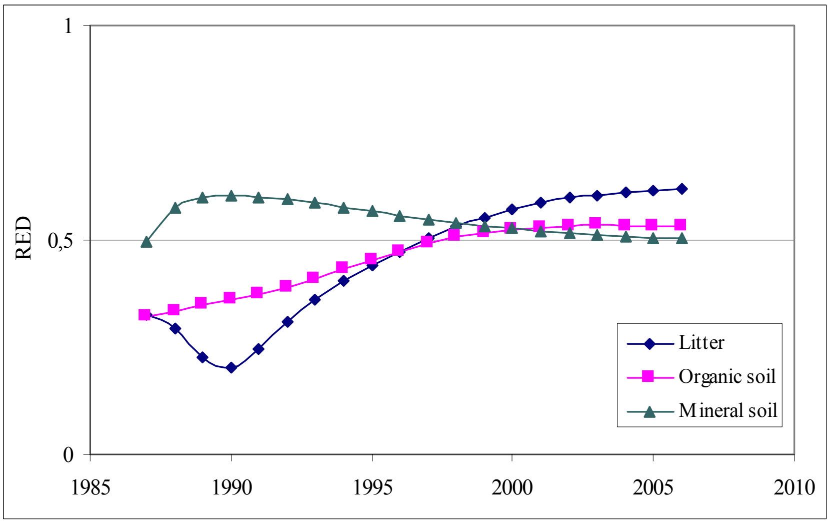


FIG. 5.16. Time variation of the RED for the litter-soil compartments.

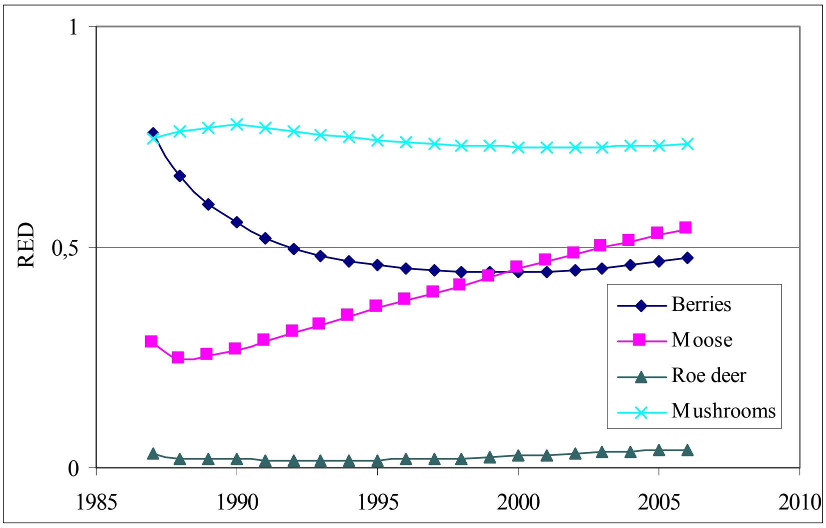


FIG. 5.17. Time variation of the RED for the understorey compartments.

Two periods can clearly be distinguished regarding the tendencies of the curves. During the first 5 years after contamination, all models converge for the tree (including litter) and understorey compartments, but diverge for the soil compartments. After this period, the estimates for all compartments diverge or show no clear tendencies.

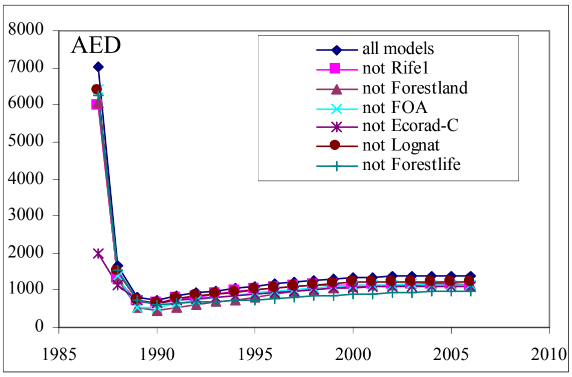
Values of the reliability index *k* are synthesised in Table 5.3. The different compartments can be classified as follows (from best to worst): roe deer >> moose, organic soil, mineral soil > total tree, wood, needles, litter >> bark, berries, mushrooms.

The bad results for berries and mushrooms are probably due to the fact that a lot of different species are considered in the comparison. On the other hand, the very good results for roe deer are likely due to the limited data sets used for calibration of the models.

TABLE 5.3. MODEL-MODEL COMPARISON: RELIABILITY INDEX

	Total tree	Wood	Needles	Bark	Litter	Organic soil	Mineral soil	Berries	Moose	Roe deer	Mush-rooms
Agreement *(k factor)*	0.21	0.19	0.22	0.11	0.24	0.30	0.25	0.14	0.30	0.92	0.13

Models clustering was carried out for needles, bark, wood and litter compartments. The results are presented in Figures 5.16 to 5.19. The models clustering for needles, bark, wood and litter is synthesised in Table 5.4.



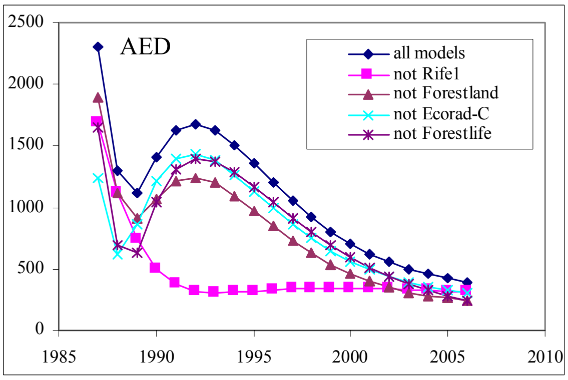
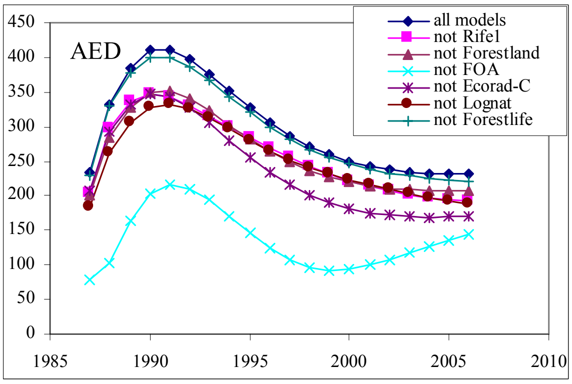


FIG. 5.16. Cluster analysis (needles).

FIG. 5.17. Cluster analysis (bark).



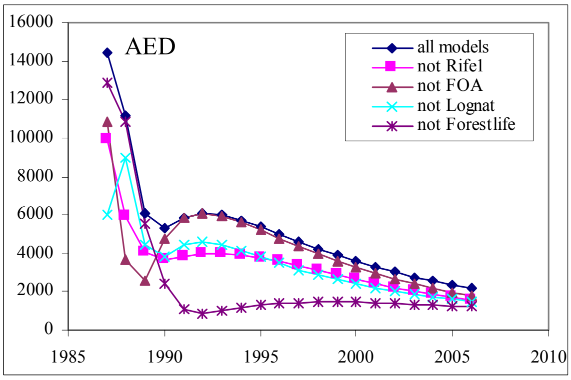


FIG. 5.18. Cluster analysis (wood).

FIG. 5.19. Cluster analysis (litter).

TABLE 5.4. MODEL-MODEL COMPARISON: CLUSTER ANALYSIS

Compartment	Models clustering and comments
Needles	2 clusters: 1) Ecorad-C 2) other models Ecorad-C significantly differs just after the contamination (the level in needles is much higher than in the other models)
Bark	2 clusters: 1) Rife 1 2) other models Rife 1significantly differs from year 3 (the level in back increases during 3 years then decreases)
Wood	2 clusters: 1) FOA 2) other models The time dynamics are similar for all the models
Litter	4 clusters: 1) FOA 2) Lognat 3) Forestlife 4) other models Only FOA shows no litter increase between year 3 and 6 In Lognat, the level in litter decreases between year 1 and 2 Forestlife significantly differs from year 3

The analysis of Figures 5.16 to 5.19 reveals that for needles, bark and litter, a rapid convergence can be obtained by removing only one model. Long-term management strategies can thus be based on a good consensus between models. On the other hand, for wood, the initial phase of divergence of the simulations cannot be attributed to one given model; although the absolute variations are reduced if FOA is not taken into account.

5.6.5. Conclusions of statistical analysis

Radioecological models simulating radiocaesium cycling in forest ecosystems have been compared for the same reference contamination scenario but with no reference to real experimental data. As the different models involved are generic or specific to some forest compartment(s) (tree, soil, understorey), simple statistical tools based on the comparison of couple of models were used, firstly, to identify the tendencies (convergence or divergence) between time series of models estimations and, secondly, for the most varying compartments, to discriminate between models through a clustering analysis.

The agreement between models (k factor) varied between 10 and 30%. The highest agreement was shown by roe deer (92%) but this value is not significant. Globally, all model estimations converge in the first five years following the contamination and show varying tendencies afterwards. Finally, the cluster analysis proved to be a powerful tool to reduce the noise of the global AED curve by isolating the models which are based on different time dynamics.

Considering the high complexity and variability of the phenomenon studied, the present models for ^{137}Cs behaviour in forest compartments are in satisfactory agreement. Nevertheless, differences in estimations of time dynamics between models show that a better understanding of the ^{137}Cs behaviour and cycling in forest ecosystems is still needed, especially for wood. Moreover, a comprehensive evaluation of these models would require a comparison of model outputs with experimental data to make the link with long-term management strategies of real cases of forest ecosystems contamination.

6. MODEL-DATA INTER-COMPARISON STUDY – 'SCENARIO 2'

6.1. INTRODUCTION

This section describes results from the second model inter-comparison study undertaken by the Forest WG. The study involved comparison of outputs from 9 different models (see Section 6.3). The contamination scenario ('Scenario 2') that was posed for this exercise was based on a data set obtained for a forest site near Kiev, Ukraine, contaminated with ^{137}Cs in 1986 following the Chernobyl accident. Scenario 1 (Section 5) was hypothetical and was intended not to validate model predictions against actual data but to compare the results produced by each modeller and his/her respective model. The primary objective of Scenario 2, however, was to allow a direct comparison of model predictions against a time series of data collected at a forest site which non of the modellers had previously seen and for which they had no prior information. In this sense the exercise was conducted 'blind' and the outcome of the exercise was not revealed to the modellers until the results obtained from individual models had been collated and plotted against the actual time series data for the forest site in question. Some problems of interpretation of results did arise due to the fact that not all of the models were constructed to make predictions of each of the endpoints specified in the scenario. Therefore, some work was required after the results of the exercise had initially been revealed to the Forest WG. In some cases, second round results were submitted, although these are not dealt with in this report. Each of these stages in the model-data inter-comparison is described below and a description of results is provided.

6.2. SUMMARY OF SCENARIO 2

The complete scenario is listed in Annex C. The scenario concerned a forest situated near Rudnya-Povcha in the Zhitomir region of Ukraine, approximately 130 km to the south-west of the Chernobyl power plant (51° 09’ N, 28° 35’ E). For convenience, the date of deposition of ^{137}Cs was taken as 1st May 1986: the total deposition at this time was 555 kBq m^{-2}. The main soil type at the site is a soddy-podzolic sandy loam which is characterised by low natural fertility and high permeability to water flow. The dominant tree species is Scots pine (*Pinus sylvestris*) with an average age of 50 years, accompanied by sparse birch (*Betula pubescens*) with an average age of 25 to 30 years. The understorey is dense and comprises several species of bilberry (*Vaccinium* spp.), purple moor grass (*Molinia caerulea*) and various other herbaceous species of lesser importance. Mushroom species to be found at the site include *Boletus edulis*, *Suillus luteus*, *Cantharellus cibarius*, *Xerocomus badius* and *Russula paludosa*. The main game species at the site is roe deer.

Participants were requested to predict activity concentrations, on a dry weight basis, of the following endpoints:

- wood of *Pinus sylvestris*;
- annual shoots of *Pinus sylvestris*;
- needles of *Pinus sylvestris*;
- total bark of *Pinus sylvestris*;
- soil profile, including litter;
- roe deer;
- ‘mushrooms’ (individual species to be modelled at the discretion of each modeller);
- bilberry.

As mentioned in the introduction, not all of the models were designed to predict radiocaesium concentrations in all of the required endpoints. For this reason, participating modellers were given the freedom to report on their own choice of endpoints selected from the above.

It was requested that each chosen endpoint should be considered as a function of time over the period 1986 to 1998, with results to be reported at one-year intervals. It was further requested that calculations be based on best estimates of parameter input values and that, if possible, results be reported as both ‘best estimates’ and/or 95% confidence intervals: in fact, only one modeller produced both best estimates and 95% uncertainty ranges.

6.3. PARTICIPANTS AND MODELS

Individual modellers and their respective models are listed in Table 2.1. Brief descriptions of each of these models are provided in Annex A.

6.4. RESULTS

The results are presented graphically as 12-year time courses (1986–1998) following initial deposition. As requested in the scenario sent out to respondents, the resolution of these predicted time courses is one year.

All the modellers reported results as ‘best estimates’. The results obtained with the FORESTPATH model were also reported as 95% confidence intervals. To simplify the graphs, normally only the median value calculated with the FORESTPATH model is plotted. The 95% confidence intervals are only plotted when they differ significantly from the median value.

6.4.1. Trees and associated components

Predictions of activity concentrations of specific components of trees (*Pinus sylvestris*) were provided by six respondents, although only the FORESTLAND and FOA models returned results for each of the endpoints requested. Figures 6.1, 6.2, 6.3 and 6.4 show plotted results for each of the specific tree-related endpoints, namely wood (without bark), total bark (including cambium), annual shoots and needles.

The first impression from Figures 6.1 to 6.4 is that the range of model predictions for each of the tree-related endpoints generally encompasses the measured data at each time interval. This observation applies particularly to predictions for wood and total bark (Figures 6.1 and 6.2) in which, at all time points, the measured data values fall within the range of values predicted by each of the models concerned. This immediately suggests that, for the Zhitomir site, the predictions provided by the FORESTLAND, RIFE, ECORAD, FOA, FORM and S-RODOS models could be used collectively to provide a reliable envelope of model predictions within which measured data on wood and bark activity concentration would be expected to fall. For predictions of annual shoots and needles (Figures 6.3 and 6.4), however, a significant number of measured data fell outside the envelope of model predictions, although in the case of annual shoots predictions were only provided by three respondents. This comparison demonstrates the value of multiple model predictions applied to a single data set: despite differences in the predictions of individual models there are evidently no ‘wild card’ predictions and it is possible that, using the models employed in this study, an averaged or ‘consensus’ prediction could be obtained which would agree quite well with the measured data.

Results for each of the individual tree-related endpoints will now be examined in turn.

Of the five predictions that were made of the time-course of contamination of wood (Figure 6.1) four indicated very similar dynamics. These were FORESTLAND, FOA, RIFE and FORM, which all showed an initial increase in activity concentration from very low (effectively zero) initial values. FORESTLAND, FOA and RIFE provided predictions that were particularly close to the measured data. FORM predicted very similar dynamics to these three models, but predicted significantly lower activities in wood for most of the 12-year period. ECORAD predicted an exponentially declining activity in wood from the very beginning of the 12-year period, suggesting a significantly different conceptual approach.

For total bark (Figure 6.2) five models provided predictions. Of these, FORESTLAND, ECORAD, S-RODOS and RIFE all provided predictions which closely bracketed the measured data over the period 1991 to 1998, although the dynamics of these models over the 12 year assessment period varied considerably. S-RODOS and RIFE both indicated an initial sharply declining activity in bark, suggesting an initially important component of external contamination immediately after deposition of ^{137}Cs from the atmosphere. ECORAD and FORESTLAND both indicated an exponentially declining trend over the whole assessment period, although without the initially very high external activities predicted by FINNFOOD and RIFE. FOA consistently predicted higher and more-or-less constant bark activities than the other four models.

Only three modellers provided predictions for annual shoot activities (Figure 6.3). These were FORESTLAND, FOA and S-RODOS. The scatter in the measured data was considerable for this endpoint, and each of these models provided predictions which lay more-or-less centrally within this scatter of data, even though each model predicted different 'early' dynamics immediately after deposition.

Six modellers provided predictions for needle activities (Figure 6.4). The models concerned were FORESTLAND, ECORAD, RIFE, FOA, S-RODOS and FORM. During the period 1991 to 1998, for which measured data were available, the dynamics and range of predicted values of each of the models was remarkably consistent, with less than one order of magnitude spanning the highest (FORESTLAND) and lowest (FORM/ECORAD) predictions. 50% of the measured data points were within the range of predicted values provided by the models.

It should be noted that, while the agreement between model predictions and data was impressive for the period 1991 to 1998, for which data are available, there are large discrepancies in model predictions for the 'early phase', approximately 1986 to 1988, for which no data are available from the Zhitomir site.

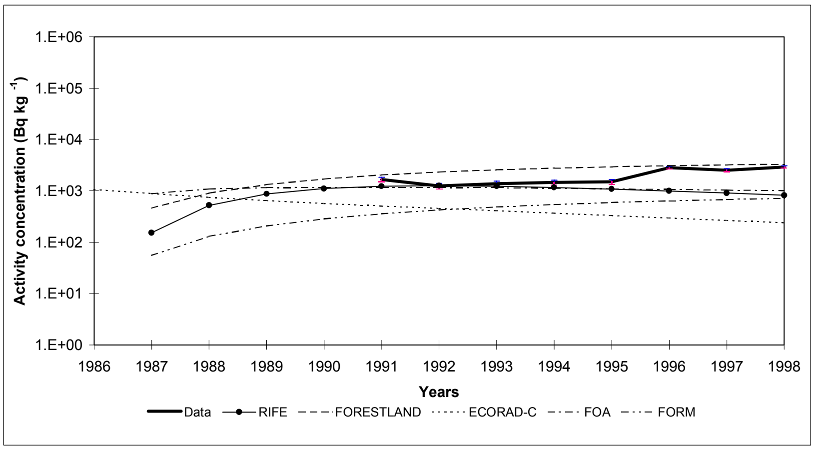


FIG. 6.1. ^{137}Cs activity concentration in wood (without bark, Bq kg^{-1}).

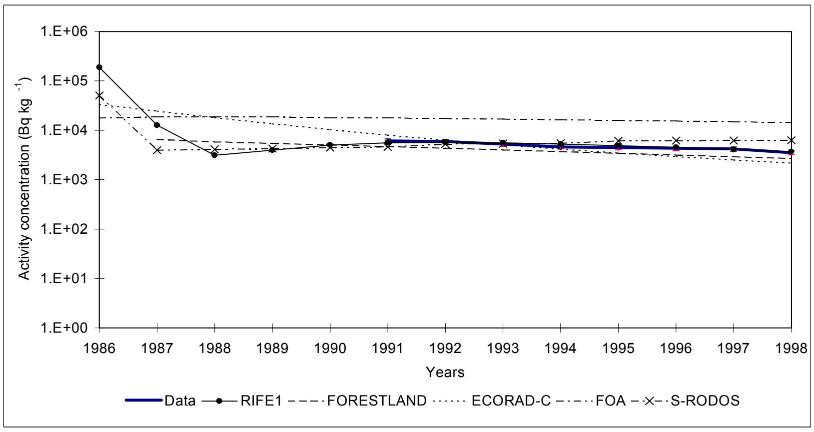


FIG. 6.2. ^{137}Cs activity concentration in bark (including cambium) (Bq kg^{-1}).

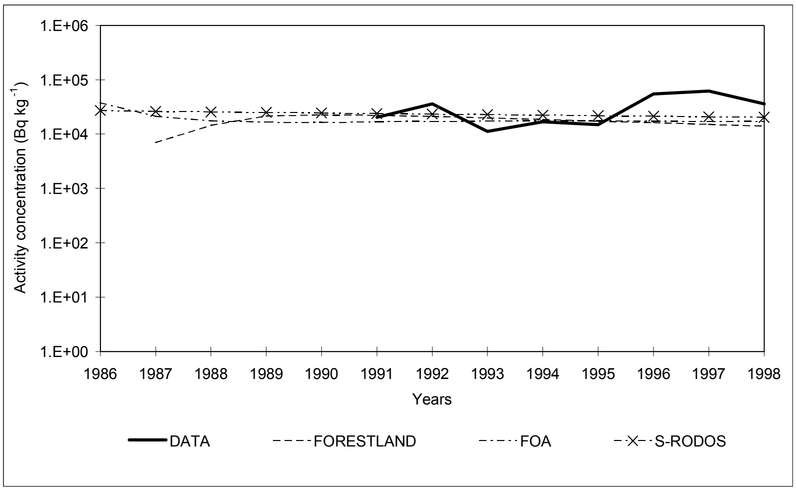


FIG. 6.3. ^{137}Cs activity concentration in annual shoots (Bq kg^{-1}).

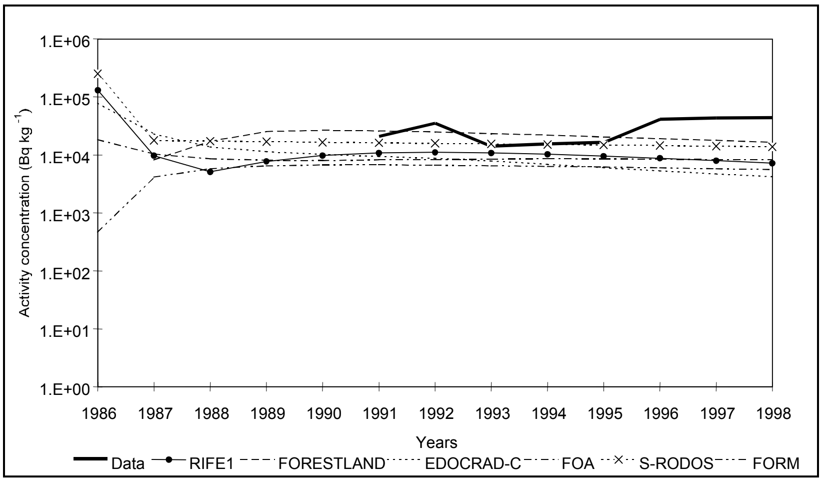


FIG. 6.4. ^{137}Cs activity concentration in needles (Bq kg^{-1}).

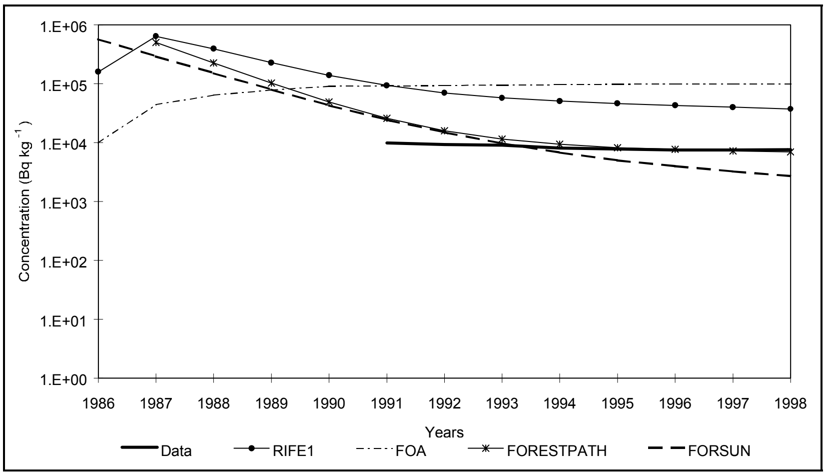


FIG. 6.5. ^{137}Cs activity concentration in soil profile (layer AoL) (Bq kg^{-1} dry weight).

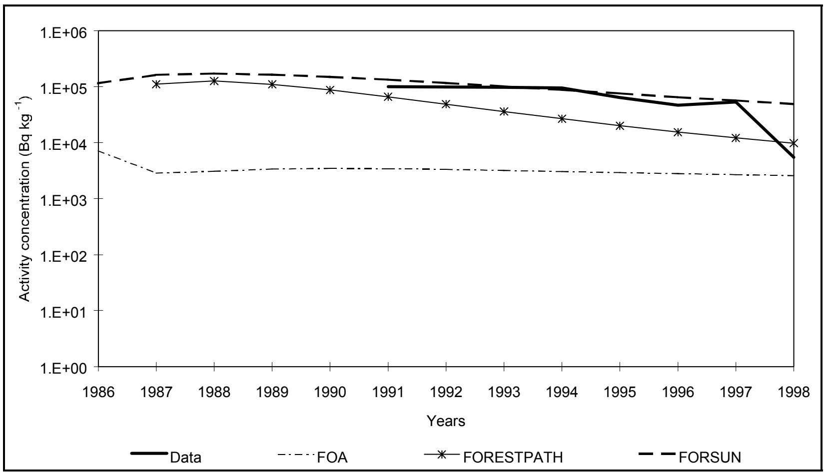


FIG. 6.6. ^{137}Cs activity concentration in soil profile (layer AoF) (Bq kg^{-1} dry weight).

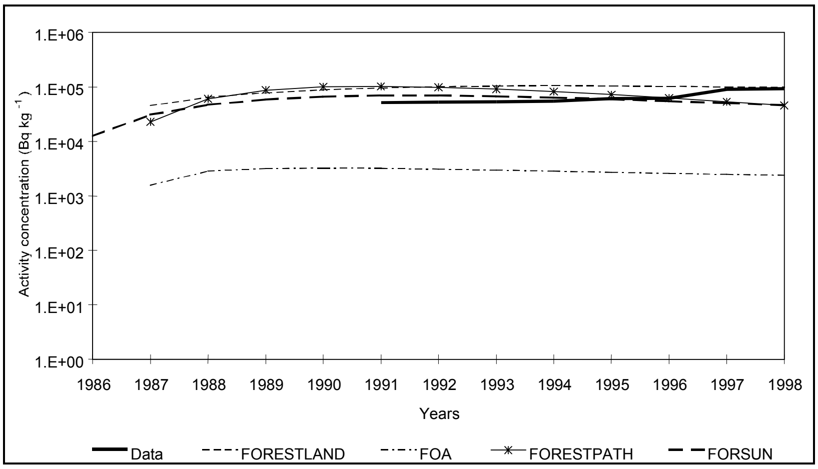


FIG. 6.7. ^{137}Cs activity concentration in soil profile (layer AoH) (Bq kg^{-1} dry weight).

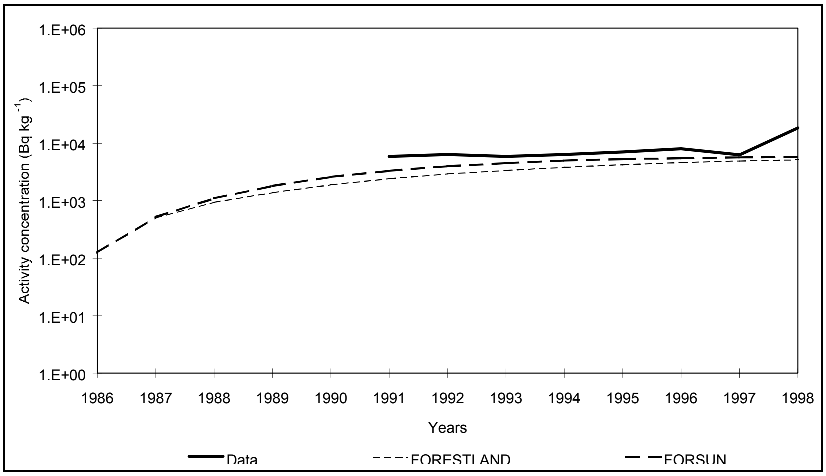


FIG. 6.8. ^{137}Cs activity concentration in soil profile (first slice of Layer Ah) (Bq kg^{-1} dry weight).

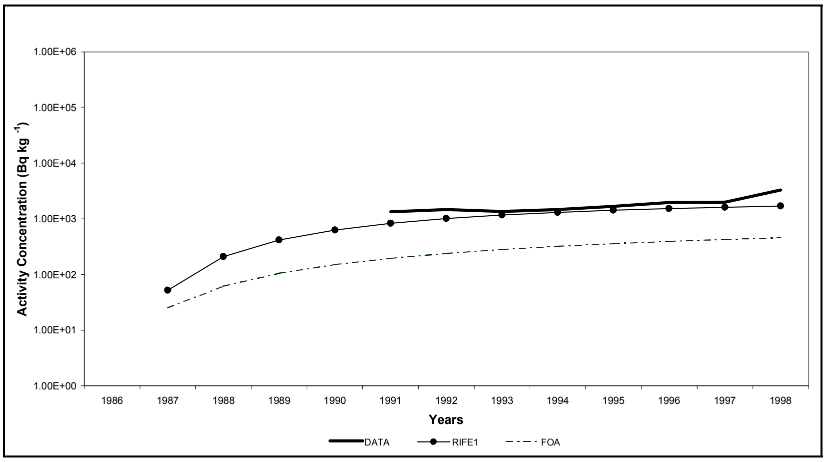


FIG. 6.9. Averaged ^{137}*Cs activity concentration in soil profile (total layer Ah) (Bq kg*$^{-1}$ *dry weight).*

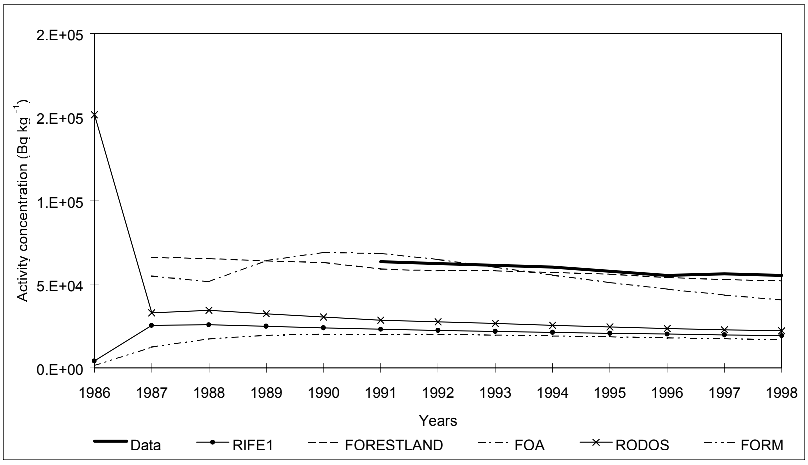


FIG. 6.10. ^{137}*Cs activity concentration in roe deer (Bq kg*$^{-1}$ *dry weight).*

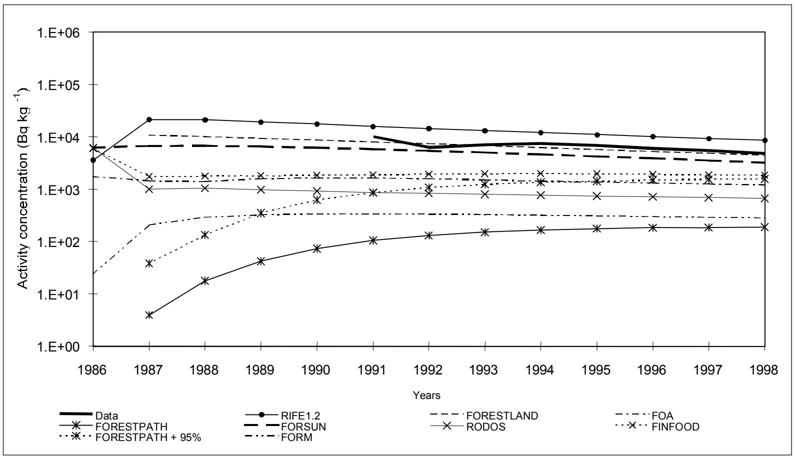


FIG. 6.11. ^{137}*Cs activity concentration in bilberries (Bq kg*$^{-1}$*fresh weight).*

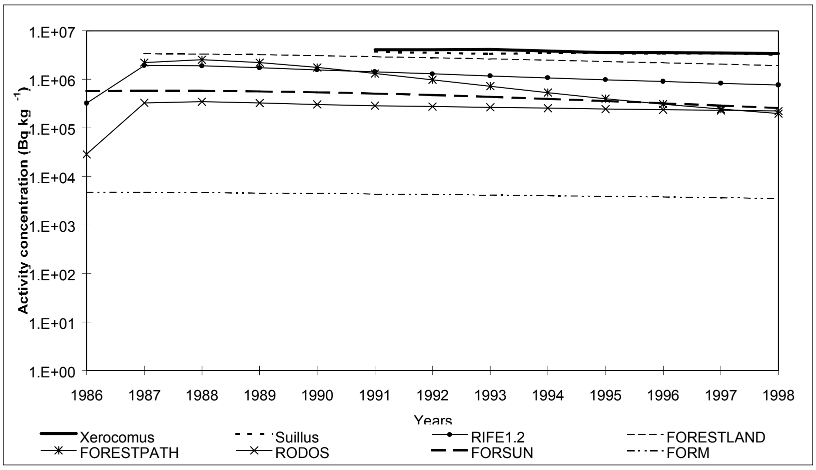


FIG. 6.12. ^{137}Cs activity concentration in mushrooms, Xerocomus badius and Suillus luteus (Bq kg^{-1}dry weight).

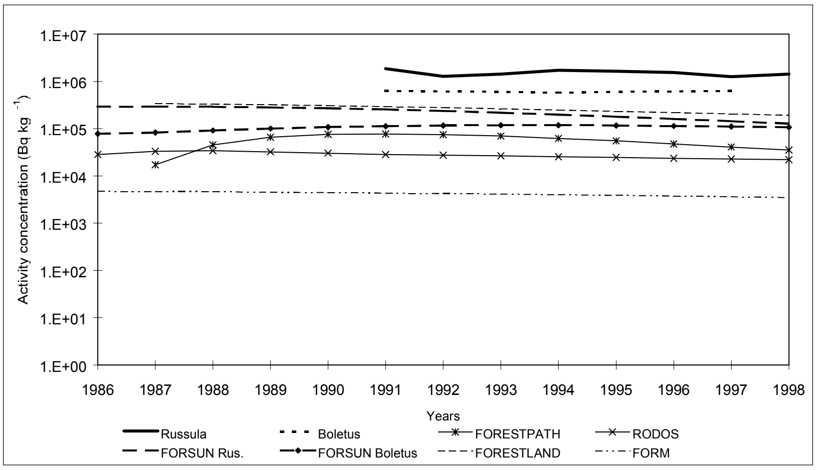


FIG. 6.13. ^{137}Cs activity concentration in mushrooms, Russula paludosa and Boletus edulis (Bq kg^{-1}dry weight).

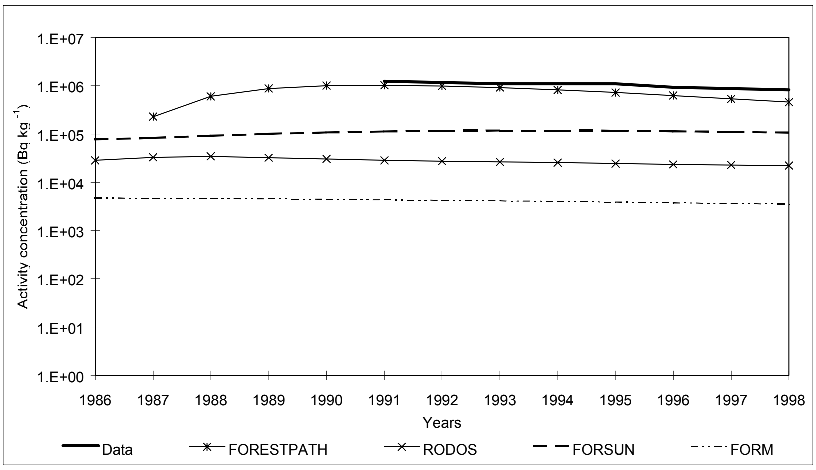


FIG. 6.14. ^{137}Cs activity concentration in mushrooms, Cantharellus cibarius (Bq kg^{-1} dry weight).

6.4.2. Soils

Model predictions for soil contamination over the 12 year assessment period were submitted for five models, FORESTLAND, FORESTPATH, FOA, FORSUN and RIFE. Direct comparison of all models for all soil endpoints was made impossible by the fact that some models are designed to return averaged predictions for some soil layers. Data from the Zhitomir site were available for the following soil layers:

—	AoL	litter;
—	AoF	fermentation horizon;
—	AoH	humified horizon;
—	Ah	dark, mineral-humic horizon;
—	E	bleached, eluvial horizon;

Each of the models provided either horizon-specific results or results which are averaged over more than one horizon.

Figure 6.5 shows predictions of RIFE, FOA, FORESTPATH and FORSUN for the AoL layer. Both RIFE and FOA overpredict the measured data by approximately one order of magnitude, whereas FORESTPATH and FORSUN both make more accurate predictions of litter activities, especially over the period 1993 to 1998. The dynamics of litter activity predicted by FORSUN, FORESTPATH and RIFE are all quite similar: each model predicts a decline in litter activity after the first year following contamination. FOA predicts a continuing build-up of activity with a plateau establishing after approximately 5 years.

Figure 6.6 shows predictions of FOA, FORESTPATH and FORSUN of activity in the AoF layer. Again, FORESTPATH and FORSUN show similar dynamics and are both quite accurate in their predictions. FOA shows rather different dynamics and tends to underpredict the measured data, possibly as a result of overpredicting the litter activities.

Figure 6.7 shows predictions of FOA, FORESTPATH, FORESTLAND and FORSUN of activity in the AoH horizon. FORESTPATH, FORESTLAND and FORSUN each show very similar dynamics and accuracy in reproducing the measured data. FOA again underpredicts the measured data by approximately one order of magnitude.

Data for the 10-cm thick Ah horizon were presented for individual 2cm thick slices (5 in all). Only FORESTLAND and FORSUN provide predictions for these individual 2 cm slices and, as an illustration, the result of FORESTLAND and FORSUN predictions for the uppermost 2 cm slice of the Ah horizon is shown in Figure 6.8. As for previous soil endpoints, both of these models performed extremely consistently and accurately with both predicting both the magnitude and time course of Ah activity very accurately.

The RIFE and FOA models provided averaged predictions for the Ah horizon, shown in Figure 6.9. Both models predicted the dynamic accumulation of activity within the averaged Ah horizon, although the FOA model underpredicted the Ah activities by slightly less than one order of magnitude.

The conclusions from this part of the exercise show that some extremely accurate predictions of soil contamination are possible with the models put forward in this study. One major problem in a model inter-comparison such as this, however, is that different model structures do not always allow a direct comparison between predictions. This is particularly evident in the case of soils in which conceptual subdivisions of the soil column are artificial and reflect (a) the individual requirements of a model; and (b) the individual conceptual approach of the modeller concerned.

6.4.3. Other biological endpoints

The biological endpoints of concern in Scenario 2 are roe deer, bilberries and mushrooms (various species). Model predictions of contamination time courses for each of these components of the forest ecosystem are shown in Figures 6.10, 6.11 and 6.12 respectively.

6.4.3.1. Roe deer

Five modellers provided predictions for time courses of contamination in roe deer. Of the five models, two (FORM and RIFE) interpret game in a generic sense while the remaining three specify the species of interest. The spread of predictions produced by these models (Figure 6.10) is relatively large, spanning two to three orders of magnitude for the period for which measured data are available. Both FOA and FORESTLAND produced accurate predictions of ^{137}Cs activity in roe deer, both with similar dynamics. The other three models (FORM, RIFE and RODOS) produced predictions which were consistent with each other but which were approximately one to two orders of magnitude below the measured values. Only RODOS predicted a significantly higher activity in roe deer immediately following contamination in 1986, but no measured data are available to verify this prediction.

6.4.3.2. Bilberry

Predictions of time courses of contamination of the understorey focussed on bilberries since the dominant herbaceous species present at the Zhitomir site are *Vaccinium* species. Results were provided by eight of the modellers. The range of predictions was large, spanning 3 to 4 orders of magnitude (Figure 6.11). This range of predicted values was asymmetrically distributed around the measured values, with most of the models underpredicting the measured data significantly. RIFE, FORESTLAND and FORSUN each made predictions significantly less than one order of magnitude different from the measured data. It is noticeable that the dynamics of bilberry contamination predicted by each model are very similar for the 1991 – 1998 period for which data are available, suggesting that most modellers agree on the processes that contribute to ^{137}Cs contamination of bilberry, and that discrepancies between models are most likely due to differences in calibration.

6.4.3.3. Mushrooms

Results were returned for *Xerocomus badius* and *Suillus luteus* (Figure 6.12), *Russula paludosa* and *Boletus edulis* (Figure 6.13) and *Cantharellus cibarius* (Figure 6.14) using each of the models with the exception of ECORAD. For brevity all of these results can be described as an ensemble, since the predictions for each of the mushroom species were very similar. Each of the predictions was characterised by a) a broad spread of predicted values (generally three to four orders of magnitude) and b) a generally significant underprediction of the measured values by each of the models. Individual models did perform well for single species (for instance FORESTLAND performed well for *Xerocomus badius* and *Suillus luteus* and FORESTPATH performed well for *Cantharellus cibarius*) but in general the performance of the models was not as good as it was for other end points.

It was concluded in Scenario 1 (see Section 5) that a high degree of variability between the predictions for mushrooms indicates that the modelling of mushroom contamination remains probably the most problematic aspect of forest modelling. The results of the blind predictions in Scenario 2 confirm this view.

6.5. GENERAL SUMMARY

The results obtained for Scenario 2 in general support the conclusions drawn from Scenario 1. Hence, for many of the conceptual compartments represented within the forest models available to the Forest WG there is generally a high level of consistency between predictions made by the models tested. Furthermore, when compared with the Rudnya-Povcha data set, many of the model predictions proved rather accurate, although admittedly over a period some 5 to 12 years after initial contamination when the 'early' dynamics of ^{137}Cs within the forest ecosystem are likely to have been superseded by slower, long-term rates of redistribution within the forest. Particularly accurate and consistent predictions were made for the tree-related compartments and for certain soil compartments.

Yet again, however, the biological endpoints (roe deer, bilberries and mushrooms) proved more difficult to model. Predictions of all these endpoints proved more variable and, sometimes, consistently inaccurate (especially predictions for mushroom contamination).

6.6. STATISTICAL ANALYSIS

6.6.1. Objective

The objective of the analysis presented here was to apply the statistical method described in Section 5.6.2 to the results of the model-data inter-comparison. The specific questions addressed in this analysis were as follows.

— What is the degree of agreement between model predictions and experimental data?

— For which endpoint (wood, bark, needles) is the degree of agreement the best or the worst?

6.6.2. Statistical method

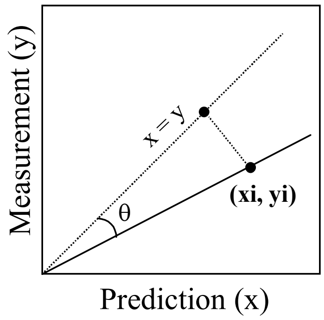
The methodology applied is derived from an original statistical method proposed by Williams & Leggett [1984] for comparison of predictive models against experimental data. When comparing model predictions with experimental data, two distinct types of uncertainties have to be considered:

(a) uncertainty associated solely with the model, which usually arises from incomplete understanding of the phenomenon being modelled; and

(b) uncertainty associated with the observations, arising from the inherent variability of the phenomenon being measured and from imprecision in the measurement procedure.

According to Williams and Leggett [1984], two reliability indices can be used as a measure of the accuracy of a model: a geometrically intuitive reliability index, kg, and a statistically rigorous reliability index, ks. For reasonably accurate models ($k<2$), k_g and k_s can be used interchangeably as reliability index (it was show that, in these conditions, $0.989 < k_g/k_s < 1.027$), but the geometrical definition of the reliability index is easier to conceptualise. If there is a time set $\{y_1, ..., y_n\}$ of observations corresponding to a time set $\{x_1, ..., x_n\}$ of model predictions, the goodness of the prediction x_i is determined by the relative proximity of the point (x_i, y_i) to the line x=y.

The definition of the reliability index k_g is then obtained as follows:



$$\mathrm{Tan}(\theta) = \frac{\sqrt{(x_i - y_i)^2}}{\sqrt{(x_i + y_i)^2}}$$

$$s = \sqrt{\frac{1}{n}\sum_{i=1}^{n} \tan^2 \theta_i} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{x_i - y_i}{x_i + y_i}\right)^2}$$

$$k_g = \frac{1+s}{1-s}$$

Therefore:

$$k_g = \frac{1 + \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{1-\frac{y_i}{x_i}}{1+\frac{y_i}{x_i}}\right)}}{1 - \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{1-\frac{y_i}{x_i}}{1+\frac{y_i}{x_i}}\right)}}$$

6.6.3. Results

The model predictions were compared with yearly means of experimental on-site values measured between 1991 to 1998. In the case of roe deer and mushrooms experimental values were missing for some years.

The results of the comparison are summarised in Tables 6.2 to 6.6.

TABLE 6.2. MODEL-DATA COMPARISON: k_s VALUES FOR TREE COMPARTMENTS

Tree parts					
	Wood	Bark	Needles	Shoots	Average
Ecorad-C	5,75	1,37	4,73	5,08	4,23
Forestland	1,60	1,33	1,83	2,21	1,74
FOA	1,88	3,41	3,40	2,09	2,70
Forestpath	5,18	12,74	65,07		27,66
S-Rodos		1,38	2,10	1,91	1,80
Rife1	2,03	1,08	3,21		2,11
Form	3,47		4,57		4,02
Average	3,32	3,55	12,13	2,82	
St. Dev	1,50	4,98	25,35	0,15	

TABLE 6.3. MODEL-DATA COMPARISON: k_s VALUES FOR SOIL COMPARTMENTS

Soil profile							
	Ol (0–2 cm)	Of (2–6 cm)	Ol+Of (0–6 cm)	Oh (6–8 cm)	Ah (8–10 cm)	Ah (average)	Average
Forestland			1,62	1,69	2,07		1,79
FOA	11,70	20,70	2,21	23,16		5,45	12,64
Forestpath	1,50	2,78	2,21	1,67			2,04
Forsun	1,93	2,21	1,68	1,45	1,70		1,79
Rife1	6,39					1,42	3,91
Average	8,22	10,37	3,50	9,36	1,89	3,43	
St. Dev	5,36	9,27	2,98	10,18	0,26	2,85	

TABLE 6.4. MODEL-DATA COMPARISON: k_s VALUES FOR MUSHROOMS

Mushrooms						
	Cantharellus	Russula	Boletus	Xerocomus	Suillus	Average
Forestland		6,34	2,52	1,58	1,49	2,98
Forestpath	1,47	27,25	10,70	8,38	8,50	11,26
Rodos	41,08	6,06	24,12	14,99	13,87	20,02
Forsun	9,05	8,25	5,40	10,18	9,67	8,51
Rife1				9,33		9,33
Form		385,42	154,43	956,91	883,52	595,07
Average	17,20	86,66	39,43	166,90	183,41	
St. Dev	21,03	167,25	64,82	387,05	391,40	

TABLE 6.5. MODEL-DATA COMPARISON: k_s VALUES FOR BILBERRIES

Bilberries	
Forestland	1,17
Forestpath	43,22
Rodos	13,25
Forsun	1,54
Rife1	1,76
S-Rodos	3,53
Form	21,33
Average	12,09
St. Dev	17,12

TABLE 6.6. MODEL-DATA COMPARISON: k_s VALUES FOR ROE DEER

Roe deer	
Forestland	1,06
FOA	1,20
Rodos	2,36
Rife1	2,81
Form	3,17
Average	2,12
St. Dev	0,95

A synthesised estimation of the models accuracy for all studied endpoints is presented in Table 6.7. All tree compartments are gathered under “Tree parts” and the soil compartments are subdivided into an organic and a mineral part. In this table, “3”, “2” and “1” means that model predictions and the experimental data agree within a factor of 2, 10 and >10, respectively. If the endpoint was not estimated by the model them the corresponding cell remains blank. The last column is obtained by summing the values for all endpoints and can be seen as a global index of the model predictive capacity and accuracy. According to this index, the models can be ranked as follows (from best to worst):

FORESTLAND, RIFE1 > FOA, FORESTPATH, FORSUN,
S-RODOS > ECORAD-C, RODOS, FORM

TABLE 6.7. MODEL-DATA COMPARISON: SYNTHESIS

	Tree parts	Organic soil	Mineral soil	Mushrooms	Bilberries	Roe deer	Total
Ecorad-C	2						2
Forestland	3	3	3	3	3	3	18
FOA	3	1	2		3	3	12
Forestpath	1	3		2	1		7
Rodos				1	1	3	5
Forsun		3	3	2	3		11
Rife1	3	2	3	2	3	3	16
S-Rodos	3				3		6
Form	2			1	1	3	7

6.6.4. Conclusions of statistical analysis

Except for mushrooms and to a lesser extent for berries, the predictions made by the models were in good agreement with the experimental data (the reliability index was less than 5 in all cases). Overall, FORESTLAND and RIFE1 give the best results (cf. Table 6.7). Notwithstanding the quality of these models, this is probably partly due to the fact that they were calibrated with measurements from the same, or similar, geographical area as the reference scenario. Appropriate calibration is a powerful determinant of model accuracy and it might be expected that the results of the model inter-comparison would have been different if data obtained from another region had been selected as the reference scenario. Finally, the conclusions about model accuracy have to be treated with care because they are based on measurements over a relatively short period (8 years) compared to the time scale of the biological and geochemical processes in a pine forest ecosystem.

This inter-comparison exercise has shown that, in general, there is a satisfactory agreement between the blind predictions of nine existing models and measured data on ^{137}Cs behaviour in multiple forest compartments. Differences in time dynamics estimations between models show, nevertheless, that a better understanding of the ^{137}Cs behaviour and cycling in forest ecosystems is still needed. This requires continued collection of experimental data, which will also assist in improving representation of this behaviour in models.

7. SECOND MODEL-MODEL INTER-COMPARISON STUDY – ‘SCENARIO 3’

7.1. INTRODUCTION

This section describes results from the third model inter-comparison study undertaken by the Forest WG. This study, ‘Scenario 3’, involved a model-model comparison exercise similar to Scenario 1. However, Scenario 3 was radically different from the previous two scenarios in that a subterranean source of ^{137}Cs was considered. One of the conclusions of the Scenario 2 exercise was that the generally high success of each of the models in predicting the dynamics of ^{137}Cs at the Rudnya-Povcha site could be due in part to the fact that all models had been developed and calibrated for a Chernobyl-type source term. Discussion within the Forest WG revealed that very few participants had considered a scenario which, instead of involving a discrete deposition event from the atmosphere, consisted of a prolonged or ‘chronic’ release to the soil from an underground source.

The scenario developed for the third inter-comparison was based on a hypothetical, though realistic, case in which a shallow waste repository containing ^{137}Cs had been capped by a clean 1-m-thick cover. The details of the repository, described in Section 7.2, were based on a previous IAEA study on Quantitative Acceptance Criteria for Near Surface Disposal of Radioactive Waste [IAEA, 1999]. It was assumed that, at the time of capping of the waste-filled trenches, no vegetation existed on the soil surface and, therefore, modellers had to take into account the development of a forest cover over a period of 50 to 200 years through the process of natural regeneration. A summary of Scenario 3 is provided in the next few paragraphs.

7.2. SUMMARY OF SCENARIO 3

The complete scenario is listed in Annex D. The source term is a series of ten minimally engineered trenches filled with loosely tipped radioactive waste containing ^{137}Cs. The waste material is covered with a 1-m-thick layer of clean soil; though this soil is initially devoid of vegetation, a tree cover becomes naturally established following closure of the repository. It was requested that participants provide predictions of ^{137}Cs activity concentrations in the end points listed below over a period over 50 to 200 years following repository closure. A maximum simulation period of 200 years takes ^{137}Cs through 6.7 physical half-lives, which still leaves a significant activity within the system. One of the interesting questions to be addressed by this scenario was whether the ^{137}Cs activity concentrations within the soil and biological endpoints achieved steady state over this period.

The dominant tree species assumed was Scots pine (*Pinus sylvestris*) with sparse examples of birch. Two hundred years after closure the average height of the trees was assumed to be 20-25 m and the average density of wood biomass on the trenches between 10 and 20 kg m^{-2}. No information on tree growth rates is available over this period. Of particular importance to this scenario were assumptions concerning tree root distributions within the trench caps, which were as follows:

- pine root growth rate decreases with age;
- pine roots reach half maximum depth after 10-15 years;
- root distribution of pine trees older than 40-60 years do not change significantly with time.

Full details of pine root distribution for 12 year old trees were provided in the scenario, described in Annex D. The end points for which model predictions of activity concentrations (dry weight) were requested were as follows:

- total tree, *Pinus sylvestris*;
- total wood, *Pinus sylvestris*;
- needles, *Pinus sylvestris*;
- total bark, *Pinus sylvestris*;
- 'mushrooms' (specifically *Xerocomus badius* and Boletus species);
- bilberry *(Vaccinum macrocarpon)*;
- soil – 10 cm depth increments from 0 to 110 cm from the surface.

As in previous scenarios, not all of the models were designed to predict radiocaesium concentrations in all of the required endpoints. One specific problem was that some models made predictions of activity concentrations for soil depths averaged over relatively larger increments, rather than from specific depth increments.

7.3. PARTICIPANTS AND MODELS

The individual modellers, and their respective models, which participated in Scenario 3 are listed in Table 7.1. The reduction in the number of participants compared with previous scenarios reflects the fact that this scenario was novel and some of the models which had been designed and written to address a Chernobyl-type scenario could not easily be modified to simulate tree uptake from a subterranean source term.

TABLE 7.1. MODELLERS AND MODELS PARTICIPATING IN SCENARIO 3 OF THE FOREST WG MODEL INTER-COMPARISON. BRIEF DESCRIPTIONS OF EACH OF THESE MODELS ARE PROVIDED IN ANNEX A

Modeller(s)	Model	Institute
R. Avila and L. Moberg	FORESTLAND	SSI, Stockholm, Sweden
S. Fesenko and S. Spiridonov	FORESTLAND	RIARAE, Russia
R. Bergman	FOA	NDRE, Umea, Sweden
M. Frissel	FORM	IAEA, Vienna, Austria
A. Konoplev and A. Bulgakov	FORWASTE	Typhoon, Obninsk, Russia
I. Linkov	FORESTPATH	Harvard University, USA
S. Mamikhin	ECORAD-C	MSU, Moscow, Russia
G. Shaw	RIFE	Imperial College, UK

7.4. RESULTS

The results are presented graphically (Figures 7.1 to 7.10) as time courses over 50 to 200 years following initial trench closure. All the results are reported as 'best estimates' only. For some endpoints some modellers provided results for a 50-year simulation period only.

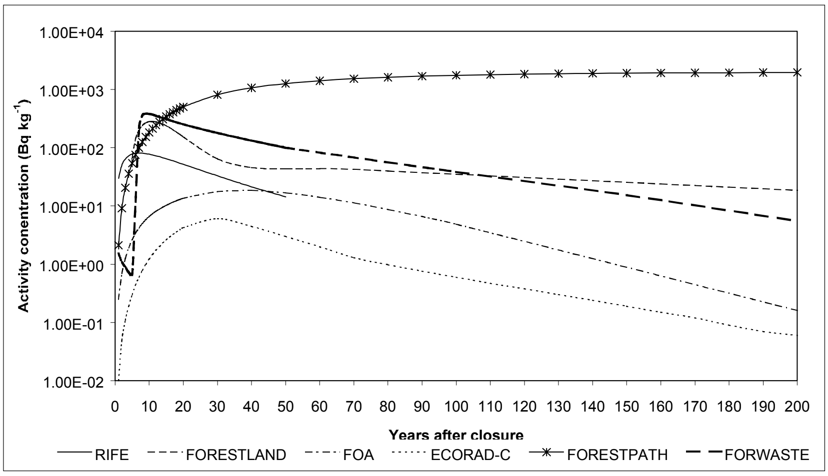


FIG. 7.1. ^{137}Cs activity concentration in total tree (Bq kg^{-1}).

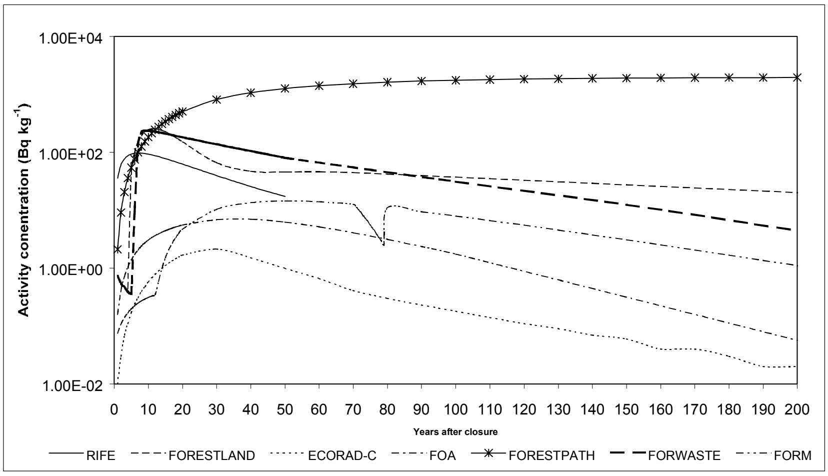


FIG. 7.2. ^{137}Cs activity concentration in wood without bark (Bq kg^{-1}).

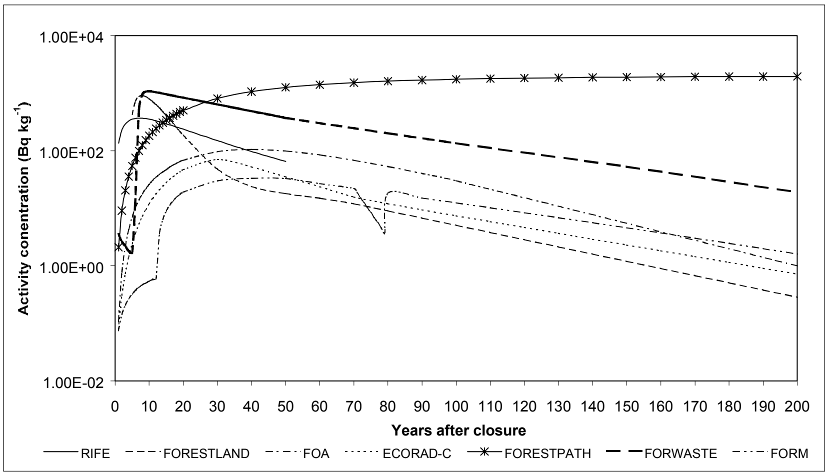


FIG. 7.3. ^{137}Cs activity concentration in needles (Bq kg^{-1}).

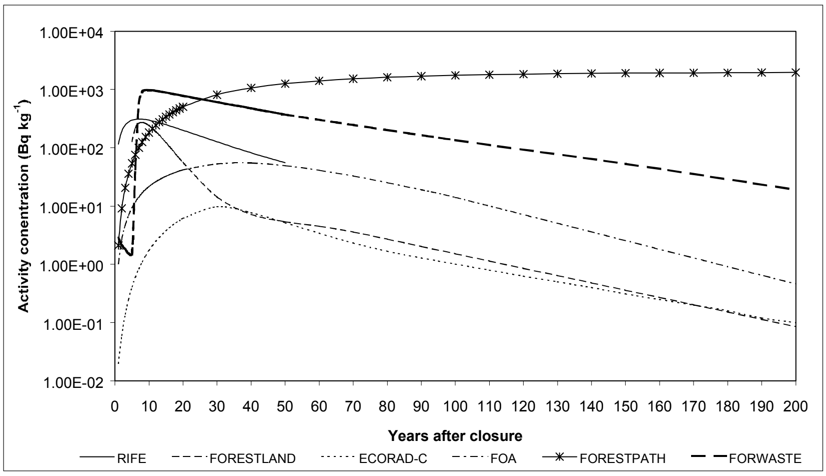


FIG. 7.4. ^{137}Cs activity concentration in total bark (Bq kg^{-1}).

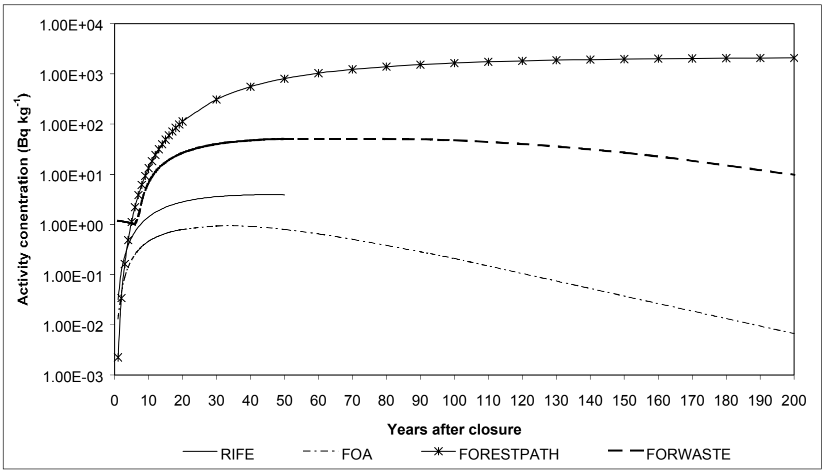


FIG. 7.5. ^{137}Cs activity concentration soil cover, (dry weight, Bq kg^{-1}), 0–10 cm from surface.

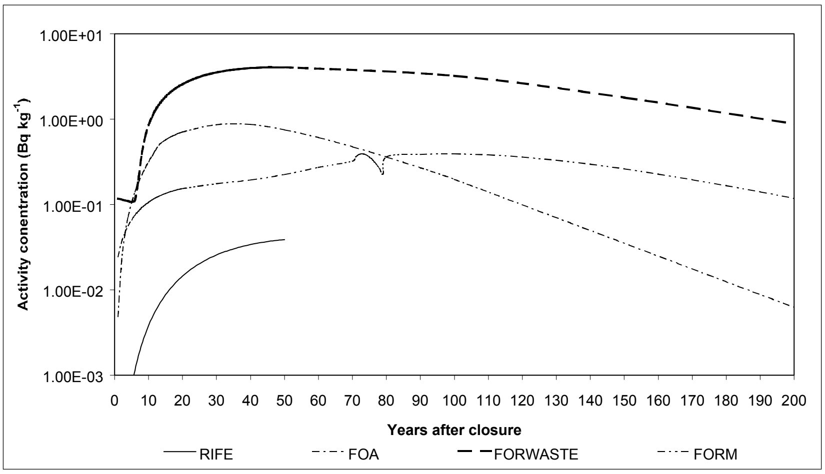


FIG. 7.6. ^{137}Cs activity concentration in soil cover (dry weight Bq kg^{-1}), 30–40 cm from surface.

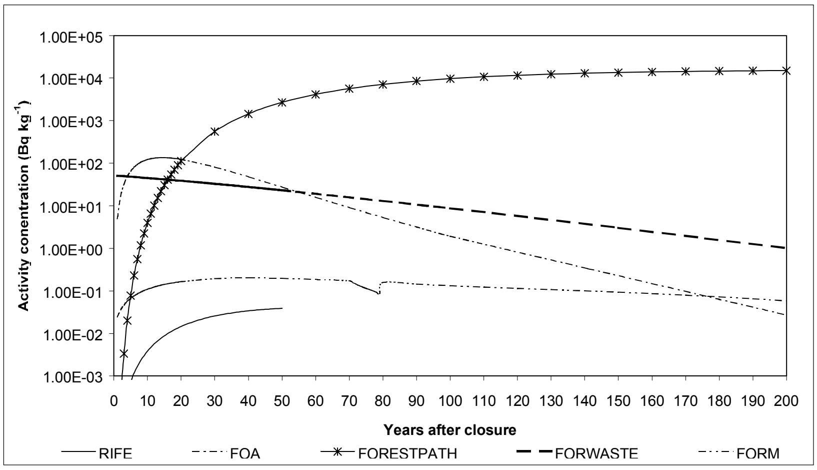


FIG. 7.7. ^{137}Cs activity concentration soil cover, (dry weight, Bq kg$^{-1)}$, 90–100 cm from surface.

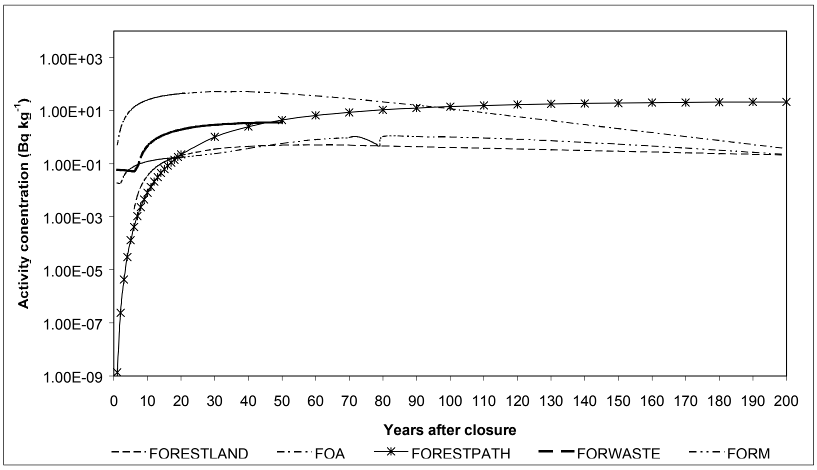


FIG. 7.8. ^{137}Cs activity concentration in bilberries, (fresh weight, Bq kg^{-1}).

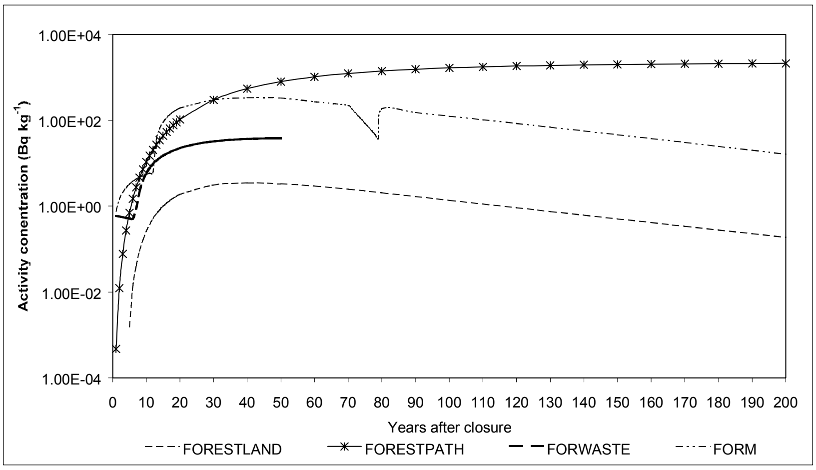


FIG. 7.9. ^{137}Cs activity concentration in Xerocomus badius (fresh weight, Bq kg^{-1}).

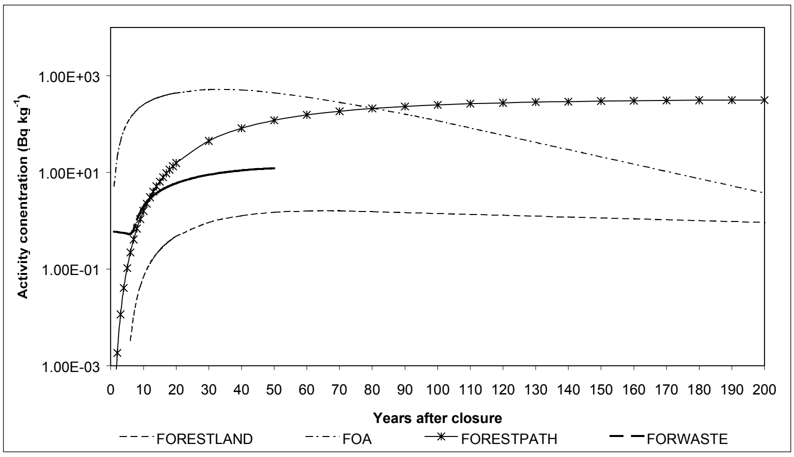


FIG. 7.10. ^{137}Cs activity concentration in Boletus edulis (fresh weight, Bq kg^{-1}).

7.4.1. Trees and associated components

Predictions of activity concentrations of specific components of trees (*Pinus sylvestris*) were provided by all seven respondents, although the FORM model did not return results for the total tree or bark. Figures 7.1, 7.2, 7.3 and 7.4 show plotted results for each of the specific tree-related endpoints, namely total tree, wood (without bark), needles and total bark (including cambium).

For each of these endpoints all models except FORESTPATH predicted very similar contamination dynamics over the 50 to 200 year simulation period. Broadly, the consensus of these simulations is that an initially rapid increase in ^{137}Cs activity concentrations in all tree tissues occurs over a period of 10 to 40 years, followed by a steady decline to the end of the 200 year simulation period which can probably be accounted for by radioactive decay. FORESTPATH also predicted a rapid increase in activity concentrations of all tissues up to approximately 50 years following trench capping, but thereafter, despite accounting for radioactive decay, this model predicted a levelling off of the tree activity concentrations, suggesting a balance between tree uptake and radioactive decay.

Examining the time trends predicted by the individual models reveals some interesting nuances within the simulations. The tree uptake dynamics predicted by the FORWASTE model were characterised by an initial decline in ^{137}Cs activity concentration over a period of approximately five years before significant uptake of ^{137}Cs by the tree tissues began. It is unclear why this pattern of uptake emerged but might be due to an assumption that the ‘clean’ soil contained residual ^{137}Cs from atmospheric fallout. The FORM model also produced interesting simulations of activity concentrations in wood and needles in which discrete inflections in the simulated time trends were evident at 10 and 70 years. These times correspond to tree ages at which thinning or harvesting might normally be expected to occur in a managed forest and the FORM model takes this into account. As described above, however, the overall time trend predicted by FORM was similar to that predicted by most of the other models.

In a model-model inter-comparison based on such a novel and hypothetical scenario it is difficult to draw hard conclusions concerning the absolute variability between the predictions of different models. Nevertheless, it is striking that, in general, there was agreement within two orders of magnitude between most models after the commencement of the steady state phase of predictions after 10 to 50 years. This is particularly interesting considering that none of the models had previously been calibrated for this type of scenario and the modellers relied solely on their own judgement when making the necessary modifications to their models to represent, conceptually, the system to be modelled in Scenario 3. Selection of appropriate model parameters also relied on this judgement. A key conceptual problem for all the modellers was how to represent the process of upwards transport of ^{137}Cs through the soil profile and into the trees and vegetation. Solutions to this problem were critical to the simulated vertical distributions of ^{137}Cs which are described in the next section.

7.4.2. Soils

The initial soil profile in Scenario 3 was completely homogeneous in structure, so the scenario description did not include any details of soil horizons. For this reason it was requested that soil distributions of ^{137}Cs were presented for arbitrary 10-cm soil layers. Some models (*e.g.* FOA and RIFE) only consider a limited number of averaged soil depths, so only selected results produced by the models are considered here. Time course simulations of ^{137}Cs activity concentration at soil depths of 0 – 10, 30 – 40 and 90 – 100 cm are shown in Figures 7.5, 7.6 and 7.7, respectively. The simulations for the 0 – 10 cm depth (Figure 7.5) show very similar time trends but the range of absolute activity concentrations predicted spans up to five orders of magnitude. The envelope of absolute activity concentrations predicted for the mid-section of the soil profile (Figure 7.6) also spans a similar range of values, but one of the models (FORM) predicts a rather different time trend when compared to the others for which data are available at this depth. Finally, predictions for the deepest part of the originally clean soil profile (90–100 cm, Figure 7.7) were highly divergent both in the nature of the time trends predicted and the magnitude of soil contamination.

Upwards transport of a relatively highly sorbed radionuclide such as ^{137}Cs can occur as a result of advection-diffusion mechanisms, but also by biological mechanisms such as root uptake and translocation and the bulk turnover of soil by soil fauna (bioturbation). The differences in model simulations of the soil distribution of ^{137}Cs in Scenario 3 may be largely due to different assumptions being made by modellers about the relative importance of these various transport mechanisms. However, though the results presented here suggest that this problem is worthy of further study they do not suggest a clearly preferred methodology for this type of scenario.

7.4.3. Other biological endpoints

The biological endpoints of concern in Scenario 3 were limited to bilberries and two species of ‘mushrooms’. The results for these are shown in Figures 7.8, 7.9 and 7.10, respectively.

7.4.3.1. Bilberries

Five modellers provided predictions for time courses of contamination in bilberries. The time courses of bilberry contamination predicted by each of these models were similar in both form and magnitude. The contamination of plant species and mushrooms over the long term is principally dependent on soil contamination and, given the conceptual difficulties in modelling soil transport of ^{137}Cs which were identified in the previous section, it is striking that the bilberry predictions are so similar. The main difference between the five models was that, as in the case of tree contamination, FORESTPATH predicted that a plateau of contamination would be reached after some 50 years, whereas the other models predicted a general decline in bilberry activity after 50 to 70 years. The overall discrepancy in absolute predicted activity concentrations was two to three orders of magnitude.

7.4.3.2. 'Mushrooms'

Predictions were provided by four modellers for *Xerocomus badius* and by four modellers for *Boletus* species, though only three modellers provided predictions for both species. Once again, the general trends of each of the predictions were similar and the overall uncertainty in absolute predicted activity concentrations was approximately three orders of magnitude. This slightly greater uncertainty of the absolute predicted values, compared with those for bilberry, probably reflects the generally greater uncertainty associated with transfer coefficients for 'mushroom' species than for green plants. The similar form of the predicted time trends, again compared with the results for trees and bilberries, no doubt reflects the fact that it is the activity concentrations predicted for the upper soil layer(s) which is the main determinant of contamination of biological endpoints.

7.5. GENERAL SUMMARY

Scenario 3 is dramatically different from both the preceding scenarios and is sufficiently novel, insofar as the modellers' experience is concerned, that the results obtained should be treated with some caution. Nevertheless, the scenario provided some fascinating results that may be very useful in guiding the design and selection of future forest modelling inter-comparison scenarios.

Given the fact that none of the modellers who participated in Scenario 3 has previously had any opportunity to develop or calibrate a model for the uptake and redistribution of ^{137}Cs from a subterranean source, the degree of agreement between predictions for tree components and for biological endpoints has been striking. What is not clear from these results, however, is why there should be such generally good agreement between models. Perhaps the key question to be addressed is the way in which individual modellers represented vertical, upward soil transport. From the selected results shown here for predicted soil distributions there is evidently a degree of dissimilarity in the manner and extent by which soil transport has been approached by each of the models. A key question arising from the results of Scenario 3 is whether physical or biological transport of ^{137}Cs through the soil is likely to dominate a) when the ^{137}Cs is below the surface and b) when deep-rooted plants such as trees are allowed to access such subterranean sources.

8. CONCLUSIONS

8.1. INTRODUCTION

During the period 1998–2000, the BIOMASS Forest WG has addressed a number of issues relevant to the improvement and validation of existing models which are designed to predict the behaviour and fate of radionuclides, principally ^{137}Cs, in forest ecosystems. The Forest WG has provided an active forum within which 11 modelling groups have put forward their models for testing and inter-comparison in three scenarios. In addition, reviews have been undertaken concerning our fundamental understanding of biogeochemical cycling in forest ecosystems, the use of interaction matrices in model design and process identification, and the definition and application of the transfer factor concept in forest ecosystems. The main conclusions from each of these components of the Forest WG work programme are now summarised.

8.2. CONCLUSIONS FROM REVIEWS

- Research into the dynamics of radiocaesium cycling in forests has often been limited to understanding single biogeochemical pathways rather than examining the system *in toto*.
- More data sets addressing radiocaesium cycling within forests on a whole ecosystem basis would be useful to aid model development and validation.
- To achieve the above, there is still a need for a standardisation of current approaches to data acquisition in a format which can be used to facilitate the comparison of radiocaesium cycling at different geographical locations and, preferably, on a whole-ecosystem basis.
- An alternative way to improve the development of conceptual models is the application of a systematic method of identifying dominant features, events and processes (FEPs) using an ‘interaction matrix’ approach.
- Carried out correctly, the interaction matrix method should introduce a higher level of objectivity into model design and development.
- When modelling a complex system, such as a forest, the interaction matrix method should enhance the ability of modellers to determine what level of aggregation is optimal and what interactions or pathways should be included.
- Transfer factors, of one type or another, are likely to remain key parameters within dynamic forest models.
- It is impossible to recommend a ‘best option’ for the type of transfer factor which should be used in any particular model since the choice of transfer factors will usually depend on the purpose of the model being constructed.
- Irrespective of the purpose of the model or the type of transfer factor used, however, the precise definition of the transfer factor adopted, and how it used in a model, should be clearly described by the modeller.
- The definition of transfer factors suitable for application to perennial woody vegetation such as trees remains problematic because the radionuclide burden of wood may have been accumulated over a period of several decades and single TF values may not adequately reflect this.
- A novel method to determine the wood interception potential (WIP) for radiocaesium has been proposed.

8.3. CONCLUSIONS FROM MODEL INTER-COMPARISON EXERCISES

- For the soil and tree compartments examined in Scenario 1 there was generally a high level of consistency between predictions made by the 11 models tested.
- A high level of agreement between predictions for the soil compartments was particularly evident in Scenario 1 and this doubtless reflects the research efforts which have been made over the last 10 years in understanding processes controlling radiocaesium migration in soils.
- Predictions of all the biological endpoints proved more variable, especially predictions for mushroom contamination.

- This raises the question of whether our current (mainly deterministic) modelling approaches are suitable to predict the behaviour of biological entities which will inevitably exhibit a high degree of variability.
- The results of the Scenario 2 inter-comparison generally confirmed those of Scenario 1.
- Particularly accurate and consistent predictions were made for the tree-related compartments and for certain soil compartments at the Rudnya-Povcha site.
- The biological endpoints (roe deer, bilberries and mushrooms), however, proved more difficult to model with certainty.
- A major limitation of the Scenario 2 inter-comparison was that the time-scale for which data were available was limited to the period from 1991 to 1998, which excludes both short-term and genuinely long-term trends.
- There is clearly still a need to keep adding to existing data sets to ensure that the genuinely long term trends of ^{137}Cs distribution in forests are recorded and understood.
- Scenario 3 was dramatically different from the preceding two scenarios and provided a stern test of the ability of modellers to adapt both their conceptual ideas and parameters when considering a subterranean source term of ^{137}Cs.
- The key conceptual question which arose as a result of Scenario 3 was whether vertically upward transport of ^{137}Cs in a soil profile is best considered as a physical process or as a biologically mediated process, especially in the presence of deep-rooting trees.

9. RECOMMENDATIONS AND SUGGESTIONS FOR FUTURE WORK

The following recommendations and suggestions for future work are based on a comprehensive discussion between members of the Forest WG during the final BIOMASS meeting in November 2000. This discussion centred on three main issues, as follows:

- the design and management of model inter-comparison studies;
- improvements in the way radionuclide transfers in forest ecosystems are modelled; and
- suggestions for future work by the BIOMASS Forest WG.

Recommendations and suggestions in each of these categories are presented below.

9.1. RECOMMENDATIONS CONCERNING THE DESIGN AND MANAGEMENT OF MODEL INTER-COMPARISON STUDIES

- Establishment of ground rules for model inter-comparisons. The primary aim of the Forest WG was, from the outset, to conduct model-model and model-data inter-comparisons using available forest radioecology models. While the choice of scenarios was somewhat limited (see next point) it was clear that there were several different ways in which the inter-comparisons could be designed and implemented. The manner in which the three inter-comparisons reported in Sections 5, 6 and 7 of this report were conducted is not presumed to be optimal, but in the absence of any generally accepted guidelines for such exercises it was difficult to judge how best to manage each scenario. Questions arose within the Forest WG as to whether all modellers should make predictions for all endpoints, should model-data inter-comparisons be ‘double blind’ (which effectively excludes a modeller who has a suitable data set on which to base an inter-comparison scenario since he/she might already have used those data to calibrate their model), how many modelling iterations should be allowed and should modellers be allowed to make second-round predictions after seeing the data set they first tried to model unseen? These and several other questions lead to the recommendation that a generally accepted set of ground rules should be established (or at least agreed upon by any group of modellers wishing to compare and test their models) to simplify the management of model inter-comparison studies and to assist in quality assurance of these exercises.

- Choice of ‘challenging’ scenarios. One general criticism of the three inter-comparisons conducted by the Forest WG was that they were too similar to scenarios with which most of the modellers were familiar. In other words, the chosen scenarios were not challenging enough. Scenario 3 (Section 7) was the most challenging, involving a sub-surface source term which none of the modellers had previously addressed. However, even this scenario prescribed a forest type which had been the focus of most of the modellers’ previous efforts. It is recommended, perhaps as part of establishing a set of ground rules as described above, that scenarios are designed which test the available models to the reasonable limits of their performance.
- Provision of single ‘calibration’ datum points in blind model-data inter-comparisons. One of the problems in conducting a truly blind model-data test is that the modellers receive no feedback on the performance of their model until they see the full data set after submission of their simulation. Provision to the modellers of a single datum point from the blind data set as part of the inter-comparison scenario has several advantages. The main advantage for the modeller is that he/she can determine how well his/her model is performing with respect to that datum point, thereby avoiding order of magnitude discrepancies between data and model predictions. This benefits both modeller and the inter-comparison in general since it ensures that discrepancies between the performance of individual models are not due to ‘wildcard’ errors in the calibration of individual models for the specific scenario being considered. The further benefit for the inter-comparison is that, even if all models ‘predict’ the single calibration datum perfectly, the simulated kinetics of each model before and after that datum point can be compared to provide valuable information on the discrepancies between simulations. Selection of a suitable datum point from a blind data set which may contain tens or even hundreds of data is a difficult question, but it is recommended that such a datum is provided when conducting model-data inter-comparisons.
- The need to give estimates of uncertainty in model simulations. Despite the request for modellers to provide estimates of 95% uncertainty bounds around predictions made in each of the three model inter-comparisons conducted by the Forest WG, only one model (FORESTPATH) consistently provided such estimates. Both Type A (stochastic) and Type B (system/process) uncertainties contribute to the overall uncertainty of a model simulation. To some extent, the potential for Type B uncertainties in individual models can be gauged by comparing the representation of forest compartments and transfer processes in that model against an idealised system representation, such as that shown in the matrix in Figure 2.2, although this comparison is qualitative. It is, however, becoming increasingly more straightforward to represent stochastic uncertainties of parameters in models and it is recommended that co-ordinators of model inter-comparison exercises make it mandatory for participants to provide uncertainty estimates, even if this dictates that scenarios are kept simple to avoid overloading participating modellers.

- Model versus modeller uncertainty. An interesting question is that of model versus modeller uncertainty. This was not a real issue in the Forest WG since each modeller was using a model which he/she had developed themselves. The question arises, however, if two modellers are each using the same model independent of each other. This question has been addressed in detail by Linkov and Burmistrov [2001].
- Model-data comparisons are most useful. It was the generally agreed conclusion of the Forest WG that scenario 2, the model-data inter-comparison, proved to be the most interesting and the most useful test of model performance. This is in accord with the conclusions of Hoffman and Hofer [1988], who concluded that the applicability of model-model inter-comparisons was questionable because this method "offers no measure of accuracy without independent test data". It is recommended that, where possible (i.e. when genuinely independent and unseen data sets are available), model-data inter-comparisons should be conducted in preference to model-model inter-comparisons.
- Are forest models and modellers fit for purpose? Since the inter-comparison exercises addressed by the Forest WG did not deal with specific applications (e.g. dose assessment, evaluation of countermeasure effectiveness) the question of 'fitness for purpose' of participating models is difficult to answer. However, recommendations for future studies in Section 9.3 include the suggestion that specific case studies be addressed. It is recommended that such studies be pursued to determine whether the current generation of forest radioecology models is fit for purpose.

9.2. RECOMMENDATIONS CONCERNING MODELLING OF RADIONUCLIDE CYCLING IN FOREST ECOSYSTEMS

- Time dependency in forest models should include tree growth. One of the fundamental characteristics of trees is that their biomass changes with time. This change is generally referred to as 'growth', which implies an increase in biomass with time although, in the later stages of a tree's life cycle, its biomass can decline significantly. The complex natural cycle of biomass change in forests is closely linked to elemental cycling, especially the cycling of carbon which constitutes the major proportion of biomass of trees. Models of ecosystem development proposed by authors such as Aagren and Bosatta [1996] incorporate growth as an integral and fundamental component of the models. However, in many current radioecological models growth is not explicitly (or even implicitly) represented – no mention is made of tree growth in Figure 2.2, for instance! It is recommended that in any future BIOMASS exercises involving forests that tree biomass be considered as a process of fundamental radioecological significance.
- High versus low contamination scenarios. It was evident after the Chernobyl accident that differences in the initial deposited activity of a radionuclide can vary by several orders of magnitude, especially if the near-field and far-field are compared. The point was raised during the Forest WG discussions that almost all the models participating in the inter-comparison exercises represent transfer processes as linear phenomena, hence differences in the total activity burden within a forest make no difference to the model simulations. Over the potential order-of-magnitude range of activity concentrations, however, it is possible that this assumption is invalid. Furthermore, the existence of large concentration ranges of naturally occurring analogues such as ^{133}Cs and K for ^{137}Cs, ^{88}Sr and Ca for ^{90}Sr, can lead to non-linear behaviour, especially during biological uptake and sorption in soils. One of the Forest WG models (FORSUN) represents

competition between the Cs and K ions during soil sorption as a non-linear process. However, it is recommended that other modellers begin to represent key mechanisms in their models which might be inherently non-linear

- Process-oriented models are preferable for long-term predictions. Leading on from the last point, it was concluded that process-orientated (i.e. mechanistic) models, which would include key non-linear processes such as diffusion, are preferable for long-term predictions since site-specific calibration for the far future is impossible. This approach would also facilitate the inclusion of other pollutants in forest models since the use of empirical transfer functions and coefficients would be avoided as much as possible. This approach is recommended, but the practical difficulties involved are acknowledged and the reader is referred to the discussion of transfer factors in Section 3 of this report.
- Integration of forest radioecology models with models for other ecosystems. With the advent of improved software for modelling and the ability to integrate models with Geographical Information Systems, it is recommended that forest radioecology models be designed in future as part of general landscape models. The consideration of ecosystems such as forests and adjacent aquatic systems as separate ecological entities is artificial, though greatly simplifies the modeller's task. However, for the future development of forest radioecology models it would be preferable, and particularly useful for particular assessment problems as radioactive waste disposal, to focus at a landscape level.
- Is it possible to construct a generic model for any ecosystem based on fundamental ecological properties? This final and challenging question was raised by the Forest WG. It is well established that variations in ecosystem type between climatically distinct locations, for instance, are due to variations in primary production, biological decomposition rates etc. The ability to model radionuclide behaviour based on such fundamental properties of an ecosystem would enable the model to be scaled to warm/cold and wet/dry climates. This characteristic would help to address the potential effects of climate change on radionuclide behaviour in the environment, especially over the long time scales which need to be addressed in the context of waste disposal.

9.3. SUGGESTIONS FOR FUTURE WORK BY THE BIOMASS FOREST WORKING GROUP

The following list of suggestions is proposed on the basis of detailed discussion within the Forest WG, although only brief discussions and justifications for each suggestion are provided here:

- Long-term and retrospective predictions of radionuclide behaviour in forests. As already discussed above, certain assessment problems, such as radioactive waste disposal, demand that long-term simulations of radionuclides in natural ecosystems such as forests are made. The validity of model simulations over periods of several decades could be determined using retrospective case studies for which calibration data could be obtained for the present day. One good candidate for such a study would be the forested area affected by the Kyshtym accident in 1957.
- The problem of forest edges. Forest edges are important since it is known that enhanced wet and dry deposition occurs here and they are often exploited for food by man and animals in a significantly different way than the 'deep' forest. Forest edges are common and extensive landscape features and consideration of forest edges might necessitate the landscape approach described in Section 9.2.

- The geosphere/forest interface. An important interface of significance to the problem of migration of radionuclides from underground repositories is the geosphere/forest interface. Scenario 3 (Section 7) made a start in addressing the role of forest vegetation in enhancing the migration of radiocaesium from the sub-surface to the surface soil horizons. However, further consideration of this type of scenario might assist the work of BIOMASS Theme I.
- The effect of forest fires on radionuclide distribution. Forest fires are an extremely important influence in the long-term development of forest world-wide, although specific information on the effects of such fires on re-distribution of radionuclides is almost non-existent.
- Application of existing forest models to case studies such as countermeasure/remediation effectiveness and establishment of reference levels for forest products such as timber/timber products.
- Integration of Forest WG activities with those of BIOMASS Themes I and II. The relevance of forest modelling to the problem of radioactive waste disposal has been mentioned in Sections 9.1 and 9.2. Dose reconstruction activities, especially for accidents such as Kyshtym and Chernobyl, could also benefit from the inclusion of the expertise of the Forest WG members.

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