Radioecological Sensitivity

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After the release of radionuclides into the environment it is important to be able to readily identify major routes of radiation exposure, the most highly exposed individuals or populations and the geographical areas of most concern. Radioecological sensitivity can be broadly defined as the extent to which an ecosystem contributes to an enhanced radiation exposure to Man and biota. The concept can be applied to humans and other biota, but the focus for the forum has been on considering radioecological sensitivity with respect to humans.

There have been significant recent improvements in our capability to estimate spatial variation in the environmental behaviour of radionuclides through better understanding of the underlying processes. In addition, there has been an improved ability to apply this knowledge by integrating relevant spatial information in the form of geo-referenced data sets using Geographical Information Systems. Thus, it is now possible to refine the estimation of spatial variation in radiation exposure, both for routine releases and in accident situations.

Radioecological sensitivity analysis attempts to integrate current knowledge on pathways, spatially attribute the underlying processes determining transfer and thereby identify the most radioecologically sensitive areas. This identifies where high exposure may occur and why. Single food products or species can be considered, or a number of key parameters to identify areas where sensitive pathways may occur together.

A consideration of the concept of radioecological sensitivity is given in this report, based on a series of forum meetings. The relevant issues with respect to radioecological sensitivity for terrestrial and aquatic ecosystems were considered and suitable, generally applicable indicators for radioecological sensitivity analysis discussed. The conclusions are outlined, considering the potential usefulness of the concept, and methods by which it can be applied. A framework for the estimation of radioecological sensitivity is proposed and the various indicators by which it can be considered have been identified. These are (i) aggregated transfer coefficients (Tag), (ii) action (and critical) loads, (iii) fluxes and (iv) individual exposure of humans. The importance of spatial and temporal consideration of each of these outputs is emphasized. It is important to be able to provide information on the extent of radionuclide transfer and exposure to humans at different spatial scales, to reflect the sometimes large spatial differences which occur. Single values for large areas, such as countries, can often mask huge variation within the country. Similarly, the relative importance of different pathways can change with time and therefore assessments of radiological sensitivity are needed over different time periods after contamination. In general, terrestrial and freshwater ecosystems are more radioecologically sensitive to atmospheric radioactive contamination than marine systems.

Due to the availability of data on radionuclides in the environment, and for various practical reasons, the radioecological sensitivity concept is foreseen to be primarily applicable to accidental situations and to a limited number of radionuclides. However, the radioecological sensitivity concept is not a concept that will be used in an acute situation such as directly after an accident. It is rather meant to be used in radiation protection, nuclear safety and emergency preparedness when there is a need to identify areas that have the potential of being of particular concern from a risk perspective. Prior identification of radioecologically sensitive areas and exposed individuals should improve the focus of emergency preparedness and planning, and contribute to environmental impact assessment for future facilities.

Further work is needed on the uncertainties associated with the use of spatial data in radioecological sensitivity analysis. The concept of radioecological sensitivity should be extended to a consideration of doses to biota. The desirability of being able to do this was one reason for identifying the intermediary indicator of the aggregated transfer coefficient.

1. INTRODUCTION

Certain components of ecosystems can accumulate large amounts of radionuclides. The extent of variation depends upon the radionuclide and the type of ecosystem. For some radionuclides (especially Cs, Sr and I isotopes), there is now a good understanding of the underlying environmental factors leading to high exposure of humans and also improved information on variation in dietary and social habits. This enables an improvement in the identification of critical groups and quantification of the extent of their exposure. Furthermore, we know that many factors leading to high exposure can vary both spatially and temporally, and that this can be important in determining individual doses. Many post Chernobyl studies have demonstrated that the highest exposures do not necessarily coincide with the most contaminated areas, especially in the mid-long term after an accident.

A consideration of time and space can therefore help to identify not only the key exposure routes and associated critical groups, but also the locations where high exposure will occur and where it will be sustained for longer periods of time. These analyses have been facilitated by the increasing use of Geographical Information Systems (GIS) combining dynamic models with spatially varying information. This has, in turn, prompted a reconsideration of the concept of radioecological sensitivity, which was first proposed by Aarkrog (1979), who outlined an approach to estimating radioecological sensitivity in a study quantifying doses arising from global fallout of ^{137}Cs and ^{90}Sr in Denmark and the Faroe Islands.

The concerted action has functioned by holding a series of meetings to discuss the terms of reference for the forum. When considering radioecological sensitivity, it has discussed the issues which should be resolved and the criteria which need to be considered. The forum has then considered which quantities can be used as indicators of radioecological sensitivity with a particular focus on identifying sensitive areas as well as sensitive processes and communities in both terrestrial and aquatic ecosystems.

There is the potential to consider sensitivity with respect to doses to biota, but the focus within this forum has been restricted to a consideration of human exposure. It is only in recent years that the effect of radiation on biota has become of concern, and a consideration of radioecological sensitivity of biota is needed, building on the progress now achieved under EU-financed research projects, a separate concerted action by the International Union of Radioecologists (IUR) and other international and national bodies.

2. TERMS OF REFERENCE

Forum members agreed the following terms of reference during the first two meetings.

The Forum will provide a sound basis on which radioecology can proceed with respect to the concept and application of radioecological sensitivity. It should help in identifying and quantifying the radionuclide transfer capacity in different ecosystems and by doing so reducing the radioecological sensitivity through better emergency preparedness planning.

The goal for the Forum is to develop a system for making an assessment of radioecological sensitivity. This should be done based on development of a range of parameters and a method by which they can be compared, contrasted and combined. Radioecological sensitivity should be considered over both the short term and long term since radionuclide behaviour varies with time in different ecosystems. The following parameters should be the main focus for initial assessment.

- ◊ Action load
- ◊ Flux
- ◊ Human exposure

Each of the above factors needs to be addressed and defined in the context of radioecological sensitivity. Their usefulness, appropriateness and quantification should provide a focus to the working group studies.

The reference for the forums work is to focus on spatially varying conditions and how these affect the end point, namely Man, either directly or indirectly. Geographical factors can be both environmental parameters and man's interaction and management of the land. The consequences for man should be looked at with respect to a range of different criteria. Initially, the following criteria should be addressed:

- ◊ Health risk (collective and individual dose)
- ◊ Economic impacts (intervention levels)
- ◊ Emergency preparedness

3. OBJECTIVES

The overall aim of this concerted action is to provide a standardised system, i.e. consensus, in relation to the methodology and terminology used to define «sensitivity» and «resilience». The forum was established to achieve the following objectives:

- to promote the dissemination of information on methodology and scientific results between researchers working in related radioecological disciplines, where a definition of radioecological sensitivity is needed.
- to provide a forum for discussion and synthesis of the concept of radioecological sensitivity by a combination of working groups and discussion groups.
- to provide a definition of radioecological sensitivity in the context of radioecology and radiation protection, which will aid in the implementation of policy
- to prepare a consensual document clearly defining the terminology and methodology to be used in future radioecological sensitivity assessments
- to provide an input into describing potential studies which might be justified under the fifth framework.

Subsidiary objectives with respect to methodology were also defined during the forum discussions as follows:

- to define radioecological sensitivity with reference to areas of small spatial scale with well defined geographical and population characteristics.
- to identify physical quantities that will represent radioecological sensitivity in a well-defined, conceptually simple, easy to measure and generally applicable way.
- to identify sensitive pathways on the basis of expert knowledge in the absence of quantitative information. To provide physical quantities to define radioecological sensitivity.
- to identify the potential users of the assessment of radioecological sensitivity.
- to identify relevant characteristics of exposed populations that may influence radioecological sensitivity.

4. RADIATION PROTECTION ISSUES FOR RADIOECOLOGICAL SENSITIVITY[1]

The International Commission on Radiological Protection (ICRP), makes a distinction between what is called practices and interventions (ICRP 1991 (Publication 60)). A practice is a human activity that is undertaken by choice but which increases the overall exposure of man (and biota). Practices are controlled to restrict the additional radiation doses. Intervention, on the other hand, is an action against radiation exposures that already exist with the intention to reduce the exposures. According to the ICRP, both practices and interventions are justified when they cause more good than harm. The main protection principle in both cases is that protection should be optimised which means that all doses should be kept as low as reasonably achievable, economic and social factors being taken into account. In addition, dose limitation and constraints are necessary to prevent the optimised situation from being one where a few individuals receive inappropriately high doses.

The sources of radioactivity may be routine or accidental and they include both man-made and natural sources of radioactivity. The routine situations include releases from nuclear installations (mining, milling, fuel fabrication, reactor operation, reprocessing, radioactive waste handling, conditioning and disposal) but also releases from other sources like hospitals, research establishments and phosphate plants. The routine releases are regulated as practices according to ICRP. Accidents may lead to substantial contamination of the environment and are classified as interventions.

In principle, radioecological sensitivity is a generic concept and, in that sense, source-independent. It is applicable to practices as well as interventions. However, the practical application is different for these two different situations. For practices, the radioecological sensitivity of an area or ecosystem can be taken into account as part of a pre-planning and an optimisation procedure of a particular source.

For accidents, knowledge about radioecological sensitivity incorporated as part of emergency preparedness can assist in prioritising (e.g. the identification of what areas should be considered first, in what way and when) after an accident has occurred.

Due to the availability of data on radionuclides in the environment, and for various practical reasons, the radioecological sensitivity concept is foreseen to be primarily applicable to accidental situations and to a limited number of radionuclides. Normally, a practice should be handled according to the established and accepted radiation protection principles and standards which include protection of man (and the environment).

The radioecological sensitivity concept is applicable to all radionuclides, but sensitivity is radionuclide specific. Depending on the situation the radionuclides of concern varies. For routine releases from nuclear installations, radioactive waste and accidents, respectively, the radionuclides given in Table 4.1 are most relevant. In addition, radioecological sensitivity can also include naturally occurring radionuclides such as ^{210}Po.

Table 4.1 Most relevant radionuclides for a range of release types

Routine releases	Radioactive waste	Accidents
Cs, Sr, I, Co, Pu, Am, Np, Tc, S-35, C-14, H-3, Eu	Cl-36, Np-237	Cs, Sr, I, Pu, Ru, Ag

There are a number of radionuclides that are of potential interest for their radiological significance in various situation. In this report, we consider ^{131}I, ^{137}Cs and ^{90}Sr for the illustration of the concept. These three radionuclides are of particular importance in the case

[1] Background text on radiation protection principles is provided in Appendix A

of accidents and routine releases. They are chosen as examples in this report primarily due to the availability of field data. Thus, the intention is to give illustrative examples on the applicability of radioecological sensitivity, not to give a comprehensive description, which is outside the scope of this study.

5. BACKGROUND TO RADIOECOLOGICAL SENSITIVITY

5.1 Previous approaches to estimating sensitivity

Since the first use of nuclear weapons, there has been a considerable effort devoted to understanding and quantify the environmental behaviour of anthropogenic radionuclides. Certain pathways were identified as accumulating radionuclides in the 1960s, the most well known example being that of the atmosphere-lichen-reindeer-reindeer herder. Estuarine systems were shown to accumulate radionuclides discharged from the Sellafield reprocessing plant which were bound onto sediments. Critical exposure routes in the Irish sea included consumers of winkles and external exposure to people who spent many hours in close proximity to contaminated sediments (such as houseboat dwellers).

5.1.1 Critical group analysis

The emphasis of much of the previous work regarding identification of potentially high exposure to radiation has been on identifying critical groups. The use of the concept of critical groups has been widely used in radiation protection. Critical groups represent those members of the public who are most exposed from a particular practice (ICRP publication 60). The predicted mean dose to individuals within the critical group is compared with dose limits and constraints on the basis that if the dose is below the dose criteria then other members of the public will also be adequately protected. Analysis with respect to critical groups often involves site specific environmental monitoring data and habit information to provide best estimates of their doses, taking account of all relevant exposure pathways.

In the UK critical groups that have been identified include those (i) living closest to nuclear sites and receiving the greatest exposures from direct radiation and atmospheric pathways, (ii) consuming relatively high amounts of local seafoods and (iii) consuming wild foods from the local countryside. A study of the sensitivity of predicted critical group doses to changes in key input parameters has been conducted by Robinson *et al.*, (1996). Examples of their findings included that for Sellafield, the doses to high rate seafood consumers varied by a factor of three using the range of site-specific seafood intakes defined by the Ministry of Agriculture, Fisheries and Food (MAFF). The doses from terrestrial food ingestion were sensitive to assumptions about the chemical form and deposition velocity of iodine.

5.1.2 Radioecological sensitivity

In his treatise *Environmental Studies on Radioecological Sensitivity and Variability with Special Emphasis on the Fallout Nuclides ^{90}Sr and ^{137}Cs*, published in 1979, Aarkrog defines *Radioecological Sensitivity* as:

> … the infinite time-integrated radionuclide concentration in the environmental sample considered, arising from a deposition of 1 mCi km^{-2} of the radionuclide in question.

This definition is further elaborated in the General Introduction of his report:

> The radioecological sensitivity of a sample is the infinite time integral of appropriate quantities of the sample from an appropriate quantity of the radionuclide deposited. The radioecological sensitivity equals the steady state concentration in the sample of the radionuclide considered from a constant annual deposition rate of the radionuclide distributed like global fallout throughout the year.

Finally, the definition of *Radioecological Sensitivity* is repeated in the Concluding Remarks with the phrase "In this study radioecological sensitivity has been defined as the transfer factor from deposition to the environmental sample". Thus, *Radioecological Sensitivity*, as used by Aarkrog is closely related to the physical quantity of the *Aggregated Transfer Coefficient* (Tag). However, there is one important difference from the definition of *Tag* - as usually defined today - due to the form of deposition considered by Aarkrog. In the period between 1950 and 1970 considered by Aarkrog, the main concern was fallout from nuclear weapons tests in the atmosphere, Aarkrog considers the long-term steady deposition of radionuclides and adopts as unit deposition the accumulation of 1 mCi per km^2 of a certain radionuclide over a period of 1 year. In contrast, the aggregated transfer coefficient is usually defined for a pulse contamination.

The unit[2] of Radioecological Sensitivity that emerges from Aarkrog's definition is (e.g. for ^{90}Sr)

$$\frac{pCi\ ^{90}Sr\ kg^{-1}}{mCi\ ^{90}Sr\ km^{-2}\ y^{-1}} = \frac{pCi\ ^{90}Sr\ kg^{-1}\ y}{mCi\ ^{90}Sr\ km^{-2}}\ . \tag{1}$$

Alternatively, the numerator in eq. (1) may be expressed in terms of radioactivity ratio against a relevant stable isotope(s) (e.g. Ca or K) and the unit becomes

$$\frac{pCi\ ^{90}Sr\ (g\ Ca)^{-1}\ y}{mCi\ ^{90}Sr\ km^{-2}}\ . \tag{2}$$

Examples of the usage of this definition are given in Table 5.1.

Table 5.1 Examples of radioecological sensitivity estimates as defined by Aarkrog (1979) for products or humans contaminated by global fallout.

Environmental sample	Isotope	Sensitivity Value
Rye	^{90}Sr	32 pCi ^{90}Sr kg^{-1} y per mCi ^{90}Sr km^{-2}
Milk	^{137}Cs	3.43 pCi ^{137}Cs $(g\ K)^{-1}$ y per mCi ^{137}Cs km^{-2}
Beef	^{90}Sr	1.4 pCi ^{90}Sr kg^{-1} y per mCi ^{90}Sr km^{-2}
Milk (Faroe Islands)	^{90}Sr	9 pCi ^{90}Sr $(g\ Ca)^{-1}$ y per mCi ^{90}Sr km^{-2}
Human bone	^{90}Sr	5.1 pCi ^{90}Sr $(g\ Ca)^{-1}$ y per mCi ^{90}Sr km^{-2}
Danish total diet	^{137}Cs	4.2 pCi ^{137}Cs $(g\ K)^{-1}$ y per mCi ^{137}Cs km^{-2}
Danish human body	^{137}Cs	11.5 pCi ^{137}Cs $(g\ K)^{-1}$ y per mCi ^{137}Cs km^{-2}

In the Conclusions of his treatise , Aarkrog estimates doses to the population arising from the presence of ^{137}Cs and ^{90}Sr in foodstuffs. These dose estimates, expressed in mrad per ^{137}Cs $(g\ K)^{-1}$ y or mrad per pCi ^{90}Sr $(g\ Ca)^{-1}$ y.

Aarkrog estimated time integrated activity concentrations in milk over a long time period giving an estimate of radioecological sensitivity with the units[3] Bq ^{137}Cs (or ^{90}Sr) l^{-1} y per Bq m^{-2}. For ^{137}Cs, the value for the Faroe Islands was much higher for milk (at 60) than that for Denmark (of 4), but the Islands produced much less milk. Thus, the Faroe Islands were more radioecologically sensitive with respect to the individual consumption of ^{137}Cs in milk but less with respect to collective doses.

[2] the equations and data have been deliberately left in the original curie units
[3] The units in the original Aarkrog report have been converted into SI units.

Arctic Monitoring and Assessment Programme (AMAP)

In AMAP, an analysis was undertaken of the vulnerability of Arctic ecosystems to radionuclide contamination (Strand *et al.,* 1999). Compared to temperate ecosystems, the arctic was shown to be much more vulnerable to radionuclide contamination. The expert group on radioactivity concluded that arctic terrestrial and freshwater ecosystems were more vulnerable to radiocaesium contamination than were marine ecosystems. The main contributing factors to the enhanced vulnerability were the high transfer rates to semi-natural products in the arctic, the long ecological half-lives and the consumption of relatively large amounts of these products by arctic inhabitants, especially indigenous groups of reindeer herders. The most vulnerable pathway was the transfer of radiocaesium to humans via lichen and reindeer, however, other products such as milk, freshwater fish and lamb could also be important.

5.1.3 Critical Load Concept

Radioecological sensitivity can be quantified in terms of critical loads, which were originally developed in response to the impacts of anthropogenic acidifying emissions. Critical loads provide a practical approach for the development of controls of acidifying emissions strategies at national and international scales. The critical load approach has been developed to cover a wide range of both pollutants and receptors and can be defined as:

> *'a quantitative estimate of an exposure to one or more pollutants below which significantly harmful effects on specified sensitive elements of the environment do not occur according to present knowledge'*

Critical loads are damage thresholds for pollutant deposition, and imply that if deposition is below the threshold then there is no effect and thus no problem whereas if it is above the threshold then harm will occur. The concept is shown figuratively in Figure 5.1, although the type of effect/load relationship can obviously vary from that shown.

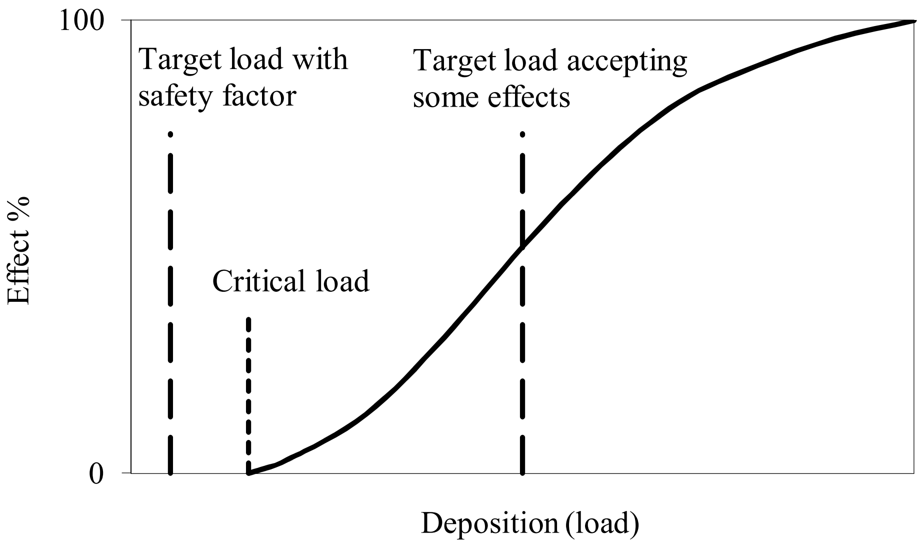


Figure 5.1. Theoretical dose-response curve showing comparison of target loads with critical load for previously considered pollutants.

From a radioecological perspective, the critical load for a food product has previously been defined as the level of radionuclide deposition (Bq m^{-2}) which leads to activity concentrations in a food product above intervention limits. They have been applied only for the medium to long term phase after deposition, and have not been used for the early phase when interception and weathering are important factors determining contamination. This approach was initially developed for radiocaesium using empirically derived aggregated transfer coefficients for clay, loam, sand and peat soil groups in the mid- to long-term after a radiocaesium deposition event with respect to ^{137}Cs transfer to cow milk (Wright *et al*., 1998). Estimation of critical loads has also been incorporated within semi-mechanistic models, allowing the dynamic quantification of radiocaesium critical loads for many food products after deposition. Since critical loads relate to soil to plant uptake, they are not relevant to the early phase after deposition, when surface contamination dominates.

In Sweden, the use of "critical deposition" values has been developed, based on an analysis of radionuclide behaviour in different agricultural Swedish areas by Eriksson (1997). The sensitivity for various crops and milk to accumulate caesium was estimated for different soil types, with or without fertilization. The critical deposition (expressed as Bq m^{-2}) needed to reach a specified caesium activity in a food product (for example 300 Bq kg^{-1} for milk) was used as a measure of the sensitivity. The effects of a critical deposition were also estimated for the years following deposition.

5.2. Definition of an area and ecosystems

An important part of the concept of radioecological sensitivity is to allow comparisons to be made between areas, identifying areas that may be more sensitive, or conversely resilient, to radioactive contamination. Identification of such areas both prior to and following a nuclear accident would allow for the cost-effective implementation of countermeasures.

Models provide representations of the real world. Increasingly, Geographical Information Systems, providing tools for the manipulation and analysis of geo referenced information, are being used within radioecological models to allow the incorporation of spatial variability in radionuclide behaviour. When quantifying radioecological sensitivity, it is therefore important to understand the behaviour of radionuclides within the environment, identifying those factors influencing transfer and its geographical variation. The spatial units used to quantify radioecological sensitivity should therefore be chosen to reflect the behaviour of the radionuclide in the environment, but may be constrained by the availability of suitable spatial data. In essence, spatial units can be considered as discrete geographic entities for which the variation in various characteristics can be identified. For example, the SAVE project (Howard *et al.,* 1999) estimated critical loads, the deposition necessary to achieve activity concentrations in a food product above its maximum permitted level, for ^{137}Cs in cow milk in western Europe. The spatial variation in critical loads in western Europe was estimated for a period starting two months after deposition when contamination of vegetation is dominated by the transfer of ^{137}Cs from soil to plant. Therefore, critical loads were estimated for spatial units based upon dominant soil type, further classified according to soil texture, for which variation in the transfer of ^{137}Cs was estimated. However, the identification of spatial units for assessing radioecological sensitivity based upon environmental characteristics may be inappropriate for some radionuclides. In the case of ^{131}I, because of its short half-life, it may be more appropriate to assess radioecological sensitivity using spatial units used to record the production of cow milk. The size and scale of the spatial units chosen may also require compromises to be made between the size of the area of interest, the amount of detail required and the information available.

If there is adequate data and understanding, radioecological models incorporating the spatial variation in input parameters may be implemented by imposing spatial units over the area of interest. For example, the SAVE-IT software package (Howard *et al.,* 1999) developed during the SAVE project, uses a raster data structure, with over 143 000 uniform spatial units (grid cells or pixels) with dimensions of 5 × 5 km. Essentially, input data sets of soil properties, ^{137}Cs deposition, agricultural production and diet are considered as flat Cartesian surfaces subdivided into uniform pixels. Input raster spatial data for the SAVE-IT system were derived using GIS to identify properties at the centre of each 5 × 5 km pixel; properties for each pixel could have been determined according to area within each pixel, but this may have biased the input data sets to those with the greatest area. Outputs from the SAVE-IT system, including critical load values, use the same data structure. A raster data structure was adopted as it is easily implemented allowing spatial data to be readily stored, manipulated and displayed. The size of the pixels, and consequently their number, further represents a compromise between the scale of analysis, data storage and model run time. The SAVE-IT system used uniform spatial units, but it may be more appropriate to use imposed spatial units that are not uniform. Such an approach was adopted for predictions of the impact of releases of radioactivity into the North Atlantic and Arctic seas in a box model (Iosjpe *et al*., 2002) developed under the ARMARA project. Spatial units have been delineated according to the physical characteristics of different marine areas such as position of landmasses and depth. Flows of water between these different marine areas and the behaviour of different radionuclides within them are used to predict the temporal variation in activity concentrations in different environmental compartments and consequent collective doses.

6. MEASURES OF RADIOECOLOGICAL SENSITIVITY

6.1. Radioecological sensitivity and exposure pathways

The definition of radioecological sensitivity should be as broad as possible because many different factors can influence the rate of exposure. Some factors are generically applicable to all radioactive contaminants whereas others would be relatively more important for just a few radionuclides.

A terrestrial ecosystem can be considered as radioecologically sensitive if it retains radionuclides for a long time in an available form to either Man or other biota. It may also act as a secondary source, disseminating radionuclides to surrounding ecosystems. Radioecological sensitivity can thus be broadly defined as the extent to which an ecosystem contributes to an enhanced radiation exposure to Man and biota.

Enhanced exposure can arise for a number of different reasons:

- an ecosystem collects and retains more contamination (high biomass concentration, high precipitation rates, proximity to nuclear sources, ..)
- retention or other characteristics enhance external exposure to Man and biota,
- an ecosystem makes the radioactive pollutants readily available to Man and biota (directly or via other ecosystems),
- Man's utilisation of an ecosystem (dietary habits, occupancy, agricultural practices).

Radioecological sensitivity is a very general term, which can be considered from a wide range of different perspectives. For most pollutants, sensitivity is assessed as the effects of pollutant input on various aspects of ecological functioning such as biochemical, physiological, morphological and behavioural responses.

In radiation protection terms, the primary aim has been to provide an appropriate standard of protection to Man as the final receptor of the radioactive pollutant. Therefore, the focus of sensitivity analysis is different, taking into account the behaviour/transfer of the pollutant via different compartments of the ecosystem which lead finally to doses to Man. In this context, radioecological sensitivity relates to the potential of the ecosystem to contribute to radiation exposure to Man.

Radioecological sensitivity in ecosystems is related to the following exposure routes to humans:

a) ingestion, which is dependent on environmental mobility, determined by rates of transfer between different ecosystem compartments and/or ecosystems themselves and on dietary habits,
b) the external exposure, which varies in different habitats, and which depends on social factors such as occupancy rate,
c) inhalation, which depends largely on the size of the contaminant radionuclide particles and on climatic conditions.

In the forum, the focus has been on a discussion of radioecological sensitivity with respect to the pathways leading to ingestion dose, but the criteria considered would also be relevant to the other three routes. To consider radioecological sensitivity, we need to define the appropriate quantities necessary to define sensitivity and to consider their temporal and spatial variation

6.2 Quantification of radiological sensitivity

The conclusions from forum discussions on how to quantify radioecological sensitivity are summarised in Figure 6.1. Even though the concept is applicable to all types of contamination, we have mainly considered atmospheric deposition as a contamination pathway. Four quantities were identified, three of which have been commonly used in radioecology or radiation protection, namely aggregated transfer coefficients, fluxes and individual exposure of humans. In addition, a fourth quantity, the action load, was identified as a useful sensitivity measure, which defines the deposition at which the activity concentration in a food product would equal maximum permitted levels in the period following deposition.

These four quantities (in rectangular boxes), are each influenced or defined by different processes or factors (shown in the oval boxes). For each of these quantities, both temporal and spatial variability need to be considered. Temporal considerations are potentially important and need to be considered from three perspectives:

- physical half-lives of radionuclides
- biological half-lives in various ecosystem components
- ecological half-lives in different ecosystem compartments and types of ecosystems.

Spatial variability will depend on factors such as:

- ecosystem characteristics,
- variation in human utilisation of terrestrial and aquatic resources
- climatic variation

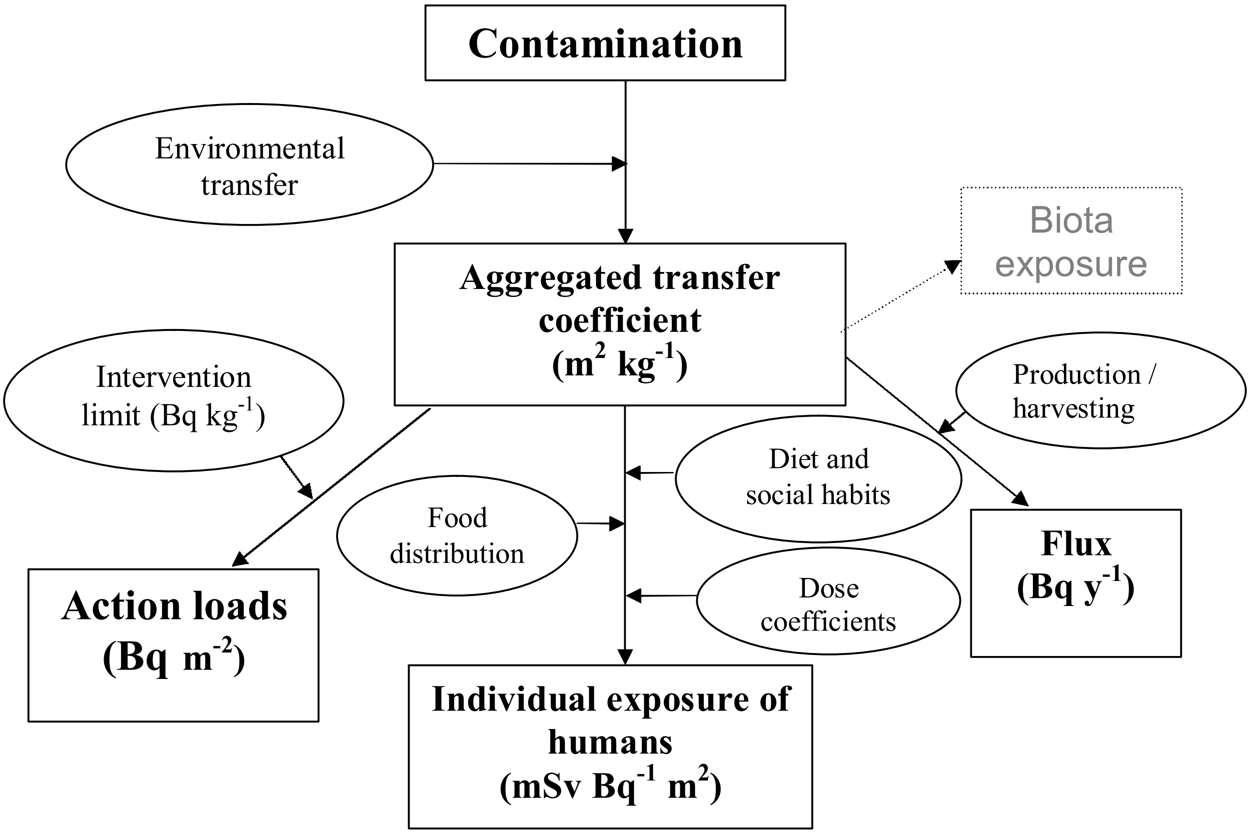


Figure 6.1 Scheme showing derivation and relationship between the four radioecological sensitivity indicators. All outputs are temporally and spatially variable.

It is important to consider radioecological sensitivity for different times after deposition to estimate doses, (e.g. 1 y, 5 y and 50 y) and also for different age groups. Spatial variability will depend on variation in both ecosystem characteristics and human utilisation of terrestrial and aquatic resources.

6.2.1 Transfer to environmental compartments

The extent of transfer to different environmental compartments, including food products, has been quantified using a variety of different transfer functions, including concentration ratios for soil-plant and transfer coefficients for plants to animals. In the above figure, 6.1, we have based the assessment on an area basis, and therefore the most appropriate transfer function to consider is the aggregated transfer coefficient, (Tag), defined as the activity concentration in a food product (Bq kg^{-1}) divided by the corresponding radionuclide deposition (Bq m^{-2}); with units of m^2 kg^{-1}. The application of Tag values is most suitable for terrestrial ecosystems, for freshwater and marine systems they are more difficult to quantify and apply appropriately. Different environmental factors influence the extent to which deposited radionuclides are transferred to food products. The various factors are discussed below for terrestrial, freshwater and marine ecosystems.

The relative importance of different factors obviously varies with the physical half-life of each radionuclide. Thus, for most radioiodine isotopes, only factors influencing short-term exposure need to be considered.

Terrestrial ecosystems

There are three major categories of factors determining exposure in terrestrial ecosystems: physical and chemical, ecological and anthropogenic.

Physical and Chemical factors

The amount, types and chemical forms of contaminating radionuclides will be dependent upon the origin and route of contamination (controlled release, accidental discharges, testing or use of nuclear weapons, migration from underground sources such as mining and nuclear waste disposal). For atmospheric discharges, meteorological conditions are of prime importance in determining the pattern of dispersion (cf. Chernobyl accident) as is the type of deposition (wet or dry). Chemical and physical forms of contaminating radionuclides have the potential to greatly affect bioavailability and its change in time. For instance, radionuclides in fuel particles deposited after the Chernobyl accident were much less mobile than those on condensed particles. If environmental mobility is reduced due to the chemical form of the contaminating radionuclides, their net export from any ecosystem will also be low. The most radioecologically sensitive source will be one which emits radionuclides in a highly bioavailable form, or where bioavailability increases in the environment with time (eg Chernobyl particle dissolution releasing ^{90}Sr).

Ecological factors

There are a large number of different factors associated with different ecosystem components which can affect radionuclide mobility and subsequent exposure. They can be grouped according to the ecosystem compartment they are associated with, as shown in Table 6.1.

Table 6.1 Factors which contribute to radionuclide mobility

Vegetation-associated characteristics	Leaf Area Index (LAI) , density, species composition (e.g. coniferous vs. deciduous trees), above-ground biomass, weathering rate, growth rate, plant surface type, foliar absorption, translocation rates, plant uptake processes, root density with depth
Soil characteristics	Texture, Structure, Bulk Density, Particle Size Distribution, Organic Matter Content, Permeability, CEC, pH, exchangeable K Content, exchangeable Ca Content, soil microbial community
Animal characteristics	species, gut absorption, body distribution, biological half-lives, dietary selection

The above factors can influence environmental transfer of radionuclides via three major processes:

◊ Surface contamination: The extent of interception by vegetation surfaces is greatly dependent upon vegetation density, leaf area index and plant surface characteristics. Subsequent weathering of radioactive contamination from plant surfaces will depend not only on meteorological conditions but also on plant surface characteristics. A high interception by plants and short weathering rates will make an ecosystem radioecologically sensitive (e.g. coniferous forest)

◊ Transfer to plants: The rate of foliar absorption is affected by the chemical form of the deposited radionuclide (radionuclides in a soluble form are much more readily absorbed than in an insoluble form). Subsequent translocation will also depend on the amount intercepted and its chemical form. Radionuclide uptake by plants occurs via the soil solution. Therefore, the processes controlling radionuclide transfer between various soil components and the soil solution are critical for radionuclide bioavailability. For example, fixation of many radionuclides on non-specific cationic exchangeable soil components is weaker than on more specific sites such as clay minerals. In addition, soil solution composition is of prime importance because of the natural competition between radionuclides and their stable analogues e.g. strontium and calcium, caesium and potassium, etc. Therefore, soils with low potassium and clay mineral contents will be more radioecologically sensitive to radiocaesium than soils with high potassium and clay minerals contents. Strong fixation will enhance retention of radionuclides in upper soil layers. However, other factors will also influence soil migration such as biological activity of the soil, meteorological conditions during deposition and infiltration capacity of the soils. In general, a high soil migration of radionuclides will reduce the radioecological sensitivity of the soil since many plants have their roots located in the upper layers of the soil. However, for deeper-rooting species the reverse will apply for a certain period of time. The degree of resuspension of soil and its subsequent adhesion to plant surfaces is also a factor to consider, especially for relatively immobile radionuclides such as actinides.

◊ transfer to animals: Transfer to animals varies with species (e.g. smaller ruminants are more radioecologically sensitive than larger ruminants). Rates of gut absorption vary between radionuclides from <1% for plutonium to 100% for radioiodine. Subsequent tissue distribution varies and radioecological sensitivity depends on the target organs. In terms of exposure of the animal itself, the accumulation of radioiodine on the thyroid is of primary concern. For food products, the transfer of radioiodine, radiostrontium and radiocaesium to milk (and of the latter to muscle) is also important.

Most of the above processes are relevant to any time after contamination has occurred and for most radionuclides. However, a notable exception is that short-term exposure is particularly dependent on interception and weathering processes. The relative importance of soil type increases with time in particular with regard to change in bioavailability.

Some of the above factors are time-dependent and vary greatly with seasons. Many biota exhibit changes in growth rates with season. Deciduous plants only have a significant above-ground biomass from spring to autumn. Dietary selection by animals can also be highly seasonally variable.

Many of the factors listed above vary spatially; of these, the most important include climate, vegetation species, vegetation growth rate, soil types, food production rates and dietary habits. Therefore, modelling of radioecological sensitivity must incorporate spatial variation in key parameters.

A radioecologically sensitive region could be one where the radionuclide is mobile and readily transferred from soil to foodstuffs or biota.

Freshwater ecosystems

Rivers

The activity concentration of radionuclides in rivers resulting from a surface deposition of activity to the catchment is described by a time-dependent runoff coefficient, $R_c(t)$ (m^{-1}), defined as:

$$\frac{Activity\ concentration\ in\ runoff\ water\ Bq\,m^{-3}}{Activity\ deposited\ to\ catchment\ Bq\,m^{-2}} \tag{1}$$

An exponential "transfer function" model is used for the runoff of activity from contaminated catchments (Monte 1995; Smith *et al.*, 1997; Smith *et al.,* 2001). This model assumes three components to the transfer: a short term (timescale, $\tau \sim 30$ d) transfer of recently deposited activity, a medium term exponential decline ($\tau \sim$ years) as a result of changing availability of the radionuclide in catchment soils and a long-term (τ ~ decades) slow decline. The runoff coefficient may be estimated by

$$R_c(t) = \alpha.e^{-(\lambda+k_1)t} + \beta.e^{-(\lambda+k_2)t} + \gamma.e^{-(\lambda+k_3)t} \quad (2)$$

where λ is the decay constant of the radionuclide and α, β, γ (m^{-1}) and k_1, k_2, k_3 (y^{-1}) are empirically determined constants. Estimates of the initial rate of decline in activity concentrations in rivers, k_1, (Monte 1995; Smith *et al.,* 2000) are shown in Table 6.2. The estimates correspond to effective half-lives of around 2 weeks for ^{90}Sr and ^{137}Cs and 6 days for ^{131}I. The values of the decay constants, k_2, k_3 are estimated as 0.41 y^{-1} and 0.02 y^{-1} respectively (Smith *et al.*, 1999, 2001) for ^{137}Cs. Sr-90 appears not to have a very long term component, but declines over the medium to long term with $k_2 \approx 0.1\ y^{-1}$.

Table 6.2. Initial rates of reduction in activity concentrations in different rivers after the Chernobyl accident.

River	**k_1 (y^{-1})***		
	I-131	**Sr-90**	**Cs-137**
Pripyat	18	24	23
Dnieper	-	16	28
Po	9	-	-

* Rate constants are calculated after accounting for physical decay. Effective half lives include physical decay.

Estimates of the initial activity concentration in rivers ($\alpha + \beta + \gamma \approx \alpha$) can be made by assuming dilution of activity directly deposited on the river surface to give:

$$\alpha \approx \frac{1}{\delta} \quad (3)$$

where δ is river mean depth. However, this is expected to be only a rough guide, since deposition times can be significant compared to river water transit times, and catchment runoff makes a contribution to river activity concentrations.

Empirical studies after Chernobyl give $\alpha \approx 0.2 - 0.4\ m^{-1}$ for ^{90}Sr (from Helton *et al.,* 1985, Monte 1996); $\alpha \approx 0.3\ m^{-1}$ (from data in Helton *et al.,* 1985) for ^{137}Cs and $\alpha \approx 1.0\ m^{-1}$ (estimated from data in Jackson & Jones (1990)) for ^{131}I. There is a lack of good empirical data on estimates of α, hence models for initial runoff have high uncertainty. Long-term runoff coefficients (β, γ) are much better quantified, and have been shown to be predictable, for ^{137}Cs, using soil characteristics (Smith *et al.,* 2000). Organic, boggy catchments have much higher ^{137}Cs runoff coefficients than catchments with high coverage of mineral soils (Hilton *et al.*, 1993; Kudelsky *et al.*, 1996).

Lakes and reservoirs- The initial activity concentration in water of a lake or reservoir, $C_L(0)$ ($Bq\ m^{-3}$) may be determined by the average areal deposition of radionuclide to the lake or reservoir surface as follows:

$$C_L(0) = \frac{D}{\delta} \quad Bq\,m^{-3} \quad (4)$$

where D is aerial deposition and δ is lake mean depth. This gives a reasonably accurate estimate of total initial mean concentration in the lake water. If the lake is stratified, δ can be taken as the mean depth of the epilimnion. More complex models for in-lake mixing require site specific studies. Simple physical characteristics of lakes – for example, mean depth, water residence time, can be estimated from map data. Removal of the initial deposit of radioactivity from the lake water may be estimated using the water residence time of the lake, and estimates of transfer rates of different radionuclides to the sediment. The distribution of the radionuclide between solid and liquid phases can be calculated using the K_d for the particular radionuclide.

Long term activity concentrations in lakes with relatively short water residence times are primarily controlled by inputs of radioactivity from the surrounding catchment. Estimates for these lakes may therefore be made using the runoff coefficient approach described above. "Closed" lake systems are defined as those lakes where there is relatively low turnover of water, often because the catchment is flat and has little input of surface water to the lake. Long term activity concentrations in closed lakes are controlled by transfers of radioactivity to and from bottom sediment deposits. Such systems have been studied extensively in the AQUASCOPE project (Smith *et al*., 2000). Initial results show that the long term activity concentration of ^{137}Cs in these lakes is typically one order of magnitude higher than in open lake systems, being estimated using a (time dependent) runoff coefficient approach (i.e. from the ratio of activity concentration observed in the lake water per unit of fallout to the lake).

Freshwater fish

All other things being equal, the radiocaesium activity concentration in fish is inversely proportional to the potassium concentration of the surrounding water (e.g. Fleishman 1973; Blaylock 1982; *et al*., 2000b). The fish-water concentration factor (CF, activity concentration in fish/activity concentration in water) for ^{137}Cs may be estimated using the empirically determined equation given by Rowan and Rasmussen (1994):

$$\log[\mathrm{CF}] = 3.320 - 0.718\log[\mathrm{K}^+] + 0.292[\text{trophic level}] - 0.233\log[s] \qquad (5)$$

where $[\mathrm{K}^+]$ is the potassium concentration in mg l^{-1}, s is the suspended sediment concentration in mg l^{-1} and [trophic level] takes a value of 0 for non-predatory fish and 1 for predatory fish.

Similarly, a relationship has been determined between ^{90}Sr activity concentrations in fish and water calcium concentration (IAEA, 1994):

$$CF = \exp(5.18 - 1.21.\ln(Ca_w)) \qquad (6)$$

where Ca_w (mg l^{-1}) is the concentration of Ca in the water.

Tag values may be estimated for fish using the concentration factor and the estimated activity concentration in water. These will obviously be time dependent and must be estimated using models such as those outlined above. In addition, for larger, predatory fish, there may be a significant delay in the peak activity concentration observed in the fish, due to relatively slow uptake rates. Thus, after Chernobyl, maximum ^{137}Cs activity concentrations in pike perch and brown trout were observed approximately one year after the accident (Elliot *et al.,* 1992).

Marine ecosystems

Since about 70% of the world's surface consists of water, the oceans received a substantial proportion of global fallout. Furthermore, the sea is the ultimate recipient of run-off from catchments. In coastal waters, contamination from nuclear energy production may predominate over global fallout. This has been the case in the Irish and North Seas, which are contaminated with radionuclides released from the reprocessing plants and, most recently, from the Chernobyl accident.

Marine ecosystems are relatively less radioecologically sensitive (ie. more resilient) compared to freshwater and terrestrial environments with respect to atmospheric radionuclide deposition. Such insensitivity is a result of the capacity of marine ecosystems to quickly dilute an input of radioactive pollutant through processes such as advective currents and waves, coupled with the large volumes involved. Thus, short term consequences are likely to be more important, in marine ecosystems as dilution will occur over the long term.

Radioecological sensitivity in marine ecosystem with respect to doses will be affected by a number of key factors which are considered below.

(i) Dispersion of radionuclides in the marine environment

Radioactive contaminants released to the marine environment are transported and dispersed by advective and turbulent processes. Dispersion of radionuclides in oceans can be considered with regard to both water and sediment layers and can be described by a range of processes including advection of the soluble fraction of the radioactivity within the water mass, the adsorbed fraction of radioactivity with suspended particles in the water column, interactions between water and bottom sediment phases through sedimentation and resuspension processes, diffusion of radioactivity through pore water, bioturbation and burial in bottom sediment and radioactive decay.

(ii) Residence times of radionuclides in the water column

Fish and other marine foods are mainly produced in coastal seas, some of which are rather closed systems, and the residence time of the water in such systems is relatively long. Other coastal waters have a more direct connection to the open ocean, and therefore the mean residence time is shorter. High residence times of radionuclides in the first 50 m depth of the water column may partially be due to its uptake in living biota in this layer.

(iii) Sedimentation rates

Sedimentary particles and suspended matter may remove radionuclides from seawater and deposited them onto the seabed. Certain radionuclides, such as ^{137}Cs, are more strongly sorbed onto sediment particles than others. Where sedimentation is rapid, radionuclide accumulation on the seabed is also rapid and radioactive particles may subsequently be covered by later sedimentary deposits before significant diffusion can occur. However, other transport processes may displace radionuclides such as burrowing of benthic organisms, marine currents near the bottom, sediment translocation, floods, inputs from land and ice movements. In the absence of significant disruption from these factors, it may be possible to identify single years of deposition in the sediment layers and thus estimate the sedimentation rate. Variation in sedimentation rates of global fallout radionuclides in coastal areas has been studied by several countries. This has identified some areas where accumulation of radionuclides is high and therefore where radionuclides may be available for long periods to biota if they are bioavailable when absorbed to sediments. This variability, and the characteristics of the sedimentation basin, will vary spatially.

(iv) Concentration factors of marine biota

Concentration factors, which are used to quantify radionuclides transfer from sea water/sediment phases to the biota are important parameters which can have a considerable effect on the extent of contamination of marine foodstuffs.

Phytoplankton concentrate several radionuclides from water. Well known examples of high transfer include the accumulation of ^{106}Ru from the Windscale reprocessing plant by porphyra and the high transfer of ^{99}Tc to brown seaweeds.

Few systematically collected measurements have been made of the activity concentration of radionuclides in phytoplankton in coastal areas. Higher plants and algae should be also considered as recent, novel uses of such marine products have been discovered such as components of creams, dietary supplements, direct consumption by vegetarians and bread.

Marine fish obtain accumulate radionuclides via two main pathways: through absorption from the surrounding water by their gills and from the ingestion of food. The latter is the most important. In shallow waters, direct contamination from sediments may be a third exposure route. Molluscs obtain radionuclides via filtration of seawater and suspended particles. In France, Italy, Belgium and Spain there are estuarine areas dedicated exclusively to farming these species, which would automatically designate these areas as being of high radioecological sensitivity. Crustaceans concentrate radionuclides from marine-plants, small sea animals, particles in sea-water and carrion. The higher levels of marine food chains are occupied by birds and marine mammals, but there are few data on which to assess factors affecting transfer to these species.

The adsorption of radionuclides to inorganic suspended particles depends on their chemical form and morphology. The temporal variation in the proportion of a contaminant radionuclide fixed to inorganic suspended particles is difficult to assess, as there is no historical record of its variability. Several recent studies have confirmed the preferential fixation of certain radionuclides to inorganic suspended particles (Carvalho 1997). In areas where this occurs, radionuclides could be scavenged from the water column, making these areas less radioecologically sensitive.

(v) Velocity of interchange within estuarine areas.

In contrast to freshwater systems, where the composition of the water shows great variation in chemical composition, marine waters generally have a similar composition of minerals. In waters close to river outlets, namely estuaries, the salinity of the water is lower than in the open ocean. This influences K_d values and thus the sedimentation of radionuclides, which is generally higher in estuaries than in the open sea. Similarly, concentration factors between fish and water are higher in brackish waters, and this counterbalances, to some extent, the lower radionuclide activity concentrations in low salinity waters compared to waters with normal salinity (UNSCEAR 2000).

Deposition of contaminated sediments in estuaries can lead to high exposure. An example is the Esk estuary in Cumbria, where highly contaminated sediments have been deposited in the estuary rivers and on tide-washed pastures. Transfer to biota in these pastures is, however, low due to the strong absorption of radionuclides to the sediments and consequent low bioavailability (Howard *et al.*, 1996).

(vi) Location and harvesting rate

Sensitive marine ecosystems will include those into which liquid discharges are released (eg Irish sea affected by Sellafield Reprocessing Plant activities, coastal areas close to the influence zones of La Hague and Marcoule, emissions of natural radioactivity of non-nuclear industries, areas affected by accidents).

In addition, estuarine areas are used extensively for farming molluscs (e.g. Galicia produces 90% of Spanish molluscs and production is high in some French Atlantic coasts), fish (e.g. Nordic estuarine areas) and crustaceans and also marine zones of high biological productivity such as the Barents sea may be considered as radioecologically sensitive.

Overall Conclusion for Transfer to environmental compartments

In general, terrestrial and freshwater ecosystems are much more radioecologically sensitive to atmospheric radiocaesium contamination than marine systems. Because of the wealth of experimental and observation data now available (particularly since the Chernobyl accident), it has been possible to identify and quantify many of the factors contributing to this enhanced radioecological sensitivity for radiocaesium. Because many of these factors vary spatially, it is important to quantify radioecological sensitivity incorporating spatial variations. Such models are now becoming available but are currently restricted to only few radionuclides (e.g. SAVE-IT which only considers radiocaesium).

6.2.2 Action loads

In section 4.1.3 the use of critical loads was discussed for the medium-long term after deposition. The potential application of the critical loads approach has been considered in the forum. It is a potentially useful approach for identifying radioecologically sensitive areas for emergency planning and has the advantage of simplicity and ability to be presented in a spatial format. Assessment of critical loads in the Arctic has emphasised that ecological half-lives are an important aspect of critical loads, since previously deposited fallout may have a significant long term contamination effect for food products or environmental compartments with long ecological half-lives (Howard *et al.*, 2002).

Previously, the use of critical loads was developed only for the mid-long term phase. Within the forum, it was decided that a similar approach would be useful for the acute phase after an accident. For this application, critical loads were renamed as action loads. In such circumstances, short term "action loads" can be defined in the same way as critical loads, but would depend on processes which are most important in the initial stages of an accident. Such action loads were felt to be particularly useful since maps or tables of action loads for different food products can be combined with maps of deposition following any future nuclear accidents for the rapid identification of areas that are either sensitive or resilient to radiocaesium deposition, and targeting use of resources.

The relevant maximum permitted levels for foodstuffs are given in Table 6.3. It should be emphasized that these values are not in force now, they will enter into force if an accident occurs after a decision by the Commission. They will be evaluated and possibly changed within 3 months.

Table 6.3 Maximum permitted levels for foodstuffs

Radionuclides	**Maximum Permissible activity levels in foodstuffs (Bq kg^{-1})**			
	Baby Food	**Dairy Products**	**Other Products**	**Liquid food**
Caesium	400	1000	1250	1000
Iodine	150	500	2000	500
Strontium	75	125	750	125
Plutonium	1	20	80	20

Terrestrial action loads

The action load is defined as the amount of radioactivity (Bq m^{-2}) which needs to be deposited to produce activity concentrations in a food product which equals the maximum permitted level for that product. Action loads for terrestrial systems were quantified in the forum using three different models: ECOSYS which has been developed by GSF for Southern Germany conditions, AGROLAND which has been developed by SSI, Sweden and SAVE-IT, developed under the SAVE EC project.

The scenario originally considered for estimation of action loads are given below:

1. Radionuclides: ^{137}Cs, ^{90}Sr, ^{131}I, Pu.
2. Target foodstuffs: cow milk
3. Time of deposition: May 1st, August 1st, October 1st
4. Fallout deposited as (i) dry and (ii) wet deposition
5. Three soil types: clay, sand and peat

The requested outputs were:

- Time variation in action load over (i) 0-2 months for ^{131}I and 0-6 months for ^{137}Cs, ^{90}Sr and Pu; and (ii) 5 years after deposition
- Agricultural management: ruminants grazing outdoors on (i) improved pasture and (ii) unimproved extensive upland grazing conditions

Strictly, the action load should only refer to the initial period after deposition, and changes in the load needed to put product contamination over intervention levels at later periods would then be referred to as critical loads. However, for simplicity, we have referred to the action load throughout when discussing changes with time. A description of the three models used giving information on their format and parameter values are given below with their respective outputs.

Description of model output

ECOSYS

ECOSYS calculates the deposition of radionuclides from air concentration and accounts for dry deposition and wet deposition taking into account precipitation rates which influences the intercepted fractions. Dry deposition needs an input of time integrated activity concentration in air. Dry deposition velocities are dependent on the plant development stage and are derived from the biomass. The Leaf Area Index (LAI) is used as a measure to calculate the biomass. The surface characteristics of various plant species are also taken into account in the derivation of dry deposition velocities. The chemical form of the radionuclide are also considered; e.g. for iodine this has a major effect.

For wet deposition, the model needs an input of deposited activity per unit area. The intercepted fraction on vegetation is dependent on the amount of rain, the fraction retention due to water retention capacity, the biomass (using the LAI), and the radionuclide.

Radionuclide availability for plants is derived from an assumed migration rate (40 years for Cs) and a fixation half-life (9 years for Cs) which are compounded to give an overall half-life (5.9 years for Cs). The soil in pasture is assumed to be a 10 cm layer with the radionuclides being homogeneously distributed.

For the scenarios considered in this context, the relevant detailed parameter assumptions are given in Appendix B.

For the comparison of results with those of AGROLAND and SAVE-IT, ECOSYS was run for a 3 mm rainfall event and for deposition on grass. If instead of this an amount of rainfall of 1 mm or 10 mm was assumed, the resulting action loads would be lower by a factor of about 2, or higher by a factor of 3 respectively, due to different intercepted fractions. This holds for all of the following scenarios.

ECOSYS Model predictions

Initial comparisons of the predicted action loads show that action loads for ^{90}Sr are always lower compared to the other radionuclides. A contributory factor to this is the low intervention limit (125 Bq kg^{-1}) for ^{90}Sr in cow milk.

For the deposition there is a general trend in the action load as follows:

$$^{90}\mathrm{Sr} < {}^{131}\mathrm{I} < {}^{137}\mathrm{Cs} < \mathrm{Pu}.$$

There is a high predicted action load for ^{131}I for the prediction for a May accident with wet deposition; the reason is that at the beginning of May a mixed feeding regime consisting of fresh grass and hay is assumed (see Appendix B). Otherwise there is only a small seasonal variation in predicted action loads for dry deposition due to small biomass differences used in ECOSYS.

The effect of interception on differing biomass may be being compensated for by growth dilution.

For Pu, higher action load values are predicted for deposition in October than for deposition in May or August because stored feed is not contaminated by a deposition in October.

Within ECOSYS, wet deposition consistently gives a higher action load than dry deposition. This is because the predicted total interception is greater for dry deposition than for wet deposition due to wash-off from leaf surfaces. There is little difference in the wet and dry deposition predicted action loads for ^{90}Sr, but the values for ^{137}Cs and ^{131}I vary by factors of 2 and 3-4 respectively. This is due to differences in the retention function used for Sr, Cs and I to model the interception of radionuclides upon plant surfaces under both wet and dry deposition. Differences in predicted action loads for iodine are also due to the chemical form of iodine assumed in the model.

Action loads as a function of time

Using ECOSYS, for most radionuclides the action load has been plotted for 6 months after deposition, except for ^{131}I for which 1 month duration is shown due to the short physical half-life. All plots are for feeding with grass from intensive cultivation. ECOSYS was run for four different radionuclides, dry and wet deposition and three different deposition dates. Because the output is extensive, most plots are provided in Appendix C. An example estimated action load variation with time (5y) for dry deposition of ^{137}Cs on 1.5.00 is given in Figure 6.2.

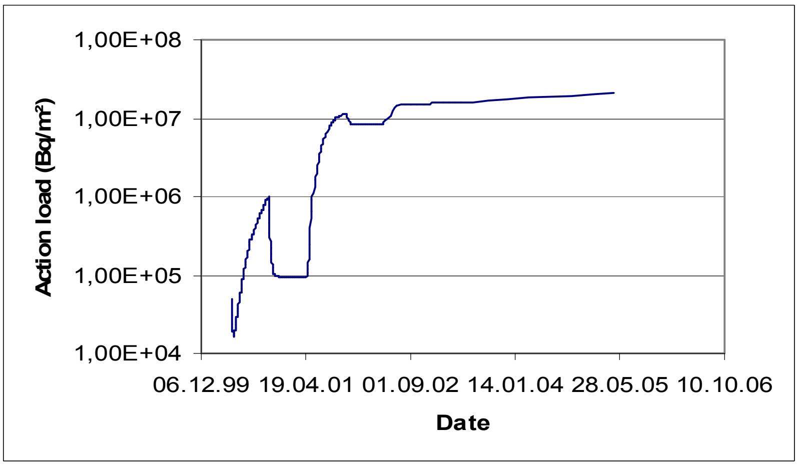


Figure 6.2 Estimated action load variation with time for ^{137}Cs assuming dry deposition on 1.5.00

AGROLAND

The action load predictions using AGROLAND have been derived for the southernmost region of Sweden where dairy farming is more common.

Predictions were carried out for ^{137}Cs, ^{90}Sr and ^{131}I but not Pu. The model reflects the short growing season seen in Sweden and hence model predictions haven't been undertaken for a scenario deposition date of 1st October.

The model has been derived from experimental data for one specific site in southern Sweden following the Chernobyl accident. The model is parameterised to experimental data measured for a single area. Differences in transfer to 3 different soil types (clay, sand and peat) from the IAEA handbook No 364 are used to model action loads for three different soil types.

The start point of the model is deposition (Bq m^{-2}) of the radionuclide, which is partitioned to the soil and plant surfaces using a relationship between biomass and interception onto plant surfaces. Deposition is assumed to occur as wet deposition. For AGROLAND, the biomass value is used directly in the equation to derive the interception. The equation has been derived from experimental data. The equation outcome is highly site specific.

Losses of radioactivity from the contaminated plant surfaces are modelled using a weathering half-life which is not constant and is a function of time – it gets progressively longer as time after deposition increases. Using a continually varying half-life for the weathering of radioactivity from plant surfaces provides a better fit to the experimental data than using a single weathering half-life.

Foliar absorption of radioactivity remaining upon plant surfaces is assumed to be instantaneous and complete. Plant biomass is estimated using a logistic growth curve (with a minimum value of 0.02 in May, a maximum value of 0.55 in August and an exponential decline with a half-life of x). In the short-term, for calculation of action load, growth dilution is not included, but for the longer-term dilution is controlled by the estimation of biomass. AGROLAND will, due to having a higher estimated interception fraction, predict more interception than ECOSYS which is compensated for by a higher biomass, giving similar activity concentrations (Bq kg^{-1}).

Transfer of radionuclides from soil to plant is modelled using a concentration ratio. Therefore, the model estimates the activity concentration of radioactivity in the soil by estimating the migration of contamination down the soil profile (relating this to porosity, permeability, k_d, and bulk density, which are variable for the three soil types considered). The depth of radionuclide migration is then used (90% of total activity), along with soil bulk density (for each of three soil types), to calculate the activity concentration in soil. The IAEA handbook 364 CR for clay, sand and peat soil types is based upon the assumption of a homogeneously contaminated soil layer of 10 cm (for pasture). These CR values are modified using the above approach to account for the change of the distribution of contamination within the upper soil layer.

Cow dry matter intake rates are derived from recommended values for the area of southern Sweden (Eriksson, pers. comm.). Transfer of radioactivity to cow milk is modelled dynamically using a biological half-life (derived from literature values) and the transfer coefficient for the area of southern Sweden. For longer-term predictions, an equilibrium transfer is used to model transfer to milk.

The model does not consider resuspension, inhalation or soil ingestion.

Time dependence of fixation is not considered in the model, although some influence of time upon soil transfer will be included from the point of view of migration of contamination down the soil profile. Hence, long-term estimates from the model are conservative, as the model doesn't explicitly incorporate changes in fixation with time.

AGROLAND Model predictions

Soil type effects

The variation in estimated action loads for ^{137}Cs and ^{90}Sr deposited in May are shown in Figure 6.3. Action loads increase with soil depth for all cases except for ^{90}Sr in sandy soil, where the action load is lower in the 10 cm depth, presumably due to migration of ^{90}Sr out of the rooting zone in this soil type.

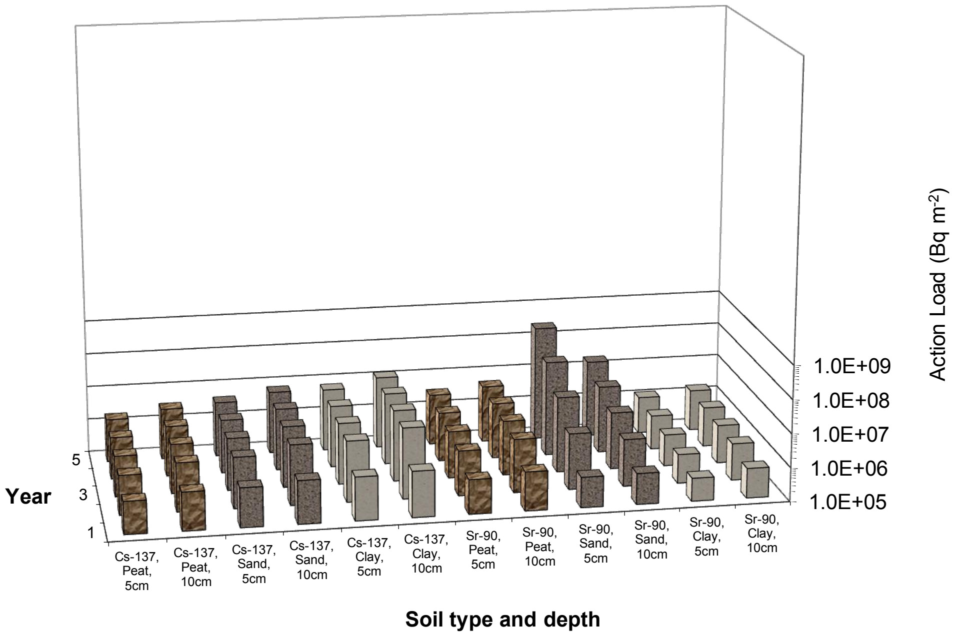


Figure 6.3 Estimated action loads for three soils types using two different soil depths over a five year period

For AGROLAND, action loads for ^{137}Cs for the three soil types modelled are, as expected:

Peat < Sand < Clay.

However, for ^{90}Sr, the predicted action loads do not reflect the soil-to-plant transfer factors used which are in the order:

Peat < Clay < Sand.

AGROLAND has soil type specific K_d values, which are used in modelling the migration of contamination down a soil profile and hence modify the *actual* soil-to-plant transfer factors used, which is probably leading to the observed pattern of predicted action loads for ^{90}Sr of:

Clay < Peat < Sand.

Biomass effects

The effect of different plant biomass is shown in Figure 6.4 for ^{131}I at 14 d after deposition in May and for ^{137}Cs at 14 d and 180 d after deposition in May. As biomass increases, the action load also increases due to growth dilution.

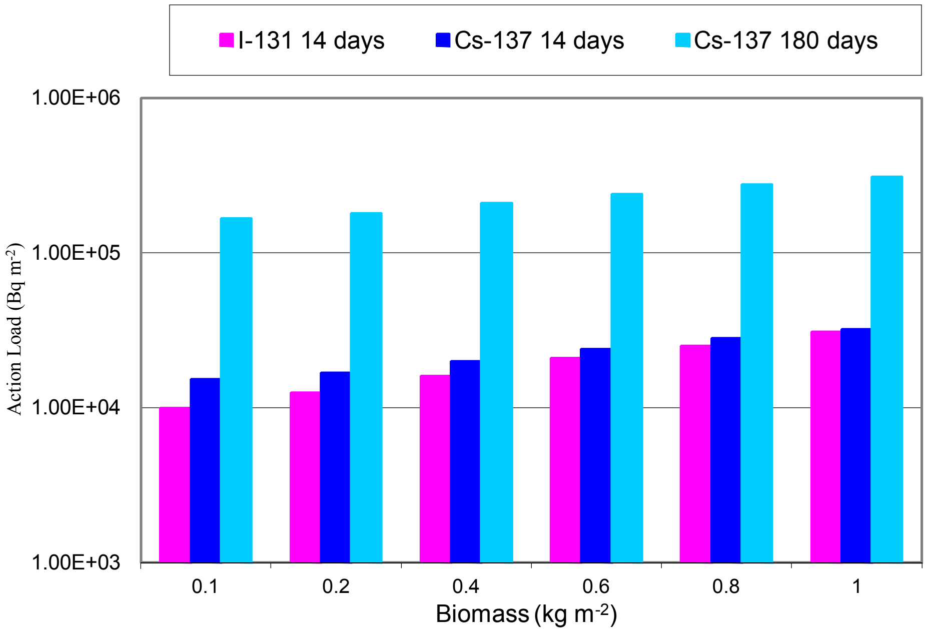
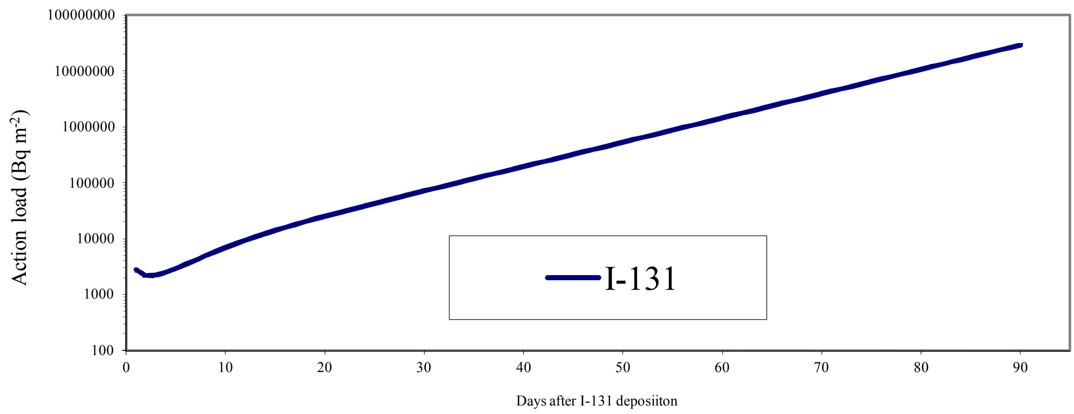


Figure 6.4 Changes in action loads with increasing assumed plant biomass

Changes with time

Changes over the short term in action loads for the three radionuclides are shown in Figure 6.5. Changes with time in the action load for ^{131}I are faster because of the short physical half-life. The action loads increase in the order:

$$^{131}I < {}^{90}Sr < {}^{137}Cs$$



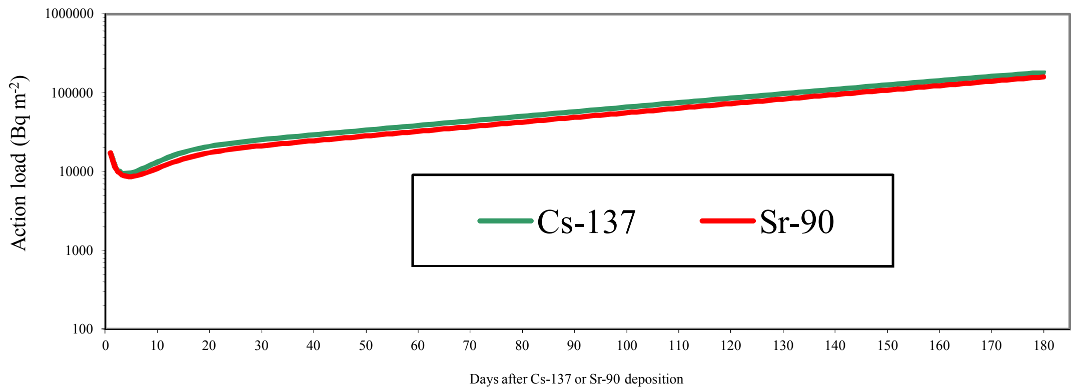


Figure 6.5 Changes with time in the action loads for ^{131}I, ^{137}Cs and ^{90}Sr.

SAVE-IT

The SAVE-IT system can be used to predict the impacts of ^{137}Cs deposition across western Europe. Its underlying structure is based upon the ECOSYS model, but at its core, a semi-mechanistic ^{137}Cs soil-to-plant transfer model with four input soil properties (% clay, exchangeable K, pH and % organic matter) is used to predict ^{137}Cs transfer across all soil types. SAVE-IT contains a spatial database for western Europe (resolution 5 × 5 km) of the variation in the four input soil properties and a regional database with information on the regional variation in agricultural management practices.

The SAVE-IT system has been used to compare predicted action loads for cow milk for the UK and France. Action loads were predicted for 0, 6 and 60 months after a uniform ^{137}Cs deposition of 10 000 Bq m^{-2} occurring on the 1 May, 1 August and 1 October.

In SAVE-IT, ^{137}Cs interception is dependent upon plant biomass at the time of deposition. The annual variation of pasture biomass for different regions is estimated using an exponential growth function from the initial biomass, yield and times of minimum and maximum biomass. For the UK and France, times of minimum and maximum biomass are assumed to be the same (1 November and 1 July respectively) with a greater pasture yield assumed for France. Cs-137 uptake to pasture grass is predicted using the semi-mechanistic ^{137}Cs soil-to-plant transfer model and transfer to animal products is estimated from dry matter and ^{137}Cs intake rates assuming equilibrium. Animal diets in SAVE-IT are modelled assuming four types of feed (pasture, stored grass, maize silage and concentrates), the proportions of which can be varied throughout the year using 6 bi-monthly intervals. To compare predicted action loads for the UK and France, dairy cow diets were assumed to consist entirely of pasture grass, with intake rates of 17 kg (dry weight) d^{-1}. A ^{137}Cs transfer coefficient to cow milk of 0.0079 d kg^{-1} was used.

SAVE-IT Model predictions

Predicted action loads for cow, sheep and goat milk for the UK and France for 0, 6 and 60 months after the three accident scenarios are summarised in Table 6.4 and shown in Figures 6.6 and 6.7 respectively.

At 0 months for each of the three deposition scenarios, predicted action loads are similar for the UK and France; this would be expected as, in the early phase, surface contamination will dominant food product contamination.

Predicted action loads are lowest for deposition occurring on the 1 October and highest for 1 August. This reflects the prediction of pasture biomass with maximum pasture biomass occurring on 1 July. For the three accident scenarios, the greatest pasture biomass occurs for an accident on 1 August, when a larger proportion of ^{137}Cs deposition will be intercepted, but pasture grass ^{137}Cs activity concentrations will be lower due to growth dilution.

With time after deposition, plant and animal product contamination will become dominated

Figure 6.6 Predicted cow milk critical load in the UK assuming deposition 1 May 2000

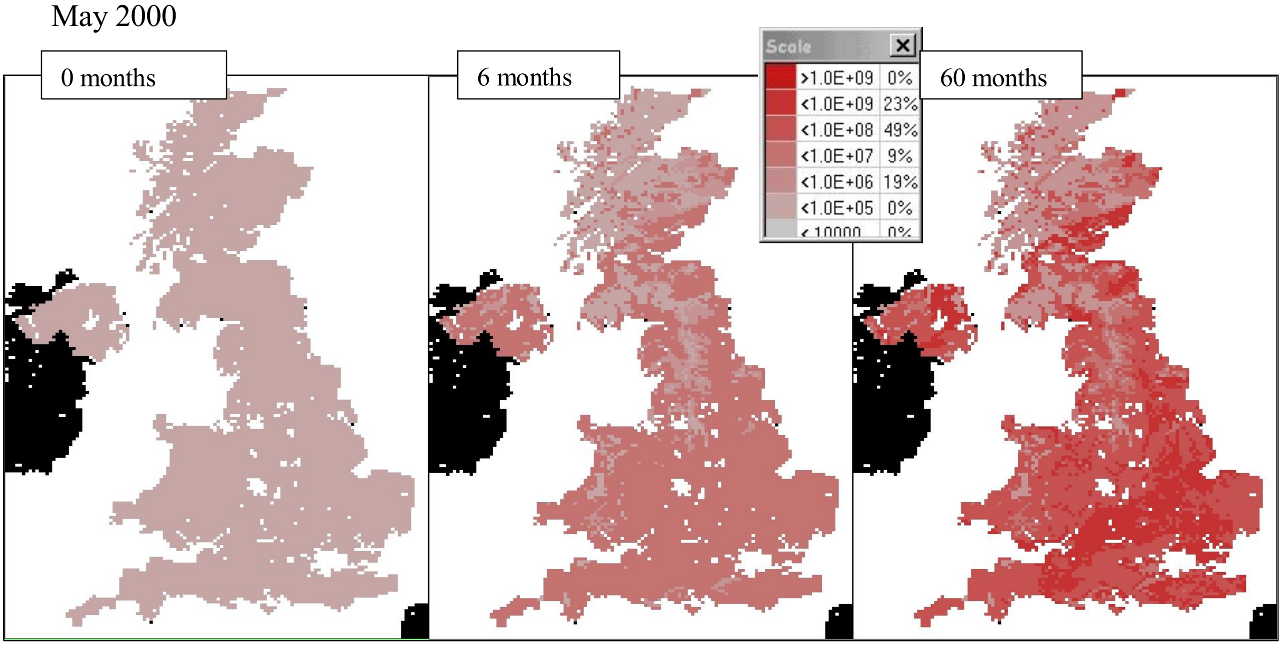
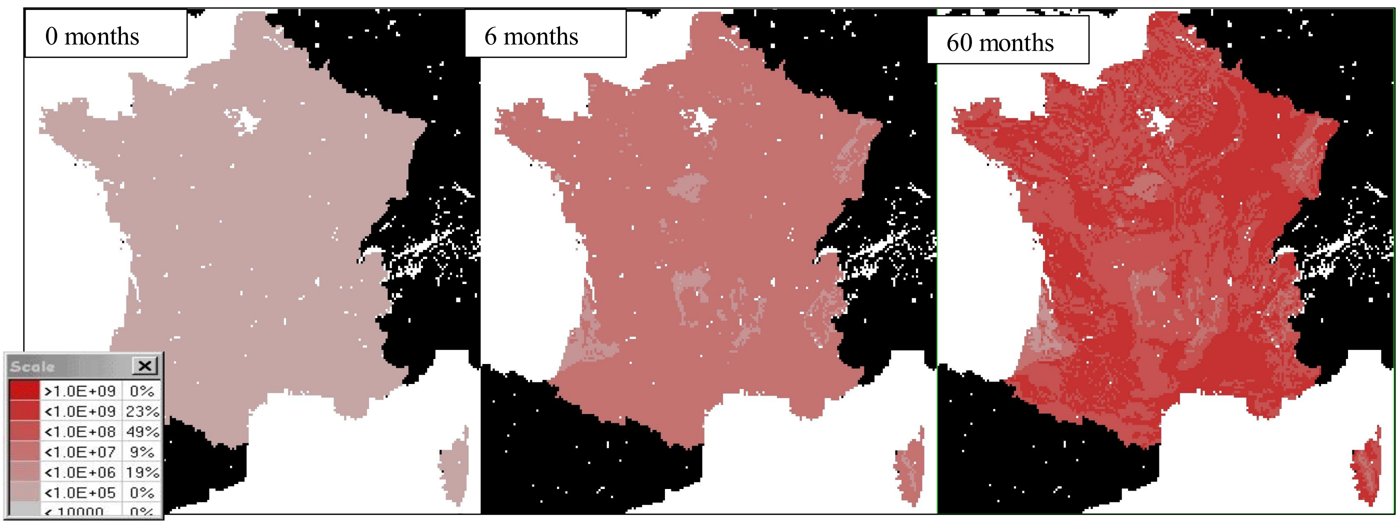


Figure 6.7 Predicted cow milk critical load in France assuming deposition 1 May 2000



by ^{137}Cs soil-to-plant transfer and predicted action loads are lower for the UK than for France. This is due to soils in the UK generally having lower % clay, exchangeable K, pH and higher % organic matter than soils in France.

Table 6.4. Action loads for cow milk in the UK and France for 0, 6 and 60 months after deposition occurring on 1 May, 1 August and 1 October predicted using SAVE-IT.

Country	Months after deposition	Predicted action load (Bq m^{-2})								
		Deposition 1 May			Deposition 1 August			Deposition 1 October		
		Min	Mean	Max	Min	Mean	Max	Min	Mean	Max
UK	0	1.3×10^{4}	1.4×10^{4}	1.4×10^{4}	1.5×10^{4}	1.6×10^{4}	1.6×10^{4}	6.2×10^{3}	7.4×10^{3}	7.7×10^{3}
	6	3.8×10^{4}	2.7×10^{6}	4.4×10^{6}	3.8×10^{4}	3.0×10^{6}	5.1×10^{6}	3.8×10^{4}	1.7×10^{6}	2.6×10^{6}
	60	2.0×10^{5}	5.9×10^{7}	2.1×10^{8}	2.0×10^{5}	5.9×10^{7}	2.1×10^{8}	2.0×10^{5}	5.9×10^{7}	2.1×10^{8}
France	0	1.3×10^{4}	1.4×10^{4}	1.4×10^{4}	1.5×10^{4}	1.7×10^{4}	1.7×10^{4}	6.2×10^{3}	7.7×10^{3}	7.7×10^{3}
	6	3.8×10^{4}	3.9×10^{6}	4.8×10^{6}	3.8×10^{4}	4.0×10^{6}	5.6×10^{6}	3.8×10^{4}	2.2×10^{6}	2.7×10^{6}
	60	2.0×10^{5}	9.7×10^{7}	4.9×10^{8}	2.0×10^{5}	9.7×10^{7}	4.9×10^{8}	2.0×10^{5}	9.7×10^{7}	4.9×10^{8}

General Comparison of the different model results

The models applied for the different countries suggest that in terms of sensitivity to ^{137}Cs, action loads increase in the order:

Southern Germany < France < UK < Sweden.

However, it should be realised that there will be substantial within country variability in the factors influencing action loads, especially in periods of rapid growth such as the spring when spatial variability in growth will occur. Furthermore, the underlying structure and assumptions in the different models used differs, therefore the resulting comparisons should be treated with caution.

Using AGROLAND (wet deposition) action loads increase as follows:

$$^{131}I < ^{90}Sr < ^{137}Cs$$

In contrast, using ECOSYS (for wet deposition):

$$^{90}Sr < ^{131}I < ^{137}Cs.$$

For ^{90}Sr, AGROLAND and ECOSYS give similar predicted action loads.

May and August predicted action loads for SAVE-IT (France and UK) are similar; differences between SAVE-IT and ECOSYS predictions could be explained by the differences in the cow milk transfer coefficient used.

For AGROLAND and ECOSYS, action loads for radionuclide deposited in May are higher than those for August which are higher than those for October. However, for SAVE-IT, action loads are lower in May than in August.

The consideration of action loads using the three models has been useful and shown some interesting differences. However, it must be realised that the models used for the action load estimation are general transfer and assessment models. For the first fourteen days after deposition, these models are not specifically designed to cope with all the relevant process involved in interception and weathering in a mechanistic way. The validity of the results is better for longer time periods when the critical load is being calculated.

Freshwater action loads

Drinking water

A conservative estimate of the action level for drinking water may be made by assuming that the activity concentration in drinking water is equal to that in the contaminated reservoir or river. We can then estimate an action level of deposition to the catchment leading to an initial concentration in water exceeding, in this example, the NRPB Generalised Derived Limit (GDL) for drinking water (NRPB 1996). Obviously, different radionuclides and different situations will require different action levels.

For rivers, using the above estimates for the initial runoff coefficient in the river, the initial activity concentration in river water is expected to exceed the GDL if the deposition to river and catchment exceeds the action levels shown in Table 6.5.

Table 6.5. Action levels for initial exceedence of the GDL for drinking water.

Radionuclide	GDL Bq l^{-1}	Action Level (Bq m^{-2})
Cs-137	100	300 kBq m^{-2}
Sr-90	50	125 kBq m^{-2}
I-131	20	20 kBq m^{-2}

Since the initial activity concentration in rivers declines relatively rapidly, critical loads for mid-long term countermeasures are expected to be much higher. Drinking water supplies from rainwater collectors and small streams may require more stringent action loads since activity concentrations per unit of deposition are likely to be significantly higher than in rivers. A conservative method of estimating the action load for streams/rainwater collectors could be based on exceedence of the drinking water GDL in rainwater which may be predicted using estimates of activity concentration in air and empirical washout factors.

For lakes and reservoirs, the action load may be estimated using the mean depth (or, if appropriate, the mean epilimnion depth). Using Equation 4, and assuming a maximum permitted level of 100 Bq l^{-1}, the action load, A_L (Bq m^{-2}) may be estimated using:

$$A_L = 10^5 . \delta \qquad \text{Bq m}^{-2} \qquad (7)$$

where δ is the lake, or epilimnion, mean depth. For highly particle reactive radionuclides, this is expected to be a conservative estimate since sediment removal during water treatment will reduce activity concentrations significantly. For ^{137}Cs, ^{90}Sr and ^{131}I, however, this will be only a slight underestimate of the action load required.

Action loads for fish

Because activity concentrations in fish are time-dependent (depending on rates of change of radioactivity in the surrounding water, and on biological uptake rates) it is difficult at present to give accurate short-term action loads, which would need site-specific information and may not be appropriate. Long term (months-years after fallout) critical loads for ^{137}Cs, however, may be calculated using the concentration factor approach, as shown in the following example.

The concentration factor of ^{137}Cs in systems of low (1 mg l^{-1}) and high (10 mg l^{-1}) may be estimated from results given in, for example, Blaylock (1982):

$K^+ = 1$ mg l^{-1}; CF = 10^4 l kg^{-1}

$K^+ = 10$ mg l^{-1}; CF = 10^3 l kg^{-1}

Assuming a maximum permitted level of ^{137}Cs in fish of 1000 Bq kg^{-1} (wet weight), we require intervention at:

0.1 Bq l^{-1} for low K^+ waters

1.0 Bq l^{-1} for high K^+ waters

Using models developed in AQUASCOPE (Smith *et al.,* 2000), we have estimated (Table 6.6) the action load for three example systems: a lake or river with a catchment composed mainly of mineral soils; a lake or river with a catchment composed of a mix of 33% organic boggy soils and 67% mineral soils; and a "closed" lake system.

Table 6.6. Action loads for ^{137}Cs in large, predatory fish for different scenarios.

Scenario	Low [K^+]	High [K^+]
Mineral catchment	3×10^4 Bq m^{-2}	3×10^5 Bq m^{-2}
33% organic boggy catchment	1×10^4 Bq m^{-2}	1×10^5 Bq m^{-2}
"Closed" lake system	3×10^3 Bq m^{-2}	3×10^4 Bq m^{-2}

Marine critical loads

Most contamination of marine systems arises from point sources or planned direct discharges with subsequent dispersion, which make take long periods of time. Therefore, the action load approach is less useful to decision makers considering marine systems than it is for terrestrial and freshwater ecosystems and critical loads have been considered instead.

We have interpreted the use of critical loads as referring only to an atmospheric deposition onto marine ecosystems. This is the deposition (Bq m^{-2}) over the marine area considered that will lead to activity concentrations in marine food exceeding the intervention level, and has been applied here with respect to the activity concentration intervention level of 600 Bq kg^{-1} ^{137}Cs in fish. The critical load approach has been applied by the NRPA to the Atlantic waters surrounding the European Atlantic coast. Applying simple models of dilution in different boxes, and quantification of the main parameters of this ecosystem such as transfer coefficients and sedimentation rates, a critical load map was derived for 1 year after contamination (Figure 6.8).

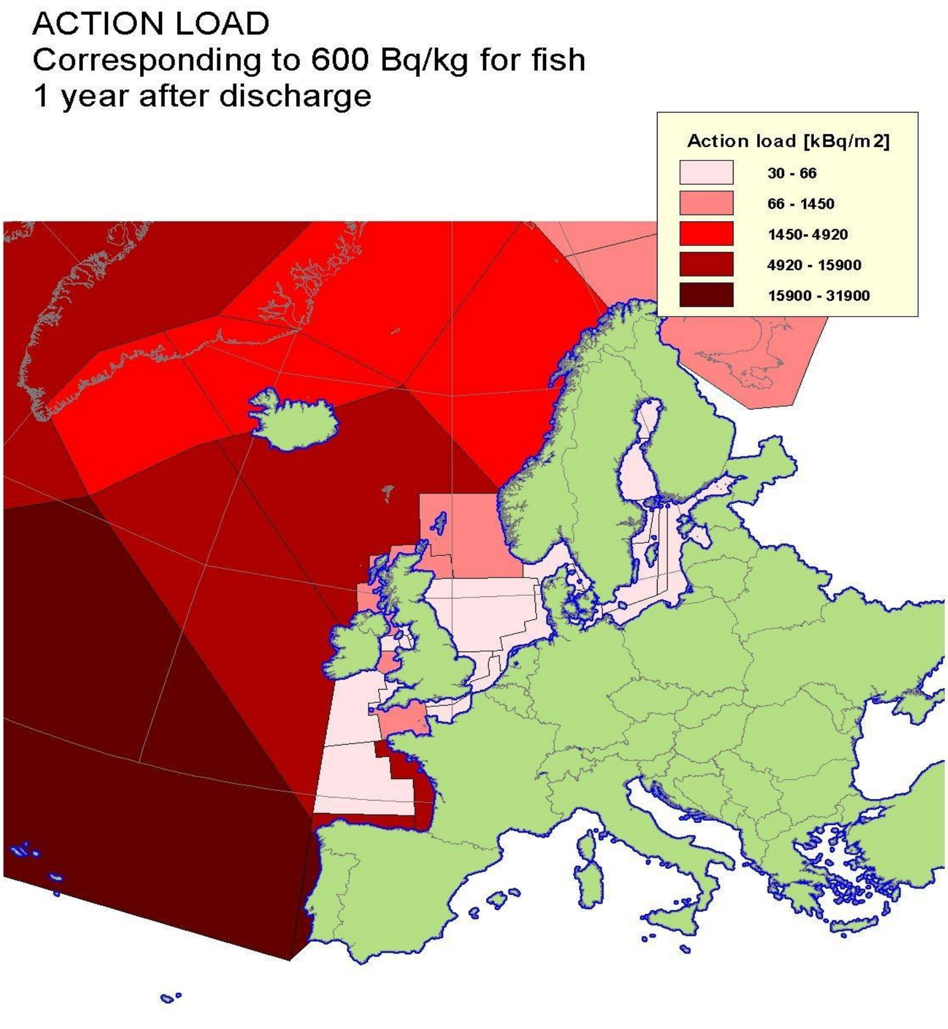


Figure 6.8 Estimated critical load for ^{137}Cs in fish 1 year after contamination

Overall Conclusion of action loads in both terrestrial and aquatic environments

The quantification of action loads provides a useful initial spatially variable reference guide concerning the amount of a specified radionuclide that would need to be deposited to give rise to concentrations which exceed intervention levels. For terrestrial ecosystems, they are determined by interception and weathering rates. Action loads will be site and season specific and are best estimated using models which describe deposition and weathering mechanistically. In freshwater ecosystems, action loads for drinking water (from lakes and reservoirs) depend on the initial runoff and the mean depth, for fish it is more difficult to estimate and probably not meaningful, but critical loads for later periods will be affected by transfer rates from water and nutrient levels in water. The concept is less easily applicable for marine systems where considerable dilution of a surface deposit occurs, and where concern is often more directed to point source continuous releases into marine ecosystems.

6.2.3 Fluxes

The flux is defined as the total amount of radioactivity produced in a specified environmental product over a given time period (e.g. Bq y^{-1}) which is transferred from one compartment to another. For collective dose estimation, agricultural production statistics need to be incorporated so that fluxes of radionuclides can be quantified. To improve the quantification of the collective doses, it is important to assess the spatial dimension of the key parameters defining radioecological sensitivity. Identification of high fluxes requires a consideration of both variation in environmental transfer pathways (discussed above) and of rates of production and harvesting of food. For agricultural products, national statistics for many countries are compiled into international sources. Such data can therefore be used relatively easily in assessments of flux. However, some of the categories of food products consumed are not consistent with radiological measurements, for example, dairy products are sub-divided into different categories which would have different radionuclide activity concentrations. A greater problem is connected with the estimation of the extent of harvesting of wild or free food products from extensive ecosystems such as forests and upland areas, which are rarely quantified.

If there are areas producing food with a low level of contamination, but with a high production, these areas may contribute significantly to the collective dose. It is valuable to identify these areas because countermeasures can also be applied with the objective of reducing collective as well as individual dose. This will be dependent on the most cost effective action from scenario to scenario.

Terrestrial fluxes

Key products from terrestrial ecosystems, especially in the short term after an accident and for routine releases, include milk (and dairy products), leafy green vegetables, meat and cereals. Thus, the intensity of milk production per unit area can be considered a sensitive criteria for estimation of collective dose. For certain countries, the production of milk from sheep or goats is important due to both production rates (Figure 6.9) and the high observed transfer of radionuclides compared with cattle.

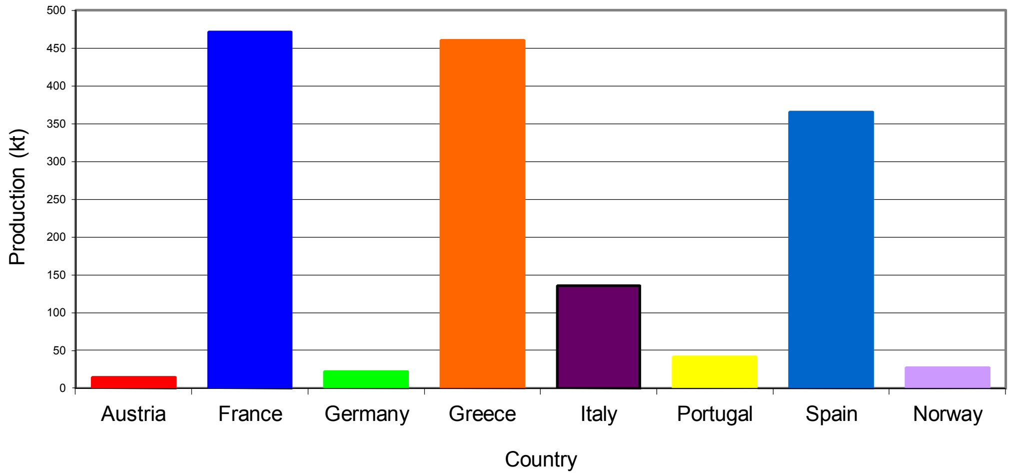


Figure 6.9 Annual production of goat milk in different European countries (FAO statistics, 1994)

The change with time in production of food products can be significant and is highly affected by food safety considerations. For example, production and export of ruminant products in the UK have been severely disrupted by BSE and foot and mouth. Substantial yearly variation in fluxes can also occur in extensive systems, a notable example being the highly variable rate of production of fungal fruiting bodies (mushrooms).

Freshwater fluxes

Fluxes of radioactivity from freshwater ecosystems could be defined in terms of numbers of people supplied by drinking water from a reservoir or river abstraction. Fluxes from consumption of fish are more difficult to quantify, since consumption of freshwater fish varies significantly and harvesting of freshwater fish is poorly quantified. In most parts of western Europe, consumption of freshwater fish is relatively low, but in Scandinavia and the CIS consumption rates may be high. In their post-Chernobyl assessment of radioactivity in freshwater systems in Cumbria (Camplin *et al.,* 1989), the UK Ministry of Agriculture used a critical group fish consumption rate of 36.5 kg y^{-1}, based on a survey of fishermen on Lake Trawsfynydd. The National Radiation Protection Board recommend a value of 20 kg y^{-1} (Robinson 1996) for critical group consumption in the UK. In Scandinavia and the CIS, consumption rates may be significantly higher than this.

Although estimates of productivity (kg of fish produced per unit surface area per year) of different lakes and rivers are available, the uncertainty in these estimates is high. For example, fish production of lakes in Belarus is on average 16 kg ha^{-1} (Ryabov[4]). Fish farms are not likely to be an important source of transfers of radioactivity to Man since activity concentrations in farmed fish (if they are fed on uncontaminated food) are typically very low.

Marine fluxes

Fish and shellfish provide about one-sixth of the animal protein consumed by people worldwide. A billion people, mostly in developing countries, depend on fish for their primary source of protein. Global marine production has increased six-fold since 1950, but the rate of increase annually for fish caught in the wild has declined from 6 % in the 1950s and 1960s to 0.6 % in 1995-1996. The catch of low-value species has risen as the harvest from higher – value species has plateaued or declined, masking some effects of overfishing. Some of the

[4] Severtsov Institute, pers. Comm..

recent increase in the marine fish harvest comes from aquaculture which has more than doubled in production since 1990.

Flux in a marine context can be considered to be the annual amount of deposited radionuclide which has been transferred to fish through marine foodchains and then distributed in the market as contaminated fish. Thus, flux [Bq y^{-1}] is calculated as the multiple of the yearly production of fish (kg y^{-1}) by the activity concentration (Bq kg^{-1}) produced in a specified area which contributes to a collective dose.

The production of marine foodstuffs is well documented by FAO and other organisation (and can be consulted online for many countries), but this general picture usually conflicts with information obtained from local authorities. The production may be well known, but not where this production is consumed and in what quantity. Furthermore, the total production received by local markets generally does not agree with that estimated from known consumption by the population. The differences between production and market available and consumption is probably due to the use of fish in industrial products (fish oil etc.).

Example of estimation of flux radioecological sensitivity for marine ecosystems

An example calculation is given of analysis of radiological sensitivity of marine areas using a compartment model (Iosjpe *et al.*, 2002) for radiological assessment of the Arctic Ocean and Northern Seas. The calculations correspond to an assumed uniform deposition over all marine areas of 10 kBq m^{-2} ^{137}Cs and assumes a CF for fish of 100 and of 30 for crustacea and molluscs. Collective doses, based on fluxes have been estimated for a period of 300 years after deposition (Figure 6.10) and show that coastal regions and shelf seas are most sensitive for this hypothetical atmospheric release scenario. The reason for this is that the initial dilution in coastal and shelf areas is much lower due to the smaller volume of water and because of the much higher rate of seafood harvesting in these areas.

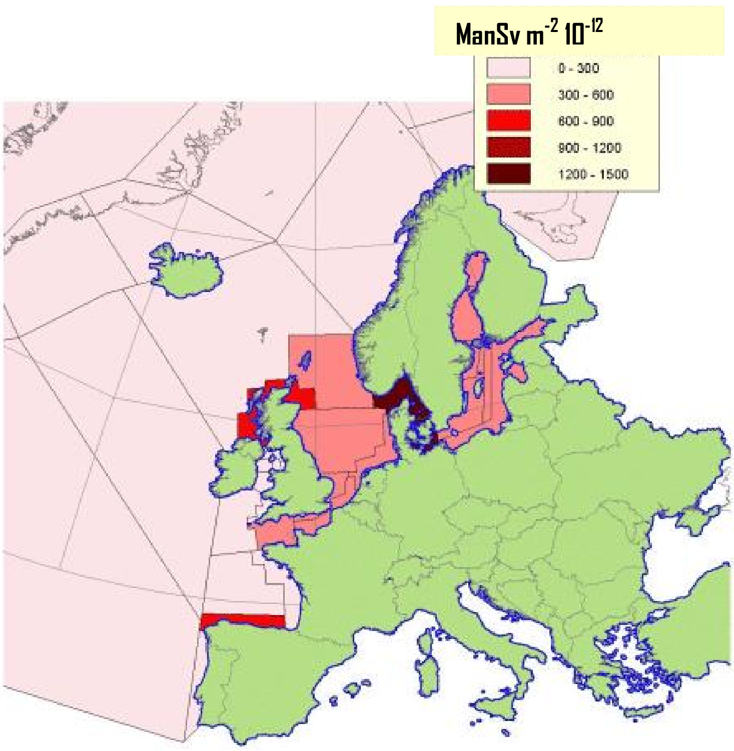


Figure 6.10 Estimated collective dose after 300 years from different compartments of the marine environment per area unit (m^2) after an assumed uniform deposition of ^{137}Cs at 10 kBq/m^2.

6.2.4 Individual exposure of humans

The individual indicator with the units mSv per Bq^{-1} m^2 is the end point which is most similar to that suggested by Aarkrog although the forum considered it better to estimate this quantity for a number of different time periods rather than to infinity. The value requires an estimate of the amount of radioactivity consumed, estimated from activity concentration in food products and information on dietary habits. This is then converted to Sv using the appropriate dose coefficient. The estimate should include both internal and external dose.

As part of the discussions surrounding this issue, Panayotis Assimakopoulos proposed a method of estimating exposure to communities which is provided as Appendix D.

Dietary habits

The types and amount of food which are eaten are a critical factor influencing ingested dose. The concept of identifying critical groups with respect to one or a few food products has been well developed.

Several methods have been used to estimate the radionuclide content of the human diet. From analysis of individual diet constituents collected at the production location, combined with information on the composition of human diet, the total intake of the various radionuclides in the diet may be calculated. Diet components may, however, also be collected at the location of consumption and pooled into one sample before analysis.

The composition of the diet may be estimated from statistical information on the consumption of foods, or from interviews on food habits in selected population groups. Dietary habits of a population change with time, especially during periods with rapid changes in the socio-economic structure of society (Howard *et al.,* 1999).

For estimating the influence of the diet, the range of radionuclide activity concentrations of components of the diet has to be evaluated. It is possible to calculate the radioecological sensitivity of the total diet in each region from the individual diet components.

Consumption of terrestrial products

A well known example of a terrestrial critical group are reindeer meat consumers who enhance their radiocaesium exposure by the inclusion of other semi-natural products in their diet. Thus, radioecologically sensitive groups due to consumption of a range of products have been identified such as hunters and gatherers in forests. In general, self-sufficiency with respect to diet and food production tends to make people more radioecologically sensitive.

Consumption of aquatic products

The average consumption of seafood in the world is given in the UNSCEAR report (2000). Average fish plus seafood consumption per individual is about 8 kg y^{-1}, ranging from 4 to 6 kg y^{-1} in the Near East and Africa to 10-14 kg y^{-1} in the Far East and Europe. It may be assumed that the annual consumption is 6 kg y^{-1} ocean fish, 1 kg y^{-1} freshwater fish and 1 kg y^{-1} shellfish. Total freshwater fish consumption by the world population is thus 6 10^9 kg y^{-1}, which, when a correction is made for edible weight of 50%, agrees with the estimated annual global harvest of 10^{10} kg landed weight. The catch mostly takes place within the continental shelf over an area of 27.5 10^6 km^2 and with a mean depth of approximately 50 m. The volume of these waters is thus 1.4 10^{18} l. The mean residence time of the water over the continental shelf is assumed to be about 3 years for ^{90}Sr and ^{137}Cs and 3.5 years for $^{239+240}Pu$, which is the same as that observed for the North Sea.

A well known example of an aquatic critical group are shellfish consumers in Cumbria, UK.

Whole body burdens

Here, we focus on the possibility of employing the whole-body ^{137}Cs burden acquired by inhabitants in an area as a measure of the area's Radioecological Sensitivity. The mechanism of whole-body ^{137}Cs acquisition following a contamination is depicted in Figure 6.11.

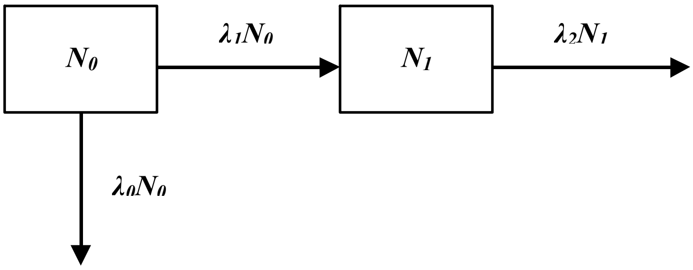


Figure 6.11. A model for whole-body ^{137}Cs accumulation following a radio-contamination event.

This is a two-compartment model describing the time evolution of the area's ^{137}Cs contamination $N_0(t)$ [measured, e.g., in Bq m^{-2}] and the corresponding whole-body ^{137}Cs accumulation of an inhabitant of the area $N_1(t)$ [measured in Bq] at time t after the arrival of the fallout. As shown in Appendix D, the differential equations that govern this model lead to the solutions

$$N_0(t) = N_0(0)\, e^{-\lambda_0 t} \tag{1}$$

and

$$N_1(t) = \frac{\lambda_1 N_0(0)}{\lambda_2 - \lambda_0}\left[e^{-\lambda_0 t} - e^{-\lambda_2 t}\right]. \tag{2}$$

in which $N_0(0)$ is the fallout at arrival time $t = 0$. According to this model, the environmental contamination is a monotonically decreasing function with a half-life

$$T^0_{1/2} = \frac{\ell n 2}{\lambda_0} \tag{3}$$

whereas the body-burden, starting with an initial value of $N_1 = 0$, reaches a maximum and thereafter decreases exponentially. This behaviour is depicted in Figure 6.12.

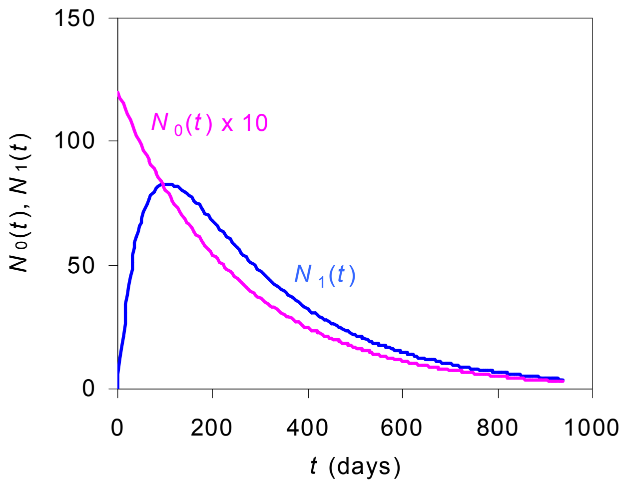


Figure 6.12. Predicted evolution of ^{137}Cs environmental contamination and whole-body burden

$$N_{1\,\max} = N_0(0)\,\frac{\lambda_1}{\lambda_2}\left(\frac{\lambda_0}{\lambda_2}\right)^{\frac{\lambda_0}{\lambda_2-\lambda_0}} \tag{4}$$

at time

$$t_{\max} = \frac{1}{\lambda_2-\lambda_0}\,\ell n\left(\frac{\lambda_2}{\lambda_0}\right) \tag{5}$$

The solutions in eqs. (1) and (2) depend on the three decay parameters λ_0, λ_1 and λ_2. The value of λ_1 affects essentially only the maximum reached by the whole-body burden [see eq. (4)] and plays little part in shaping the overall behaviour of N_0 and N_1. Its value is determined by the level of contamination of foodstuffs resulting from the environmental contamination N_0 and hence by soil-to-plant and plant-to-animal transfer ratios. It also depends on the dietary habits of the population in the area.

The decay constant λ_0 depends solely on the characteristics of the area under consideration. It accounts for removal of contamination by such mechanisms as binding of ^{137}Cs in clays, wash-off by rain and agricultural practices. The corresponding half-life will be in the order of years. On the contrary, the decay constant λ_2, also known as the "biological decay constant", depends primarily on the biological half-life in the human body, which varies from 60-140 days, with 100-110 days being commonly reported. Since $\lambda_0 << \lambda_2$, very soon (after two or three biological half-lives) the decay curves for N_0 and N_1 reach a state of secular equilibrium and thereafter decay essentially with the half-life of eq. (3). This behaviour is evident in Figure. 6.12.

From the above discussion it emerges that *prima facie* the maximum amplitude $N_{1\,\max}$, reached by the whole-body burden N_1, is a more representative quantity of the overall radioecological sensitivity of an area than the experimentally determined decay rate. This is because $N_{1\,\max}$ involves all decay constants in the process and thus takes into account both the area characteristics and the dietary habits of the population. The decay constant extracted from whole-body measurements is related primarily to the geographical characteristics of the area. This premise is investigated below by examining data available in the literature.

Analysis of data on whole-body measurements

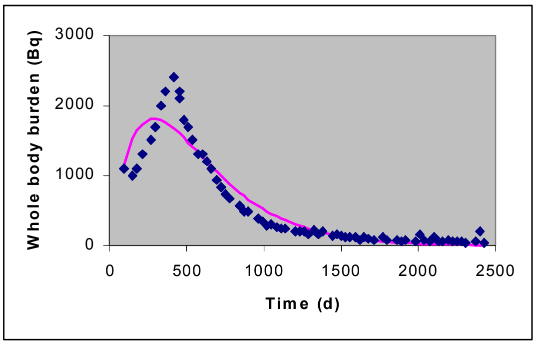
Several investigations involving the evolution of ^{137}Cs accumulation through periodic whole-body measurements have been reported in the literature, especially after the Chernobyl accident in 1986. For the purposes of the analysis presented here, only studies with a time span of more than four years and involving a data set of measurements with more than five points were considered. These studies are summarised in Table 6.7.

Table 6.7 Compilation of various studies of ^{137}Cs whole-body and respective deposition measurements. Fallout values are given by the authors mentioned in reference column or are calculated from values taken from other studies.

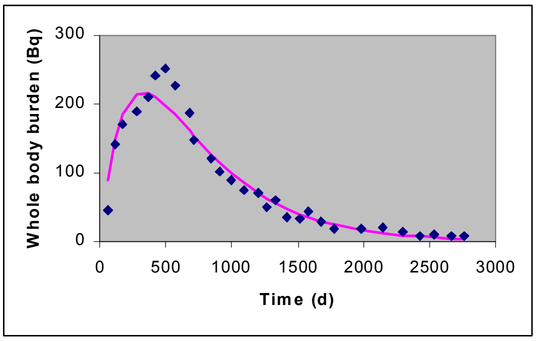
Area	Reference	Fallout (kBq m^{-2})	t_{max} (days)	$N_{1\ max}$ (Bq)	$N_{1\ max}$ / Fallout	$T_{1/2}^{eff}$ (d)
Bavaria –Germany	Ruhm, W. *et al.*, 1999	100	365	1520	15.2	270
Belgium	Genicot, J. & Hardeman, F. 1994	2[a]	365	219	110	415
Ioannina -Greece	Kalef- Ezra, J., *et al.*, 1992	10[b]	304	6775	678	190
Viitasaari-Finland	Rahola, T. *et al.*, 1998	29				851
Ammansaari-Finland	Rahola, T. *et al.*, 1998	1.5				1324
Sweden	Johansson, L. *et al.*, 1999	4				1746
Helsinki - Finland	Rahola, T. *et al.*, 1998	3				1190

[a]Mean value calculated from Salvadori *et al.*, (1996).
[b]Value taken from Petropoulos *et al.,* (2001).

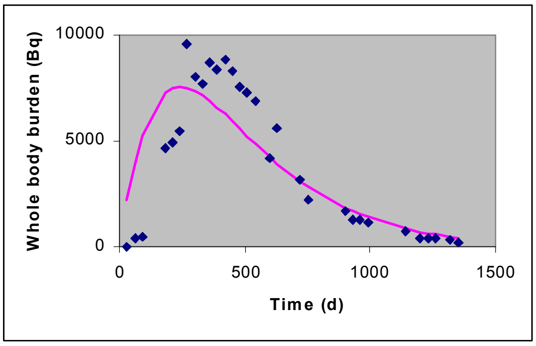
There are only a few studies in which the ^{137}Cs burden has been monitored with an early enough starting point so that the entire temporal evolution predicted in Figure 6.12 can be seen. In many studies, data collection commenced at a point beyond the maximum of $N_1(t)$, essentially in the region of secular equilibrium between N_0 and N_1. Furthermore, as shown in Figure 6.13, an attempt to fit eq. (2) to these three sets of data produced very poor results. The explanation for this may be that the data in Figure 6.13 are biased, at least during the immediate post-Chernobyl period, by countermeasures taken to safeguard the population from internal contamination. Thus, in the countries concerned, instructions were given to the population to avoid certain heavily contaminated dietary items. However, as time went by these measures were gradually relaxed. The general trend of such countermeasures would be to both diminish and delay the occurrence of the maximum ^{137}Cs body burden. In view of this, analysis of the data with regard to the maximum by means of the maximum ^{137}Cs body burden in eq. (4) or the functional form of eq. (2) was not further pursued.



(a)



(b)



(c)

Figure 6.13 Cs-137 whole body burdens of the general population in three European countries following the Chernobyl accident (a) Germany (b) Belgium (c) Greece

In an alternative type of analysis, all data in Table 6.7 were fitted with a single exponential decay function

$$N_1(t) = N_{eff} e^{-\lambda_{eff} t} \qquad (6)$$

for t > 2 y, where N_{eff} is an arbitrary normalisation factor. As discussed earlier, in this region where $N_0(t)$ and $N_1(t)$ decay essentially with the same rate, λ_{eff} is for all purposes equal to λ_0. The results of this analysis are presented in Figure 6.14 (below) and are seen to reproduce the experimental data rather well; the corresponding "effective" half-lives

$$T_{1/2}^{eff} = \frac{\ell n 2}{\lambda_{eff}} \qquad (7)$$

are contained in the last column of Table 6.7.

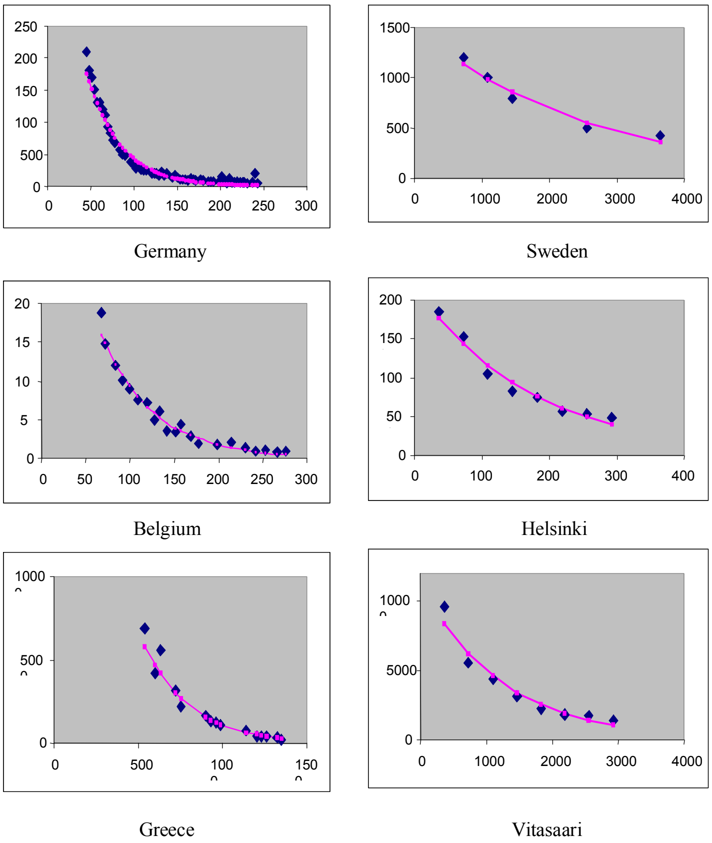


Figure 6.14 Whole body ^{137}Cs burden as a function of time for the general population in seven regions of Europe. The region of secular equilibrium is considered and the data have been fitted with a single exponential function

The data in the last column of Table 6.7, and also given in graphical form in Figure 6.15 faithfully reproduce existing measurements and their values cover a wide enough range to differentiate between the various regions considered with regard to their radioecological sensitivity. The data for regions in Scandinavia show the highest sensitivity with effective half lives for whole-body ^{137}Cs decontamination between 3.5 and 6 years. This may be attributed to the high transfer of radiocaesium to semi-natural products in these countries and their subsequent consumption. On the other hand, the region of northern Greece from which the corresponding measurements are taken, with a half life of 0.5 y, appears to be an extremely resilient region. Again, this may be explained by the predominantly clay soils of northern Greece and the dietary habits of the local population.

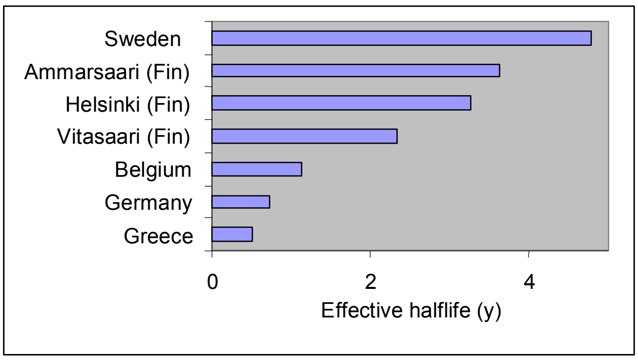


Fig. 6.15. Effective half life of ^{137}Cs whole -body decontamination in seven European regions.

To construct a scale, based on the effective half life of ^{137}Cs whole-body decontamination, that will meaningfully reflect the radioecological sensitivity of a region, all the factors mentioned above should be taken into account. The parameters entering a model that aims to calculate the value of $T_{1/2}^{eff}$ for a given region should be the soil composition, aggregate transfer coefficients and the dietary habits of the population. The data presented in this study indicate that an effort in this direction could lead to useful results.

Swedish studies

There have been detailed studies of effective ecological half-lives in different Swedish communities which allow a detailed assessment of within-country variation in effective ecological half lives in humans, summarized in Ågren (1998). In a paper on hunters, he observed that the effective ecological half life was longer than other southern Swedish groups and more comparable to people in Northern Sweden and Saami people. He attributed long effective ecological half-lives to ingestion of food products which had high persistent ^{137}Cs activity concentrations (including reindeer meat, game, mushrooms and berries). In contrast, Rääf (2000) estimated the effective ecological half-lives in a low deposition area in Southern Sweden to be 1.8 years after correction for the presence of pre-Chernobyl caesium. Here, the contribution from forest products are considerably less than in Northern Sweden.

7 MODIFYING FACTORS

7.1 Countermeasures

The application of countermeasures is only considered here with respect to dose reductions for humans. Both collective dose and individual doses (including critical population groups) have to be considered. Countermeasures can be considered with respect to two perspectives:

1. Emergency response - identification of sensitive areas, products or communities for emergency preparedness after an accident
2. Persistent sensitivity - identification of sensitive areas, products or communities where countermeasures may not be effective or difficult to implement

7.1.1 Emergency response

The application of radioecological sensitivity analysis using GIS will help to identify areas where countermeasures are needed (e.g. areas where maximum permitted levels are likely to be exceeded) and to prioritise between areas or communities on the basis of their environmental and behavioural/social conditions. By considering spatial variation in transfer rates, potential countermeasures can be identified which are most effective with respect to both cost and dose reduction.

Internal dose reductions

Countermeasures can reduce radionuclide activity concentrations in food products and thus ingestion of radioactivity. This can be achieved by implementation of soil and/or animal based countermeasures in agricultural and semi-natural production systems, but also by provision of advice to change special consumption habits (e.g. sheep/goat milk and mushroom consumption). Catalogues of countermeasures are available which can be used to select suitable measures with respect to effectiveness and potential environmental and social-ethical side effects (e.g. Voigt *et al.,* 2000). To apply these measures most effectively, information is needed on spatial and temporal variation in radionuclide soil to plant transfer (via Tag values and/or semi-mechanistically models), its transfer in foodchains, bioavailability of radionuclides and consumption behaviour. For individual doses, high transfer rates, bioavailability and consumption of highly contaminated products would each contribute to high radioecological sensitivity. Additionally, for collective dose, production rate of the most contaminated food types and extent of exported radioactivity from contaminated areas and consumption behaviour of the different population groups are major radioecological sensitivity determining factors. A change in production to potentially less contaminated products or a reduction or cessation in the production or harvesting of identified 'sensitive' foodstuffs will reduce collective dose. Thus targeted advice on the basis of identifying radioecologically sensitive criteria has considerable potential to reduce both individual and collective doses.

External dose reductions

For external doses, radioactivity in or on vegetation (especially forest canopies) or present on surfaces such as soils, roof tiles, walls of houses or urban surfaces (sideways and streets) need to be considered. An important sensitivity criterion with respect to external dose is the living habits of people. Thus, time spent in areas with higher external dose (by people such as forestry workers) will be a radioecologically sensitive aspect. The most radioecologically sensitive ecosystem components in terms of being major contributors to external doses in an urban environment are gardens and trees; other components make only minor contributions to external dose (TEMAS 1999).

There are fewer data available on countermeasures which are potentially effective for external dose reduction compared with internal dose measures. Urban countermeasures are largely directed to decontamination or covering of surfaces. The rate at which radioactivity moves down soil profiles influences the effectiveness of different types of ploughing or of procedures such as triple digging (Roed *et al*., 1999). For radioactivity deposited on soils procedures such as removal of soil layers will generate a considerable amount of waste.

The extent of external dose and of the potential reduction is highly sensitive to people's living habits, such as the amount of time spent in different types of environment. Therefore, appropriate recommendations on occupation times will help to reduce external doses. This might be especially applicable for children, for whom a much higher radiation risk than adults is attributed. Furthermore, those countermeasures that are known to be effective might not be socially acceptable (e.g. highly contaminated soil stored in piles around gardens).

7.1.2 Persistent radioecological sensitivity

Radioecological sensitivity will change upon the application of countermeasures. In a situation where countermeasures are being used, sensitive areas might be defined as those where it is difficult to implement effective countermeasures, for whatever reason. An example is semi-natural ecosystems, where standard agricultural measures are either not technically feasible or inappropriate.

7.2 Redistribution, runoff

Deposited radionuclides may move laterally in catchments, and estimation of runoff is an important component of evaluating freshwater contamination with time. However, substantial movement of radionuclides has been shown to occur from one part of a catchment to another, as reported during erosion process in the French alps and solution and particulate transport in upland catchments by Tyler & Heal (2000).

7.3. Bioavailability

A key factor for all ecosystems will be the bioavailability of the contaminant radionuclide. Environmentally mobile forms of radionuclides will transfer to a greater extent than those which are immobile, such as radionuclides strongly attached to particles. However, bioavailability may change with time. For instance, radionuclides associated with fuel particles released in the Chernobyl accident have gradually disintegrated in the environment, releasing radionuclides such as ^{90}Sr. Hence, uptake of ^{90}Sr has actually increased with time after the accident in areas where there were fuel particles deposited.

The extent which chemical form influences environmental transfer rates varies with different radionuclides. The transfer of radioactivity to the milk and meat of farm animals is likely to be a major exposure pathway of human populations, following an environmental release of radioactivity. Radioiodine absorption is complete and has not been shown to vary with source. Radiocaesium absorption varies over a 50-fold range, depending upon dietary source. Source dependent bioavailability is therefore an important factor determining the radiocaesium contamination of ruminant derived food products and reliable *in-vitro* techniques have been developed to rapidly determine bioavailability. In contrast, under conditions of adequate calcium intake, the absorption of radiostrontium will not be greatly influenced by the dietary source.

7.4. The Relevance of multiple pollution to the concept of radioecological sensitivity

Nowadays, contamination of the environment by anthropogenic pollutants occurs all over the world, thereby pushing the concept of « pure nature » into history. Contaminants are present everywhere, to various concentrations ranges in most ecosystem compartments of the biosphere. Through their growing occurrence, both locally or more widespread, they have promoted an increasing concern with respect to the risks associated with their toxicity, both for human health and for the functioning of ecosystems themselves. An important point to be made here is that contaminants, including radionuclides, have invaded the environment as complex mixtures. Consequently, the behaviour of radionuclides can potentially be affected, through synergistic or antagonistic effects, by the concomitant presence of other pollutants (PCB, HAP, heavy metals, phytosanitary chemicals, etc), either due to the presence of different contaminants in the polluting source or the contamination by a range of different anthropogenic sources. This leads to the necessity for reassessing the environmental behaviour of radionuclides and associated risks in a broader context than has generally been considered before. Achieving this goal requires a reassessment of traditional radioecology (which deals with ultra-trace amounts) into the general field of ecotoxicology (which deals with traces or more). Preliminary investigations have shown that radionuclides fluxes and compartmental loads are indeed affected by the presence of non-radioactive pollutants, both in terrestrial and aquatic ecosystems. These studies have shown that:

- The distribution of radionuclides in the various horizons of podzolic soils from boreal forests is modified by an increasing load of the heavy metals, Cu and Ni. Concomitantly, the soil-to-plant transfer factors are influenced, effects which are currently described but not yet appropriately explained (EPORA project Final Report (Suomela *et al*., 1999).
- Experiments in controlled conditions on biological models representing various cellular organisation levels (bivalves, fishes) have demonstrated that a chronic stress induced by Cd and Zn, at concentrations representative of those occurring in the environment, promote a decrease of radionuclide bioaccumulation in organisms exposed to these heavy metals. (Ausseil *et al.,* 2000, Fraysse *et al.,* 2000).

The implication for radioecological sensitivity analysis is that Tag values may be modified by the presence of other pollutants, but that currently, there is inadequate evidence to provide guidance on these effects for most pollutants.

8. RECOMMENDATIONS FOR USERS

Who is the user of the radioecological sensitivity concept? This may vary from one situation to another and between countries. Some identified potential user groups are:

1 Those planning a practice giving rise to releases of radioactive substances and who wish to take into account the environmental impact of the practice (including Environmental Impact Assessment (EIA), for future facilities).
2 Regulatory authorities who have to assess the impact of the practice including accident scenarios.
3 Emergency preparedness planners and those involved in managing intervention situations.
4 (Political) decision makers in charge of large-scale countermeasures.

It is obvious that the radioecological sensitivity concept is not a concept that will be used in an acute situation such as directly after an accident. It is rather meant to be used in radiation protection, nuclear safety and emergency preparedness when there is a need to identify areas that have the potential of being of particular concern from a risk perspective. One major user group is the third group above. If radioecologically sensitive areas are identified in advance in an emergency preparedness plan, the efforts to improve the situation can be implemented faster and be more efficiently focussed on the appropriate countermeasures in these areas. Identification of sensitive areas may also lead to that identified problems can be reduced by actions taken in advance, before an accident occurs.

The radioecological sensitivity concept is based on four quantifiable end-points or radioecological indicators – aggregated transfer coefficients, action loads, exposure of individuals and flux. All endpoints are time dependent and spatially variable. It is of major importance to know the resulting doses to individuals, as these are often the limiting parameter for regulations. In practice, however, these doses are seldom possible to measure. Dose estimates must therefore usually be based on estimates of diet, social habits, food availability and dose coefficients. A more direct, and measurable quantity is action load. This can be related to the intervention limits set by the authorities. With knowledge of the transfer of radionuclides in the ecosystem, for example expressed in terms of aggregated transfer coefficients, these relations can be established beforehand in an emergency preparedness plan. In principle, then, some of the identified problems can be reduced beforehand, before an accident occurs, if this is deemed to be economically and socially acceptable.

It is generally recommended that:

- The practical use (model predictions, ecological input, pathway analysis, habits, exposures) of the radioecological sensitivity concept and in particular the (scientific) interpretation of the results should be made by a person educated for this purpose.
- The results of the interpretation should preferably be presented in terms of maps and texts together with any associated uncertainties in a way that is understandable for a non-expert
- The results and the various assumptions made should be transparent.

It is further recommended that:

- The decision maker who is not an expert in radiation protection and radioecology is given the results in terms of maps and texts, and is
- Informed about the uncertainties (in space and time)

9. WHAT IS STILL NEEDED?

Further work is needed on the uncertainties of radioecological sensitivity analysis, particularly on those associated with the use of spatial data. Recent studies of predicted ^{137}Cs activity concentrations in cow milk in Hungary using different input data for soil type have shown substantial differences in the spatial distribution of ^{137}Cs contamination (Wright *et al.,* in press). It is clear that spatial modelling requires reliable, representative input data, and this is just as important as constructing and parameterising the model. Estimation of spatial variation in radioecological sensitivity relies upon good input data and further work is needed to address the issue of spatial data quality.

The prime focus of this forum was to identify the factors determining radioecological sensitivity mainly with respect to Man in response to the user group. Nevertheless, it is advisable to widen our approach by considering radioecological sensitivity through the dose to biota in addition to the dose to humans. Therefore, any new approach would incorporate effects on biodiversity and would allow us to make a judgement on the suitability of a given ecosystem to host human activities related to and/or utilising radionuclides (e.g. a nuclear power plant, a nuclear waste processing plant, etc).

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APPENDICES

A. Relevant Radiation Protection Issues,

Brechignac, F.

Low level irradiation of an organism with ionising radiation may lead to harmful effects.. The measure used to quantify the amount of radiation received is the dose. It is dose estimation, and hence dosimetry, which allows determination of the detrimental effect on man as a result of the different types of exposure. The main concepts developed by ICRP for protection purposes are briefly summarised in the following.

Absorbed dose:

Radiation emitted by radioactive substances interact with matter by releasing energy. The quantity of energy released is called the “absorbed dose”. Absorbed dose is expressed in Gray (Gy) which corresponds to the energy of 1 Joule released in 1 kg of matter (1 Gy = 1 J/kg). This energy release is the source of perturbations within the exposed matter, by ionisation of its constituting atoms and are therefore called ionising radiation. For living matter, the absorbed dose alone does not describe the overall risk. This is why two other concepts have been created: the “equivalent dose” and the “effective dose”.

Equivalent dose:

All types of radiation (for example alpha, beta and gamma) do not produce the same effects. If energy is released in a small volume of tissue, the hazard will be greater than if this same energy is released in a larger volume ?. Alpha radiation, with a mean travel distance within living matter which is very small, is more deleterious than beta radiation which is only partially retained within the human body. The “equivalent dose” concept expresses these effects in an equivalent manner, using the Sievert unit (Sv), by taking into account a weighting factor characteristic of each radiation type. As an example, the equivalent dose for alpha radiation is 20 times the absorbed dose, whereas for gamma radiation, it is equal to the absorbed dose.

Effective dose:

The risk of cancer induction is subordinated to the dose, but all tissues do not present the same susceptibility to radiations. The “effective dose” allows estimation of the risk of cancer induction in the overall body by taking into account the various radiosensitivities of the different tissues concerned. Hence, the effective dose is the sum of the equivalent doses for each tissue (or organ) weighted by a coefficient which depends on the susceptibility of the irradiated tissues to stochastic/detrimental effects. These coefficients being normalized, their sum is equal to 1.

Individual and collective dose:

According to the ICRP the total individual dose, which is the sum of external and internal doses, should be below 20 mSv/year (averaged over 5 years, 50 mSv for a single year) for workers, and below 1 mSv/year for members of the public. The collective dose addresses the overall risk for public health, and constitutes mainly a management/regulatory indicator for comparative purposes (eg for selecting one process among many), and is expressed in manSv.

Time considerations:

The “committed dose” is related to internal contamination. For adults, it is calculated for a period of 50 years from the initial time considered whatever the age of the exposed individuals. For children, it is calculated for an overall period up to the age of 70 years.

B Details of ECOSYS model relevant to scenarios for action loads

For the scenarios considered in this context, the following relevant detailed parameter assumptions are given.

1. Locations assumed

The calculations have been performed with the standard parameter set which was developed for Southern German conditions.

2. Interception rates assumed

Dry deposition scenario: For deposition onto pasture a yield dependent deposition velocity has been applied. In addition, deposition to the underlying soil has been assumed. See page "Deposition" for deposition to pasture and for deposition to pasture+soil. The ratio of these two numbers can be interpreted as a dry deposition interception rate.

Wet deposition scenario: Wet deposition has been assumed to occur with 3 mm of precipitation (= 3 liters per m²). Depending on this amount of rainfall, and on the yield of pasture an interception fraction is derived which is given in the second column of table "Deposition".

3. Weathering rates assumed

A weathering rate according to a half-life of 25 days has been assumed. In addition, decrease of activity concentration due to increase of biomass has been considered; this is taken into account by a dilution rate which is dependent on the month of the year according to the following half-lives (in days; no growth dilution during the months not given in the table):

	Grass (Intensive cultivation)	Grass (Extensive cultivation)
March	9	12
April	24	24
May	20	45
June	20	60
July	20	60
August	20	60
September	30	90
October	40	120

4. Biomass assumed

The season dependent biomass of grass is given by the following tables (between the given dates linear interpolation is made):

Grass (Intensive cultivation):

Date	**1.Jan.**	**15. March**	**16. May**	**31. Oct.**	**1. Nov.**	**31. Dec**
Yield (kg m²)	0.03	0.05	1.5	1.5	0.05	0.03

Grass (Extensive cultivation):

Date	**1.Jan.**	**15. March**	**30. June**	**31. Oct.**	**1. Nov.**	**31. Dec**
Yield (kg m²)	0.03	0.05	1.5	1.5	0.05	0.03

5. Transfer coefficient to milk

Animal product	**Transfer factor in d l-1**			
	Caesium	**Strontium**	**Iodine**	**Plutonium**
Cow milk	$3 \cdot 10^{-3}$	$2 \cdot 10^{-3}$	$3 \cdot 10^{-3}$	$6 \cdot 10^{-5}$

Sheep and goat milk	$6 \cdot 10^{-2}$	$1.4 \cdot 10^{-2}$	$5 \cdot 10^{-1}$	$4 \cdot 10^{-4}$

6. Biological half-lives in milk

Animal product	Caesium		Strontium		Iodine		Plutonium	
	a_1, a_2	$T_{1/2}$ (d)	a_1, a_2	$T_{1/2}$ (d)	a_1, a_2	$T_{1/2}$ (d)	a_1, a_2	$T_{1/2}$ (d)
Cow milk	0.8	1.5	0.9	3	1	0.7	1	7000
Sheep, and goat milk	0.2	15	0.1	100	-	-	-	-

7. Soil-plant uptake and effective half-life

Reduction of root uptake is due to migration to deep soil, fixation, and radioactive decay.

Factor	Caesium	Strontium	Iodine	Plutonium
Transfer factor for pasture (Intensive cultivation)	5×10^{-2}	5×10^{-1}	1×10^{-1}	2×10^{-4}
Transfer factor for pasture (Extensive cultivation)	1	1	1×10^{-1}	2×10^{-4}
Half life (a) for				
migration	40	20	40	40
fixation	9	20	1000	10000
⇒ effective	5.9 (^{137}Cs)	7.5 (^{90}Sr)	0.022 (^{131}I)	

8. Additional information

Seasonal dependent feeding rates (kg d^{-1}) (between the given dates linear interpolation is made):

	Cow		Sheep		Goat	
Date	**Pasture**	**Hay**	**Pasture**	**Hay**	**Pasture**	**Hay**
1. Jan.	0	14	0	1.8	0	2.6
20. April	0	14	0	1.8	0	2.6
10. May	70	0	9	0	13	0
20. Oct.	70	0	9	0	13	0
9. Nov.	0	14	0	1.8	0	2.6
31. Dec.	0	14	0	1.8	0	2.6

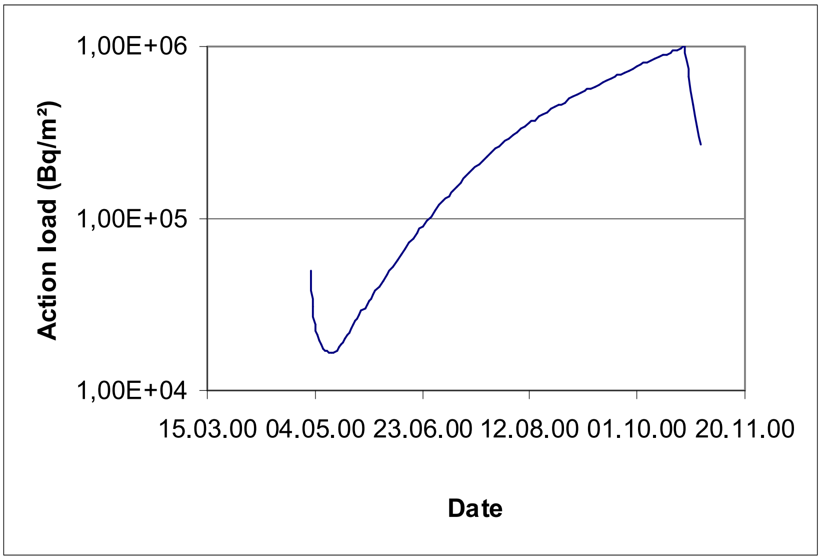
C Action load estimation using ECOSYS

ECOSYS: Action loads as a function of time

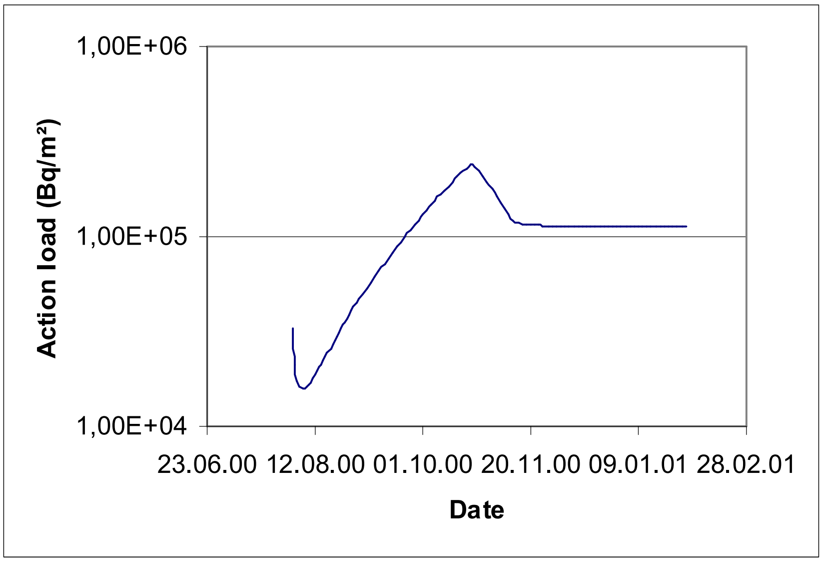
For most cases, the action load is given for 6 months, except for ^{131}I where 1 month is shown (on a 6 month plot the values during the first months – which are most interesting - cannot be distinguished from the x-axis). All plots are for feeding with grass from intensive cultivation and three different deposition dates are used. The action loads refer to a maximum permitted level of activity in cow milk of

1000 Bq/l	for Cs-137
500 Bq/l	for I-131
125 Bq/l	for Sr-90
20 Bq/l	for Pu-239

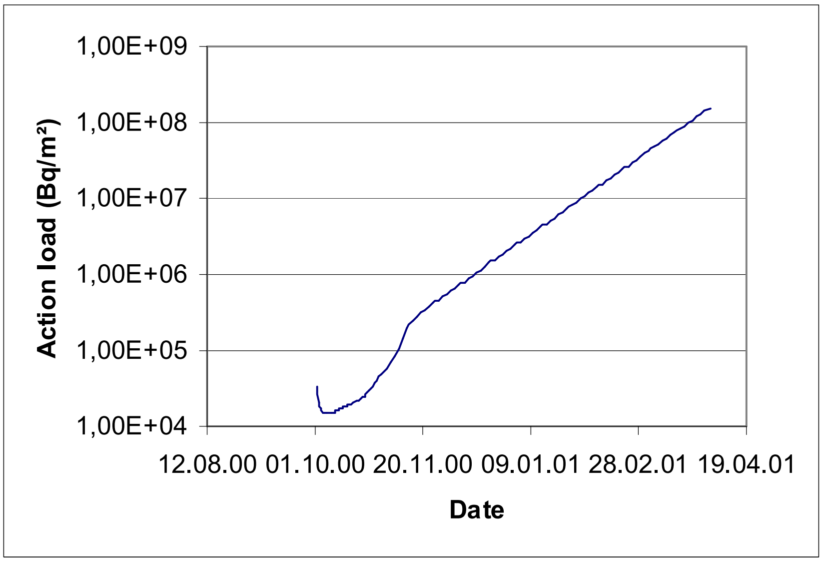
CAESIUM-137



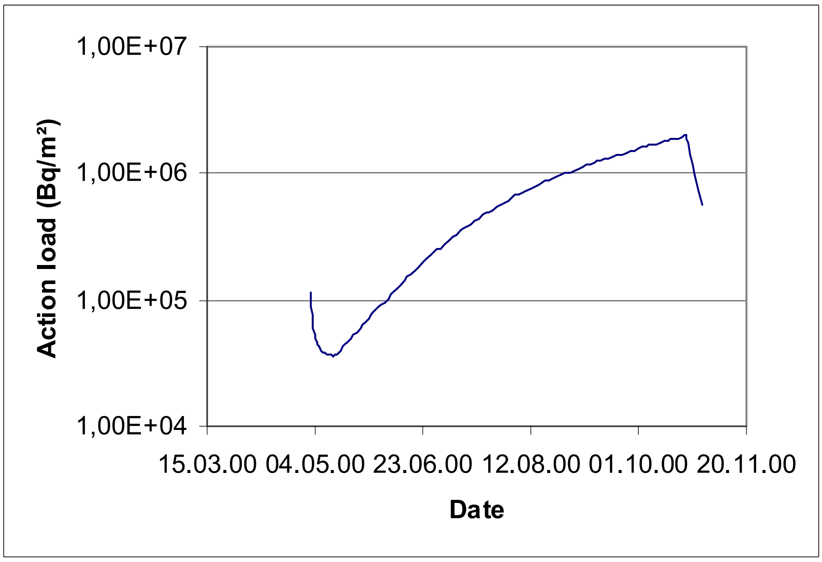
Cs-137, dry, 1.5.00



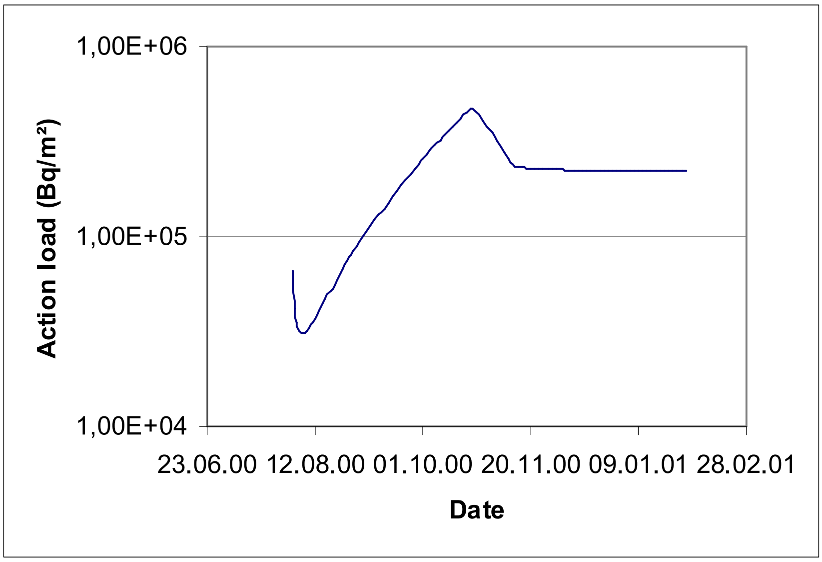
Cs-137, dry, 1.8.00



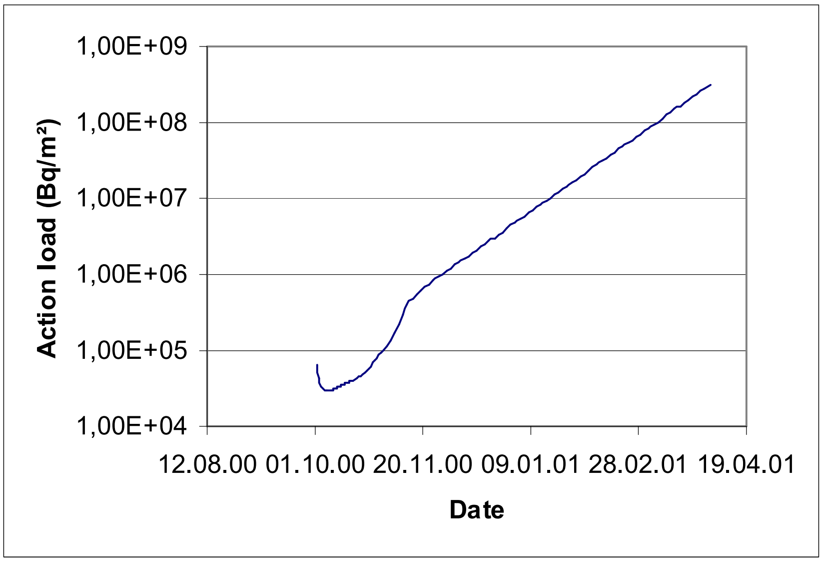
Cs-137, dry, 1.10.00



Cs-137, wet, 1.5.00

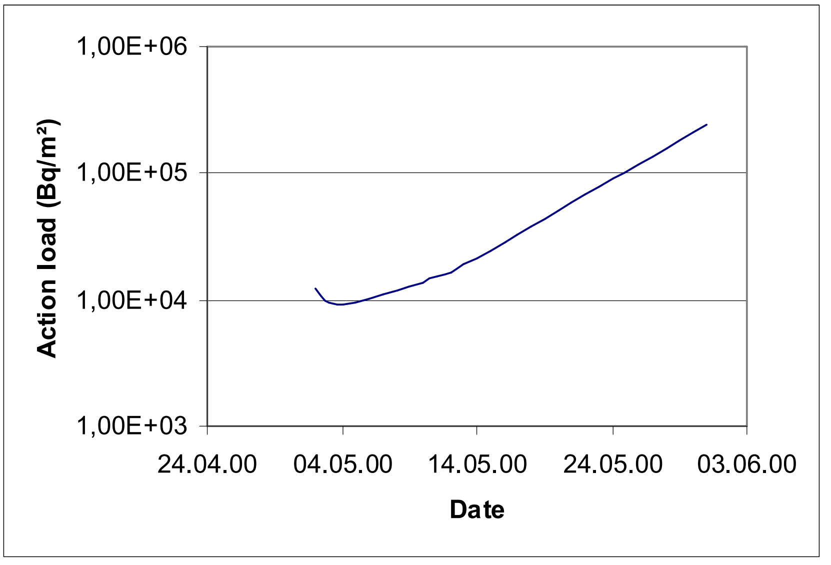


Cs-137, wet, 1.8.00

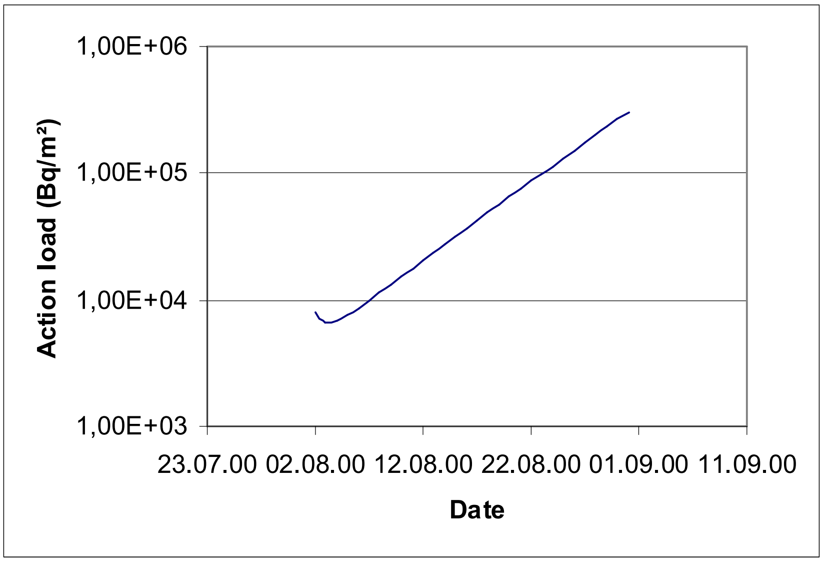


Cs-137, wet, 1.10.00

IODINE-131



I-131, dry, 1.5.00



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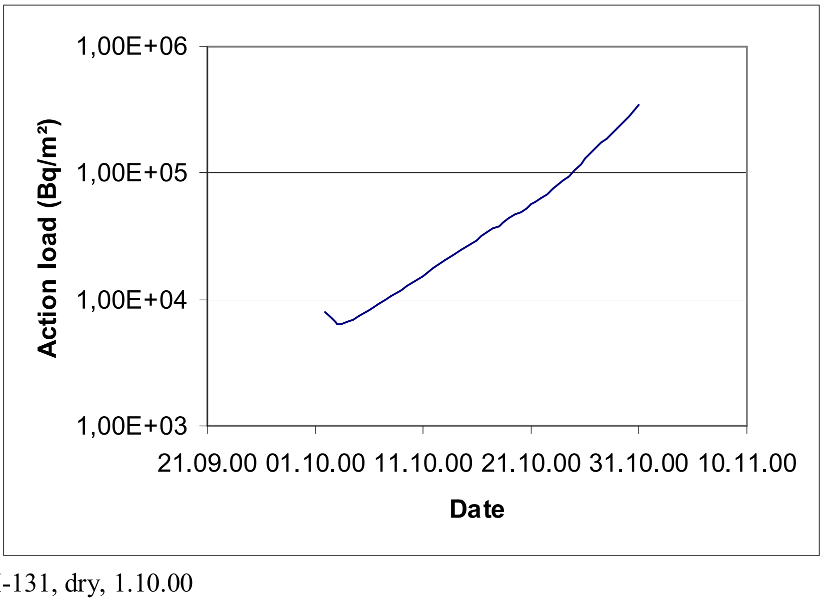
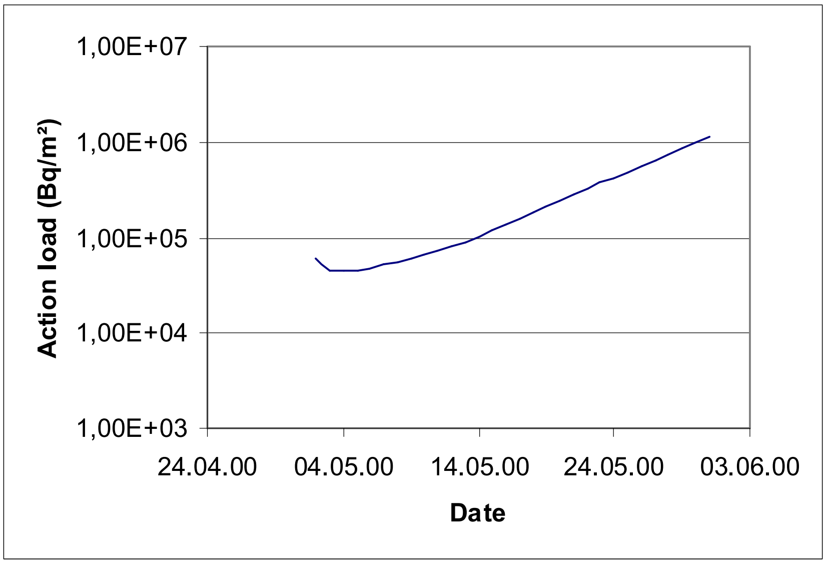


I-131, dry, 1.10.00



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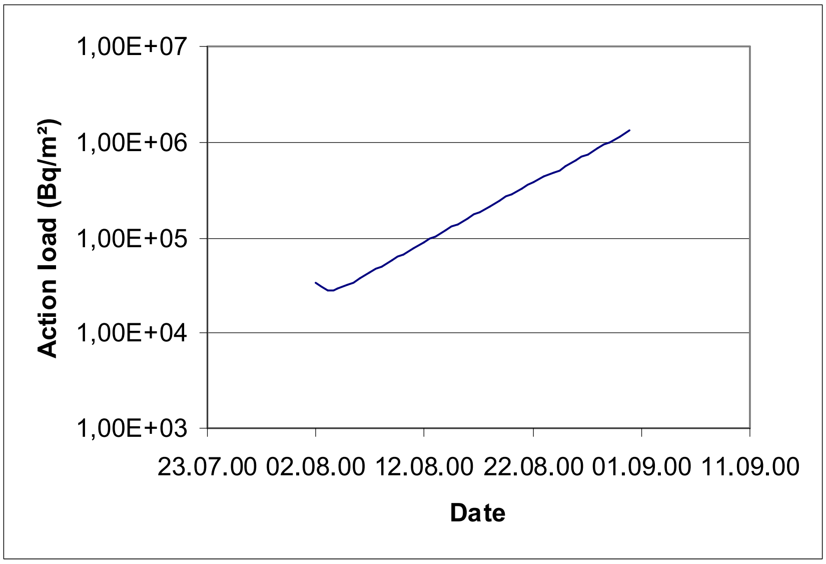

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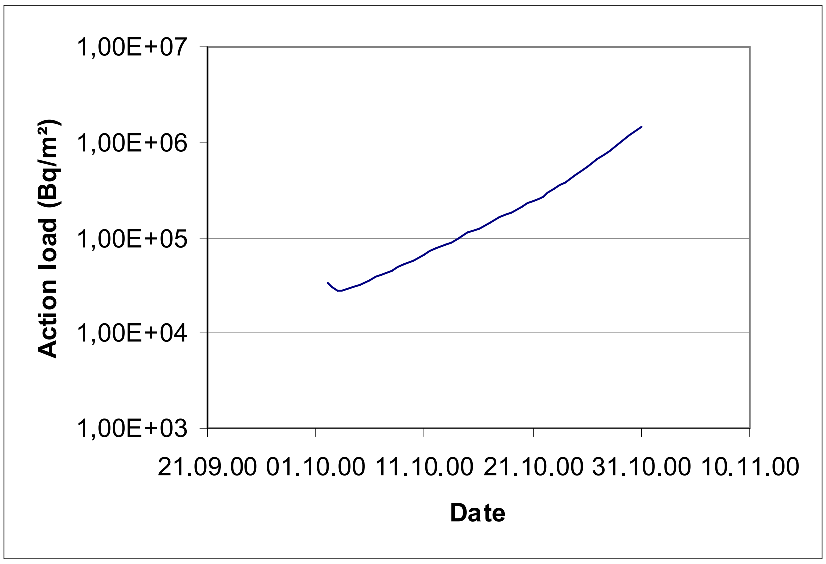
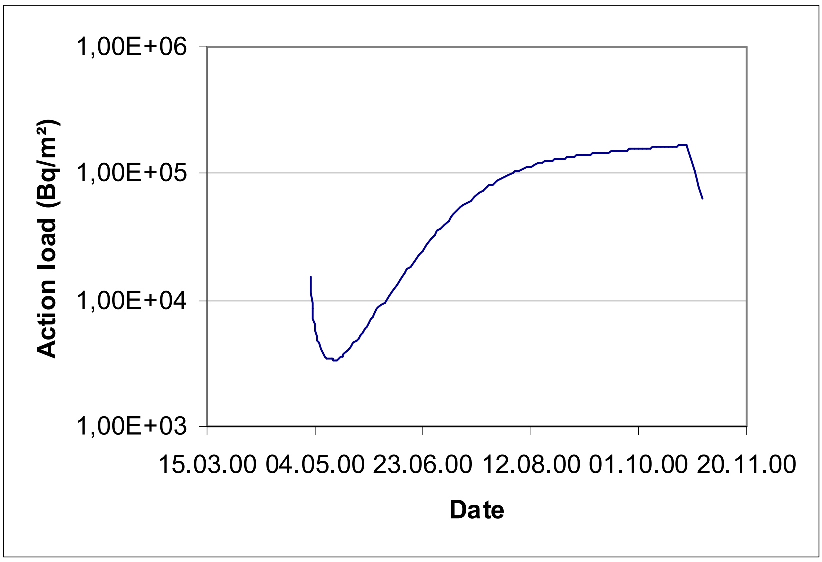
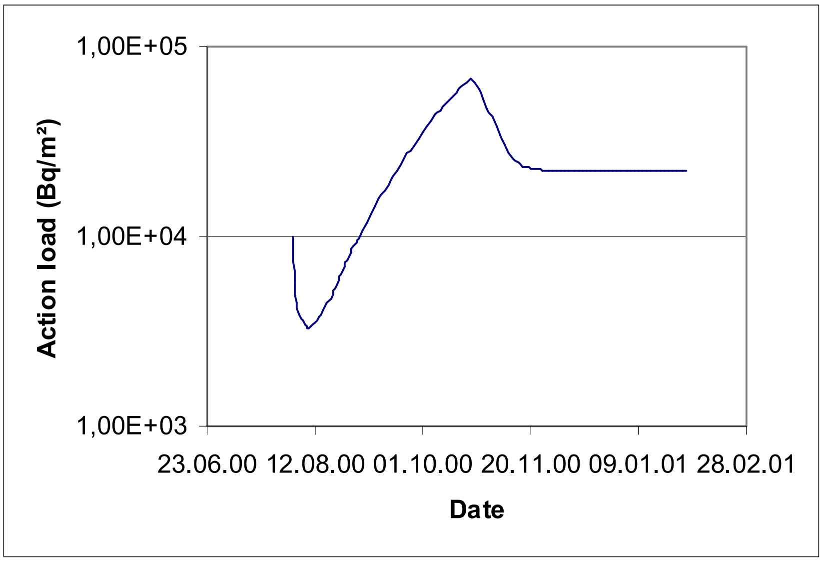


I-131, wet, 1.10.00

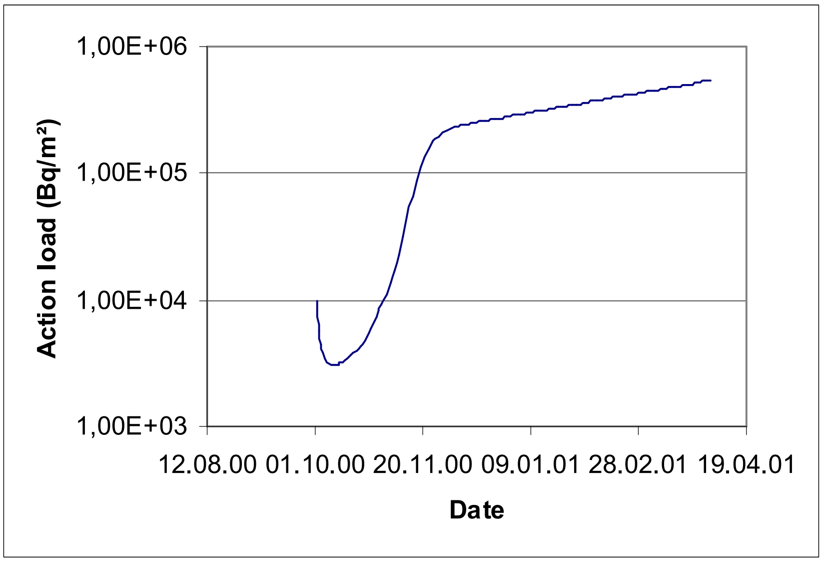
STRONTIUM-90



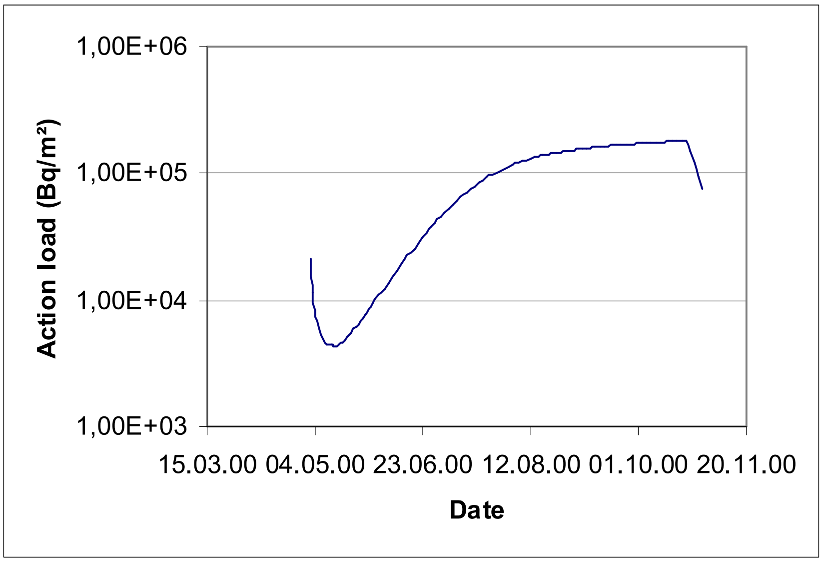
Sr-90, dry, 1.5.00



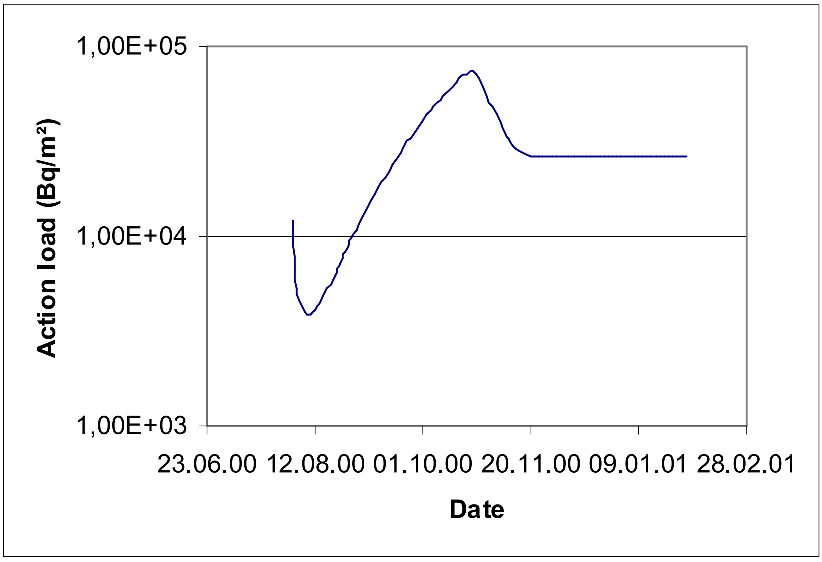
Sr-90, dry, 1.8.00



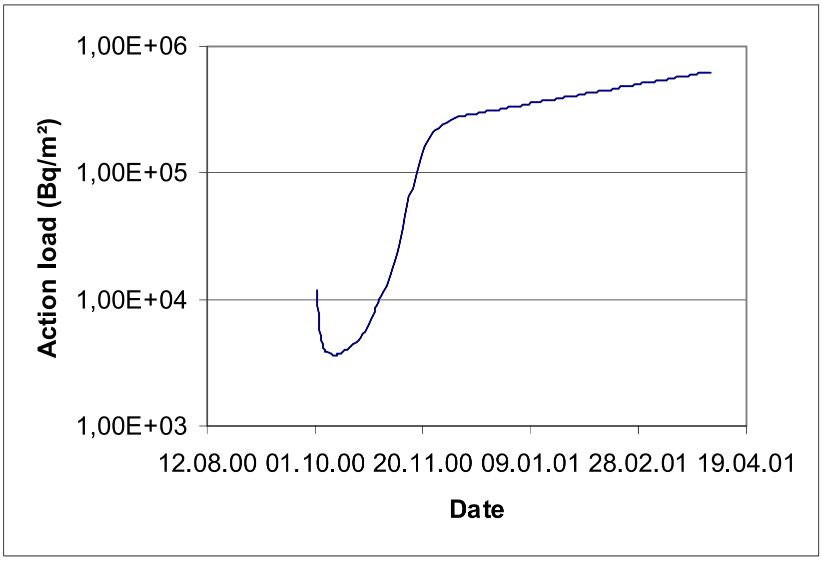
Sr-90, dry, 1.10.00



Sr-90, wet, 1.5.00

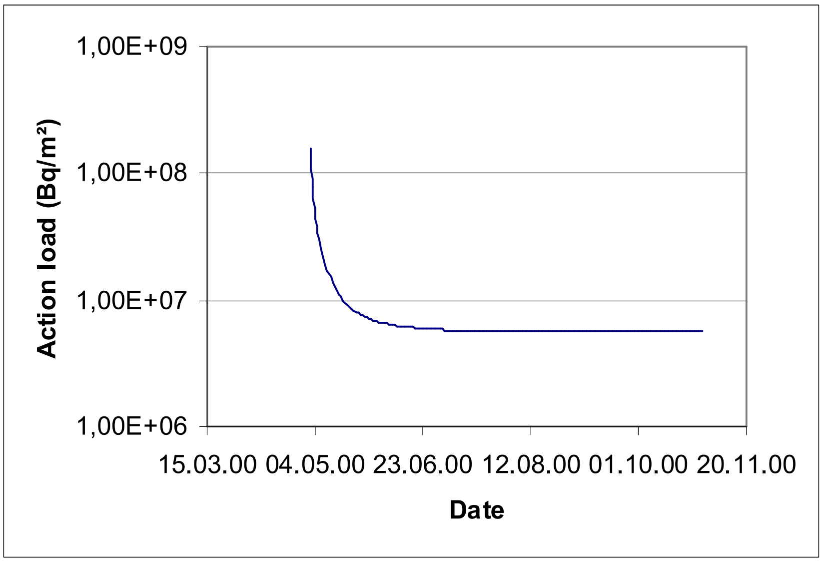


Sr-90, wet, 1.8.00

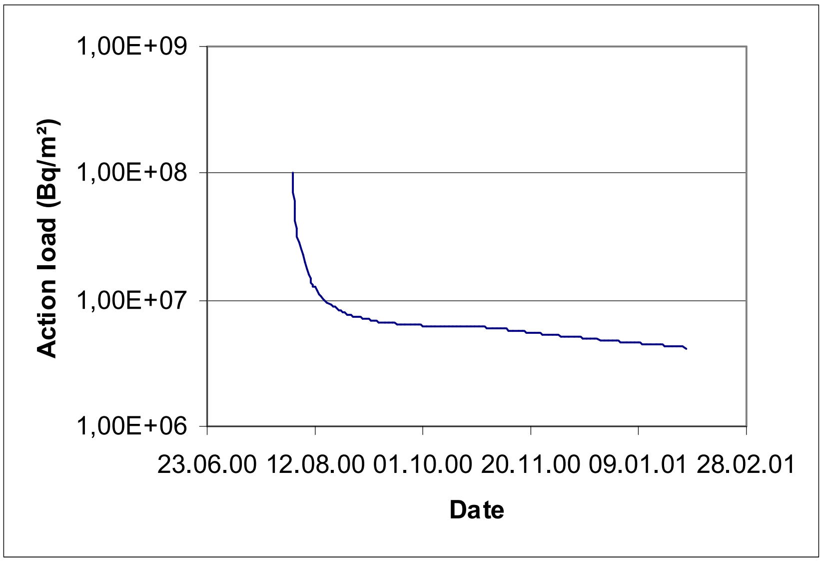


Sr-90, wet, 1.10.00

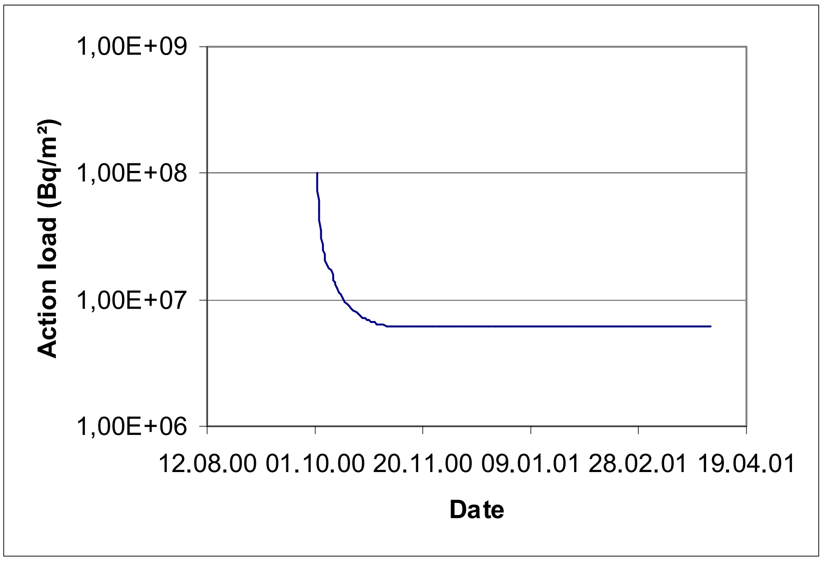
PLUTONIUM



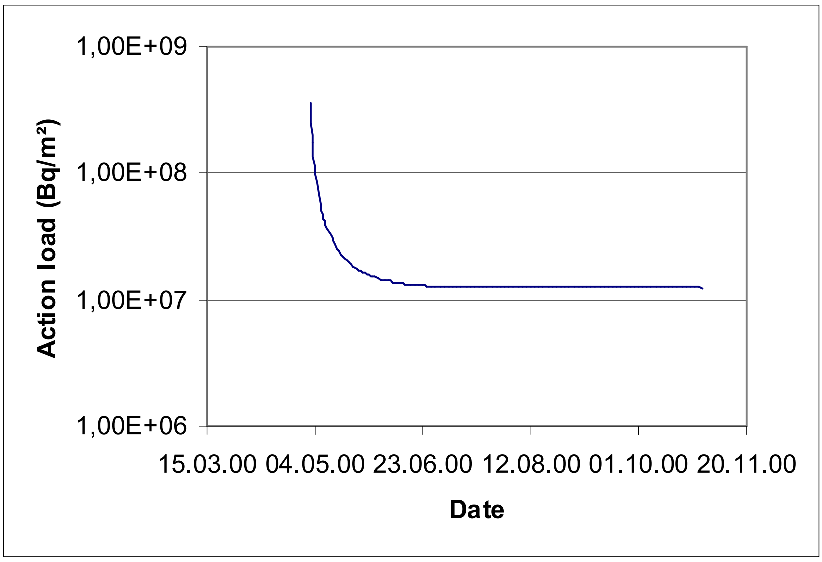
Pu-239, dry, 1.5.00



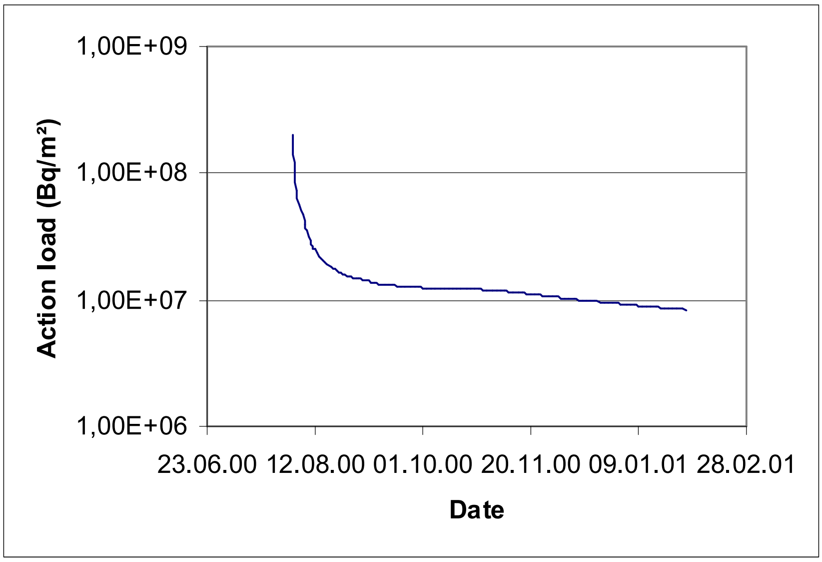
Pu-239, dry, 1.8.00



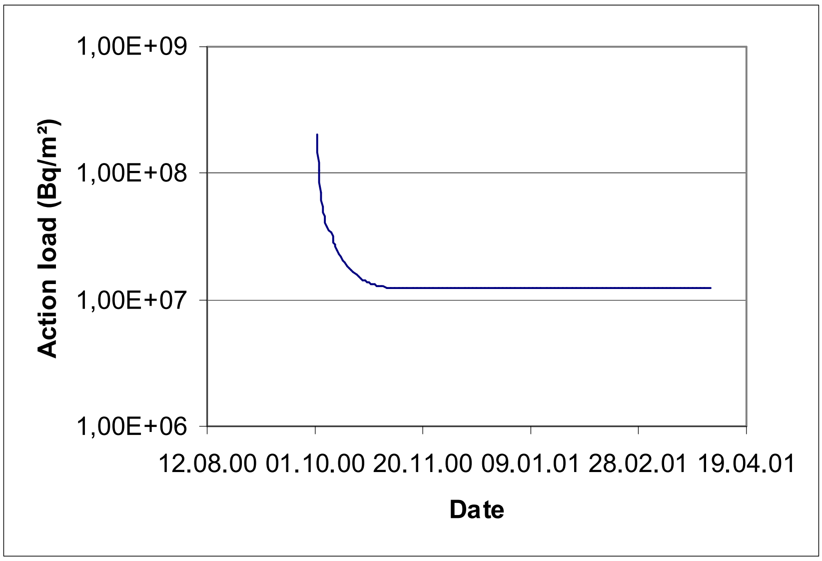
Pu-239, dry, 1.10.00



Pu-239, wet, 1.5.00



Pu-239, wet, 1.8.00



Pu-239, wet, 1.10.00

D. A possible approach to estimation of radioecological sensitivity with respect to individual doses for communities

P.A. Assimakopoulos

According to the schematic in Fig. 6.1, a endpoint of the Radioecological Sensitivity of an area is the effective dose that an individual receives over a period of time, T_0, after the incidence of the fallout. This quantity depends both on the isotope I that causes the dose and the age of the individual T at the time of onset of the deposition. Subtracting all factors of geometry (the area of the region, the intensity of the deposition and the composition of the population) we may define a quantity, which we may call Partial Radioecological sensitivity $S_I(T,T_0)$ as

> The effective dose, accumulated by an individual of age T at the time of the incident until they reach the age $T + T_0$, if the deposition of isotope I in the region is 1 Bq m^{-2}.

This approach has the advantage of flexibility in defining the time period considered and the age of the person exposed.

It would be possible to extend the concept of Radioecological Sensitivity to a region, by estimating the average effect of the exposure to the deposition of all ages represented in the population. Thus, one can define formally the Radioecological Sensitivity of a region to isotope I as

$$S_I(T_0) = \int_0^{T_0} S_I(T,T_0) f(T) dT \qquad (1)$$

in which f(T) is the age distribution function of the population, normalised to unity, i.e.

$$\int_0^{T_0} f(T) dT = 1 . \qquad (2)$$

For a mixture of isotopes in the deposition one can go one step further and define a quantity that describes the Radioecological Sensitivity of a region with a single number. If a_I is the fractional population of isotope I in the deposition, one can introduce the quantity

$$S(T_0) = \sum_I a_I S_I(T_0) \qquad (3)$$

with the condition, for normalisation purposes,

$$\sum_I a_I = 1 . \qquad (4)$$

As indicated in eqs. (1) and (3) the physical quantities $S_I(T_0)$ and $S(T_0)$ depend on the time period T_0 considered (e.g. T_0 = 1, 5, or 25 y). Both quantities are measured in Sv Bq^{-1} m^2.

Integral Quantities

The integral physical quantity that emerges naturally in this scenario is what may be termed the Expected Detriment (due to a specific isotope I) $D_I(T_0)$, as evaluated for a specified time interval T_0. Again, intuitively, the Expected Detriment to a region should be proportional to the:

- Radioecological Sensitivity $S_I(T_0)$ of the region
- Deposition P_I of isotope I in the region
- Population N affected

and inversely proportional to the

- Area A of the affected region.

Folding in the geometry of the region we may thus write

$$D_I(T_0) = \frac{PNS_I(T_0)}{A} = pNS_I(T_0) \qquad (5)$$

where in the last step of eqn (5) we have used the specific deposition p (Bq m^{-2}) in the region

$$p = \frac{P}{A}. \qquad (6)$$

Finally, following the practice reflected in eqn (3), one can define the Detriment to a region due to all isotopes in the deposition, expressed with a single number, through the relation

$$D(T_0) = \sum_I a_I D_I(T_0) \qquad (7)$$

under the normalisation condition of eqn (4). The units of quantities $D_I(T_0)$ and $D(T_0)$ are man Sv.

Calculation of Radioecological sensitivity

The quantity that needs to be calculated first according to the above scenario is the Partial Radioecological sensitivity $S_I(T,T_0)$. Such a calculation may be described with the schematic contained in Fig. A1. The steps in the calculation proposed by this scheme are as follows:

1. The dietary habits of the population in the region are determined and food products of primary importance in the diet (e.g. milk, meat, wheat) are identified. A 'model diet' for the region is determined in the form of a set of coefficients $\{b_m\}$, each of which represents the daily, monthly or yearly intake of food product m (e.g. in kg d^{-1}).
2. By definition, the aggregate transfer coefficients Tag(I,m) for the transfer of isotope I to food product m in the model diet is the contamination concentration in the product resulting from a deposition of 1 Bq m^{-2}. Numerical values of Tag's will depend primarily on soil type, agricultural practices, diet of animals etc and may be estimated from values measured in regions with similar characteristics. Thus, the total activity intake R(I) of an individual in the region, resulting from a deposition of 1 Bq m^{-2} of isotope I, will be given by

$$R(I) = \sum_m T_{ag}(I,m) b_m \qquad (8)$$

   where the sum extends over all items in the model diet.
3. Given the daily intake R(I), the Partial Radioecological Sensitivity $S_I(T,T_0)$ may be calculated according to the method prescribed in ICRP Publication 30 (ICRP 1980).

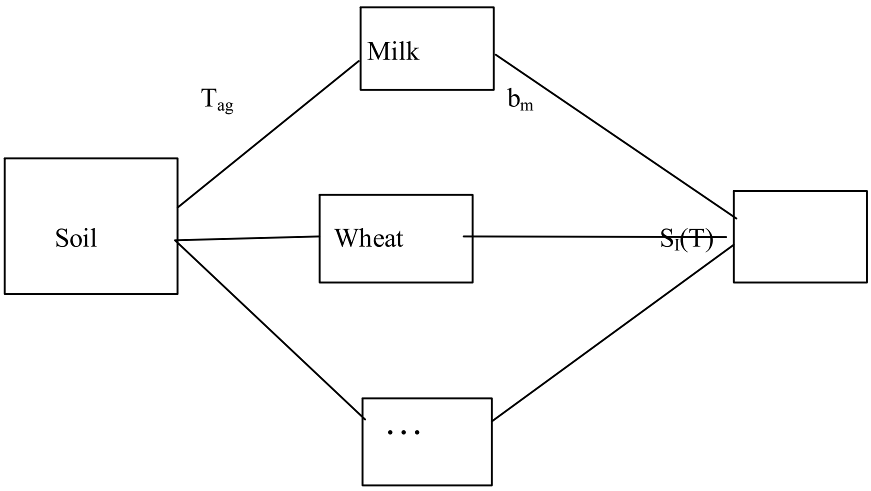


Figure 1. Successive steps in the calculation of Partial Radioecological

Figure A1

Following this calculation, one could proceed to calculate the quantity in eqn (1), which further requires knowledge of the age distribution function f(T). The shape of f(T) will significantly affect the radioecological sensitivity of a region. Since activity intake is expected to affect younger age-groups to a much greater degree, a skewed age distribution curve, such as an area dominated by retired, old people will result in a lower numerical value of $S_I(T_0)$.

The Radioecological Sensitivity scale

The quantity defined in eqn (3) can be evaluated to express Radioecological sensitivity through a single number and thus create, for purposes of comparison, a Radioecological Sensitivity Scale, such as the Richter scale in seismology. To do this, one needs the coefficients a_I in eqns (3) and (4). These may be taken from the double-humped fission curve or could be calculated in a variety of accident scenarios.

Whole Body Analysis

Let $N_0(t)$ be the ^{137}Cs fallout in an area [e.g. Bq m^{-2}] and $N_1(t)$ the corresponding average whole-body burden [Bq] of an inhabitant of the area at time t after the arrival of the fallout. The time evolution of the fallout may be assumed to follow an exponential decay

$$N_0(t) = N_0(0)\, e^{-\lambda_0 t} \qquad \text{(A.1)}$$

in which $N_0(0)$ is the fallout at arrival time. Then, according to the model in Fig. 1, the rate of change in the body-burden N_1 is governed by the differential equation [Assimakopoulos, P.A. *et* al., 1991, Crout, N.M.J. *et al.*, 1996]

$$\frac{dN_1}{dt} = \lambda_1 N_0 - \lambda_2' \frac{N_1}{V_1} \qquad \text{(A.2)}$$

in which V_1 is the average weight of an inhabitant and N_1/V_1 the ^{137}Cs concentration in his body. In order to simplify the notation, we may absorb V_1 into λ_2' and write eq. (A.2) as

$$\frac{dN_1}{dt} = \lambda_1 N_0 - \lambda_2 N_1 \ . \qquad \text{(A.3)}$$

With the help of eq. (A.1), eq. (A.3) may be then written as

$$\frac{dN_1}{dt} + \lambda_2 N_1 = \lambda_1 N_0(0) e^{-\lambda_0 t} \qquad \text{(A.4)}$$

which, if we multiply both sides with $e^{\lambda_2 t}$, may be cast in the form of the total differential

$$\frac{d}{dt}\left(N_1 e^{\lambda_2 t}\right) = \lambda_1 N_0(0) e^{(\lambda_2 - \lambda_1)t} \qquad \text{(A.5)}$$

yielding

$$N_1(t) = \frac{\lambda_1 N_0)0)}{\lambda_2 - \lambda_0}\, e^{-\lambda_0 t} + K e^{-\lambda_2 t} \qquad \text{(A.6)}$$

The constant of integration K in the last expression may be determined from the boundary condition $N_1(0) = 0$ as

$$K = -\frac{\lambda_1 N_0(0)}{\lambda_2 - \lambda_0} \qquad \text{(A.7)}$$

which yields the solution for $N_1(t)$ in eq. (2)

$$N_1(t) = \frac{\lambda_1 N_0(0)}{\lambda_2 - \lambda_0}\left[e^{-\lambda_0 t} - e^{-\lambda_2 t}\right]. \tag{A.8}$$

The time evolution of eq. (A.7) is depicted in Fig. 2. Solving $\frac{dN_1}{dt} = 0$ for t, gives the time t_{max} at which the maximum of $N_1(t)$ occurs as

$$t_{max} = \frac{1}{\lambda_2 - \lambda_0}\,\ell n\left(\frac{\lambda_2}{\lambda_0}\right) \tag{A.9}$$

and substitution of the last expression into eq. (A.7) gives the maximum amplitude of N_1 as

$$N_{1\,max} \equiv N_1(t_{max}) = N_0(0)\,\frac{\lambda_1}{\lambda_2}\left(\frac{\lambda_0}{\lambda_2}\right)^{\frac{\lambda_0}{\lambda_2 - \lambda_0}}. \tag{A.10}$$