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Individual radiation doses from unit releases of long lived radionuclides

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Studsvik Nuclear

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INDIVIDUAL RADIATION DOSES FROM UNIT RELEASES OF LONG LIVED RADIONUCLIDES

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This report concerns a study which was conducted for SKB. The conclusions and viewpoints presented in the report are those of the author(s) and do not necessarily coincide with those of the client.

Information on SKB technical reports from 1977-1978 (TR 121), 1979 (TR 79-28), 1980 (TR 80-26), 1981 (TR 81-17), 1982 (TR 82-28), 1983 (TR 83-77), 1984 (TR 85-01), 1985 (TR 85-20), 1986 (TR 86-31), 1987 (TR 87-33), 1988 (TR 88-32) and 1989 (TR 89-40) is available through SKB.

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ABSTRACT

The turn-over in a standard biosphere of radionuclides, disposed in a repository for high level waste was studied from a dose point of view. A multi-compartment model with unit releases to the biosphere was designed and solved by the BIOPATH-code. The uncertainty in the results due to the uncertainty in input parameter values were examined for all nuclides with the PRISM-system.

Adults and five year old children were exposed from 10 different exposure pathways originating from activity in well and lake water. The results given as total doses per year and Bq release (conversion factors) can be used in combination with leakage rates from the geosphere for safety analysis of a repository. The conversion factors obtained (arithmetic mean values), are given below.

Nuclide	Adults	Children
C-14	1.6E-14	1.2E-14
Se-79	1.1E-13	4.5E-14
Tc-99	2.6E-15	6.8E-16
Sn-126	5.5E-14	2.2E-14
I-129	8.3E-13	5.0E-13
Cs-135	4.7E-14	1.9E-14
Pb-210	9.6E-12	2.5E-12
Ra-225	1.3E-13	4.3E-14
Ra-226	2.6E-12	8.9E-13
Ac-227	2.4E-11	6.1E-12
Th-229	6.3E-12	1.7E-12
Th-230	1.1E-12	2.9E-13
Pa-231	1.5E-10	3.9E-11
U-233	2.4E-12	6.6E-13
U-234	2.3E-12	6.6E-13
U-235	2.2E-12	5.9E-13
U-236	2.2E-12	6.1E-13
U-238	2.1E-12	5.6E-13
Np-237	3.3E-12	8.5E-13
Pu-239	6.2E-12	1.8E-12

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1 INTRODUCTION

The biosphere constitutes the last link of transfer of nuclides from a repository to man. The dispersion in the biosphere and the resulting doses are therefore an essential part of the performance of a designed repository. This report handles the doses to individuals from unit releases to a biosphere of long-lived radioactive nuclides contained within high-level waste. The nuclides reach a standard biosphere via the ground water. This standard biosphere which constitutes of a well and lake with adjacent farming land is supposed to support a small group of individuals with the annual demand of food and fresh water.

The objectives of the work described in this report are:

- To present conversion factors between unit releases to a standard biosphere and doses to adults and children for all nuclides of radiological importance.

- To show the uncertainty in the doses predicted due to the uncertainty in parameter values.

The code BIOPATH (Bergström et al, 1982) was used for calculations of the turnover of radionuclides in the biosphere and the PRISM-system (Gardner et al, 1983) for obtaining ranges and information about parameters dominating the uncertainty.

2 BACKGROUND

The concept for Swedish disposal of high-level waste is to enclose the waste in durable canisters and place it into a deep geological formation with multiple barriers, both manmade and natural. There are two main paths from which man can be exposed from radioactive nuclides contained within such a repository. Firstly, direct exposure from intrusion into the repository may be one possibility. Secondly, leakage of nuclides from the repository to the biosphere may occur due to some initial failure of the barriers or due to degrading of them. Discussions concerning intrusion scenarios are going on internationally (NEA, 1989). It is, however, out of the scope of this report to handle the former scenarios. The emphasis in this report is instead to give a base for evaluation of the consequences to man from long-term leakages into the biosphere.

For judgement of the feasibility of nuclear energy facilities the ICRP's general ALARA principle is valid. This implies that the doses should be as low as reasonably achievable, taking account of social and economical factors. One specific problem concerning the safety assessments of the disposal of high-level waste is the long-term perspective during which considerable changes of the current ecosystem may occur. However, all radiological protection standards are based upon the dose or risk to people who have the current habits and metabolic characteristics. This is done because of the problems associated with predictions about how human beings will develop in the future. The most straight forward course of action is therefore to ensure that the consequences from waste disposal do not exceed those that would be the case today. This philosophy was adopted in the calculations shown in this report.

Besides, the evolution of the biosphere, which is impossible to predict with accuracy, there are different types of recipients for contaminated ground water reaching the biosphere. Typical such recipients are different types of water, soil, sediments and peat. Concerning water recipients earlier assessments have logically shown that releases to freshwater recipients cause higher exposure than to brackish and saline water bodies (Bergman et al, 1979). In this study a standard biosphere with the entrance of nuclides into fresh water recipients was applied for obtaining the factors describing the relation between amount of activity released and individual doses. This standard biosphere is representative for the middle part of Sweden and a basic assumption is that the unit releases occur during a period of 500 years. This is adopted as a reasonable conservative time where no major changes of the ecosystem may occur. The change of the exposure to man during the evolution of such ecosystem was studied earlier for some typical radionuclides contained within high-level waste (Sundblad et al, 1988).

Doses from the nuclides in Table 2-1 were assessed.

Table 2-1 Nuclides assessed and half-lives.

Nuclide	Half-life (years)
C-14	5 730
Se-79	6.4E4
Tc-99	2.1E5
Sn-126	1.0E5
I-129	1.6E7
Cs-135	2.3E6
Pb-210	22.3
Ra-225	14.8*
Ra-226	1 602
Ac-227	2.18
Th-229	7 340
Th-230	7.7E4
Pa-231	3.2E4
U-233	1.6E5
U-234	2.5E4
U-235	2.0E8
U-236	2.3E7
U-238	4.5E9
Np-237	2.1E6
Pu-239	2.4E4

* Given in days.

The calculations were performed for unit releases of the nuclides given in Table 2.1. These nuclides may reach the biosphere from a repository dependent on their long half-life or by generation from parent-nuclide. The turnover in the biosphere was studied with a compartment model. The differential equations were solved with the BIOPATH-code. The total dose from ten exposure pathways to adults and five year old children were calculated.

Because models are simplifications of our perception of reality their results are affected by an inherent uncertainty. This uncertainty originates from different phases in the modelling and can be divided into the following head lines:

- interpretation of the scenario
- model structure
- parameter values
- human factors

The emphasis in this work was to examine the uncertainty in the results for all nuclides studied due to the uncertainty in the parameter values describing the dilution, transport, accumulation, uptake and exposure pathways.

3 DESCRIPTION OF CODES

The mathematical method included in the BIOPATH code is based on compartment theory with first-order kinetics. Therefore, the cycling and content of radioactive matter in different ecosystems are described by a system of first-order linear differential equations with constant or time varying transfer coefficients and a number of physically defined areas or volumes (which in this report are called reservoirs or compartments).

An integrated system of computer programs, PRISM was used for the uncertainty analysis. Within the PRISM-code, sets of model parameter values are generated from given distributions of each parameter. The responses of the model are calculated for each set of those generated parameter values and the results are analysed statistically. The results show the distribution of the responses in this case doses and give valuable information about which parameters dominate the uncertainty for respective exposure pathway. For further description of the two codes see Appendix A.

4 MODEL OF ECOSYSTEM

A conceptual model was designed for the ecosystem to be studied. This conceptual model should consider all processes and transfers which are important for estimating the total exposure to man. Due to the complexity of reality this conceptual model is a simplification of our perception. Detailed description of the model and assumptions are given in the subchapters below.

4.1 INFLOW TO THE BIOSPHERE

The radionuclides reach the recipient area via the transport with groundwater. It is assumed that the nuclides, in soluble form in the water enter a lake. Before reaching the lake it is possible that a well could be located so that some of the contaminated groundwater is extracted for the demand of water for a limited number of people and their livestock. The basis for this is given in Appendix B. The part of the nuclides reaching the well was assumed to vary loguniformally between 10 % and 0.01 %. This range covers the dispersion in groundwater used in earlier assessments (Bergström, 1983).

Consequently, the major part of the nuclides reaches the lake directly. Outflow of groundwater to a lake may occur by seepage through the sediments. Most of this seepage occurs usually in the near-shore areas with an exponential decrease towards the middle of the lake. There is also a possibility that fracture zones are in direct contact with the sediments. These interface problems are addressed in several ongoing studies (BIOMOVS TR, Draft, 1990, Sundblad et al, 1990 and Ehlert et al, 1984). Changes of the chemical conditions may cause accumulation in such areas and result in a delay before nuclides reach the lake water. These effects are not considered in this study.

4.2 CRITICAL GROUPS

According to the radiological definition a critical group shall consist of a limited number of persons who can be expected to obtain higher radiation doses than average. In this case it is assumed that the critical group lives, produces and consumes all the necessary food stuff where the nuclides first reach the biosphere. This implies that the hypothetical critical group is situated in the area adjacent to the well and lake.

4.3 ECOSYSTEM

A typical current Swedish ecosystem according to the earlier discussion was studied. It is representative for the middle part of Sweden, with mostly wood-lands but containing agricultural areas. The lake is affected by the surrounding agricultural area implying that the lake will be of eutrophic character. Data for this lake were taken from the studies of lake Trobbofjärden (Evans, 1986 and Sundblad, 1986). This lake was earlier intensively studied and modelled for the purpose of estimating how the exposure to man was changed during the turnover of nuclides during the evolution of a lake. It seems to give a proper representation of most Swedish eutrophic lakes.

4.4 COMPARTMENT STRUCTURE

Application of the BIOPATH-code implies that according to the compartment theory the biosphere to be studied is divided into appropriate reservoirs or compartments. The number and structure of compartments represent a compromise between:

- a sufficiently differentiated system in order to encompass all important reservoirs and exposure pathways
- simplicity of design in order to facilitate uncertainty analyses and comparison of model predictions with measurements of turnover and elemental balance in nature or calculations using other models, and
- available information on dispersal mechanisms.

In general compartments can be designed so that they with satisfactory precision fulfil the condition of momentaneous homogeneous mixing. In nature, however, such reservoirs are often connected to areas with gradients or where the probability of leaving the reservoir may vary substantially within the reservoir, like the sediments. Better realism is then obtained by further division of such reservoirs.

<u>Detailed description</u>

In this study ten reservoirs were designed for simulation of the turnover of radionuclides important for the exposure of the critical group. A schematic description of the compartment system is given in Figure 4.1. The contaminated ground water enters the compartment system, a minor part via the water taken out from the well as the first reservoir in the model and the major part reaching the lake directly. The capacity of the well is enough to support the people and necessary livestock with fresh water as well as for irrigation of a garden plot with vegetables.

The soil constituting this garden plot is the next reservoir in the system. The garden plot supports the critical group with the annual consumption of vegetables and potatoes. The nuclides reach the soil via irrigation with water from the well.

Nuclides migrate in the soil and will with time reach the deeper part of the soil, however, with different rate due to their physical and chemical properties. A deeper soil reservoir connected to the upper one is therefore used in order to simulate the processes.

The migration of the radionuclides in the soil may also lead to transport back to the ground water. Minor amounts of the radionuclides can thus reach the well once again. This circulation, though of minor importance for the doses, is simulated in the model by the transport from local deep soil back to the well.

The water in the lake is described with only one compartment. This is of course a simplification due to inhomogeneous mixing especially during time periods when there is a stratification in the water column. However, for a shallow lake like this one and for annual average conditions this is an acceptable generalization.

The sediments in the lake are represented by two reservoirs, one upper aerated zone where bioturbation mostly occurs and one deeper with reducing conditions acting in this case as a sink for a part of the nuclides. With the drainage of the lake there is a transport out from the compartment system of the part of nuclides remaining dissolved in the water.

Further, it is assumed that the water in this lake is used for irrigation of farming land in the vicinity of the lake. This area constitutes of three reservoirs describing the plough-layer, deeper soil and groundwater respectively. This top soil consists of the soil which is irrigated with water from the lake. Similar to the processes described above the nuclides migrate to deeper layer and back to the lake. In this case this recirculation is described with two compartments, deep soil and ground water. This ground water may have connection with the initial inflow of groundwater but in this study the aim of this compartment is only to describe the delay in the recirculation in this model.

The last compartment in the recipient area is the local atmosphere over the land where the critical group lives. The transport to the atmosphere is via resuspension of dust from the two top soils and a transport out of the compartment system by the winds. No deposition is considered in this case due to the small local air volume and fast turn-over time.

In Appendix B the basis are given for how the masses of the compartments are obtained.

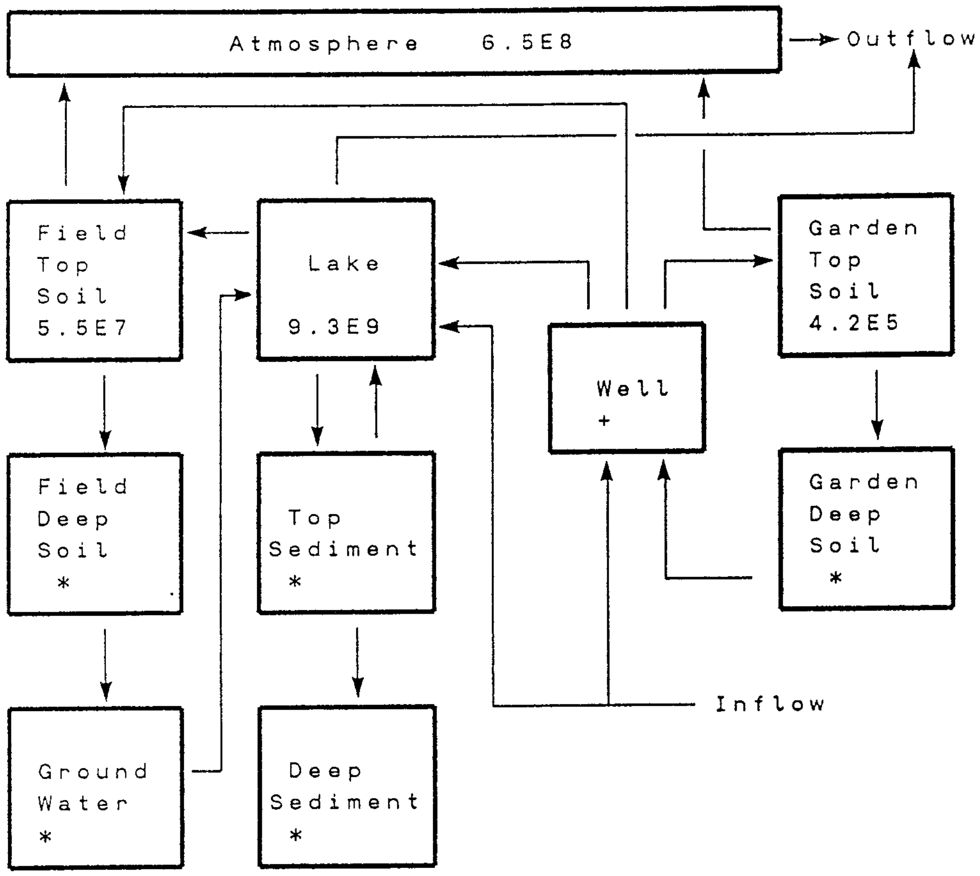

Figure 4.1 The structure of the compartment model with masses (kg).

* Not of importance for the results.

\+ The volume is varied within the calculations.

4.5 TRANSFER PROCESSES

The transfer processes considered are mathematically described as transfer coefficients or rate constants (turnover per year) in the code. There are two groups of them, namely the general and the element dependent. The general coefficients are depending on physical processes only, while the element dependent also are due to chemical processes. For the former, information about the turnover of different carriers in each media may be used for obtaining the necessary coefficients, like the turnover of water in the lake or the amount of water used for irrigation. For the latter, additional information about the solubility of each element must be considered. In general this is described by a distribution factor K_d taking into account different processes implicitly.

It was assumed that the nuclides are in soluble form in the ground water when entering the biosphere.

Nuclides in the well water are transported out via three different paths. The main transport is via the demand of water for the household of the critical group. Minor parts are taken out for irrigation of the plot and as drinking water for the live stock.

The nuclides taken out from the well by using the well water for the demands of the house-holds are transported to the lake by the flow water. It is a basic assumption that the flow water reaches the lake directly. Of course the amount of activity is somewhat reduced due to the uptake into man. However, this is not taken into account. Nuclides contained within the water consumed by cattle are transported to the fields by excretion when grazing or by manuring when stabled.

The nuclides migrate in the soil both in the garden plot and the field. Main processes are transport with infiltrating water, diffusion and bioturbation which redistribute activity in the soil. This result in a transport to deeper layers and to the ground water. This transport is dependent on environmental conditions as well as on the physical and chemical behaviour of the radionuclides.

When radionuclides reach the water in the lake depletion of the nuclides in the water will occur due to interaction with mineral and organic particles causing a transport to the sediment. This process is dependent on the chemical form of the radionuclides and the environmental conditions. In addition diffusion and bioturbation may cause a transport from the water column to the sediments. Expressions for obtaining transfer coefficients describing these processes are given in Klos et al, 1989. Investigation of this expression with data for the actual lake in this study was performed with the PRISM-system. However, the diffusion and bioturbation values were taken from the reference. Furthermore, they are about the same values as reported in Sundblad et al, 1988. The results showed that for nuclides with Kd-values greater than 1, the only transfer process of importance is the one caused by the settling of particles. If Kd is less than one some minor contribution arises from the diffusion and bioturbation process. However, it is still the Kd value which contributes most to the uncertainty. The results from Sundblad et al, 1988 support these findings.

As well as for sedimentation several processes are involved in the transport within the sediments. These will give a transport of nuclides back to the water as well as to deeper parts of the sediments. To describe these phenomena in the compartment model of the recipient the sediment is simplificated with two compartments, one upper where transport back to the water will occur and one deeper which will act as a sink for the nuclides. The processes influencing the transport of nuclides within the sediments can be considered to be more or less as external and internal processes. Mechanical resuspension as well as the growth of the sediments may transport nuclides back to the water column or to deeper situated parts, respectively. Processes within the sediments like bioturbation, advection and diffusion may redistribute the activity. The latter process is of course dependent upon the concentration difference between the pore water in the sediment and the overlying water mass.

There are several methods reported to describe these transports within the sediments (Klos et al, 1989 and Sundblad et al, 1988). The rate constants reported in these references for the transport from sediment to the water column seem to be to low for the actual type of lake and compartment system used. Within a study of the turn over of cesium from the Chernobyl fallout in lake Hillesjön (Bergström et al, 1989), which is the same type of lake as considered in this study, the transfer back to water was estimated by steady state conditions from fall-out cesium from the sixties. The results indicated a considerable large transport back to the water column. The main process is probably due to mechanical resuspension. It is well known that waves caused by wind will effect the sediment in a shallow lake.

However, in a report handling estimation of major uncertainties coupled to modelling of the turnover of Cs-137 and Ra-226 in a lake ecosystem the uncertainty in this transfer had a low impact on the resulting doses for continuous release (Bergström et al, 1984). If the transport back to the water from the sediment is mostly caused by a mechanical process the transfer rate will be the same for all nuclides. According to Klos et al the transport back to the water column is quite important for nuclides with low Kd-values while for those with high values this transport will be negligible. According to our experience the transport to the water column from the top sediment is not at al negligible for shallow lakes. In this study a term of the order of 0.1 y^{-1} was added to the formula as best estimate, see Appendix B. This term represents the mechanical transport of nuclides to the water column caused by resuspension.

Nuclides in the upper sediment are also transported to deeper sediments due to the annual growth of the sediments, diffusion and other processes. The main processes are the diffusion and the build in by the sedimentation at the top of the sediment layer. Only for nuclides with high mobility (low Kd) the diffusion is in the same range as the factor caused by the growth. The rate constant describing the resultant transfer was based upon these processes, see Appendix B for further description.

Nuclides in the water column in the lake are transferred to the soil of the farming land by irrigation. In the soil they migrate in a similar way with the process described above for the garden plot. Reaching the groundwater underlying the irrigated soil there is a feedback to the lake by the groundwater discharge to the lake.

There is also a transport to the atmosphere mostly caused by erosion and resuspension. This is described as a general process independent of the element to be studied due to the negligible difference between nuclide considered. Recycling back to the soil occurs by deposition from the atmosphere. This process is not included in the compartment system due to its minor contribution to the transport of nuclides. The air exchange due to wind results in a transport of nuclides out of the system.

In Appendix B the equations and data used for obtaining these transfer coefficients are given.

4.6 CONCENTRATION AND DISTRIBUTION FACTORS

The uptake of radioactive nuclides in biota from the surrounding media, soil and water etc are dependent on many factors. For example, the uptake in plants can be quite different from sandy and clayey soil and is also dependent on the chemical conditions in the soil. For essential elements the uptake in fish may differ due to tropical level of the fish and type of lake. In general, the uptake is higher for oligotrophic lakes than for eutrophic lakes.

This uptake is for steady-state conditions expressed as a bioaccumulation factor or concentration factor. It is an empirical ratio of nuclide concentration in the tissue of an organism to that in the connected media such as water or soil. Many authors have derived these factors for dry plant material to reduce the variable caused by the water content. However, in these calculations the dry weight values are used only for the cattle's consumption. For those vegetable products eaten directly by man the fresh weight ratios were used.

A literature survey was performed in order to cover the most recent literature concerning the behaviour of the nuclides handled. However, the survey showed in many cases quite contradictionary results, e.g. bioaccumulation factors to fish varying from less than one up to a thousands. It was out of the scope of this study to review all data carefully, some differences could be due to filtered or unfiltered water or experimental studies compared to field observations. The tendency in the selection of values was to bias the values to the upper values found in order to not underestimate the possible exposure. However, it seems very doubtful that bone seeking nuclides like thorium and uranium should have uptake factors ranging up to 1000 which was found as recommended in references addressing input parameter values to dose assessment studies.

The concentration factors found in the literature are rather general, although in some cases they are related to special conditions in the environment or results from specific experiments. All the previously mentioned factors introduce a large variation between data from different sources.

Despite all these variations a reasonable value for the concentration factors of the most important food items, as well as for the pasturage for cows can be given. All the values used in this report are given in Appendix C, Table C.1 for vegetation and Table C.2 for fish, with comments about how they were selected and references.

To describe the transfer from animal feed to animal products a distribution factor is used. This factor gives the fraction of the daily intake of nuclide which at steady state is likely to be found per litre of milk or per kilo of beef. The distribution factors used in this report are given in Appendix C, Table C.3.

4.7 EXPOSURE PATHWAYS

Man receives radiation doses via different paths of exposure. The exposure pathways can be grouped into external and internal exposure. The latter is caused by inhalation and ingestion of nuclides. Internal exposure from food can take place via a number of links in the ecological transport chain such as

- uptake in crops via root uptake
- uptake via the food chain of grass-meat, grass-milk
- uptake in fish or other marine foodstuffs from surrounding water

In this study the doses are calculated separately to adults and children. The same exposure pathways for children and adults are considered. They are shown in Figure 4.2. Children representing an age of five years were chosen.

The pathways for ingestion of nuclides are via different types of food and drinking water. The intake of soil is also included. This latter pathway is valid especially for children where some ingestion may occur during playing. This intake is also valid for adults via e.g. consuming unwashed vegetables. For children this intake is adopted to 20 g/y (Kimbraugh et al 1984) and by adults to 10 g/y. The main part of the soil is from the garden plot.

Following pathways were considered for ingestion of nuclides:

- Water
- Milk
- Meat
- Vegetables
- Root vegetables (potatoes)
- Cereals
- Fish
- Soil

Consumption data used for adults and children are given in Appendix C Tables C.4 and C.5.

Earlier calculations of the doses from these long-lived nuclides showed that the internal exposure dominates the exposure for the nuclides considered.

The only external exposure considered is from ground. This represents staying on the fields and the garden plots. Other external exposure pathways can be neglected for this type of recipient area according to safety studies previously performed (Bergström, 1983). Time of external exposure was adopted to 500 hours per year for adults and 1500 hours for children. Adults are exposed during agricultural work, which has been assumed to be as average 300 and 200 hours per year from the garden plot and fields respectively. Children are exposed during playing which was assumed to occur as average during 500 and 1000 hours per year from the fields and garden plot respectively.

The expsoure via inhalation is through inhalation of nuclides in the air originating from the dust from the soil. The model gives the mean concentration of the nuclides in the air during the year. The critical group will be indoors and outdoors during the year. For adults the time for outdoors is set to 1000 hours while for children the time is set to 1500 hours. The rest of the time during the year the critical group will be indoors. The nuclides in the air will be associated with particles leading to a reduction of the amount through filtration of the air indoors. This filtration is valid even in houses with natural ventilation. The reduction of activity concentration in indoor air is adopted to be 30 % of the outdoor concentration (Tveten U, 1985).

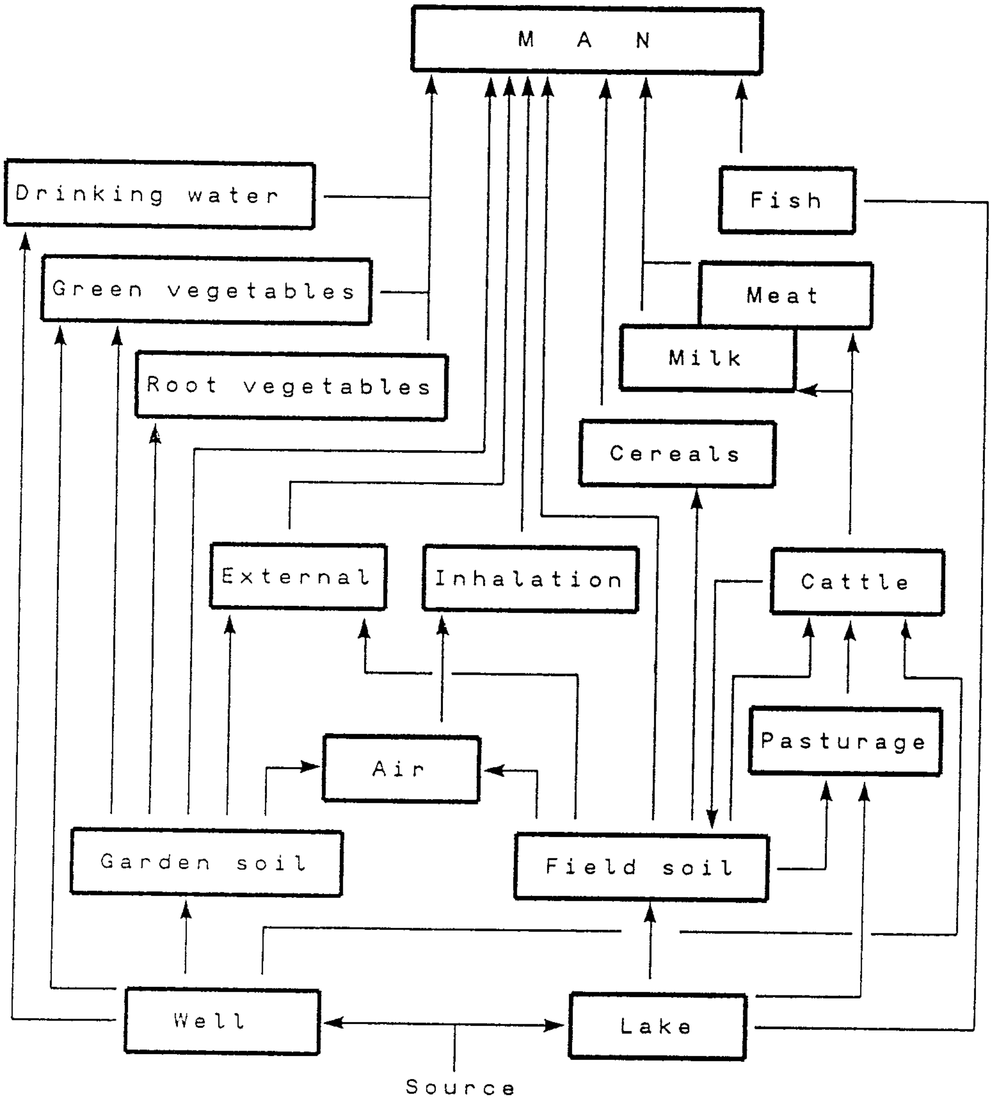

Figure 4-2 Exposure pathways for the critical group.

4.8 DOSE FACTORS

In order to transform the intake of nuclides to radiation dose, a dose conversion factor is used. This factor is dependent on the exposure situation, the decay energy of the nuclide, type of radiation, the risk of biological effects and turnover in the body.

ICRP has published ALI-values and weighted committed organ dose equivalents (ICRP30) for workers and in ICRP48 for population exposure from plutonium. However, those former values are not recommended to be used for members of the public. Johansson 1982 and 1984 performed a revision of some nuclides with respect to a chronic oral intake by members of the public. A report concerning age dependent dose factors to members of the public from ingestion is under press from ICRP. Dose factors were taken, when possible, from a draft of this report. For nuclides not handled the values from ICRP30, ICRP48, Johansson 1982 and 1984 were adopted. In general the dose factors to the public are somewhat lower than those for the workers.

The dose factors are summarised in Table 4-1 and 4-2 for inhalation and ingestion respectively. For comparison both the values from ICRP30, ICRP48, Johansson 1982 and 1984 and ICRP Draft are given. The factors used in the calculations are underlined in the table. For those nuclides not included in the draft the same dose factor was used for children as for adults.

The external dose from top soil is calculated with the dose conversion factors from Svensson 1979 with assumption of homogeneous distribution of the activity in soil. The compartment of top soil represents a depth of 30 cm. The contribution to the external dose from activity in deeper soil is only a minor part due to the absorption of the radiation in the soil. These dose factors are independent of age.

Table 4-1 Dose factors as the sum of weighted commited organ dose equivalents (Sv/Bq) for intake via inhalation for adults and children of 5 years' age.

Nuclide	ICRP30/48	ICRP DRAFT Adults	ICRP DRAFT Children
C-14	5.7E-10	5.6E-10	9.5E-10
Se-79	2.4E-09		
Tc-99	2.0E-09		
Sn-126	2.3E-08		
I-129	4.7E-08	4.0E-08	6.6E-08
Cs-135	1.2E-09		
Pb-210	3.4E-06		
Ra-225	2.0E-06		
Ra-226	2.1E-06		
Ac-227	1.8E-03		
Th-229	5.7E-04		
Th-230	8.6E-05		
Pa-231	3.4E-04		
U-233	3.6E-05		
U-234	3.6E-05		
U-235	3.3E-05		
U-236	3.4E-05		
U-238	3.2E-05		
Np-237	1.3E-04	5.5E-05	5.6E-05
Pu-239	1.4E-04	1.2E-04	1.4E-04

Table 4-2 Dose factors as the sum of weighted commited organ dose equivalents (Sv/Bq) for intake via ingestion for adults and children of 5 years' age.

Nuclide	ICRP30/48	Joh 82, 84	ICRP DRAFT Adults	ICRP DRAFT Children
C-14	5.7E-10		5.6E-10	9.5E-10
Se-79	2.3E-09			
Tc-99	3.4E-10	3.4E-10		
Sn-126	4.7E-09			
I-129	7.4E-08	9.8E-08	6.4E-08	1.0E-07
Cs-135	1.9E-09			
Pb-210	1.4E-06	1.8E-06		
Ra-225	3.1E-07			
Ra-226	3.1E-07	3.3E-07		
Ac-227	3.8E-06			
Th-229	9.4E-07			
Th-230	1.5E-07	1.6E-07		
Pa-231	2.9E-06	2.2E-05		
U-233	7.4E-07	3.1E-7*		
U-234	7.2E-07	3.0E-7*		
U-235	6.6E-07	2.8E-7		
U-236	6.9E-07	2.9E-7*		
U-238	6.4E-07	2.7E-7		
Np-237	1.1E-05	1.2E-06	4.5E-07	4.3E-07
Pu-239	1.2E-07	7.0E-07	9.7E-07	1.1E-06

* Based upon values for U-235 and U-238 given in Johansson, 1984.

5 RESULTS

Results, as arithmetic mean values of total dose are presented in Table 5-1. The results for C-14 are obtained from a simplified version of the model described in Hesböl et al, 1989.

The ranges of the total dose corresponding to 95 % of all values are presented graphically, see Figures 5-1 and 5-2. In the figures the median values are also pointed out as well as the 25th and 75th percentiles.

The percentual contributions to the total dose from dominant exposure pathways are given in Table 5-2. In the table the percentual contribution to the total uncertainty from the respective exposure pathway is given within brackets.

In Figures 5-3 to 5.6 the total dose for unit releases to the biosphere is plotted against the percentual part of nuclides reaching the well. This is shown for four nuclides which can be judged to be illustrative for different behaviour in the biosphere.

Table 5-1 Individual doses from unit releases to adults and 5 years old children, as arithmetic mean values (Sv/Bq).

Nuclide	Adults	Children
C-14	1.6E-14	1.2E-14
Se-79	1.1E-13	4.5E-14
Tc-99	2.6E-15	6.8E-16
Sn-126	5.5E-14	2.2E-14
I-129	8.3E-13	5.0E-13
Cs-135	4.7E-14	1.9E-14
Pb-210	9.6E-12	2.5E-12
Ra-225	1.3E-13	4.3E-14
Ra-226	2.6E-12	8.9E-13
Ac-227	2.4E-11	6.1E-12
Th-229	6.3E-12	1.7E-12
Th-230	1.1E-12	2.9E-13
Pa-231	1.5E-10	3.9E-11
U-233	2.4E-12	6.6E-13
U-234	2.3E-12	6.6E-13
U-235	2.2E-12	5.9E-13
U-236	2.2E-12	6.1E-13
U-238	2.1E-12	5.6E-13
Np-237	3.3E-12	8.5E-13
Pu-239	6.2E-12	1.8E-12

Table 5-2 Percentual contribution from dominant exposure pathways to total dose and, within brackets, the uncertainty.

Nuclide	Drinking water	Milk	Meat	Vegetables	Potatoes	Fish
C-14	20(38)	9(20)	4(9)	4(7)		64(26)
Se-79	13(4)	1		25(28)	51(67)	10
Tc-99	81(80)		1(1)	16(17)	1(1)	1
Sn-126	48(41)	5(5)		19(27)	15(25)	12
I-129	42(27)	20(14)	1(1)	24(50)	5(7)	
Cs-135	22(15)	5(3)	7(8)	8(7)	6(5)	51(63)
Pb-210	78(80)	1(1)		14(18)		6
Ra-225	69(61)	15(23)		13(15)		2
Ra-226	64(61)	10(9)		16(17)	8(12)	1
Ac-227	83(85)			14(15)		2
Th-229	79(78)			16(17)	3(4)	
Th-230	77(72)			19(24)	3(3)	
Pa-231	80(74)			17(23)	2(2)	
U-233	76(74)	1(1)	7(7)	15(18)		1
U-234	75(72)	1(1)	8(8)	15(18)		1
U-235	75(72)	1(1)	8(8)	15(18)		1
U-236	76(76)	1(1)	8(8)	14(15)		1
U-238	74(75)	1(1)	8(9)	15(17)		1
Np-237	79(78)		1(1)	16(17)	3(3)	1
Pu-239	84(81)			16(18)		

A continuation of this study is going on where one of the main objectives will be a detailed discussion of the results presented here.

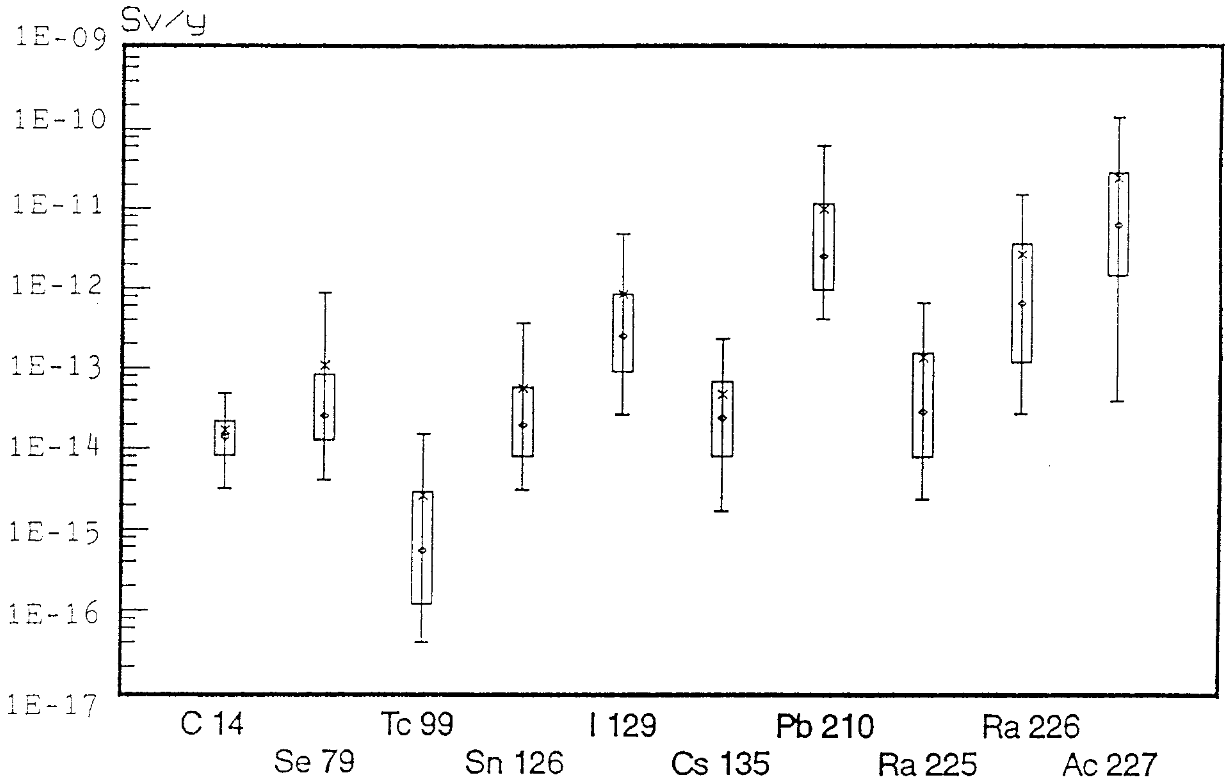

Figure 5-1 Ranges of total dose for adults corresponding to 95 % of all values obtained.

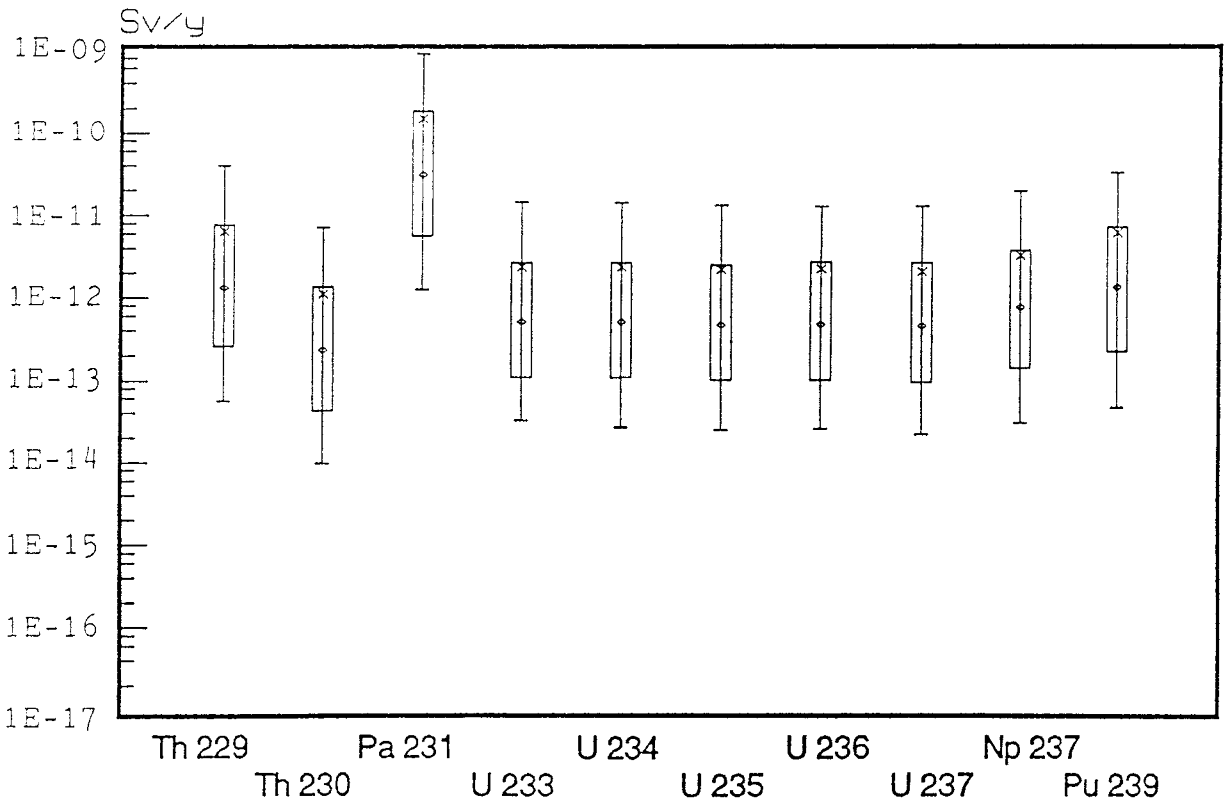

Figure 5-2 Ranges of total dose for adults corresponding to 95 % of all values obtained.

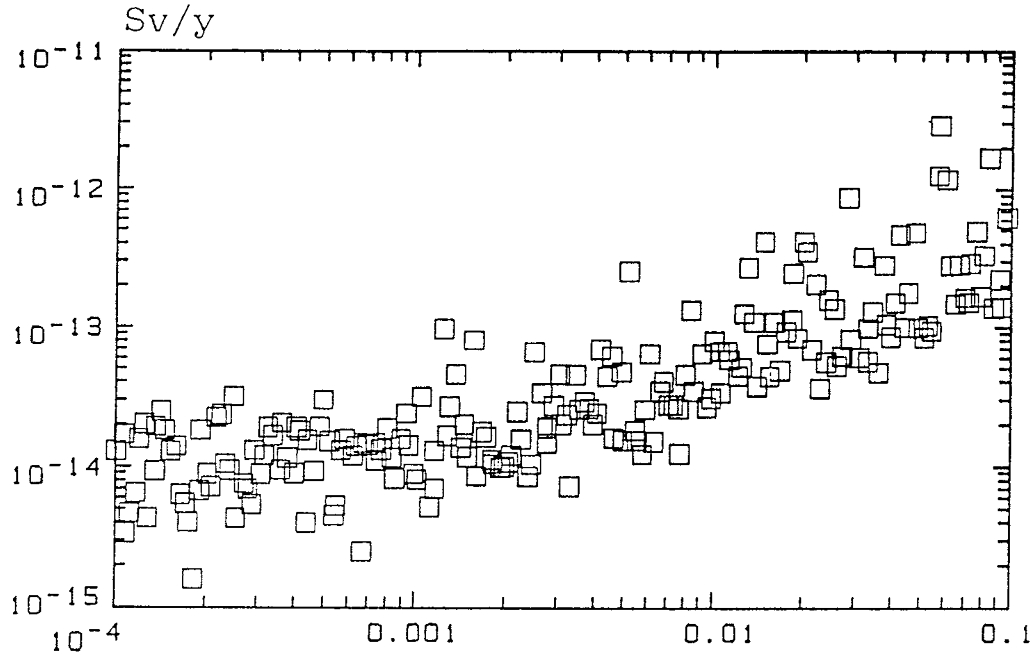

Figure 5-3 Total dose for unit release as a function of part of inflow via the well for Se-79.

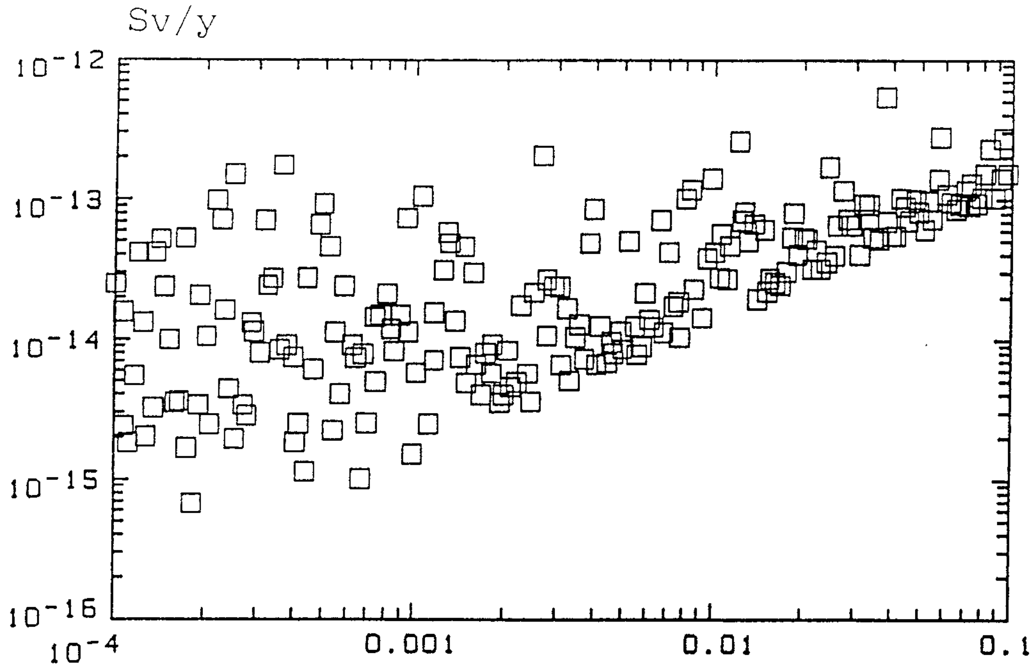

Figure 5.4 Total dose as a function part of inflow via the well for unit release for Cs-135.

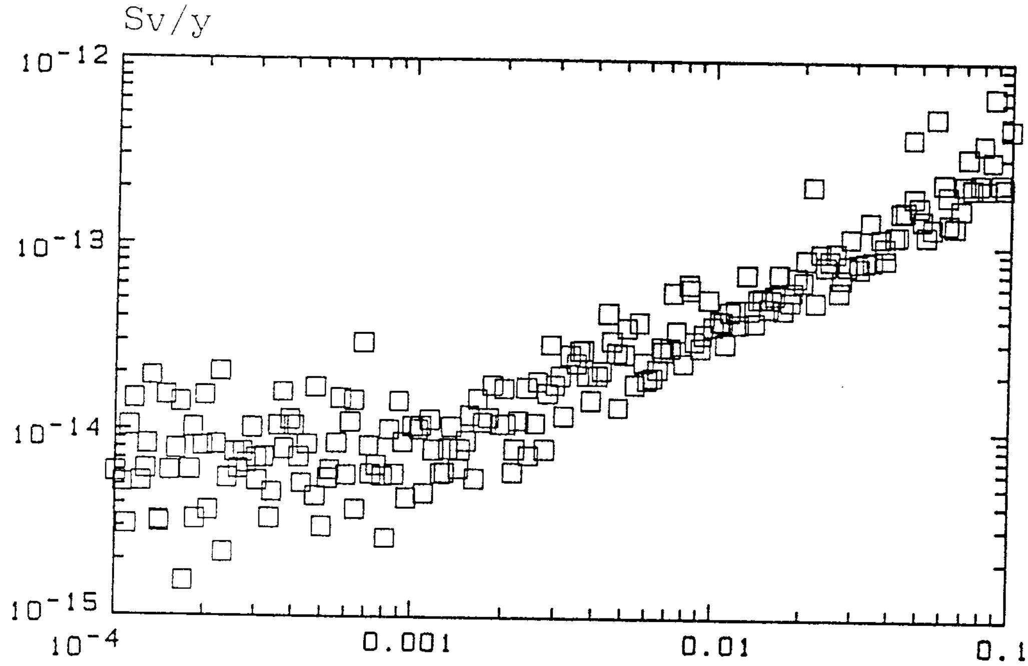

Figure 5-5 Total dose as a function of part of inflow via the well for unit release for Sn-126.

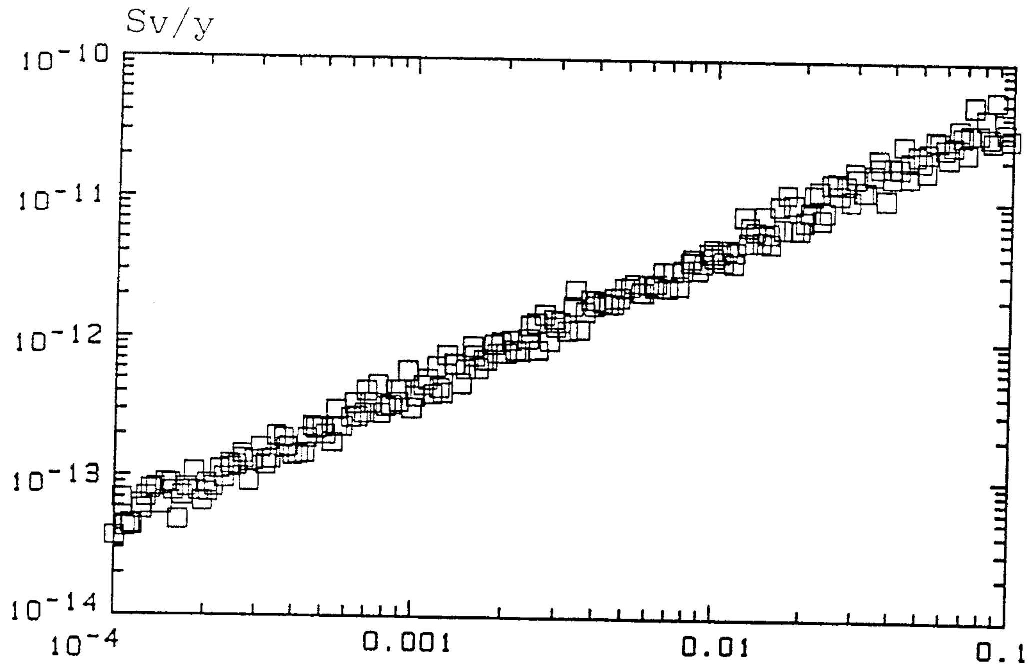

Figure 5-6 Total dose as a function of part of inflow via the well for unit release for Pu-239.

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BIOPATH AND PRISM CODES

BIOPATH-code

The mathematical method included in the BIOPATH code is based on compartment theory with first-order kinetics. Therefore, the cycling and content of radioactive matter in different ecosystems are described by a system of first-order linear differential equations with constant or time varying transfer coefficients and a number of physically defined areas or volumes (which in this report are called reservoirs or compartments).

The premises are that:

- the outflow for reservoir "j" is solely dependent upon the quantity, y_j of the radionuclide in that reservoir,
- the reservoir is instantaneously well mixed,
- all atoms, molecules or other elementary units have the same probability of leaving the reservoir

The amount of activity in a given reservoir is dependent on:

1) the outflow to and inflow from other reservoirs,
2) the source term for the reservoir, such as release to the reservoir or generation within it by decay.
3) radioactive decay

This is expressed mathematically in vector form by:

$$\dot{Y}_M(t) = K_M Y_M(t) + Q_M(t) - \lambda_M Y_M(t)$$

for parent nuclides and

$$\dot{Y}_{D_n}(t) = K_{D_n} Y_{D_n}(t) + \lambda_{D_n} Y_{D_{n-1}}(t) -$$

$$- \lambda_{D_n} Y_{D_n}(t) + Q_{D_n}(t)$$

for daughter nuclides.

The vectors Y and $\dot{Y}$ refer to activity and activity changes per unit time in the different reservoirs of the system at time t. The coefficient matrices K ($year^{-1}$) and Q(t) (activity $year^{-1}$) describe the transfer rates between the reservoirs and source-term to the reservoir, respectively. The decay constant is

$$\lambda = \ln 2 / t_{1/2}$$

where $t_{1/2}$ is the physical half-life of the nuclide.

PRISM-code

The PRISM-system consists of three main parts. Firstly in PRISM 1 random model parameters are generated by using a systematic sampling method, Latin Hyper cube. As input to PRISM 1, the mean values, type of distributions, standard deviations and upper and lower limits must be given for each parameter. These data are used to define probability density functions. The Latin Hyper cube method, used to generate sets of parameter values from the given distributions, is an efficient Monte Carlo sampling which results in random figures within the whole desired range of parameter values. In addition correlations between model parameters can be taken into account no matter what distributions they are drawn from.

Secondly, in PRISM 2 model predictions, in this report dose calculations are made for each set of parameter values.

Finally PRISM 3 statistically evaluates and summarises the joint set of model parameters and predictions.

The general statistics for the distribution of each parameter and the response of the model to this distribution contain the following:

- the arithmetic mean
- the standard deviation
- the coefficient of variation
- the geometric mean
- the percentiles (5, 25, 50, 75, and 95 %)
- the five highest values and the five lowest values, respectively

Correlations between the model parameters and responses as well as between responses are also obtained from this last part of the analysis. Two correlations coefficients are calculated: firstly, the simple Pearson correlation coefficient and secondly Spearman R which is the correlation of the ranked values of the parameters and model responses. Associated with each correlation coefficient is percent COVAR. This represents the percent variance that one variable accounts for in another variable or response. In cases of correlated model parameters and responses, percent COVAR indicates the amount of variability of the model response explained by the variability of that model parameter.

The regression procedures are used to obtain the relationship between model parameters and model uncertainties.

The selection of parameters to be entered into the regression analysis is chosen among those which have the greatest improvement in the sum of squares of regression. Default values are given in the code but may be changed if chosen.

The output from PRISM 3 consists of all statistic information (intercept, slopes and mean values) required to write the regression equation for the relationships between model parameters and response. In these calculations the relative contribution to the total uncertainty from each parameter is also obtained. In this report the total dose received is obtained from summation via seven food pathways. This means that simple analytical equations can be used to characterise the relationships between total doses and each exposure pathway.

RESERVOIR MASSES AND TRANSFER COEFFICIENTS

Masses of importances for the results

The structure of the model implies that the masses of the water and upper soil-reservoirs are a part of the dose calculations. Other reservoirs are mostly important for the turnover of nuclides e.g. the sediment and deep soil reservoirs. Because of that we have emphasised upon the former masses. The actual dispersion in groundwater is of major importance for the doses. This is related to the use of groundwater from a well located in the area. The masses of all important reservoirs with the exception for the lake are based upon a common basic assumption. This assumption is that the masses should be of enough sizes to support a critical group consisting of 25 persons with a livestock of 8 cattles with their annual demand of water and food-stuff. The live-stock will feed the critical group with milk and meat. With this as a base the following values were adopted as best estimate values:

- annual demand of water for man 50 m^3 per individual
- annual demand of drinking water for the live-stock 250 m^3
- annual irrigation of the garden plot with 150 l per m^2
- average yield of cereals 0.4 kg/m^2 (Eriksson, 1983)
- daily consumption for cattle 14 kg dry weight
- average yield of fodder including grazing 0.3 kg dw/m^2
- porosity of upper soil 0.44
- density of upper soil particles 2.5 E3 kg/m^3

As described earlier all these values were varied within the calculations. The data are summarised in Table B.1. However, the average volume for the well will be about 1600 m^3/year. According to the yield values the garden plot and the agricultural area needs to have an area of 1000 m^2 and 0.13 km^2, respectively, to be able to feed the critical group with the terrestrial foodstuffs. Consequently, with the assumption of the top soil as the upper 30 cm, the masses will be 4.2E5 and 5.5E7 kg d w soil for the reservoirs respectively. In Appendix C, Tables C4 and C5 the consumption data are given.

The volume of the lake is adopted to 9.3E6 m^3, the mean depth to 3.1 m and the residence time of water 0.75 year. These values will represent a "normal" lake in the middle of Sweden with eutrophic characteristics. The data on the lake correspond to the lake Trobbofjärden which was investigated in studies previously performed (B Sundblad, 1988).

Finally, there is a compartment for the atmosphere over the area. The area is set to 1 km^2 and the height to 500 m with an air density of 1.293 kg/m^2 the mass will be 6.5E8 kg.

Derivation of rate constants

The rate constants in this study are given in units of turnover per year which is appropriate for the scenario studied. As mentioned in the main text there are two groups of rate constants; the general independent of nuclide and those who are related to the element specific behaviour of the nuclides.

All the rate constants are calculated within the Prism-system. Data used for the general rate constants are summarised in Table B.1. The expressions used for obtaining the rate constants are given below.

Table B.1

Parameter values for obtaining the masses of important reservoirs and the general rate constants.

Parameter	B.E.	Type of distr*	Min	Max
Annual demand of water for man (m^3)	50	T	30	70
Annual demand of water for the live-stock (m^3)	250	T	200	270
Irrigation of the garden plot (l/m^2 year)	150	T	50	200
Irrigation of agri-cultural land (l/m^2 year)	150	T	50	200
Daily consumption of foodstuff for cattle (kg d w)	14	T	12	16
Residence time of water in the lake (year)	0.75	T	0.6	0.9
Fraction of acti-vity reaching the well (%)	1	LU	0.01	10
Turnover time of groundwater (year)	5	T	1	10
Area of garden plot (m^2)	1000	C		
Area of agri-cultural land (m^2)	130000	C		

* T = Triangular distribution.
LU = Loguniform distribution.
C = Constant.

Element independent rate constants

Well to garden plot:

The water from the well is used for irrigation. The best estimate irrigation rate is set to 150 mm/y and the area of the plot is 1000 m^2, this will give the transfer factor TF:

$$TF_{wg} = \frac{0.15 \cdot 1000}{TWW} \simeq 0.07\ y^{-1}$$

Where

TWW = Total amount of Water taken out from the Well

Well to agricultural soil:

It is assumed that the cattles take their drinking-water from the well. The excrement from these cattles may reach the agricultural area due to direct excretion or by manuring when stabled. The uptake into GI tract is low for most of the nuclides implying no considerable reduction of the amount of activity leaving the cattle from the intake. In addition crop rotation is assumed which is a proper assumption due to the timespan studied.

The total demand of water by the livestock is as best estimate 250 m^3/y. This gives:

$$TF_{ws} = \frac{250}{TWW} \simeq 0.1\ y^{-1}$$

Well to lake:
The flow water from the critical group is led to the lake and only a neglectable part of the original content of nuclides in the groundwater will in the end not reach the lake. This will give a rate constant factor from the well to the lake as below:

$$TF_{wl} = \frac{25 \cdot 50}{TWW} \simeq 0.8 \ y^{-1}$$

Lake to agricultural soil:
This soil is irrigated with 150 mm/y and the area of this land is 0.13 km^2. The volume of the lake is 9.3E6 m^3, this will give:

$$TF_{la} = \frac{0.15 \cdot 1.3E5}{9.3E6} \simeq 2.1E^{-3} \ y^{-1}$$

Lake to outflow:
The mean residence time for the reference lake Trobbofjärden is 0.75 year. This will give:

$$TF_{lo} = \frac{1}{0.75} \simeq 1.33 \ y^{-1}$$

Soil to atmosphere:
This transfer is mainly dependent on resuspension. The process is among other things dependent on weather situation and type of soil and vegetation. It is well recognized that understanding of the process involved and pertinent data are not sufficient. The transfer factor adopted is:

$$TF_{sa} = 1.0E\text{-}5 \ y^{-1}$$

Local to global atmosphere:
The cross section of the local atmosphere is 5.0E5 m^2 assuming a mean wind speed of 5 m/s this rate constant will be:

$$TF_{ag} = \frac{5E5 \cdot 1.6E8}{5E8} = 1.6E5\ y^{-1}$$

Element dependent rate constants

Rate constants describing the transport of nuclides from water to sediment, within the sediments and the migration in the soil and groundwater are dependent, among other things on the solubility of the element. The transfer in the lake water to the sediment and the transport within these are adopted from Klos et al 1989 in relevant parts. The processes considered are settling of particles, diffusion, bioturbation and the growth of the sediments.

The transfer factors for the migration in soil were estimated from physical-chemical data on sorption and water transport etc in the same way as used in (Bergström et al 1983). A brief description of the methods and data used are given below.

Water-sediment:

For obtaining the transfer coefficient describing the transport of radionuclides from water to sediment, the following equation was used:

$$TF_{ws} = K_d \cdot S\ [h_m \cdot (1 + K_d \cdot SS)]^{-1} +$$

$$+ \frac{1}{R} \cdot \frac{D}{h_m \cdot h_s} + \frac{R-1}{R} \cdot \frac{B}{h_m \cdot h_s}\ y^{-1}$$

where

TF_{ws}	= transfer coefficient water-sediment [y^{-1}]
K_d	= distribution coefficient water, sediment = concentration in solid/concentration in liquid [m^3/kg]
S	= sediment growth rate [kg/m^2, y]
h_m	= mean depth of lake [m]
h_s	= mean depth of top sediment [m]
SS	= concentration of suspended matter in the water [kg/m^3]
D	= sediment diffusion coefficient [m^2/y]
B	= bioturbation coefficient [m^2/y]
R	$= 1 + (1-\varepsilon)\,\frac{S_p}{\varepsilon}\,K_d$
ε	= porosity of the sediment
S_p	= density of sediment particle [kg/m^3]

The following nuclide-independent data are valid for lake Trobbofjärden:

	Best estimate	Type of distr*	Min	Max
S	$= 2.4\ kg/m^2, y$	T	0.1	6.6
h_m	$= 3.1\ m$		-	-
h_s	$= 0.1\ m$		-	-
SS	$= 0.01\ kg/m^3$	T	0.005	0.02
ε	$= 0.9$	T	0.8	0.95
S_p	$= 2.5 \cdot 10^3\ kg/m^3$	T	1.5 E3	3.0 E3
D	$= 0.03\ m^2/y$	LT	3E-3	3E-1
B	$= 3E-5\ m^2/y$	LT	3E-6	3E-4

* T = Triangular distribution.
LT = Logtriangular distribution.

The value of K_d is given in Table B.3 for each element.

Sediment to lake water:
This process is complex with several mechanisms for transfer both physical, chemical and biological. According to studies performed for the same type of lake the main process for the transport is indicated to be resuspension of sediment (Bergström et al 1989). The assumption is that nuclides in the sediment will be transported to the water phase due to mechanical effects from water flows in the lake, these effects are not included in the equations according to Klos et al 1989. This mechanical effect is only valid in

shallow lakes as Trobbofjärden. In this case a constant was added to the equation to describe this phenomenon. The rate constant is expressed as:

$$TF_{sw} = K + \frac{1}{R} \cdot \frac{D}{h_s^2} + \frac{R-1}{R} \cdot \frac{h_m \cdot B}{h_s^2} \; y^{-1}$$

where

K = mechanical resuspension

The explanation of the other parameters and values used are given above.

Value of K, logtriangular distributed

mean = 0.1 min = 0.01 max = 0.2

Top to deep sediment:
The transfer from the top sediment to the deeper sediment is given by:

$$TF_{sd} = \frac{R-1}{R} \cdot \frac{S}{h_s(1-\varepsilon) \cdot S_p} + \frac{1}{R} \cdot \frac{D}{h_s^2} \; y^{-1}$$

where the parameters are explained above.

Migration in soils:

The transfer coefficients describing the leakage from soil compartments were obtained from:

$$TF_s = \frac{U_w}{R \;\; h_i} = \frac{U_w}{h_i} \left(1 + K_d \cdot S_p \frac{(1 - \varepsilon_i)}{\varepsilon_i}\right)^{-1} \; y^{-1}$$

where

TF_s	=	transfer coefficient $[y^{-1}]$
U_w	=	water velocity through compartment $[m/y]$
h_i	=	depth of compartment i $[m]$
R	=	retention = water velocity/nuclide velocity
ε_i	=	porosity of soil i $[m^3/m^3]$
K_d	=	distribution coefficient $[m^3/kg]$
S_p	=	density of soil particles $[kg/m^3]$

The following nuclide-independent data, triangularly distributed were used:

	Best estimate	min	Max
U_w	= 3 m/y	1	10
h_1	= 0.3 m for the top soil	-	-
h_2	= 1.7 m for the deep soil	-	-
S_p	= $2.5 \cdot 10^3$ kg/m^3	1.5 E3	3.0 E3
ε_1	= 0.44 for the top soil	0.2	0.6
ε_2	= 0.2 for the deep soil	0.1	0.6

The value of K_d is given in Table B.1 for each element.

Groundwater to surface water:

The transfer coefficient describing the leakage from the groundwater compartment was obtained from:

$$TF_g = \frac{1}{RT} = \frac{1}{T}\left(1 + K_d \cdot S_p \frac{(1-\varepsilon)}{\varepsilon}\right)^{-1} y^{-1}$$

where

TF_g	=	transfer coefficient groundwater - lake $[y^{-1}]$
T	=	residence time for the groundwater in the actual compartment [y]

For other notations, see previous equation.

The following nuclide independent data were used:

	Best estimate	Min	Max
T	= 5 years	1	10

The other parameters are set to the same as for deep soil.

Table B.2

Distribution coefficients [m^3/kg] in soil,log-triangularly distributed.

Nuclide	Best estimate	Min	Max	Ref
C	0.001	0.0001	0.01	3
Se	0.01	0.001	1	1
Tc	0.005	0.001	0.01	2
Sn	0.1	0.05	0.5	4
I	0.3	0.1	1	2
Cs	1	0.1	10	1
Pb	0.1	0.01	0.01	4
Ra	0.5	0.01	0.01	5
Ac	1	0.1	0.1	4
Th	10	1	100	6
Pa	10	1	100	2
U	0.1	0.01	1	6
Np	0.1	0.01	1	2
Pu	50	1	100	3

Ref
1) Coughtrey et al, 1985
2) Bergström et al, 1983
3) Bergström et al, 1987
4) Bergström et al, 1985
5) Bergström et al, 1984
6) Grogan et al, 1985

Table B.3

Distribution coefficients [m^3/kg], sediment-lake water, log-triangularly distributed.

Nuclide	Best estimate	min	max	ref
C	0.001	0.0001	0.01	
Se	5	1	10	1
Tc	0.1	0.01	1	1
Sn	50	10	100	1
I	0.3	0.1	1	1
Cs	10	1	100	1
Pb	0.05	0.01	0.1	2
Ra	10	1	100	3
Ac	10	1	100	2
Th	100	10	1000	4
Pa	100	10	1000	4
U	10	1	100	4
Np	10	1	100	1
Pu	100	10	1000	1

Ref
1) Coughtrey et al, 1985
2) Bergström et al, 1985
3) Bergström et al, 1984
4) Bergström et al, 1983

EXPOSURE PATHWAYS

The exposure pathways are calculated based upon the concentrations in the different reservoirs. For the internal exposure from ingestion the following expression was used:

$$D_i = HC_i \cdot U_i \cdot D_{fg}$$

where

D_i = the dose (Sv/y) from exposure pathway i

HC_i = the consumption rate for intake via pathway i

U_i = the concentration in foodstuff i (Bq/kg)

D_{fg} = the dose factor for ingestion as weighted total body dose equivalent commitment (Sv/Bq)

How these concentrations due to uptake in the different foodchains are obtained is given below.

For the internal exposure from inhalation the following expression was used:

$$D_a = (T_o \cdot HA + T_i \cdot HA \cdot F) \cdot C_a \cdot D_{fa}$$

where

D_a = the dose (Sv/y) from inhalation

T_o = the time for being outside (h/year)

T_i = the time for being inside (h/year)

HA = the inhalation rate (m^3/h)

F = the filter factor

C_a = the concentration in air (Bq/m^3)

D_{fa} = the dose factor for inhalation as weighted total body dose equivalent commitment (Sv/Bq)

The external exposure was calculated as follows:

$$D_e = T_j \cdot C_j \cdot D_{fc}$$

where

D_e = the dose from external exposure (Sv/y)

T_j = the time (hours/year) for staying on the garden plot and field respectively

C_j = the concentration in soil (Bq/m^3) for the garden plot and field respectively

D_{fc} = the dose convection factor $(Sv/h)/(Bq/m^3)$

Uptake in milk and meat

Radioactive elements in meat, U_k, and milk, U_m, originating from uptake via the following ecological paths of transport:

- Root uptake to pasturage
- Retention on pasturage from irrigation water
- Drinking water
- Intake of soil during grazing

The total intake of nuclides with fodder and water are In

Thus,

$$In \text{ (in Bq per day)} =$$

$$= (MC_p \cdot B_p + MC_s) \cdot C_s + MC_l \cdot C_w + MC_p \cdot K \cdot C_l$$

Then

$$U_m \text{ (in Bq per litre)} = F_m \cdot In$$

and

$$U_k \text{ (in Bq per kg)} = F_k \cdot In$$

Uptake in green vegetables

The concentration of radioactive elements in green vegetables, U_v, originates from two sources: the uptake of radioactivity via the root system, and deposition directly on the surfaces of the leaves at irrigation.

Thus,

$$U_v \text{ (in Bq per kg)} = B_v \cdot C_s + MI \cdot R \cdot IRR \cdot C_w$$

Uptake in grain and root vegetables

Uptake in grain, U_e, and root vegetables, U_r is assumed to take place primarily through the root system.

Thus,

$$U_g \text{ (in Bq per kg)} = B_g \cdot C_s,$$

and

$$U_r \text{ (in Bq per kg)} = B_r \cdot C_s.$$

Uptake in fish

Uptake in fish, U_f, is assumed to be directly proportional to the concentration in the water of the nuclides.

Thus,

$$U_f \text{ (in Bq per kg)} = B_f \cdot C_w.$$

Where:

U_i = Uptake of one particular nuclide in food-stuff <u>i</u>. Given in Bq per unit of food (kg, litre or piece).

i = m milk
k meat
v green vegetables
g grain
r root vegetables
f fish
p pasturage
s soil

B_n = Concentration factor of a certain nuclide for uptake via pathway n (Table C.1)

n = p soil → pasturage
v soil → green vegetables
g soil → grain
r soil → root vegetables
f water → fish

F_m = Distribution factor for a given nuclide for food-stuff <u>i</u>. Given in day per unit of food (kg or litre) (Table C.2)

m = m milk
k meat

C_j = Concentration of a certain nuclide in reservoir j. Given in Bq per unit of reservoir

j = w well water
l lake water
a air
g garden soil
f field soil

MC_k = Daily consumption per individual of water, food-stuff (dry weight) and soil for for livestock. (Table C.5)

k = w water
p pasturage
s soil

MI = Mass interception factor vegetables (m^2/kg).
Best estimate 0.1 (Hoffman et al, 1983)

R = Average residence time on vegetation. Best estimate 22 days (Hoffman et al, 1979)

IRR = Irrigation rate (0.15 $m^3/m^2,y$)

K = Lumped parameter considering retention of irrigated water on vegetation (during 6 occasions of irrigation)

Table C.1

Root uptake factors for several types of nutrients (Bq/kg f w nutrient per Bq/kg d w soil).

Element	Distribution*	Pasturage*	Ranges or geom st dev	Cereals	Ranges or geom st dev	Vegetables	Ranges or geom st dev	Root vegetables	Ranges or geom st dev
Se	LT	6.5	5E-1 - 7E1	1.5	1E-1 - 8E1	6.5	5E-1 - 6E1	1.3E1	1 - 1E2
Tc	LT	1.0	1E-1 - 1E1	9E-1	1E-1 - 1E1	2E-1	1E-2 - 1.0	1E-1	1E-2 - 1.0
Sn	LT	1.0E-1	1E2 - 1.0	4E-1	1E-2 - 1.0	6E-2	1E-2 - 1.0	5E-2	1E-2 - 1.0
I	LN	6E-1	4.0	1E-1	4.0	3E-2	4.0	1E-2	4.0
Cs	LN	5E-2	2.4	1E-2	1.8	2E-2	1.9	2E-2	1.9
Pb	LT	2E-2	1E-3 - 1E-1	2E-2	1E-3 - 1E-1	2E-3	1E-4 - 1E-2	4E-3	1E-3 - 1E-2
Ra	LN	5E-2	2.5	1E-2	2.5	1E-2	2.5	1E-2	2.5
Ac	LT	5E-4	3E-5 - 7E-3	4E-4	1E-5 - 1E-3	4E-3	2E-4 - 8E-2	5E-5	2E-5 - 1E-2
Th	LT	1E-2	1E-3 - 1E-1	7E-4	1E-4 - 1E-3	2E-3	1E-4 - 1E-2	2E-3	1E-4 - 1E-3
Pa	LT	3E-3	3E-4 - 3E-2	3E-3	3E-4 - 3E-2	3E-4	3E-5 - 3E-3	6E-4	6E-5 - 6E-3
U	LT	1E-2	1E-3 - 1E-1	4E-2	4E-3 - 4E-1	1E-3	1E-4 - 1E-2	1E-3	1E-4 - 1E-2
Np	LT	1E-1	1E-2 - 1	1E-2	1E-3 - 1E-1	3E-3	5E-4 - 2E-1	3E-3	5E-4 - 2E-1
Pu	LT	1E-3	7E-5 - 1.4E-2	1E-4	1E-6 - 1E-2	2E-5	1E-7 - 4E-4	5E-5	2E-7 - 1E-3

* LT = Logtriangular distribution.

LN = Lognormal distribution.

Table C.2

Bioaccumulation factors to fish (Bq/kg f w muscle per Bq/l).

Element	Best estimate	Type of distribution*	Geom st dev	Ranges
C	4600	T		1000-10000
Se	2000	T		500 - 5000
Tc	15	T		1 - 50
Sn	3000	T		1000 - 6000
I	200	T		10 - 500
Cs	5000	LN	3.8	
Pb	100	T		50 - 200
Ra	50	T		10 - 100
Ac	100	T		10 - 1000
Th	30	T		1 - 100
Pa	10	T		1 - 100
U	50	T		10 - 100
Np	50	T		1 - 100
Pu	5	T		1 - 100

* T = Triangular distribution.
LN = Lognormal distribution.

Table C.3

Distribution factors for transfer to milk and meat, logtriangularly distributed.

Element	Distribution factor milk (day/l)	Ranges or geom st dev	Distribution factor meat (day/kg)	Ranges or geom st dev
C	1E-2	1E-3 - 1E-1	3E-2	1E-3 - 1E-1
Se	3E-3	1E-3 - 1E-2	9E-4	1E-4 - 1E-2
Tc	1E-4	1E-5 - 1E-3	2E-3	1E-4 - 1E-2
Sn	3E-3	1E-3 - 1E-2	1E-3	1E-4 - 1E-2
I*	1.3E-2	1.6	2E-2	2.1
Cs*	8E-3	1.6	3E-2	2.1
Pb	3E-4	2E-5 - 2E-3	4E-4	4E-5 - 4E-3
Ac	3E-7	3E-8 - 3E-6	1E-5	1E-6 - 1E-4
Th	5E-6	1E-7 - 1E-4	7E-4	1E-4 - 1E-3
Pa	5E-5	1E-6 - 1E-4	3E-3	2E-6 - 5E-3
U	2E-4	2E-5 - 2E-3	1E-2	1E-3 - 1E-1
Np	5E-6	1E-6 - 1E-4	3E-3	2E-4 - 5E-3
Pu	1E-7	2E-8 - 3E-7	2E-6	1E-7 - 2E-5

* Lognormal distribution.

Comments to Table C.1 and C.2

C No uptake of carbon to vegetation is considered. The accumulation factor to fish is taken from Bergström et al, 1987.

Se The literature survey performed did not give any new data while the data given in Bergström et al, 1985 were used.

Tc The uptake to plants for Tc varies considerable due to type of soil (Masson et al, 1989). High CR-values have been reported from a number of plant uptake experiments (Gast et al, 1979, Hoffman et al, 1978). Tc occurring as pertechnetate ion is rapidly taken up by plants. This anion is highly mobile in soil due to its high solubility and low sorption to soil particles. Eriksson, 1983 has studied by lysimeter experiment during five years the uptake to red clover and spring wheat. The experiments showed an immediate high uptake while after five years it was reduced by a factor of 0.0005. This confirms the importance of the chemical form of the nuclide for uptake to plants. With time the perchnetate form was reduced to less soluble forms or was moved to deeper layer. Other long-term studies showed a decrease with a factor of 0.25 to 0.5 after two years for uptake by radishes (Masson et al, 1989).In addition most of the experiments performed use unrealistic high concentrations of Tc. For releases to the biosphere from a repository the actual amounts are low compared to the experimental conditions. According to Eriksson the uptake of Tc would be about the same order of magnitude as for Cs for continuous releases of small amounts. The values used are therefore biased against the higher ranges of corresponding values for Cs.

Concerning the uptake into freshwater fish the survey did not give any new results while the value is taken from Coughtrey et al, 1983. However, laboratory experiments performed for marine fishes showed low concentration factors, about 2 (Masson et al, 1989).

Sn In similarity to Se no new data were found in the literature while the data are taken from Bergström et al, 1985.

I — An intensive study of the behaviour of I-129 released to the air from Karlsruhe reprocessing plant has been performed (Deitermann et al, 1990). The study showed that the concentration factors to vegetation were distributed lognormally. The values used are taken from that reference because the conditions ought to be quite similar to the ecosysten we study. Compared to earlier values used (Bergström et al, 1989) the uptake to pasturage is increased while the uptake to cereals are about the same. However for vegetables and potatoes the uptake has decreased. In the absence of variance data for the latter three vegetables the standard deviation given for food crops was applied.

In the aquatic ecosystem Iodine has a considerable uptake to fish muscle according to Poston et al, 1986. This is however in contrast to earlier reported values. In order to not underestimate the exposure the value recommended in Poston et al, 1986 representing piscivorous fish was applied. For omnivorous and planktivorous fishes they report even a higher value (500). However, the most common fishes for consumption belong to the piscivorous species.

Cs — In the absence of specific data for Cs-135, data for Cs-137 are used. In Mascanzoni, 1989 results from a long-term Swedish field experiment are given. The study confirmed that the root uptake among other things is dependant on the organic matter and potassium content of the soil. The values given in the paper for pasturage and cereals are used and they are in good agreement with earlier reported values. The ranges are taken from the same reference. For the remaning vegetables the survey did not give any new information while results from Bergström et al, 1987 were applied.

Cs-137 has been studied considerable in aquatic systems. Kohlemainen, 1966 has given bioaccumulation factors in the range of 200-2000 for piscivorous fish in eutrophic lake systems. Vanderploeg, 1975 gives the transfer at steady state as a function of the potassium concentration in water; 15000/[K] (piscivorous fish). However, studies of the Chernobyl fallout in the same type of lake gives values about 5000 for the species mostly consumed (Bergström et al, 1989). In contrast Neumann, 1985 gives a generic value of 200.

Pb Rounded values, upwards of the values given in Bergström et al, 1985 were used. They are in agreement with the upper ranges of values for brown beans, carrots, potatoes and corn given in Vasconcellos et al, 1987. In the paper results of a study concerning uptake of Ra-226 and Pb-210 in a region of high natural radioactivity in Brazil are given. Sheppard et al, 1990 reports values within the same range.

For fish Thompson et al, 1972 gives a mesured value of 100, which was used.

Ra The uptake to plants is among other things dependent on pH. An acidification of the soils increases the uptake. In general calculations of concentration factors based upon the total amount of Ra-226 in the soil may underestimate the transfers, Lima et al, 1988. This is because an additional transport to the soil of Ra-226 could be regarded as belonging to a fraction sorbed on the surfaces. This fraction ought to be more correlated to the exchangeable (biological available) part of the total content. An average ratio of 1.6E-2 (dry plants) is reported in Evans et al, 1982. By using data given in Linsalata et al, 1989 a ratio of 5E-2 is obtained. In Vasconcellos et al, 1989, where a region of high natural radioactivity was studied, the uptake to potatoes varied within the range of 2E-3 - 1E-1. With these references as a basis the values were chosen.

The bioaccumulation factor to fish is the one recommended in Poston et al, 1986. This high value is contradictionary to the values reported in Rope et al, 1985 where field studies of uptake in trout ranged between 0.29-2. The value used might be quite conservative due to radiums strong tropism to bone. In addition the calcium content in water is important for the uptake of Ra-226.

Ac Due to absence of data for actinium, data for americium were used. This is because of the chemical similarity between these elemnts. Best estimate and ranges for rootuptake factors were estimated from the IAEA draft 1987. The ranges used are those recommended by Frissel et al, 1986.

Concerning the bioaccumulation factor to fish Thompson et al, 1972 recommends 25. In the literature review performed by Poston et al, 1986 they recommend a value of 100. This may be too high for americium in muscle because it it will concentrate in bone and kidney. However, the value of 100 was chosen mostly because of the lack of pertinent information for Ac.

Th By using data from Linsalata et al, 1989 a root-uptake factor of 0.05 is obtained. This value is about a factor of 10 higher than in earlier compilations (Bergström et al, 1989). However, because of the low transfer to milk and meat this factor is of minor importance for the results while it was applied. The value adopted for cereals is the average value found in a Swedish investigation (Evans et al, 1982). For cerelas and root vegetables the same value as for pasturage was applied corrected to fresh weights. These values are for sure conservatively biased. Lower values are reported in Eisenbud et al, 1984.

The value of the bioaccumulation factor to fish was taken from compilations (NUREG 77 and IAEA SS-57). However, Poston et al, 1986 recommends a much higher value, 1000 in their literature survey. The recommendations are contradictionary as also can be seen in BIOMOVS TR1, 1988. The high value was not selected because it seems questionable for a bone-seeking nuclide. For a rough comparison the uptake into meat is not either supposed to be substantially for thorium.

Pa The survey did not give any new information while the root uptake values recommended as input parameter values in Bergström et al, were used.

For uptake into fish the only value found was the one given in IAEA SS-57.

U By using data from Linsalata et al, 1989 a ratio of 0.01 for pasturage is obtained. This is about the same value when converted to dry wight as recommended in Grogan 1985. The value given in the latter reference is mostly based upon Swedish investigations. For cereals best estimate value is taken from Evans et al, 1982.

The biaccumulation factor to fish is the one recommmended in Poston et al, 1986. However, in Hoffman, 1988 factors mostly under one are obtained. It was out of the scope to more precisely investigate the differece while in order to be conservativ the higher value was adopted.

Np Np is more biological available than other actinides. Cataldo et al, 1984 reports ratios for alfalfa and soybeans higher than one. Corresponding value for grain is given to 0.05. Coughtry et al, 1984 report lower values. The values used are about the average of the ranges from these references.

Concerning the uptake to fish Poston et al, 1986 recommend a factor of 50 for piscivorous fish. In earlier compilations (Bergström et al, 1983) a factor of 10 is given. However, the uptake of Np ought to be higher than for Pu while the higher value was adopted.

Pu Pu has a low biological uptake. The values given in IAEA draft are used with ranges recommeneded by Frissel et al, 1986.

For piscivorous fish Poston et al, 1986 recommend a value of 5 which was used. Agnedal, 1978 recommends a general value of 30 for fresh-water fish.

Comments to Table C.3

C	No new data were found while data from Bergström et al, 1989 were used. The transfer to milk is based upon data in Bondietti et al, 1986.
Se	The literature survey performed did not give any new data while the values used in Bergström et al, 1989 were taken. The transfer to milk is from the IAEA draft 1987 and for meat from Bergström et al, 1985.
Tc	The same values as in Bergström et al, 1989 were used due to no new information.
Sn	Data from Bergström et al, 1985 were used due to absence of new information from the literature survey.
I	The distribution factors are taken from Robens et al, 1988. The factor for meat is about the same value as previously used while the factor to milk is considerably lower. This is in agreement with results from several studies of the Chernobyl fallout (BIOMOVS Draft, 1990).
Cs	Data from Bergström et al, 1985 were used. However, a study of the transfer to milk from the Chernobyl fallout from 11 different locasions shows an average value about a factor of 0.6 lower than the value used (BIOMOVS Draft 1990).
Pb	No studies reporting values for Pb were found in the survey. Data given in the IAEA draft were therefor applied.
Ra	The values are geometric meanvalues of ranges found in the literature according to Bergström et al, 1983. The value for milk is in good agreement with the average value reported in Watson et al, 1984. In addition the value for meat given in Linsalata et al, 1989 coinsides with the value used.
Th	No new information concerning the transfer to milk was found, because of that the value given in IAEA SS-57 estimated by Ng was used. The transfer to meat can be estimated to 2.E-5 day/kg according to data given in Linsalata et al, 1989. This value is considerable lower than data previously used (Bergström et al, 1989). An estimated value based upon these references was used.

Pa	Due to lack of new information no changes of data used in studies previously performed (Ng, 1982) seemed appropriate.
U	The same value for the transfer to milk is given in Bergström et al, 1983 and IAEA draft. From the latter reference the transfer factor to meat is taken due to lack of any new information.
Np	Similar to many other elements the survey did not supply any additional information. The value used is estimated by Ng, 1981 in the absence of direct information.
Pu	Data from the IAEA draft were used in the absence of new information.

Table C.4

Consumption data for adults, triangularly distributed.

	Best estimate	Min	Max
Individuals			
Inhalation, m^3/y	8000	7000	8000
Drinking water, l/y	600	400	750
Milk, l/y	200	20	400
Meat, kg/y	55	15	100
Green vegetaables, kg/y	40	5	100
Cereals, kg/y	80	5	150
Root-fruits, kg/y	70	5	100
Fish, kg/y	30	5	100
Soil, kg/y	0.01	0.001	0.1

Table C.5

Consumption data for children, triangularly distributed.

	Best estimate	Min	Max
Individuals			
Inhalation, m^3/y	2000	1500	1500
Drinking water, l/y	150	50	200
Milk, l/y	125	50	200
Meat, kg/y	26	5	50
Green vegetaables, kg/y	8	5	50
Cereals, kg/y	44	5	70
Root-fruits, kg/y	31	5	70
Fish, kg/y	7	5	50
Soil, kg/y	0.02	0.001	0.1

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Sven Norman[1], Nils Kjellbert[2]
[1] Starprog AB
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Rolf Hesböl, Ignasi Puigdomenech, Sverker Evans
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Statistical estimation and stochastic simulation using PROPER
Lars Lovius[1], Sven Norman[1], Nils Kjellbert[2]
[1] Starprog AB
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TR 90-04
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R. S. Forsyth, U-B. Eklund, O. Mattsson, D. Schrire
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Karsten Pedersen
University of Gothenburg, Department of General and Marine Microbiology, Gothenburg
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Yngve Albinsson, Birgit Sätmark,
Ingemar Engkvist, W. Johansson
Department of Nuclear Chemistry,
Chalmers University of Technology, Gothenburg
April 1990

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R S Forsyth, T J Jonsson, O Mattsson
Studsvik Nuclear
March 1990

TR 90-08
Radiolytically induced oxidative dissolution of spent nuclear fuel
Lars Werme[1], Patrik Sellin[1], Roy Forsyth[2]
[1] Swedish Nuclear Fuel and waste Management Co (SKB)
[2] Studsvik Nuclear
May 1990