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TECHNICAL REPORT 85-48

Biosphere Modelling for a HLW Repository Scenario and Parameter Variations

Helen Grogan

October 1985

Swiss Federal Institute for Reactor Research, Würenlingen

Parkstrasse 23 5401 Baden/Schweiz Telephon 056/20 55 11

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SUMMARY

In Switzerland high-level radioactive wastes have been considered for disposal in deep-lying crystalline formations. The individual doses to man resulting from radionuclides entering the biosphere via groundwater transport are calculated. The main recipient area modelled, which constitutes the base case, is a broad gravel terrace sited along the south bank of the river Rhine. An alternative recipient region, a small valley with a well, is also modelled. A number of parameter variations are performed in order to ascertain their impact on the doses. Finally two scenario changes are modelled somewhat simplistically, these consider different prevailing climates, namely tundra and a warmer climate than present.

In the base case negligibly low doses to man in the long term, resulting from the existence of a HLW repository have been calculated. Cs-135 results in the largest dose (8.4×10^{-7} mrem/y at 6.1×10^{6} y) while Np-237 gives the largest dose from the actinides (3.6×10^{-8} mrem/y). The response of the model to parameter variations cannot be easily predicted due to non-linear coupling of many of the parameters. However, the calculated doses were negligibly low in all cases as were those resulting from the two scenario variations.

ZUSAMMENFASSUNG

Es ist in der Schweiz geplant, hochradioaktive Abfälle in tiefliegenden Kristallinformationen einzulagern. In dieser Arbeit werden Individualdosen berechnet, die sich aus dem Transport von Radionukliden durch Grundwasser an die Biosphäre ergeben können. Als wichtiges Gebiet, in welches die Tiefenwässer exfiltrieren wurde eine breite Schotterterrasse südlich des Rheines modelliert. Dies stellt den Basisfall dar. Gleichfalls modelliert wurde das Gebiet eines kleinen Tales, dessen Wasserversorgung durch eine Bohrung in geologische Sedimentschichten sichergestellt wird. In einer Reihe von Parametervariationen wird der Einfluss auf vorausgesagte Dosen untersucht. Auf einfache Weise werden schliesslich die Folgen von Klimaänderungen berücksichtigt, indem die Dosen für ein Tundraszenarium und ein wärmeres Klima berechnet werden.

Im Basisfall sind es vernachlässigbare Langzeitdosen, die als Konsequenz eines Lagers für HAA berechnet werden. Den relativ grössten Beitrag liefert mit 8.4×10^{-7} mrem/a das Isotop Cs-135 nach 6.1×10^{6} Jahren, während bei den Aktiniden der Hauptbeitrag mit 3.6×10^{-8} mrem/a von Np-237 stammt. Weil viele Parameter die Resultate auf nicht-lineare Weise beeinflussen, sind die Konsequenzen von Parametervariationen nicht einfach abzuschätzen. In allen betrachteten Variationen des Basisfalles, wie auch in den beiden Szenarienvariationen, sind die berechneten Dosen jedoch vernachlässigbar klein.

RESUME

Il est prévu en Suisse de stocker des déchets hautement radioactifs dans des formations cristallines profondes. Dans cette étude des doses individuelles qui pourraient résulter du transport des radionucléides par les eaux souterraines à la biosphère sont calculées. Comme région importante, où les eaux profondes, une large terrasse des graviers fluviaux au sud du Rhin a été modelée. Ceci représente le cas de base. La région d'une petite vallée dont l'approvisionnement en eau est assuré par un forage dans des couches géologiques sédimentaires a aussi été modelée. L'effet sur les doses prévues est étudié. Par une série de variations des paramètres. Finalement les conséquences des changements climatiques sont considérés d'une façon simplifiée, en calculant les doses pour un scénario de tundra et pour un clima plus chaud.

Dans le cas de base, ce sont des doses à long terme négligeables qui sont calculées comme conséquence d'un dépot pour DHR. La contribution la plus élévée rélativement (8.4×10^{-7} mrem/a) est livrée par l'isotope Cs-135 après 6.1×10^{6} années, tandis que pour les actinides la contribution principale provient du Np-237 (3.6×10^{-8} mrem/a). Les conséquences de la variation des paramètres ne sont pas facilement évaluables, parce que plusieurs paramètres influencent les résultats d'une façon non-linéaire. Néaumoins les doses calculées lors de toutes les variations des paramètres et aussi pour les deux variations des scénarios sont négligeables.

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1 INTRODUCTION

The present Swiss concept for the final storage of high level radioactive waste (HLW), is to construct a repository sited in deep crystalline rock (at a depth of about 1300 m) in the north of Switzerland. A complete overview of this is given in Nagra, 1985 (Vol. 1). For any safety analysis of such a repository, the biosphere must be modelled in order to calculate the consequent doses to man and the Swedish computer code BIOPATH (Edlund et al., 1981) is used for this, taking data relevant for Swiss conditions.

A total of sixteen radionuclides have been considered for the biosphere modelling and can be divided into two broad categories,

- the actinides and daughters
- the fission and activation products.

For the actinides and daughters three separate chains are studied, the Np-237 decay chain (Np-237, U-233, Th-229,...), the U-238 decay chain (U-238, U-234, Th-230, Ra-226,...) and the U-235 decay chain (U-235, Pa-231, Ac-227,...). A total of six fission and activation products are also studied, namely Ni-59, Se-79, Tc-99, Pd-107, Sn-126 and Cs-135.

2 DESCRIPTION OF THE MODEL ASSUMPTIONS

In essence a region of the biosphere is identified into which the radionuclides are likely to flow (Nagra, 1985, Vols. 4 and 5) and this forms the basis of the model. This region is divided up into physical compartments such as groundwater, soil and soilwater and the flow of radionuclides through them modelled as a function of time using BIOPATH. The source term is the activity entering the biosphere which is supplied by the geosphere transport model (Hadermann and Rösel, 1985). In the base case scenario for the biosphere, present day conditions are assumed. As a result meteorological, hydrological and agricultural statistics from Switzerland are used to calculate the precipitation and water fluxes between the compartments in the regions modelled.

The radionuclides in the separate compartments may reach man to give an internal dose by a number of different exposure pathways and these are attached to the appropriate compartments in the model, e.g. pasture - cow - man pathway is attached to the soil compartment. The pathways are reduced to include only the most important ones as it is not possible to model all potential pathways. Similarly, the

individual constituents are somewhat simplified so that crops are divided into cereals, pasture, leaf and root vegetables. The animals into cows, hens and fish, and the animal products into meat, milk and eggs. The individual parameters which specify the degree of radionuclide transfer between individual components of the foodchain are taken from the literature. In general the most realistic values pertaining to the present day situation in Switzerland were selected and are listed in full in Jiskra, 1985. A discussion of the selection of soil to plant concentration ratios is found in Grogan, 1985. The consumption values for livestock were obtained from agricultural statistics for this region of Switzerland whilst the consumption values for man were calculated assuming a 3000 kcal energy demand per day (Jiskra, 1985). It is assumed that the population of the region is totally self-subsistent, living entirely off these agricultural products and drinking locally obtained water.

Despite these simplifications the critical exposure foodchain pathways, including drinking water can be modelled and their significance to man's total dose evaluated.

A number of parameter variations have been made to the base case to ascertain their impact and importance upon the predicted doses to man and the critical pathways involved. These include a "well scenario" in which the contaminated groundwater arrives in the biosphere via a well. The Kd values were also varied and an alternative pathway of soil ingestion considered.

The time scales considered for modelling a HLW repository are, however, essentially long. Once the repository has been completed and sealed , canister failure is pessimistically assumed to occur after 1000 years. It is only at this point in time that radionuclides are potentially available to migrate through the geosphere and enter the biosphere. It is unrealistic to assume that factors in the biosphere will not change during these timescales, however the problem is in accurately predicting the nature and extent of these changes. It is for this reason that the base case scenario takes present day conditions which are well defined and process quantitative data. Despite this some effort must be made to model probable variations in the biosphere and assess their likely impact on the resultant doses to man.

For the biosphere modelling of a HLW repository these variations have been conducted on two scales. Firstly, as small scale variations to the base case i.e., parameter variations such as Kd values, and secondly, as major scenario changes i.e., a different prevailing climate. In all cases only reasonably realistic variations are entertained.

3 BIOSPHERE BASE CASE

The main safety analysis for a HLW repository assumes a gradual failure of the waste package whereby the radionuclides are released into the groundwater and are transported through the host rock (granite) up through the overlying geological layers into the biosphere (Nagra, 1985, Vol. 4 and 5). Hydrogeological studies of northern Switzerland, from a model site, show the overall groundwater flow is north towards the Rhine (Kimmeier et al., 1984). For this reason a number of different recipient geographical regions were modelled for the influx of contaminated groundwater. A more detailed description of these regions can be found in Jiskra, 1985. Of these the Laufenburg region was selected to model the base case scenario for the biosphere. The Laufenburg region is situated on a broad gravel terrace along the south bank of the Rhine, the contaminated water arrives through an aquifer and is diluted to an extent depending on the total flow through this.

Figure 3.1 shows the system of compartments used to model this region along with the relevant water fluxes between them. In general the groundwater does not flow directly into the soilwater, however this was conservatively assumed to occur in the model. As previously stated the base case is modelled for present day conditions so that the water flow rates, precipitation, compartment volumes etc., are average values taken from present day hydrological maps and various statistical reports for this region.

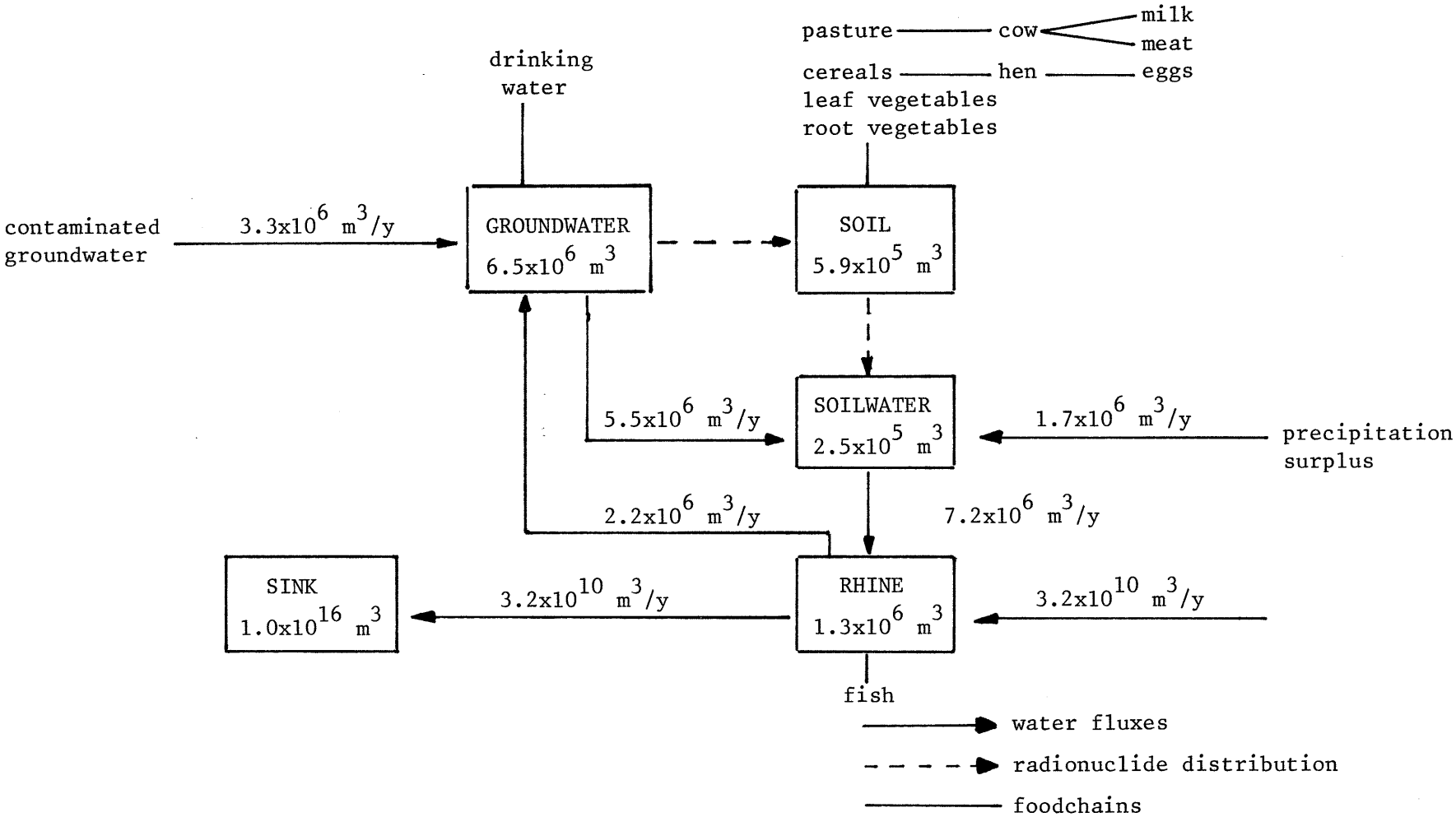

Figure 3.1: Scheme of compartments for Laufenburg.

3.1 Base Case Dose Calculations

The radionuclides source terms for the biosphere modelling are supplied by the geosphere transport model which gives the outflow from the geosphere of each radionuclide in the groundwater as a function of time. However it should be noted that the geosphere base case does not supply the input to the biosphere base case model, instead it is the results from variation 1 in which a conservative value for diffusion in the kakirites is assumed. This is because effectively no doses result from using the geosphere base case.

All the calculated doses are given for each radionuclide although it is recognised that some of these values are so small as to be meaningless. For example, a dose of 8.4×10^{-12} mrem/y from U-238 is equivalent to 4×10^{-16} moles entering the body in a year or expressed in another way, one disintegration every 25 years in the body. However, by showing the doses from each radionuclide the relative importance of each can be ascertained, so that should a situation arise where significantly less dilution occurs in the biosphere, the doses can be scaled up accordingly and thus assume greater significance.

No calculations are made for times greater than 1×10^{8} years. This cut-off point was arbitrarily chosen to restrict calculating doses at times unrealistically far into the future.

Table 3.1 gives the results for the base case calculations showing the maximum dose for each radionuclide and the time at which it occurs. These results are also presented graphically, figures 3.2, 3.3 and 3.4 show the time dependent doses for the three actinide chains, Np-237, U-238 and U-235, respectively and figure 3.5, the doses from the fission and activation products.

Table 3.1:

Base case dose calculations for the Laufenburg region.
Cut-off time is 10^8 years.

Radionuclide	LAUFENBURG REGION	
	maximum dose (mrem/y)	time (y)
Np-237	3.6×10^{-8}	2.1×10^{7}
U-233	2.2×10^{-10}	2.1×10^{7}
Th-229 *	1.0×10^{-8}	2.1×10^{7}
U-238	8.4×10^{-12}	1.0×10^{8}
U-234	8.9×10^{-12}	1.0×10^{8}
Th-230	3.1×10^{-10}	3.5×10^{5}
Ra-226 *	1.9×10^{-9}	3.5×10^{5}
U-235	4.7×10^{-12}	1.0×10^{8}
Pa-231	1.5×10^{-9}	1.0×10^{8}
Ac-227 *	6.7×10^{-10}	1.0×10^{8}
Ni-59	1.1×10^{-11}	
Se-79	3.5×10^{-8}	4.4×10^{5}
Tc-99	7.0×10^{-9}	1.1×10^{6}
Pd-107	8.7×10^{-9}	3.5×10^{7}
Sn-126 *	1.4×10^{-8}	8.8×10^{5}
Cs-135	8.4×10^{-7}	6.1×10^{6}

* includes dose contribution from daughters.

From these results it can be seen that the doses are small, Cs-135 giving the largest dose of 8.4×10^{-7} mrem/y which is significantly greater than any of the other doses. For many of the radionuclides the doses are three or more orders of magnitude less. If one assumes the maximum dose for each radionuclide occurs simultaneously the total individual dose is still only 9.5×10^{-7} mrem/y of which 88% results from Cs-135. The results show there is a general trend for the fission and activation products to give higher doses overall than the actinides and for these to occur at earlier times in the biosphere. The significance of the fission products is probably overestimated in relation to the actinides because of assumptions made in the geosphere calculations and these can be summarised as follows. For Cs-135, although a large amount of data exists showing non-linear isotherms, for simplicity a linear relationship was chosen. The sorption equilibrium distribution

constant was taken corresponding to a high geosphere input concentration, i.e., a very low value. It has been shown that by considering a non-linear isotherm the concentrations at the geosphere outlet can be considerably reduced (Hadermann and Rösel, 1983).

For the remaining fission and actinide products sorption data are very sparse, as a consequence extremely conservative (probably unrealistic) sorption equilibrium constants were chosen. This results in minimal retardation in the geosphere so that relatively large amounts are released to the biosphere.

It is worth mentioning that care should be taken in interpreting the graphs, for example, the Se-79 dose peak may at first glance appear to dominate for a comparable time interval to that of the Pd-107 peak. In fact the first peak represents around a million years whilst the second (Pd-107) represents just under seventy million years. Here one also realises the incredibly long time scales considered.

The Np-237 decay chain is the most important for the actinides with respect to dose, with Np-237 itself producing the largest dose of 8.6×10^{-8} mrem/y after about 2×10^{7} years. The dose from Th-229 (including daughters) is less than a third of this and the U-233 dose two orders of magnitude less. The uranium isotopes are extremely long-lived and strongly retarded in the geosphere so that their maximum doses in the biosphere are reached far beyond 10^{8} years.

The significance of the individual exposure pathways to the dose from each radionuclide is shown in table 3.2. Although the drinking water pathway dominates for many of the radionuclides other exposure pathways through the foodchain are also important contributors to the final dose. For example, cereals and root vegetables are equally dominant exposure pathways as drinking water for Th-229, with each contributing 28%, 35% and 30% respectively, to the total dose. Similarly, 63% of the dose from Sn-126 results from cereals and 17% from root vegetables whilst only 12% comes via the drinking water. It should be noted that the model uses a relatively high value of 730 l/y for drinking water consumption by the individual, but one can see that, for example, for Th-229 if the drinking water intake were decreased to half its present value this would have minimal impact on the overall dose but a large influence on the dose resulting from the drinking water.

Table 3.2:

Contribution of each exposure pathway to the base case dose from each radionuclide, expressed as a percentage.

Radio-nuclide	EXPOSURE PATHWAY							
	Drinking water	Milk	Meat	Leaf Vege-tables	Root Vege-tables	Cereals	Eggs	Fish
Np-237	84	-	-	1	13	2	-	-
U-233	96	1	-	-	1	2	-	-
Th-229*	30	-	1	6	35	28	-	-
U-238	96	1	-	-	1	2	-	-
U-234	96	1	-	-	1	2	-	-
Th-230	27	-	1	6	37	29	-	-
Ra-226*	64	1	1	1	8	25	-	-
U-235	96	1	-	-	1	2	-	-
Pa-231	25	-	27	4	37	7	-	-
Ac-227*	78	-	21	-	1	-	-	-
Ni-59	86	3	2	1	3	5	-	-
Se-79	30	3	66	-	-	-	1	-
Tc-99	21	54	1	1	8	15	-	-
Pd-107	86	3	2	1	3	5	-	-
Sn-126*	12	5	-	3	17	63	-	-
Cs-135	37	22	23	3	7	8	-	-

- value less than 0.5%

* includes dose contribution from daughters

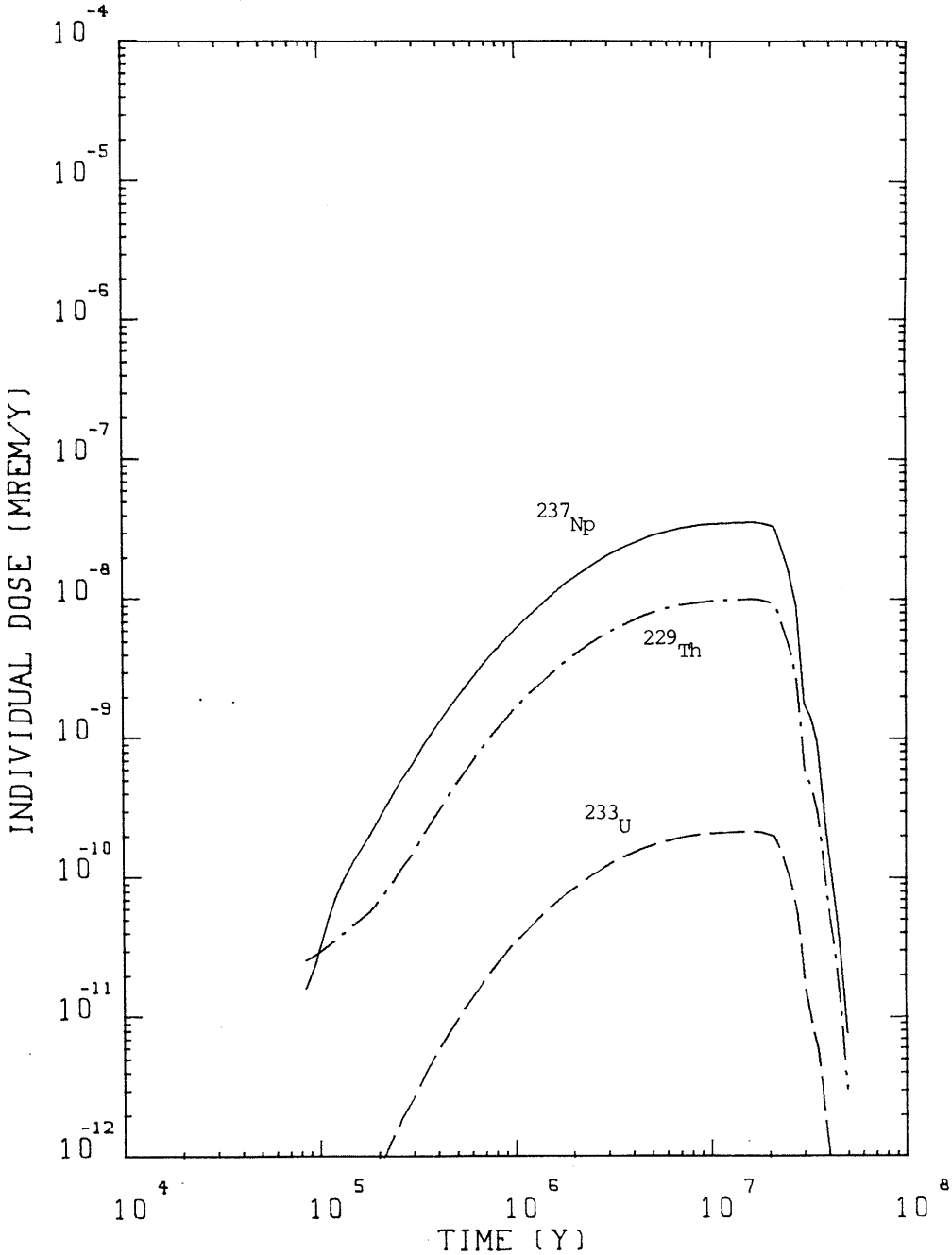

Figure 3.2: Base case doses from Np-237 decay chain.

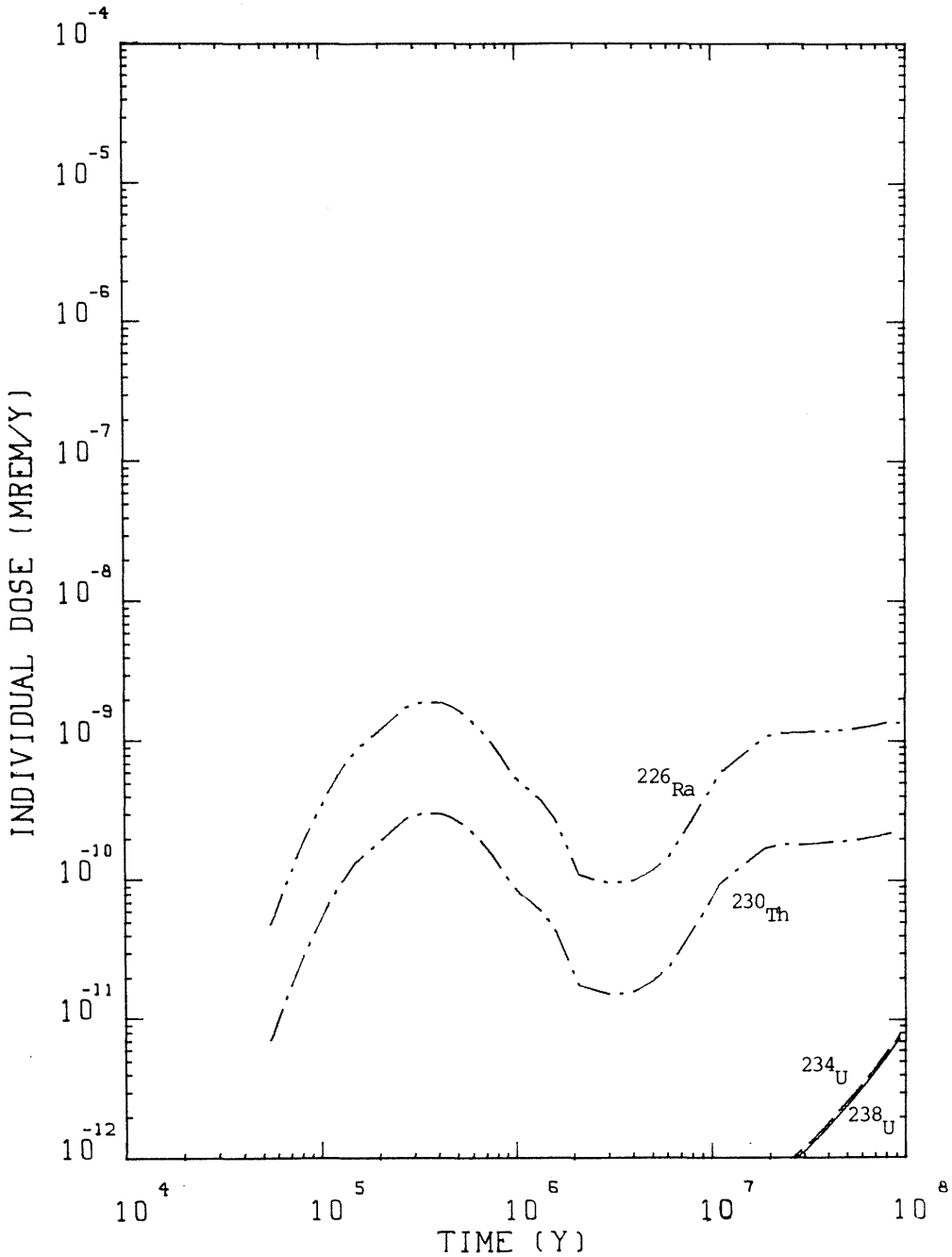

Figure 3.3: Base case doses from U-238 decay chain.

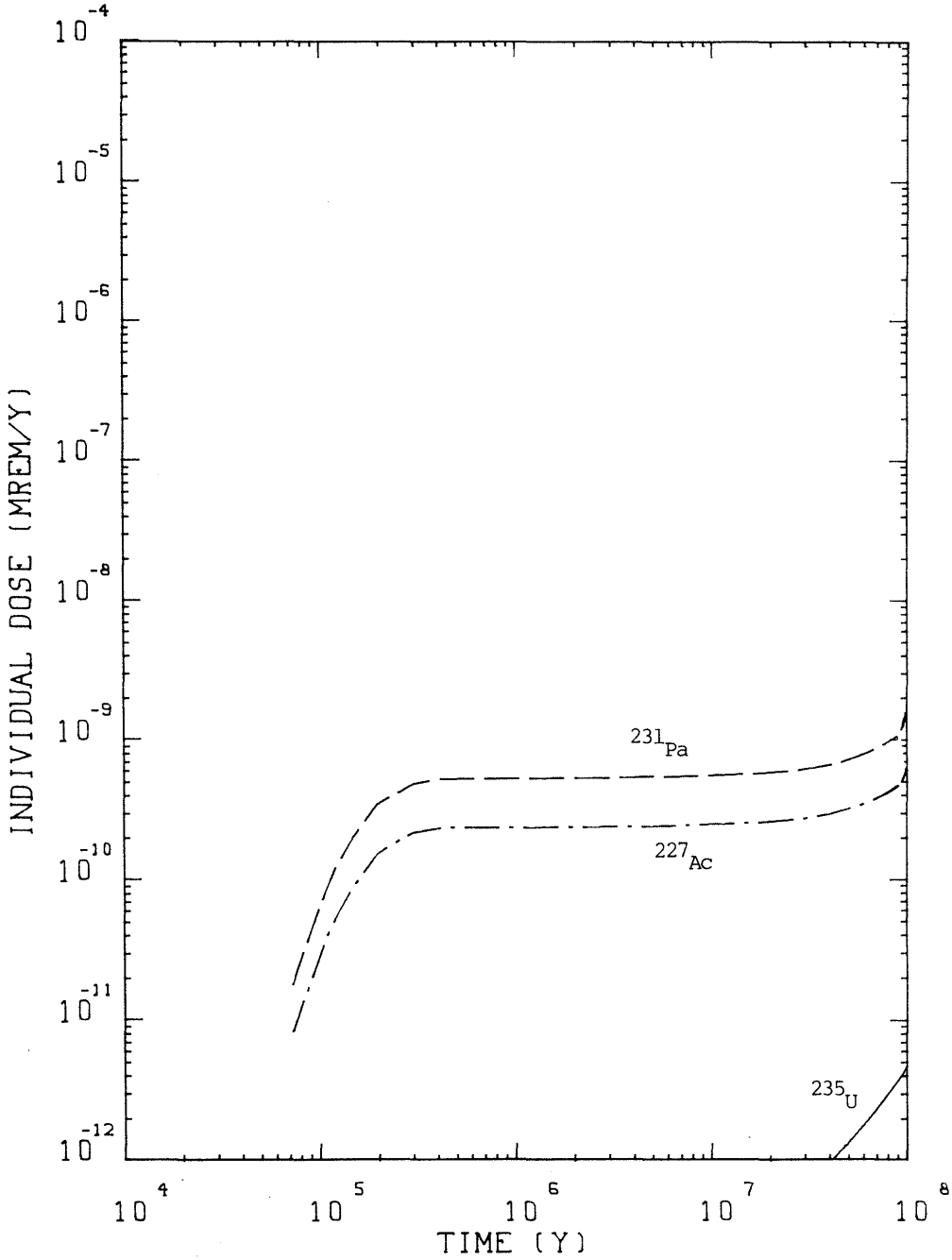

Figure 3.4: Base case doses from U-235 decay chain.

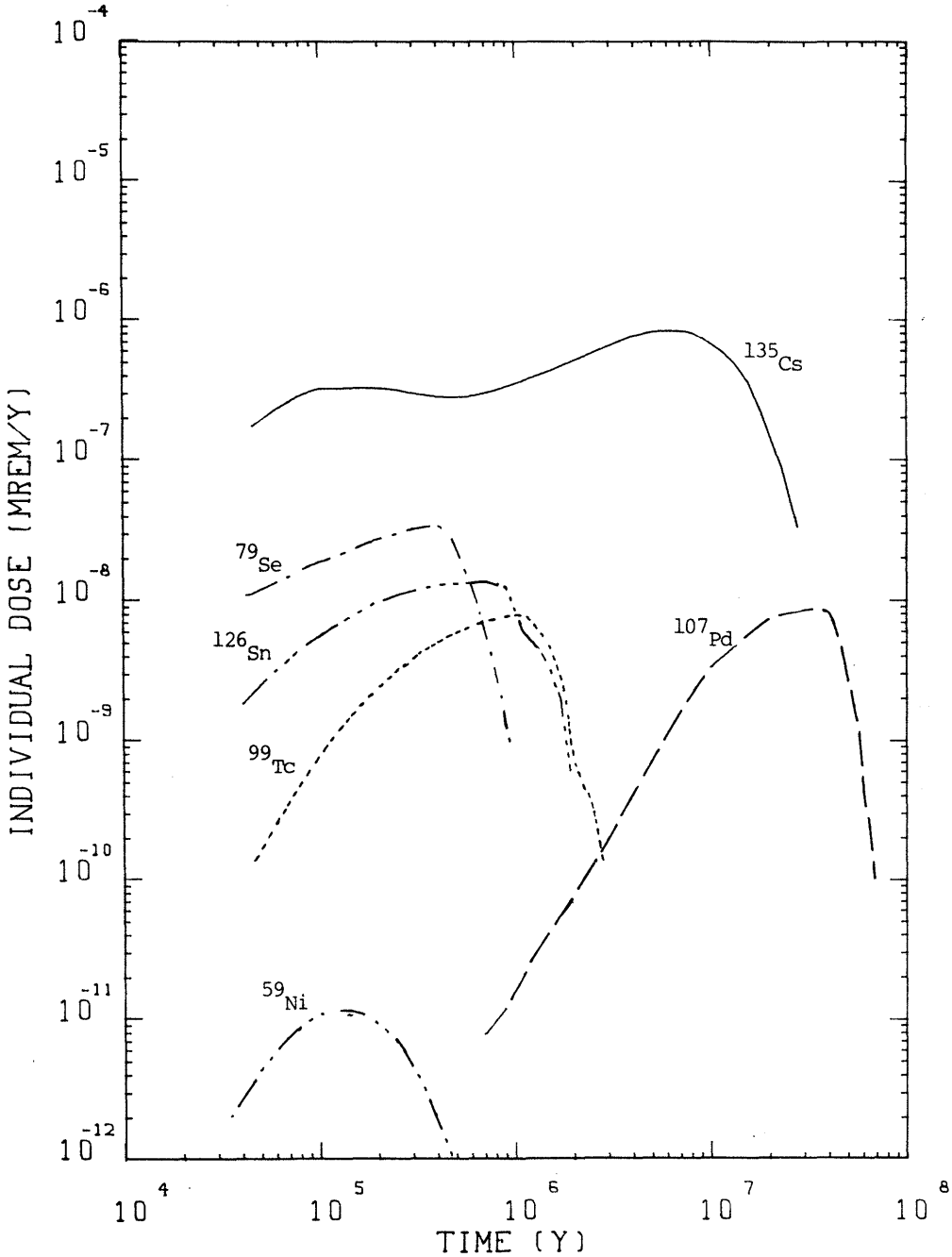

Figure 3.5: Base case doses from fission and activation products.

4 PARAMETER VARIATIONS

4.1 Well Scenario (decreased dilution in the biosphere)

A variation of the base case considered for the modelling is that of decreased dilution of the contaminated groundwater as it enters the biosphere. To model this the small narrow valley containing Hellikon in the north of Switzerland was selected. A well was assumed to be sited at the head of this valley drawing water directly from the contaminated groundwater to supply drinking water for all the community and their livestock. In order to be more conservative all the contaminated water enters the soilwater compartment from which the foodchains are derived prior to entering the groundwater compartment. It is appreciated that this does not represent the natural water path, however, it ensures minimal dilution of the contamination before entering the foodchains thus leading to conservative dose estimates. In figure 4.1 the system of compartments used to model this region are shown along with the appropriate water fluxes between them. In all other respects the model was run as for the base case, so that no alternatives were made to the concentration ratios, soil Kd's, crop consumption values etc.

In table 4.1 the results for the Hellikon region are shown in terms of the maximum dose received from each radionuclide. Figures 4.2, 4.3 and 4.4 show the time dependence of these doses for the three actinide chains, Np-237, U-238 and U-235, respectively. Figure 4.5 shows the results for the fission and activation products.

The doses which result from this parameter variation are greater than those from the base case, however, the relative significance of each radionuclide follows the same trend with the maximum doses occuring at the same time. So that Cs-135 gives the largest dose (1.8×10^{-5} mrem/y) contributing 89% to the total individual dose, assuming the maximum doses occur simultaneously for all the radionuclides. In fact, the doses from each radionuclide are all approximately twenty times greater than those from the base case which directly reflects the decreased dilution in the first modelled compartment of the region. The transport rate of water into the first compartment being twenty times less for the Hellikon region than the Laufenburg region (base case).

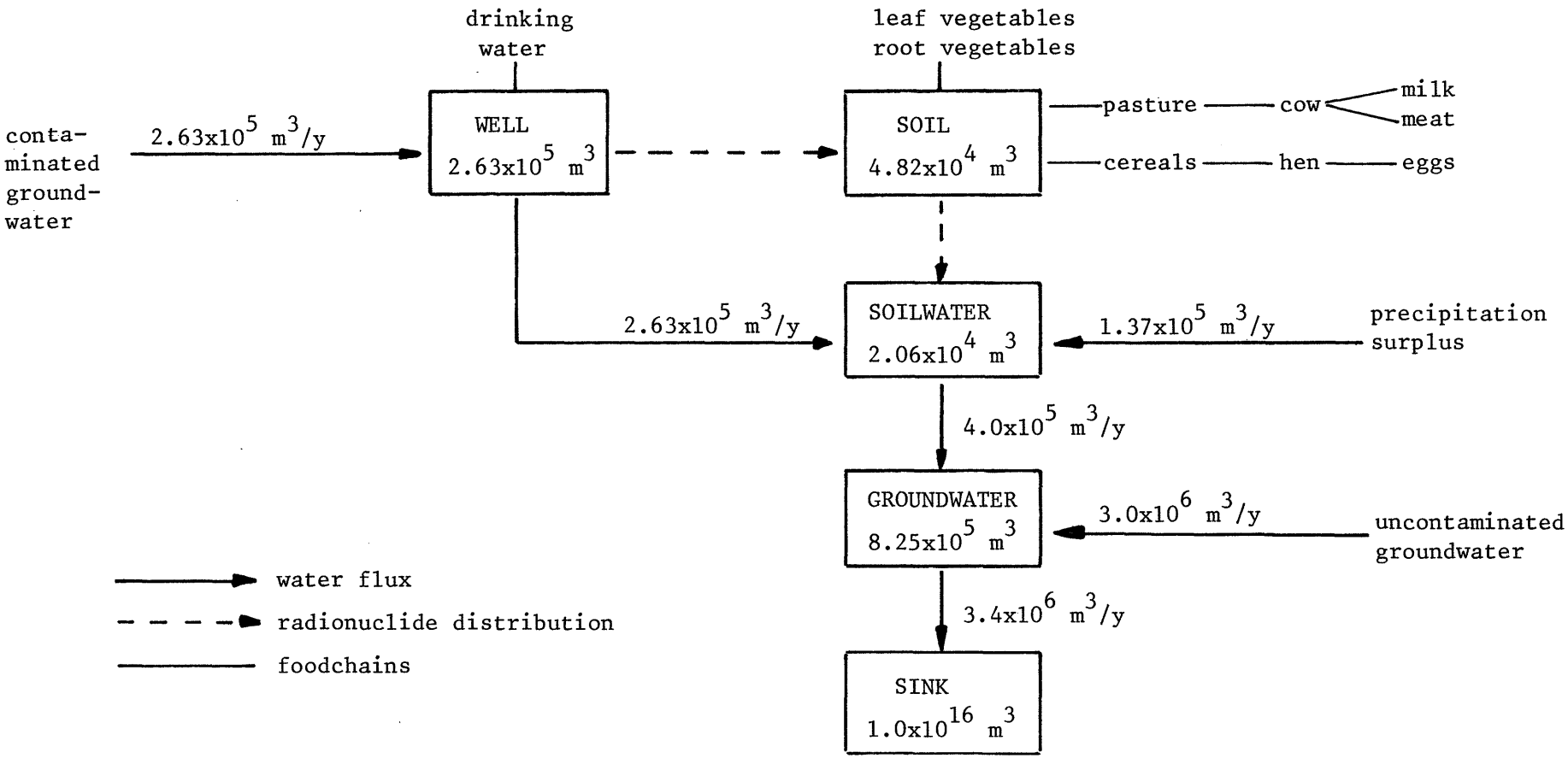

Figure 4.1: System of Compartments for Hellikon.

Table 4.1:

Dose calculations for the Hellikon region.Cut-off time is 10^8 years.

Radionuclide	HELLIKON REGION	
	maximum dose (mrem/y)	time (y)
Np-237	7.4×10^{-7}	2.1×10^{7}
U-233	4.5×10^{-9}	2.1×10^{7}
Th-229 *	1.8×10^{-7}	2.1×10^{7}
U-238	1.7×10^{-10}	1.0×10^{8}
U-234	1.8×10^{-10}	1.0×10^{8}
Th-230	5.7×10^{-9}	3.5×10^{5}
Ra-226 *	4.1×10^{-8}	3.5×10^{5}
U-235	9.7×10^{-11}	1.0×10^{8}
Pa-231	2.7×10^{-8}	1.0×10^{8}
Ac-227 *	1.3×10^{-8}	1.0×10^{8}
Ni-59	2.4×10^{-10}	
Se-79	7.1×10^{-7}	4.4×10^{5}
Tc-99	1.3×10^{-7}	1.1×10^{6}
Pd-107	1.8×10^{-7}	3.5×10^{7}
Sn-126 *	2.7×10^{-7}	8.8×10^{5}
Cs-135	1.8×10^{-5}	6.1×10^{6}

* includes dose contribution from daughters.

The importance of the individual exposure pathways (table 4.2) to the dose from each radionuclide is essentially the same as for the base case, again reflecting the linear relationship between dose and initial dilution in the biosphere. It is noticeable that the fish pathway begins to assume some significance for the fission and activation products as well as Ac-227 (U-235 decay series). This directly reflects the higher radionuclide levels in the groundwater which supply a stream in which the fish are assumed to live.

Table 4.2:

Contribution of each exposure pathway to the dose from the Hellikon region, expressed as a percentage.

Radio-nuclide	EXPOSURE PATHWAY							
	Drinking water	Milk	Meat	Leaf Vege-tables	Root Vege-tables	Cereals	Eggs	Fish
Np-237	86	-	-	1	11	2	-	-
U-233	96	1	-	-	1	2	-	-
Th-229*	35	-	-	6	33	26	-	-
U-238	96	1	-	-	1	2	-	-
U-234	96	1	-	-	1	2	-	-
Th-230	30	-	1	6	35	28	-	-
Ra-226*	64	1	1	1	8	24	-	1
U-235	96	1	-	-	1	2	-	-
Pa-231	28	-	27	4	35	6	-	-
Ac-227*	77	-	20	-	1	-	-	2
Ni-59	86	2	1	1	3	5	-	2
Se-79	31	3	64	-	-	-	1	1
Tc-99	24	53	1	1	7	14	-	-
Pd-107	86	2	1	1	3	5	-	2
Sn-126*	12	5	-	3	15	57	-	8
Cs-135	37	20	20	3	6	6	-	8

- value less than 0.5%

* includes dose contribution from daughters

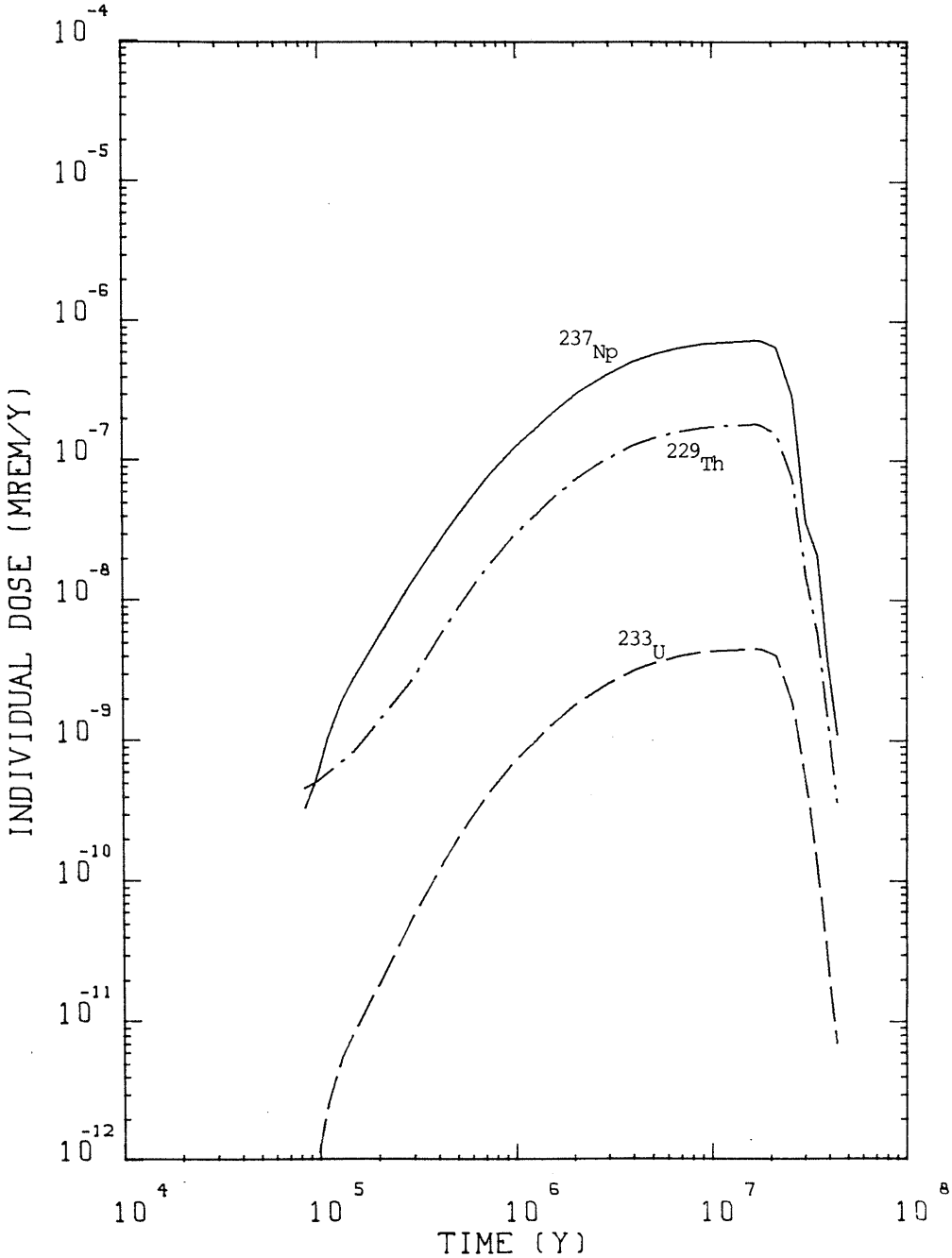

Figure 4.2: "Well scenario" doses from Np-237 decay chain.

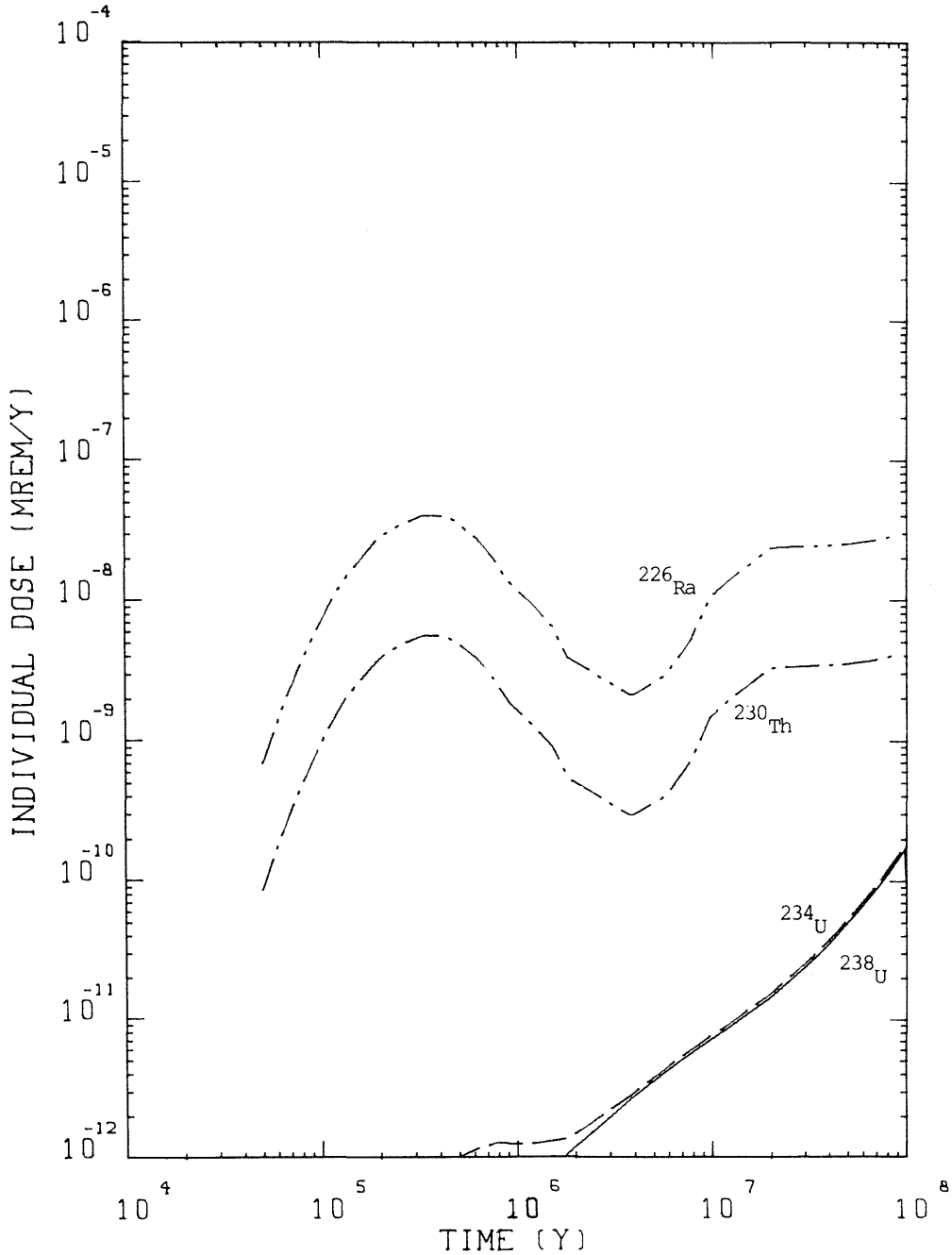

Figure 4.3: "Well scenario" doses from U-238 decay chain.

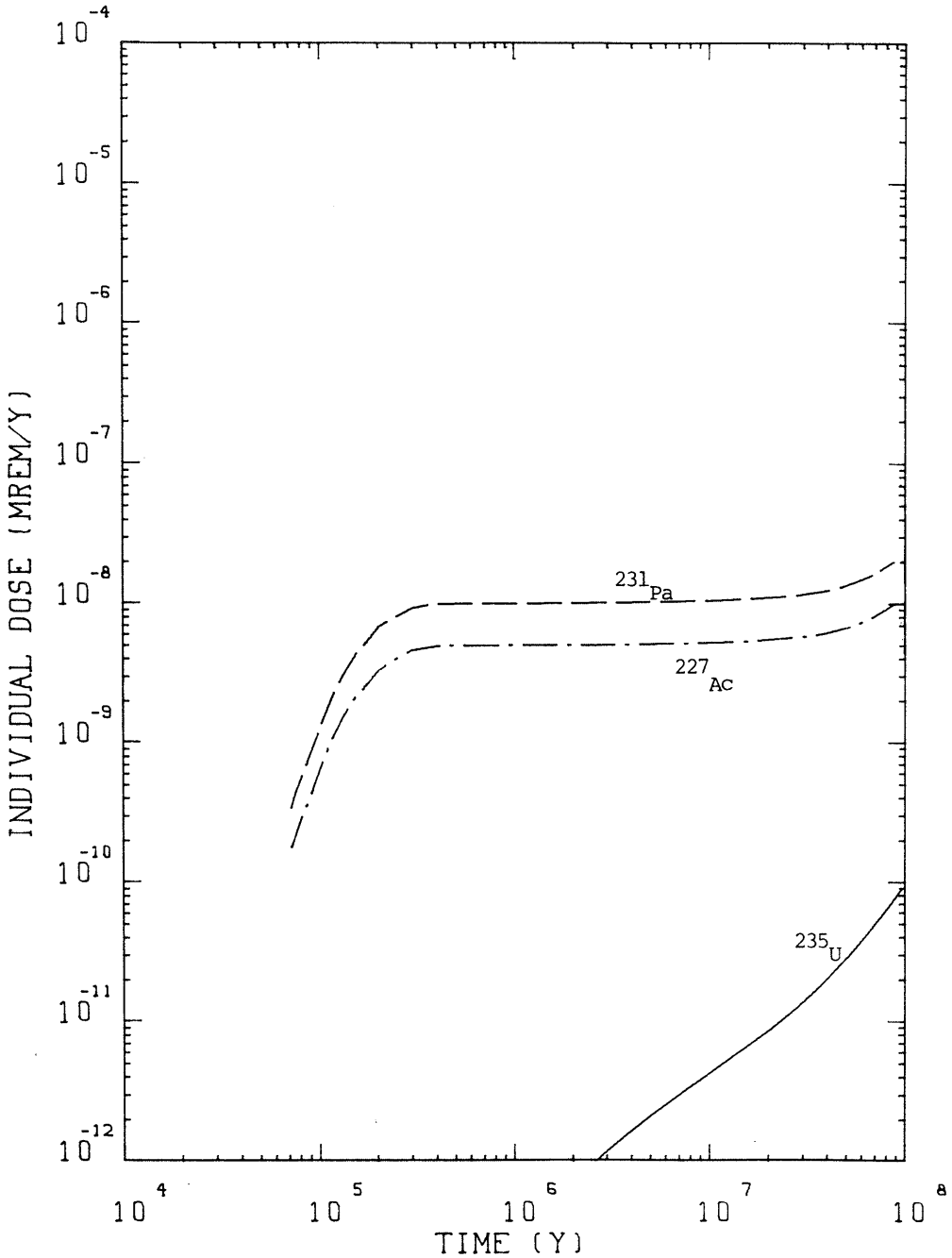

Figure 4.4: "Well scenario" doses from U-235 decay chain.

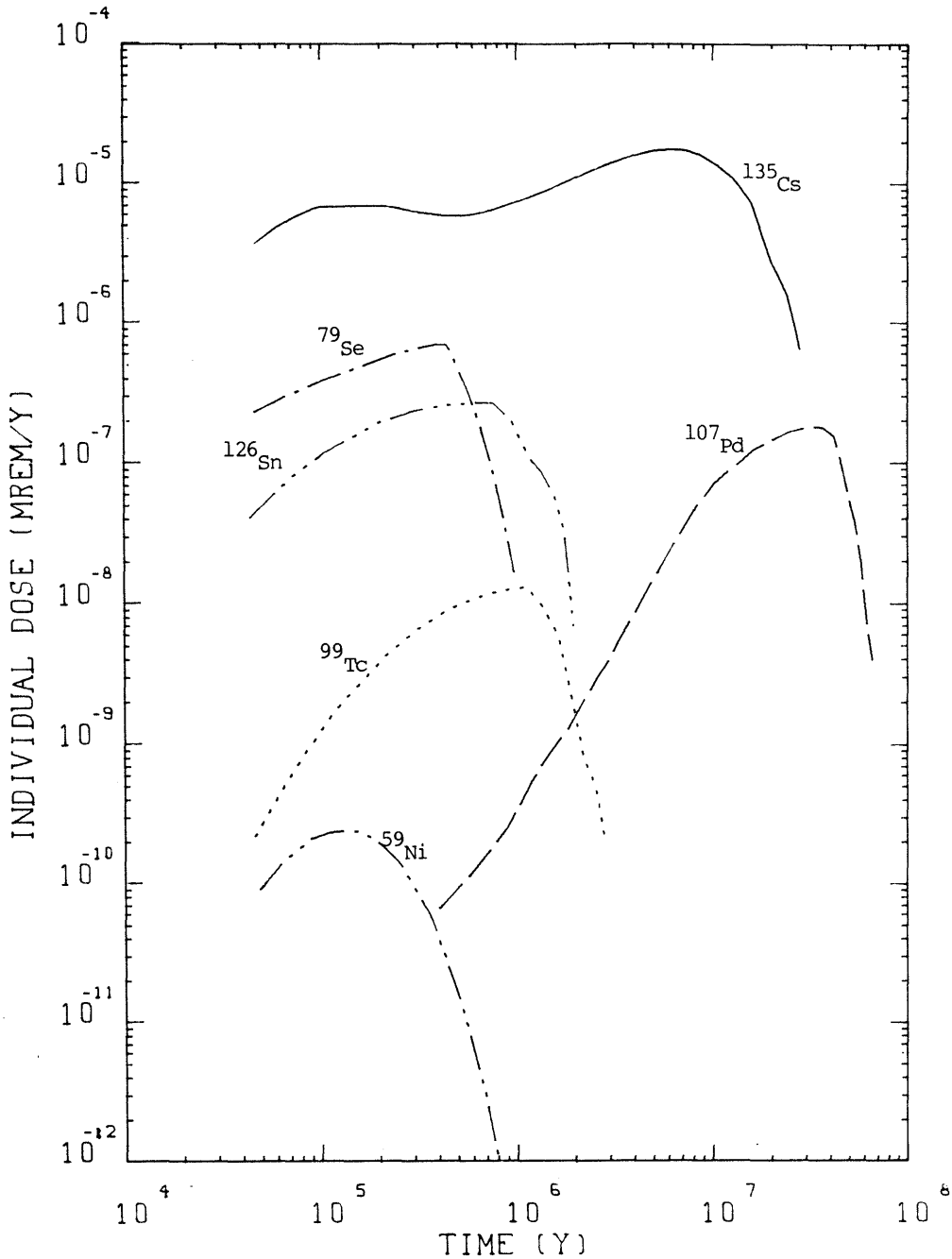

Figure 4.5: "Well scenario" doses from fission and activation products.

4.2 Conservative Kd in the biosphere

For this parameter variation the base case model was run for the base case region (Laufenburg) using conservative Kd values for the soil in the biosphere. The Kd value specifies the partitioning of an element between the soil and soilwater,so that a low Kd value, e.g. 0.001 m^3/kg, indicates the element is predominantly associated with the soilwater rather than the soil solid phase. Due to the problems encountered in determining these values as well as their variable nature in the literature, values are taken to the nearest order of magnitude for input into the model. A more detailed account of Kd values and their selection can be found in Jiskra, 1985. Table 4.3 compares the base case and conservative Kd values used in the modelling. The conservative values were chosen assuming stronger sorption or association of the element with the soil solid phase but aimed to still be reasonably realistic, consequently the extent of change in a Kd value was not the same for all elements. For example, the Kd for Th remained constant because under normal conditions it is already assumed to be very strongly sorbed to the soil solid phase. In contrast Se, which has a low Kd is not expected to become strongly sorbed to the soil solid phase even under conservative conditions, on the basis of existing experimental evidence. For this reason an increase by a factor of 5 to the base case value was considered "reasonably" conservative for Se.

The dose calculations were made in an identical manner to the base case with only the Kd parameter being varied. The results are shown in table 4.4 and presented graphically in figures 4.6, 4.7, 4.8 and 4.9 for the Np-237, U-238, U-235 chains and the fission products, respectively.

The maximum doses from each radionuclide all occur at the same time in the biosphere as in the base case and naturally the doses resulting from Th are identical since the parameter remained unchanged. In all the remaining cases the doses increased when the Kd was increased, however, this relationship was by no means linear and could not be easily predicted beforehand (Table 4.5).

Table 4.3:

Comparison of realistic and conservative Kd values used in the model.

Element	Kd Value (m^3/kg)	
	Base Case	Conservative
Ni	0.01	10.0
Se	0.001	0.005
Tc	0.001	0.5
Pd	0.01	10.0
Sn	0.1	10.0
Cs	0.1	2.0
Ra	0.1	1.0
Ac	0.1	5.0
Th	10.0	10.0
Pa	0.1	10.0
U	0.1	1.0
Np	0.01	1.0

Table 4.4:

Conservative Kd dose calculations for the Laufenburg region.

Radionuclide	LAUFENBURG REGION	
	maximum dose (mrem/y)	time (y)
Np-237	6.0×10^{-7}	2.1×10^{7}
U-233	2.9×10^{-10}	2.1×10^{7}
Th-229 *	1.0×10^{-8}	2.1×10^{7}
U-238	1.1×10^{-11}	1.0×10^{8}
U-234	1.2×10^{-11}	1.0×10^{8}
Th-230	3.1×10^{-10}	3.5×10^{5}
Ra-226 *	8.1×10^{-9}	3.5×10^{5}
U-235	6.2×10^{-12}	1.0×10^{8}
Pa-231	1.0×10^{-8}	1.0×10^{8}
Ac-227 *	8.0×10^{-10}	1.0×10^{8}
Ni-59	1.4×10^{-9}	
Se-79	7.1×10^{-8}	4.4×10^{5}
Tc-99	2.9×10^{-6}	1.1×10^{6}
Pd-107	1.1×10^{-6}	3.5×10^{7}
Sn-126 *	1.2×10^{-6}	8.8×10^{5}
Cs-135	9.7×10^{-6}	6.1×10^{6}

* includes dose contribution from daughters.

Table 4.5:

Comparison of the increase in dose from the base case relative to the increase in Kd.

Element	Factor for Increase in Kd	Factor for Increase in Dose
Np	100	17
U	10	1.3
Th	1	1
Ra	10	4
Pa	100	7
Ac	50	1.2
Ni	1000	122
Se	5	2
Tc	500	418
Pd	1000	121
Sn	100	88
Cs	20	11

This is exemplified by Pa, Np and Sn, whose Kd's were all increased by a factor of 100 whereas the corresponding doses increased by a factor of 7, 17 and 88, respectively. This reflects the relative importance of pathways in which the Kd has an influence on the total dose, for example, such as with soil-plant transfer factors of radionuclides. In all cases the factor by which the dose increased was less than the factor by which the Kd was increased. The maximum dose still resulted from Cs-135 (9.7×10^{-6} mrem/y) with the other fission products Tc-99, Pd-107 and Sn-126 resulting in doses of the same order of magnitude. For the actinides the Np-237 chain was the most significant, Np-237 and Th-229 resulting in doses of 6×10^{-7} mrem/y and 1.0×10^{-8} mrem/y, respectively. Of the remaining actinides only Pa-231 made any impact also giving a dose of 1.0×10^{-8} mrem/y.

Altering the Kd parameter has a marked effect on the importance of the individual exposure pathways to the total dose from each radionuclide as compared with the base case (table 4.6). In the base case 84% of the Np-237 dose results from the drinking water pathway whereas this accounts for only 5% of the dose when a conservative Kd is used. Instead, the crop pathways account for the remaining 95% of the dose, with root vegetables being the most important (73%). As a general trend, the cereals and root vegetables become more dominant pathways for the actinides and the drinking water pathway less significant, relative to the base case. For the fission and activation products the drinking water pathway is virtually insignificant (except for Se-79 where it contributes 15% of the dose);

the dose being distributed over the five foodchain pathways, namely milk, meat, cereals, leaf and root vegetables. The relative significance of each of these pathways depending on the particular radionuclide in question.

Table 4.6:

Contribution of each exposure pathway to the dose when a conservative Kd is used, expressed as a percentage.

Radio-nuclide	EXPOSURE PATHWAY							
	Drinking water	Milk	Meat	Leaf Vegetables	Cereals	Root Vegetables	Eggs	Fish
Np-237	5	-	-	9	13	73	-	-
U-233	72	1	-	2	15	10	-	-
Th-229*	30	-	-	6	28	36	-	-
U-238	72	1	-	2	15	10	-	-
U-234	72	1	-	2	15	10	-	-
Th-230	27	-	1	6	29	37	-	-
Ra-226*	15	2	1	3	59	20	-	-
U-235	72	1	-	2	15	10	-	-
Pa-231	4	-	28	6	9	53	-	-
Ac-227*	62	-	31	1	2	4	-	-
Ni-59	1	12	7	8	45	27	-	-
Se-79	15	4	80	-	-	1	-	-
Tc-99	-	66	1	2	20	11	-	-
Pd-107	1	12	7	8	45	27	-	-
Sn-126*	-	5	1	4	71	19	-	-
Cs-135	3	32	34	5	13	13	-	-

- value less than 0.5%

* includes dose contribution from daughters

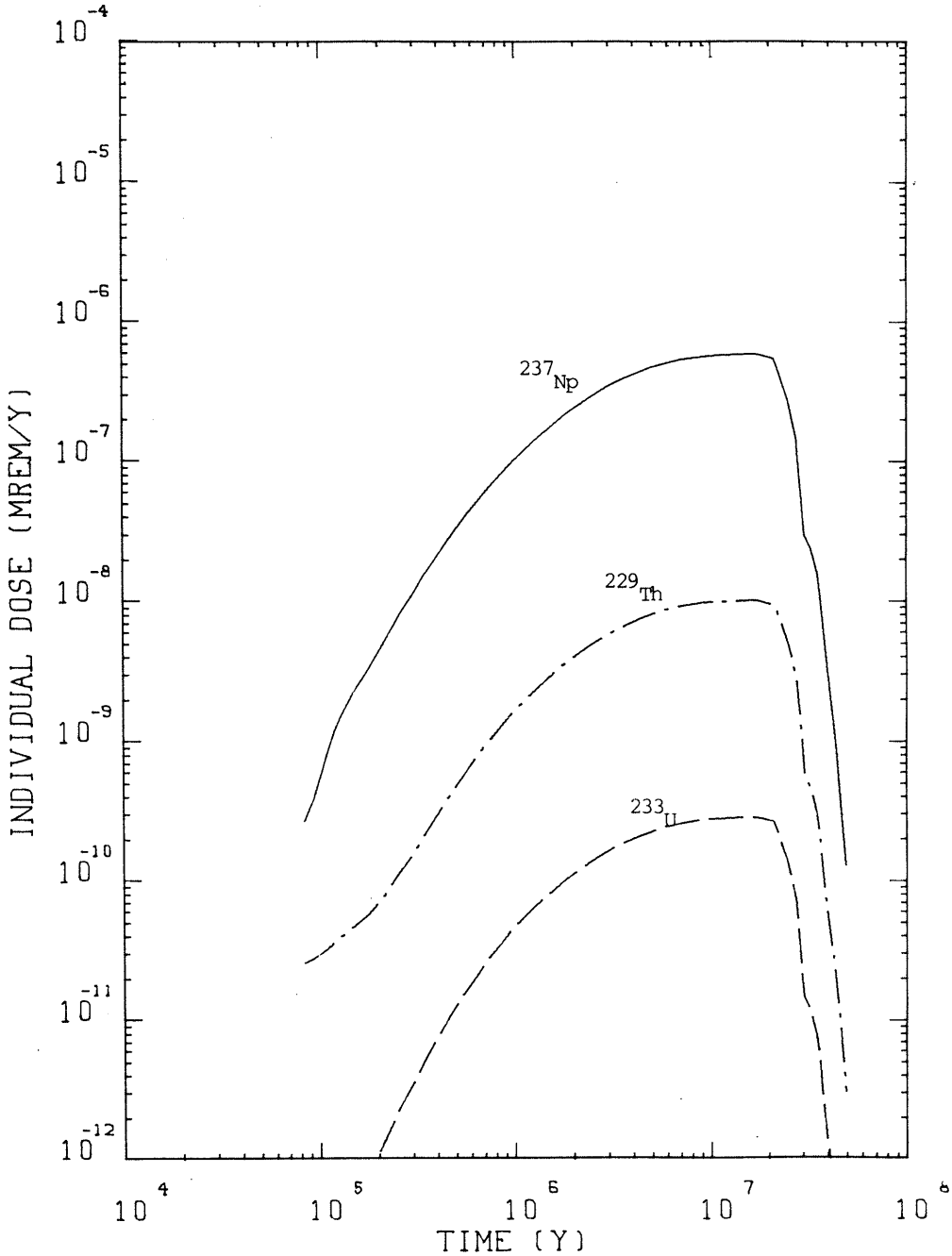

Figure 4.6: "Conservative Kd" doses from Np-237 decay chain.

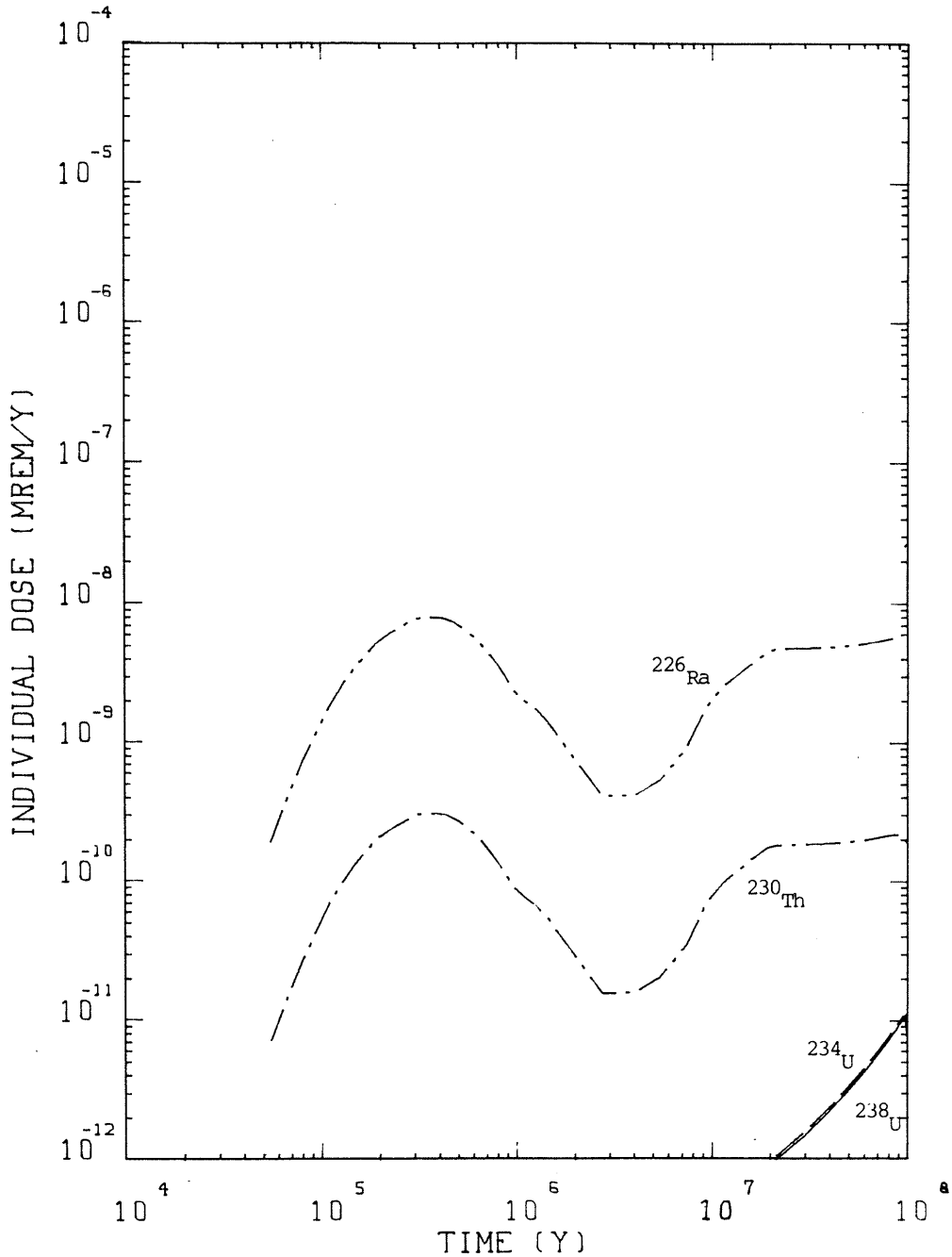

Figure 4.7: "Conservative Kd" doses from U-238 decay chain.

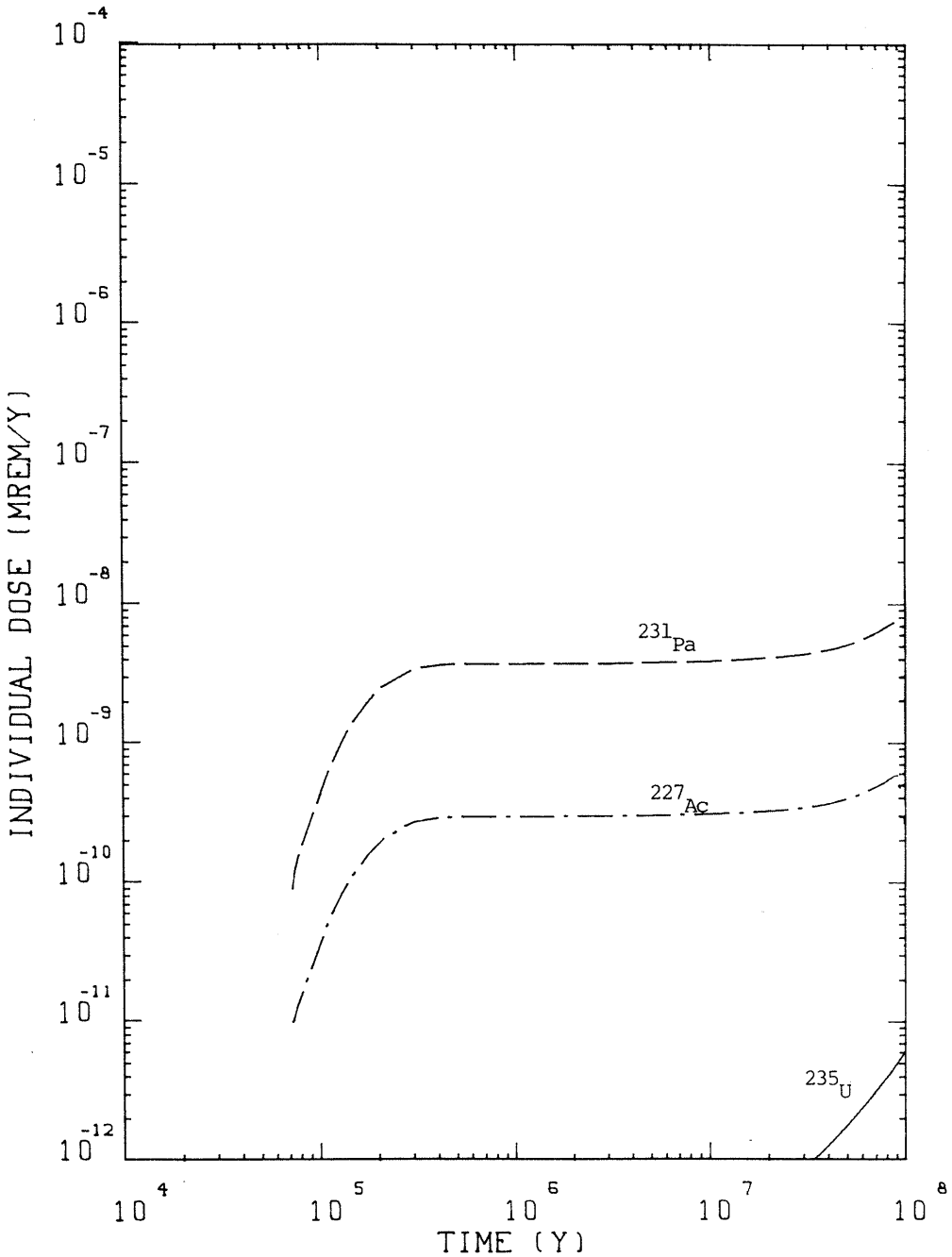

Figure 4.8: "Conservative Kd" doses from U-235 decay chain.

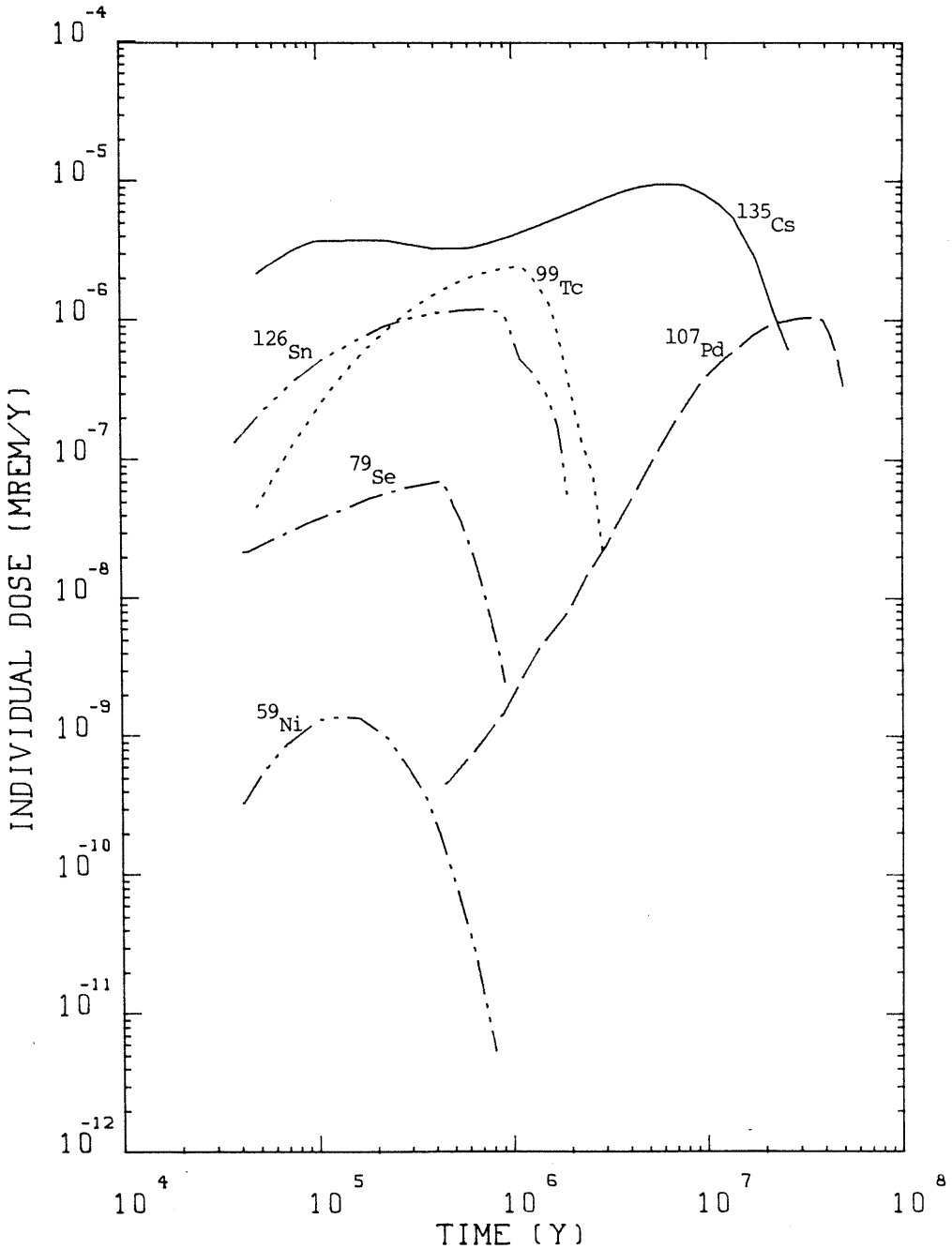

Figure 4.9: "Conservative Kd" doses from fission and activation products.

4.3 Soil Ingestion

BIOPATH, in common with a large number of other computer codes developed during the past decade, is designed to predict the environmental transport and subsequent dose to man of radionuclides released into the environment. The results tend to be somewhat similar since the transfer coefficients used to describe the transfer of radionuclides are usually derived from common literature in the absence of site specific values. There is a general consensus regarding the major exposure pathways requiring modelling. However, one exposure pathway which was initially neglected and has only been included more recently in some models, is that of soil ingestion by livestock. It has been shown that grazing animals ingest soil either involuntarily or sometimes voluntarily along with grass and that this exposure pathway can be of more importance (Thornton and Abrahams, 1983) than ingestion of feed or forage contaminated by root uptake. For this reason this exposure pathway is considered here, as a variation on the biosphere base case, where no account of it is taken.

4.3.1 Significance of Soil Ingestion

Interest in soil ingestion was initially directed towards the soil as a possible source of trace elements to livestock (Healy, 1970) and also with respect to heavy metal contamination of grazing land resulting from past mining and smelting activities (Thornton and Webb, 1975). However, in recent years its importance as an exposure pathway for radionuclide uptake via livestock has been recognized. Zach and Mayoh (1984) compared the importance of the soil-ingestion pathway with the root-uptake pathway for cattle, by calculating critical plant/soil concentration ratio values for soil-ingestion levels at which soil and feed each provide half the amount of ingested radionuclides. It was assumed that all the ingested radionuclides are equally available for absorption by the gastrointestinal tract and translocated to meat and milk, irrespective of their source. The feed ingestion rate was assumed at 50 kg (wet) per animal-day according to the USNRC77 value, with a 25% dry matter content. Assuming a soil ingestion value of 4%, soil ingestion predominated for all elements with a soil-to-plant concentration ratio less than about 0.04. The general conclusion was that for elements with low concentration ratios which indicate negligible root uptake, soil ingestion is usually the more important pathway whereas for elements with high concentration ratios the converse is true.

4.3.2 Modelling the Pathway

The soil ingestion pathway will impact on the doses to man via the consumption of livestock and livestock based produce. In the BIOPATH model these items are represented by meat and milk, respectively. To calculate the total daily intake of radionuclides by livestock as a result of soil ingestion, their concentration in the soil and the volume of soil consumed must be known. This latter factor depends on the season, quality of the pasture, as well as the pasture and livestock management. In the following section the literature relating to each of these factors are briefly reviewed to determine a suitable soil ingestion value for input into the model.

4.3.3 Review of Soil Ingestion Field Studies

4.3.3.1 Influence of Season

Soil ingestion is not uniform throughout the year. Healy (1968) studied dairy cattle in New Zealand and found the soil ingestion rate during the grazing season ranged from a low of 2% of dry matter intake during seasons of lush plant growth to a high of 14% when plant growth was poor. Similarly, Thornton and Abrahams (1983) who studied soil ingestion by cattle in Britain, found that during the early spring (april) when grass was in short supply, ingestion of soil was highest (up to 18%). In the early summer (June) soil ingestion had fallen below 2%, rising again in late summer (August). In the early summer pasture growth is very rapid and the fresh material, by the very nature of its height is subject to relatively little soil splash.

4.3.3.2 Pasture Quality and Management

On poor range conditions soil ingestion will be higher. Healy recorded ingestion rates as high as 2.2 kg per animal-day under these circumstances (average 0.5-1.2 kg per animal-day). The quality of pasture is also linked to the climate. Under semi-arid conditions in southern Idaho in 1974 soil ingestion rates of range cattle varied from about 1.2%-18.8%, with a median value of 6.2%, of dry matter intake (Mayland et al., 1975). On a more mesic wheat grass pasture in

Idaho during the same year, heifers ingested 0.73 and 0.99 kg soil per animal-day during June and August, respectively. Soil ingestion by grazing steers in Texas (1978) where pasture quality tends to be poorer, was found to depend on brush-management and was very strongly correlated to the percentage of bare ground (Kirby and Stuth, 1980). The soil ingestion as a percentage of dry-matter intake was 16.6%, 12.5% and 10.9% for mechanically tilled, chemically treated, and untreated pasture, respectively.

4.3.3.3 Livestock Management

The soil ingestion by diary cows maintained in situations where there is little or no grazing has been studied by Fries et al., (1982). In general lifestock frequently kept under these conditions ingest roughly 70% less soil than comparable cows kept in pasture. The percentage of dry matter intake ranged from 0.14% to 0.53% for cows confined to concrete, from 0.35% to 0.64% for those housed in freestall barns with soil bedding and from 0.60 to 0.96% for those with access to unpaved lots with no vegetation. Most forage and feed contained tracers of soil, which accounted for some, but not all, of the ingested soil. It does appear that animals may ingest some soil voluntarily.

Grazing sheep have also been demonstrated to ingest considerable quantities of soil which depending on the range conditions and soil structure, can make up to 30% of the dry matter intake (Thornton and Abrahams, 1983). This higher value is probably because sheep graze closer to the ground.

4.3.4 Summary

A value of 4% soil ingestion by weight of the non-soil dry matter intake for cattle would appear to be reasonable for use in BIOPATH. This is equivalent to 1 kg soil per animal-day since the pasture consumption is set at 100 kg wet weight per day (25% dry matter content), in the model.

4.3.5 Modelling Results

In table 4.7 the doses from the base case are compared with those resulting from the inclusion of the soil ingestion pathway in the base case. It can be seen that soil ingestion has a negligible influence, the doses only increase for the three radionuclides 222-Ra, Pa-231, Cs-135 and these increases are only small. However, this pathway is perhaps more significant than it would first appear. In the model the radionuclides arrive in the milk and meat via three separate pathways,

- through drinking water
- directly from soil ingestion
- indirectly from the soil, through the pasture

Table 4.8 shows the importance of each of these pathways to the final doses from milk and meat.

It must be stressed that all the calculated doses have been included in the table purely to demonstrate the significance of the different pathways and that in real terms doses less than 10^{-9} mrem/y are quite meaningless as previously demonstrated in the base case (chapter 3.1).

Despite this, the table shows that for many radionuclides direct soil ingestion results in significantly higher doses than the soil-plant pathway, e.g. U-238 gives a dose of 8.85×10^{-17} mrem/y from milk as a result of grazing pasture, whereas the dose from direct soil ingestion is nearly 3 orders of magnitude greater, being 9.32×10^{-14} mrem/y. Even so, this has either no or negligible impact to the overall dose from milk or meat in most cases because the drinking water pathway gives the greatest contribution to the overall dose of U-238.

Table 4.7:

Influence of soil ingestion to the base case doses.

Radionuclide	DOSE (mrem/y)	
	Base Case	Base Case with Soil Ingestion
Np-237	3.6×10^{-8}	3.6×10^{-8}
U-233	-	-
Th-229 *	1.0×10^{-8}	1.0×10^{-8}
U-238	-	-
U-234	-	-
Th-230	-	-
Ra-226 *	1.9×10^{-9}	2.0×10^{-9}
U-235	-	-
Pa-231	1.5×10^{-9}	1.7×10^{-9}
Ac-227 *	-	-
Ni-59	-	-
Se-79	3.5×10^{-8}	3.5×10^{-8}
Tc-99	7.0×10^{-9}	7.0×10^{-9}
Pd-107	8.7×10^{-9}	8.7×10^{-9}
Sn-126 *	1.4×10^{-8}	1.4×10^{-8}
Cs-135	8.4×10^{-7}	8.4×10^{-6}

* includes dose contribution from short-lived daughters.

- dose less than 1×10^{-9} mrem/y.

Table 4.8:

Importance of the individual pathways to the doses from milk and meat given in mrem per year.

Radio-nuclide	MILK			MEAT		
	water -milk	soil -milk (mrem/y)	soil-plant -milk	water -meat	soil -meat (mrem/y)	soil-plant -meat
Np-237	2.06E-12	5.24E-13	4.93E-15	2.35E-11	6.00E-12	5.64E-14
U-233	1.05E-12	2.68E-12	2.55E-15	2.76E-13	7.05E-13	6.70E-16
Th-229*	1.86E-13	4.05E-11	3.85E-14	2.13E-12	4.63E-10	4.40E-13
U-238	3.66E-14	9.32E-14	8.85E-17	9.61E-15	2.45E-14	2.33E-17
U-234	4.09E-14	1.04E-13	9.90E-17	1.07E-14	2.74E-14	2.60E-17
Th-230	1.88E-16	1.44E-12	1.37E-15	2.15E-15	1.65E-11	1.57E-14
Ra-226*	6.76E-12	3.08E-11	1.23E-13	4.35E-12	1.98E-11	1.98E-11
U-235	2.19E-14	5.59E-14	5.32E-17	5.76E-15	1.47E-14	1.40E-17
Pa-231	1.86E-14	4.74E-14	4.46E-16	8.50E-11	2.17E-10	2.04E-12
Ac-227*	2.43E-14	6.22E-14	3.11E-17	8.36E-11	2.14E-10	1.07E-13
Ni-59	1.35E-13	3.44E-14	1.72E-15	7.72E-14	1.97E-14	9.85E-16
Se-79	5.66E-10	1.44E-11	4.33E-12	1.30E-8	3.31E-10	9.92E-11
Tc-99	5.10E-10	1.30E-11	3.90E-11	5.84E-12	1.49E-13	4.47E-13
Pd-107	1.02E-10	2.61E-11	1.30E-12	5.85E-11	1.49E-11	7.46E-13
Sn-126*	2.57E-11	6.56E-11	6.56E-12	2.45E-12	6.26E-12	6.26E-13
Cs-135	3.04E-8	7.75E-8	1.55E-9	3.19E-8	8.12E-8	1.62E-9

* includes dose contribution from short-lived daughters.

5 SCENARIO VARIATIONS

5.1 Introduction

As has been previously stated the time scales considered for modelling a HLW repository and its consequent doses to man, are essentially long. In the safety analysis canister failure is pessimistically assumed to occur after 1000 years and transport times to the biosphere are between several thousand and tens of millions of years. On these timescales it is unrealistic to assume that factors within the biosphere will remain unchanged and for this reason some attempt has been made to model the probable changes and to assess their impact on the calculated doses to man.

During the last 10^7 years significant oscillations in our climate have occured characterised by colder glacial periods with warmer interglacial times dividing them. The extent and duration of each of these has not been constant, so that some interglacial periods have been somewhat warmer than the one being experienced at present. For this reason two main scenario variations have been considered, firstly, a tundra climate to represent peri-glacial times and secondly, a warmer climate than today representing a warmer interglacial. During the glacial periods themselves no biosphere, as modelled here, is assumed to exist. Each of these shall be discussed in the following sections with a brief discussion of how the parameters were derived from the literature to model the scenario as well as the model design. The dose calculations are then presented at the end of each section.

5.2 Tundra Scenario

It is generally agreed that a future glacial period will occur, the main uncertainty lies in predicting its onset. Considering our present knowledge of the last main glacial stage, described as the Würm in Europe, it can be divided chronologically into a number of distinct periods.

It commenced approximately 70,000 years ago after an interglacial period of high sea-level. Glaciation occured from about 70,000 to about 40,000 years ago. This was followed by an interval of warm climate resulting in a significant diminution of ice in northwestern Europe 40,000 to 30,000 years ago. A second glaciation (maximum of

the Würm stage) took place 30,000 to 10,000 years ago, with culmination at 18,000 years. The climate subsequently became somewhat warmer than today, however, 3,000 years ago present mountain glaciers began to form as a final cooler period set in, with peaks between 1000 B.C. and 500 A.D. and between 1750 and 1900 A.D. (Verhoogen et al., 1970). From this it can be seen not just one but several glacial episodes have taken place within the timescales predicted for radionuclide release from a repository.

A tundra climate would only apply to the areas around the ice sheet (periglacial) in a glacial period and will not be applicable if the entire region is covered by a permanent ice sheet (in which case the biosphere will be limited). It should also be noted that this would be a relatively temporary climate in Switzerland. At present, such a climate is typical of subarctic regions in the northern hemisphere, consequently present day studies of radionuclide transfer in those foodchains can be used to model a tundra scenario for the Swiss case.

5.2.1 Description of the Tundra Climate

The tundra is characterized by low mean annual temperatures resulting in a short growing season. The soil is often waterlogged because of a layer of permafrost (permanently frozen ground) and this will melt to some extent, depending on the summer temperatures. Because of the short growing season the vegetation is dominated by low shrubs, lichens and mosses. Work carried out in the north european tundra in Scandinavia and Greenland, has identified the lichen-reindeer-man pathway as dominant in the transfer of fallout radionuclides to man (Lidén, 1961). The major reason for this is that lichens are long-lived (50-100 years) symbiotic organisms composed of fungi and algae. They have persistent aerial parts and a relatively large surface area to volume ratio, making them efficient accumulators of radioactive debris (Aarkrog, 1979). The reindeer graze the lichen carpets directly, generally eating only the fresh top layer (Holm, 1977). The reindeer, in turn, constitute the major source of meat in the diet of the Scandinavian Lapps. This pathway results in a higher dose of radioactivity to the Lapps compared to direct inhalation. In this case the size of the dose depends on the amount of radioactive fallout, which reached a peak in 1964, when the annual radiation dose of the adult male Lapps from Cs-137 alone was approaching 200 mrem/y, and this has been decreasing steadily since that time.

5.2.2 Model Description

As the transfer of radionuclides in a tundra environment has already been studied to some extent, and the lichen-reindeer-man pathway identified as the most important with respect to man, it would seem reasonable to use this as the basis for the tundra scenario.

The movement of radionuclides in the tundra scenario is somewhat different to the base case and is shown schematically in figure 5.1. The main point to notice is that there is no root uptake of radioactivity by plants (lichen have no roots), the input is from the atmosphere. The radioactivity accumulates in an area of sediment, which subsequently becomes suspended into the atmosphere as a result of wind action. This activity is then deposited onto the lichen surface where it remains or is absorbed. The lichen is grazed by reindeer, constituting a major part of its diet. The reindeer meat is then consumed by man but the liver, blood etc. may also be utilized.

In the following section the individual processes in the scenario are discussed briefly (resuspension, deposition etc.) and the values selected for input into the model given. A comprehensive review of the literature discussing how the individual parameter values were selected can be found in Grogan (1984b).

5.2.3 Parameter Values in Tundra Model

5.2.3.1 Resuspension

Resuspension is defined as the insertion of surface contamination into the air by wind or mechanical stress such as ploughing (Sehmel, 1980). This expression is normally used to describe material deposited from the atmosphere (e.g. fallout) and then subsequently re-entrained or resuspended into the atmosphere, whereas suspension describes the subsequent insertion of particles into the atmosphere which were originally deposited onto a surface by some non-atmospheric process (e.g. industrial spillage). For convenience both processes are usually referred to as resuspension, since it is not possible to distinguish between the subsequent behaviour of the contaminants derived from the different sources.

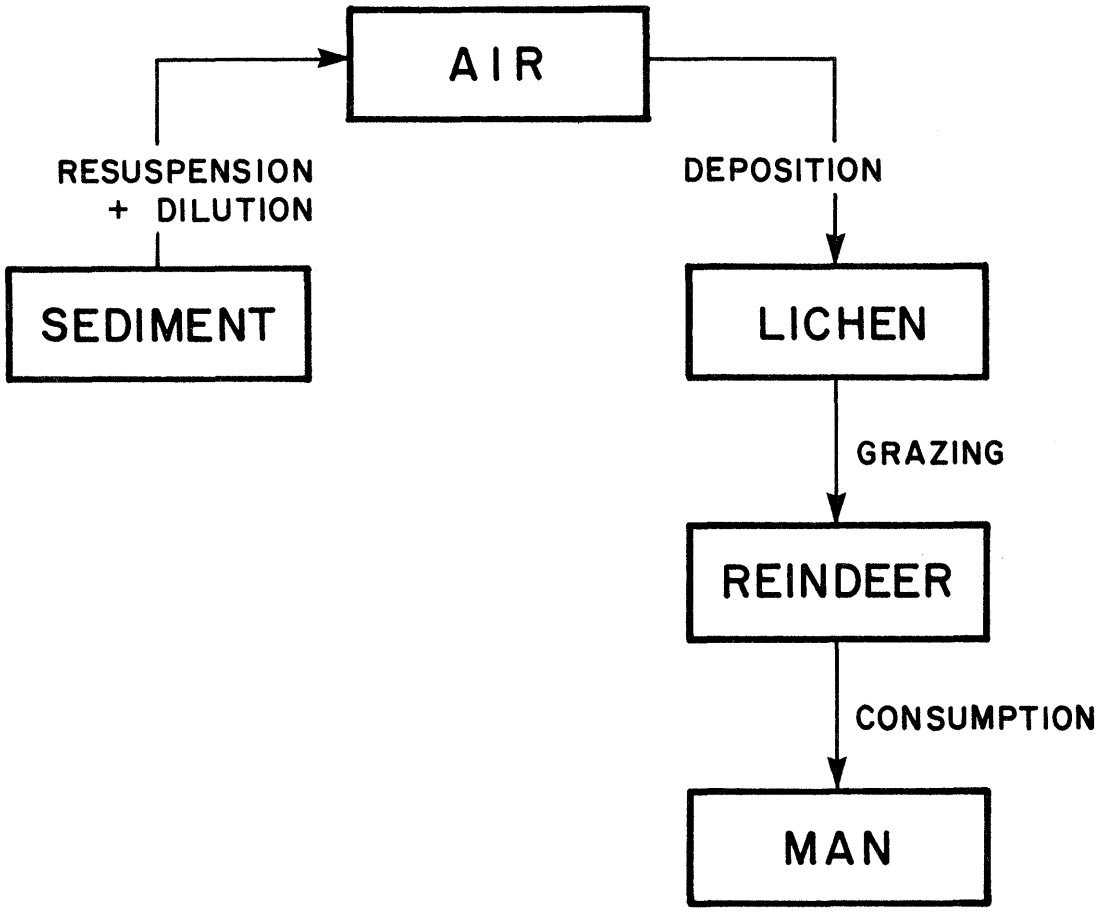

Figure 5.1: Tundra model showing the pathway of radionuclides to man

This is because after deposition, most pollutant particles lose their individual identity by becoming attached to host soil particles. The pollutants are then transported downwind attached to those soil particles. With reference to a release of radionuclides from a high level waste repository, the activity will arrive in the sediment from below, being carried up to the biosphere in the groundwater. However, once the activity reaches the soil, the radionuclides will become associated with the soil particles in a similar manner to radionuclides arriving as a result of aerial deposition. Consequently the literature values calculated for resuspension can be used; the

important factor to evaluate is the radionuclide activity in the surface soil layer, which will be subject to resuspension.

Most data on resuspension have been reported in the form of a resuspension factor K defined as,

$$K = \frac{\text{Resuspension air concentration}}{\text{surface concentration}} \quad \frac{(Ci/m^3)}{(Ci/m^2)}$$

The range of experimentally determined resuspension factors resulting from wind stress are summarised in figure 5.2, which is taken from Sehmel's review (1980). The resuspension factors range over 7 orders of magnitude with values for single experiments showing up to 4 orders of magnitude variation. Stewart (1967) has recommended a value of 10^{-6} m^{-1} for k under "quiescent conditions" outside and 10^{-5} m^{-1} for conditions of moderate activity. Kathren (1968) selected a conservative value of 10^{-4} m^{-1} for k, for fresh deposits, however, Bennett (1976) demonstrated that by using k = 10^{-5} m^{-1} the theoretical resuspended activity in the air from fallout deposits, in some cases, is greater than the actual measured concentrations, which is mainly due to activity transported down from the stratosphere. These higher values for fresh deposits are also given with half-life values, for the declining air concentrations, which vary between 35 days and 50 days. It is then assumed that it reaches a constant value of 10^{-9} m^{-1} after 1.8 years. The resuspension factors which have been evaluated from "aged" deposits tend to lie in a narrower range, $5x10^{-9}$ m^{-1} to 10^{-8} m^{-1}. In these cases the top 1 cm of soil is assumed to be resuspensible. Taking all these factors into account, a resuspension value of $1x10^{-8}$ m^{-1} would seem suitably conservative to apply to the tundra scenario.

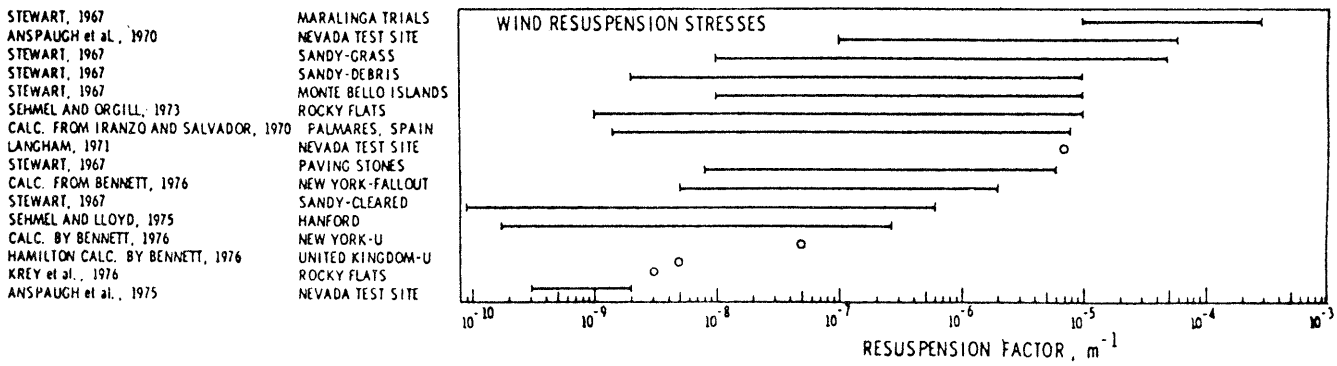

Figure 5.2: Experimentally determined windstress resuspension factors, taken from Sehmel (1980).

5.2.3.2 Dilution

The contaminated sediment that is suspended in the airstream is then carried an unknown distance, before being redeposited onto the ground surface (lichen).

The original radionuclide concentration resuspended into the atmosphere will also undergo dilution with uncontaminated air passing over the site. A figure of 1/10 dilution is selected as conservative, especially as the contaminated area will be of limited size.

5.2.3.3 Deposition

Deposition is a convenient term used to describe a complex of physical phenomena, which may result in the transfer of matter from

the atmosphere to the ground. The extent and rate of deposition depends on physico-chemical parameters of the material being deposited (e.g. particle size distribution), meteorological parameters, such as, windspeed, temperature etc., and surface parameters, such as vegetation type, density etc. (Nielsen, 1981). In 1953, Chamberlain introduced the concept of deposition velocity, Vg, which he defined in two separate ways. Either as:

$$Vg\ (z) = \frac{F}{X(x,y,z)}$$

where F is the flux to the surface (g or Ci, deposited per unit area per unit time), and X(x,y,z) is the volumetric concentration at some reference point (x,y,z) above the surface, or as:

$$Vg\ (z) = \frac{F_T}{X_T(x,y,z)}$$

where F_T is the total deposition per unit area (g or Ci), and $X_T(x,y,z)$ is the time integrated concentration. Vg is a function of reference height, however, Nielsen (1981) pointed out this height dependence is not of practical significance in numerical codes.

Two main types of deposition can be identified; dry deposition or fallout, which is the capture of particles on the ground or on the above ground plant parts, respectively, without the effect of rain; and wet deposition, the transport of particles by precipitation (rain, snow etc.) from the atmosphere to the ground or vegetation. Both types of deposition operate for gases, vapours and particles, but for the purposes of this report only the latter have been considered, as the radionuclides are assumed to be resuspended as particulates.

Considerable work has been carried out since the 1950's, on the mechanism of particle deposition. Much of the early work was carried out by Chamberlain with many other researchers contributing to our knowledge since that time. This subject is reviewed in Grogan (1984) and from this it was concluded that for the tundra scenario a value of $5x10^{-3}$ ms^{-1} is suitable to model deposition of contaminated particles to the lichen surface.

5.2.3.4 Lichen - Radionuclide Specific Data

Lichen show a marked tendency to accumulate radionuclides from atmospheric fallout and this has been a topic of study since the 1960's, Lidén, (1961). During this time considerable data have been collected on the behaviour of various radionuclides in the lichen,

mainly Pu-239, 240-Pu, Cs-137 and 90-Sr although limited data do exist for Mn-54, Tc-99, Pb-210, Am-241 and a number of other radionuclides (Holm, 1977).

Lichen tend to accumulate radionuclides because of their slow growth (live 50-100 years) in conjunction with having persistent aerial parts with a large surface area to volume ratio. It has been noted that all radionuclides are not distributed in a similar fashion in the lichen carpet (Holm, 1977; Holm and Rioseco, 1983) and also that reindeer only graze the upper layer. These two features can therefore be used as a basis to predict the rate of transfer of specific radionuclides to reindeer and hence man in the tundra scenario. This has the implicit assumption that future populations of animals will graze in a similar manner.

For the purposes of the tundra scenario the biological mean residence time (effective half-live) of each radionuclide is used as the basic parameter. This takes into account radioactive decay as well as any other physical processes resulting in a reduction of activity in the grazed layer. Experimental data do not exist for all the radionuclides of interest in a HLW repository consequently assumptions have to be made regarding their behaviour. For example, the behaviour of U and Th was assumed to be analogous to Pu whereas Np, Ra and the fission products were anticipated to be more akin to Cs in their behaviour. In view of the extremely mobile nature of Tc the upper value of 18 years recorded by Holm was chosen. Table 5.1 below shows the input values used in the model for each element.

Table 5.1:

Input values for mean residence time in lichen and fraction of ingested activity retained in reindeer.

Element	Mean biological residence time in lichen (y)	Fractional retention of ingested activity by reindeer
Np	10	1.0×10^{-3}
U	6	5.0×10^{-2}
Th	6	2.0×10^{-4}
Ra	10	2.1×10^{-1}
Ni	10	5.0×10^{-2}
Se	10	8.0×10^{-1}
Tc	18	8.0×10^{-1}
Pd	10	5.0×10^{-3}
Sn	12	2.0×10^{-2}
Cs	10	1.0

5.2.3.5 Transfer to Reindeer

By studying the feeding habits of reindeer one finds that they generally eat lichen during the winter and feed on grass, sedges, herbs, mushrooms and leaves from bushes, during the summer. However, the summer vegetation has a very low radioactive content compared to the lichen and therefore does not contribute significantly to the doses to reindeer (Aarkrog, 1979). In addition, the inhalation pathway is assumed to be negligible in comparison with the lichen consumption (Holm, 1977). Reindeer graze lichen for about 180 days each year, but it is almost impossible to measure the amount of lichen the wild animals consume per day. There are some observations that suggest reindeer consume only 0.7-1.0 kg dry lichen day^{-1} in the winter (Miettinen, 1979) but more usually a value between 2 kg and 4 kg dry weight per day is chosen. Miettinen (1979) and Holm (1977) select a value of 600 kg dry weight annually which seems adequately conservative to serve the purposes of the tundra scenario.

Plutonium uptake by reindeer has been studied in Lapland and Greenland by a number of different people (Holm, 1977; Holm and Rioseco, 1983; Miettinen, 1979; Mattsson, 1972). The highest concentrations are found in the liver (Miettinen, 1979) although measureable amounts occur in the bones. The mean residence time for plutonium in reindeer liver and cortical bone has been calculated to be about 5 years and 8 years, respectively (Holm and Rioseco, 1983; Jaakola et al., 1975). Holm (1977), calculated the fraction of ingested activity (plutonium) retained by reindeer to be about $(3\pm2)x10^{-5}$. Studies have also been made on Am-241 concentrations in reindeer bones (Miettinen, 1979) and it appears to be concentrated four times as efficiently as plutonium. Unfortunately, no measured values exist in the literature for the radionuclides of interest in the safety analysis so that values for the fractional retention of ingested activity were taken from ICRP 30 despite the fact they apply to man. This was because if one assumed elements such as Th and U had analogous behaviour to the plutonium the fractional retention value is significantly less than those actually measured for man (another large mammal).

In experimental studies it has been noted that other large animals living in the same regions as the reindeer, e.g. musk ox, elk had significantly lower activities of all radionuclides studied (Aarkrog, 1979; Miettinen, 1979). This was attributed to their different feeding habits. As a result the reindeer is a good animal to select to model the tundra foodchain since results will err on the conservative side for a dose assessment model.

5.2.4 Inhalation Pathway

In the tundra model the contaminated sediment is resuspended into the atmosphere prior to deposition onto lichen, which form the base of the critical foodchain considered. In view of the resuspended activity assumed in the environment it was felt the significance of direct inhalation by man should also be ascertained. These calculations were made very simply, taking the same activity of each radionuclide in the air as calculated in the tundra foodchain model. A mean inhalation rate for men and women was obtained from ICRP 30 (8.4×10^3 m^3/a) along with the relevant inhalation dose conversion factors. From this the maximum doses resulting from each radionuclide were calculated.

5.2.5 Dose Calculation Results

The maximum doses resulting from each radionuclide as a result of both ingestion and inhalation in the tundra climate were calculated and are presented in table 5.2. Time dependent doses were not

Table 5.2:

Maximum individual doses resulting from the ingestion and inhalation pathways in the tundra scenario.

Radionuclide	INGESTION (mrem/y)	INHALATION (mrem/y)
Np-237	1.3×10^{-11}	7.1×10^{-13}
U-233	2.7×10^{-11}	2.1×10^{-12}
Th-229 *	1.2×10^{-10}	2.3×10^{-9}
U-238	9.4×10^{-13}	7.4×10^{-14}
U-234	1.1×10^{-12}	8.2×10^{-14}
Th-230	4.3×10^{-12}	7.7×10^{-11}
Ra-226 *	2.8×10^{-10}	4.2×10^{-14}
U-235	5.6×10^{-13}	4.2×10^{-14}
Pa-231	1.2×10^{-12}	4.5×10^{-13}
Ac-227 *	9.3×10^{-13}	3.4×10^{-12}
Ni-59	2.1×10^{-13}	2.7×10^{-16}
Se-79	3.6×10^{-10}	2.2×10^{-15}
Tc-99	9.3×10^{-11}	1.8×10^{-15}
Pd-107	1.6×10^{-11}	1.4×10^{-12}
Sn-126 *	1.6×10^{-10}	1.0×10^{-13}
Cs-135	1.4×10^{-6}	4.0×10^{-12}

* includes dose contribution from daughters.

calculated as the computer code BIOPATH was not actually used directly to make these calculations. The input to the scenario was the maximum soil concentration of each radionuclide in the biosphere which was obtained from the time dependent results for the base case, calculated using BIOPATH.
For the tundra scenario the maximum doses resulting from ingestion are generally so small as to be meaningless. The doses are less than those resulting from the base case with the exception of Cs-135 where the dose increases from 8.4×10^{-7} mrem/y in the base case to 1.4×10^{-6} mrem/y in the tundra scenario, a factor of 1.7. Apart from Cs-135 the next most significant dose in the tundra scenario comes from Se-79 and is four orders of magnitude less! The inhalation pathway is generally less important than the ingestion pathway in the tundra scenario. Only the two thorium radionuclides and actinium give doses roughly an order of magnitude greater than from ingestion. But in each case the doses being considered are trivial.

In conclusion it would appear that a significant climatic change such as that produced during a glacial period will not greatly enhance the doses likely to be received by man.

5.3 Warmer Climate (Irrigation)

This is the second climatic variation considered in the biosphere and was chosen to represent an interglacial period with mean summer temperatures somewhat higher (2-3 ^{o}C) than those experienced at present. In such a situation a soil-water deficit is assumed to occur during the summer months necessitating crop irrigation for continued agricultural production. To model this system realistically careful consideration must be given to the source of the irrigation water, its salt concentration and the volumes applied, so that a stable system is defined, otherwise the soil salinity may increase to a level toxic to plants. In this case for the biosphere modelling, the irrigation water is assumed to be contaminated as a result of radionuclides entering the groundwater from a HLW repository.

It is appreciated that large uncertainties exist in predicting the agricultural practices and techniques etc., likely to dominate well into the future. For this reason irrigation as it occurs today was modelled with present day crops and foodchains so that one can gain an idea of the range of possible consequences it may have.

There are two main pathways to man resulting from irrigation which require consideration, these are: -

i) irrigation - crop - man
ii) irrigation - pasture - cow - man.

The third pathway, irrigation - soil - crop - man, is not considered here as it is essentially the same as the base case. In both instances the contamination occurs in the groundwater and enters the soil where it is available for root uptake. The process by which it enters the soil is irrelevant, whether it be artificially via irrigation or by natural transport up to the soil layer from the water table, the net effect is the same. It may be noted that the situation in ii) above is somewhat different to i), in that the pasture is grazed throughout the summer whereas the foodcrops are generally only harvested once in a season.

In order to model the impact of a warmer climate the change in dose as a result of including irrigation into the base case model was compared with the base case doses. The inclusion of irrigation into the model impacts upon the milk and meat pathways due to the pasture receiving irrigation, it also affects the cereal and leaf vegetable pathways as a direct result of their irrigation. The significance of irrigation with respect to influencing doses to man depends on a number of different factors namely, the extent of groundwater contamination by the radionuclides, the rate and duration of irrigation, the extent of foliar uptake by crops and finally the radionuclide in question. These factors are considered below in fairly general terms and the literature reviewed for data applicable to this pathway.

5.3.1 Groundwater Contamination

The flow of individual radionuclides in the groundwater comprises the source term for their activity in the irrigation water and is very low, e.g. Np-237 $4x10^{-11}$ M/yr (Hadermann and Rösel, 1985). The irrigation water will result in a continuous deposition of radionuclides onto the plant surface is available for plant uptake. It should be noted that the total deposited activity is unlikely to remain on the plant since a proportion will be washed off as further water arrives at the plant, resulting is some form of equilibrium

between the two processes.

5.3.2 Irrigation Rate

The extent of irrigation depends on the climate. For the warmer climate that is modelled, a deficit of water is envisaged during the summer months necessitating irrigation to supplement this. An annual rainfall of 800 mm is assumed, with a potential evapotranspiration of 1000 mm and a 100 mm surplus that results from the colder months and leaches down the soil profile, this leaves a 300 mm deficit that is made up by irrigation.

In this scenario it is assumed that some form of overhead irrigation occurs, such as sprinklers etc., resulting in foliar deposition of activity. Any alternative irrigation method which supplies water to the soil at the base of the crop will not directly contaminate the crop. In which case the effect of irrigation will be very similar to the base case where the contamination arrives through the soil from the groundwater.

5.3.3 Crop and Mechanism of Foliar Uptake

The extent of contamination is closely related to the crop density i.e., mass of plant material per unit area. This depends on the sowing rate for the crop as well as the stage of plant development when irrigation occurs. At the early stages of plant growth only a small fraction of the depositing activity is retained due to their small size. The interception increases with increasing leaf area which has a maximum for the major crops in June or July in Switzerland. As the plant age increases further, leaves begin to die off, reducing the effective contamination. Specific crop factors such as morphology will exert a moderating influence, e.g. crops with large flat leaves intercept more water than those with tall narrow leaves etc. In order to model this system a single crop density value was assigned to each of the three crop types (root vegetables were not considered). These values apply to mature crops and therefore overestimate the importance of irrigation in the early season when the plants are small. The values are listed below and were average values obtained from IAEA recommendations (IAEA, 1982) and agricultural statistical reports (Schweizerisches Bauernsekretariat).

Table 5.3:

Crop density values used in the model.

Crop	Density (Y) (kg/m^2)
pasture	0.85 (per cut)
cereal	0.5
leaf vegetables	2.2

Very little work has been carried out explicitly on the extent of foliar uptake of radionuclides as a result of irrigation. Most experiments are designed to study foliar uptake subsequent to a single application of radionuclides. In order to ensure almost all the activity is deposited on the plant the radionuclides are concentrated in a small volume of liquid which is applied as an aerosol with the radionuclides frequently in an ionic form. After treatment the plants are allowed to dry and are protected to prevent any washoff from rainfall.

It can be seen that contamination is generally carried out under optimal conditions for foliar uptake. Most of the experiments then study the degree of retention of the radionuclides with time and the extent of their translocation depending on the stage of plant growth at the time of contamination and harvest. No work appears to have been carried out on the effect of applying a small amount of activity in a large volume of water, as would be the case in this scenario. Despite these problems it is still possible to derive some useful data from the present literature which is applicable to the irrigation pathway. A selective literature review can be found in Grogan (1984a) from which the model parameter values were derived. Using these values a very simple irrigation model was set up as shown in the next section.

5.3.4 Irrigation Model

A very simple model was selected to model the transfer of radionuclides to crops as a result of irrigation. This was mainly because many of the foliar uptake experiments are conducted under different circumstances to those in the field and may be less applicable. Namely, the radionuclides are concentrated in a small volume of liquid whereas in the proposed scenario the radionuclide would be dispersed in a large volume of water. In addition to this very little radionuclide specific data exists for the radionuclides of

interest in a HLW, with the exception of Cs. The model is shown below and is common with that used by KBS (Bergström, 1983) and GRS (1980) and as it can be seen is not radionuclide specific

$$C = \frac{N \cdot T_V \cdot W \cdot C_W}{Y}$$

where - C activity in the crop (Bq/kg)
C_W activity in the irrigation water (Bq/l)
N retention factor
T_V retention half-life (d)
W irrigation rate (l/m^2d)
Y crop density (kg/m^2)

The values for N and T_V are shown below and were selected as average values on the strength of the experiments described in the literature.

Table 5.4:

Crop retention factors and retention half lives used in the model.

Crop	N	T_V (d)
pasture	0.3	14
cereal	0.05	30
leaf vegetable	0.3	14

The model input value was the maximum groundwater activity of each radionuclide and was obtained from the BIOPATH calculations for the base case region. The doses which resulted from including irrigation as a contamination pathway were then compared with the base case values.

5.3.5 Results

Including the irrigation pathway increases the maximum individual doses in all cases although the extent of this depends on the particular radionuclide (table 5.5). The doses increase the most for Se-79 (57%) with about 40% increase being average for most radionuclides. The Sn-126 doses are least affected showing only a 5% increase. In general the impact of this pathway was least for those radionuclides where the root vegetable pathway was important in the

base case in conjuction with the drinking water pathway being relatively unimportant (≤ 30%) (see table 3.2). The former reflects the exclusion of the root vegetables from irrigation whilst in the latter case, the drinking water is less important when the radionuclide concentration in the groundwater is low, consequently irrigation does not deliver high concentrations of the radionuclide to the plant.

Table 5.5:

Maximum individual doses with irrigation in the base case compared with the base case doses.

Radionuclide	Base Case (mrem/y)	Base Case with Irrigation (mrem/y)	% increase
Np-237	3.58×10^{-8}	4.94×10^{-8}	38
U-233	2.17×10^{-10}	3.12×10^{-10}	44
Th-229 *	1.01×10^{-8}	1.13×10^{-8}	12
U-238	8.80×10^{-12}	1.21×10^{-11}	38
U-234	9.50×10^{-12}	1.32×10^{-11}	39
Th-230	3.08×10^{-10}	3.45×10^{-10}	12
Ra-226 *	1.93×10^{-9}	2.50×10^{-9}	30
Tc-99	6.99×10^{-9}	8.23×10^{-9}	18
U-235	4.71×10^{-12}	6.70×10^{-12}	42
Pa-231	1.50×10^{-9}	1.75×10^{-9}	17
Ac-227 *	6.66×10^{-10}	9.19×10^{-10}	38
Ni-59	1.15×10^{-11}	1.62×10^{-11}	41
Se-79	3.48×10^{-8}	5.43×10^{-8}	56
Pd-107	8.74×10^{-9}	1.23×10^{-8}	41
Sn-126 *	1.37×10^{-8}	1.44×10^{-8}	5
Cs-135	8.43×10^{-7}	1.05×10^{-6}	25

* includes dose contribution from daughters

6 CONCLUSIONS

In this study negligibly low doses to man in the long term, resulting from the construction of a HLW repository, have been calculated. The largest individual dose calculated results from Cs-135 and is 8.4×10^{-7} mrem/y at 6.1×10^{6} y, while Np-237 gives the most significant actinide dose, 3.6×10^{-8} mrem/y at 2.1×10^{7} y. The relative importance of the individual radionuclides also depends strongly upon the geosphere transport calculations, for this reason it is likely that the significance of the fission and activation products has been overestimated in comparison to the actinides.

For the biosphere modelling the size of the first compartment in the model naturally proved critical in determining the magnitude of the resultant doses. This relationship is approximately linear, so that the doses in the Hellikon region are a factor of twenty greater than those in the base case region (Laufenburg). This directly reflects the decreased water flux of the first model compartment and hence decreased dilution of the radionuclides in the groundwater.

The drinking water is the dominant exposure pathway for many of the radionuclides in the base case. But, other exposure pathways through the foodchain are also important, for example 63% of the dose from Sn-126 was via cereals with only 12% resulting from drinking water. Thus, if a lower value is selected for drinking water consumption than the relatively high one of 730 l/y, the dose contribution from drinking water is reduced, but the impact to the overall dose is small.

The response of the model to a parameter variation cannot easily be predicted. This is because of non-linear coupling of many of the parameters. Altering the Kd demonstrated this; an increase by a factor of 100 in the Kd value for Np, Pa and Sn results in the doses increasing by a factor of 17, 7 and 88, respectively. The maximum dose in the base case region when conservative Kd values were input was again from Cs-135 (9.7×10^{-6} mrem/y) with Np-237 giving the highest dose from the actinides (6.0×10^{-7} mrem/y).

Two scenario variations were modelled very simplistically. These were a tundra scenario and a warmer climate necessitating irrigation for agricultural production. The doses calculated for a tundra climate were generally so small as to be meaningless; only the dose from Cs-135 (1.4×10^{-6} mrem/y) was greater than in the base case (factor 1.7). The irrigation scenario resulted in a small increase in all the doses, the maximum increase was 56% for Se-79.

In addition to the conclusions drawn directly from the modelling results, it is emphasised that validation work be carried out for the model itself. In particular the method of compartmentalisation used to describe the biosphere and the transport between the compartments. Site-specific data would help to reduce the large range in some of the

model parameters which are taken from the literature and are therefore generic in nature. Although changes in eating habits are inevitable over the long time-scales considered, by assuming a 3000 kcal demand per day such uncertainties are hopefully minimised.

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