TAME – The Terrestrial-Aquatic Model of the Environment: Model Definition

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This report was prepared on behalf of Nagra. The viewpoints presented and conclusions reached are those of the author(s) and do not necessarily represent those of Nagra.
Preface

The Waste Management Laboratory at the Paul Scherrer Institute is performing work on development and testing of models as well as on acquisition of specific data relevant to performance assessments of Swiss nuclear waste repositories. These investigations are being undertaken in close cooperation with, and with the partial financial support of, the National Co-operative for the Disposal of Radioactive Waste (Nagra). The present report is issued simultaneously as a PSI-Bericht and a Nagra Technical Report.

Vorwort


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Abstract

*TAME* - the *Terrestrial - Aquatic Model of the Environment* is a new computer model for use in assessments of the radiological impact of the release of radionuclides to the biosphere, following their disposal in underground waste repositories. Based on regulatory requirements, the end-point of the calculations is the maximum annual individual dose to members of a hypothetical population group inhabiting the biosphere region. Additional mid- and end-points in the *TAME* calculations are dose as a function of time from eleven exposure pathways, foodstuff concentrations and the distribution of radionuclides in the modelled biosphere.

A complete description of the mathematical representations of the biosphere in *TAME* is given in this document, based on a detailed review of the underlying conceptual framework for the model. Example results are used to illustrate features of the conceptual and mathematical models and to highlight areas where simplifications used to define the example biosphere must be revised in future applications. The end-point of dose is shown to be robust for the simplifying model assumptions used to define the biosphere for the example calculations.

*TAME* comprises two distinct sub-models - one representing the transport of radionuclides in the near-surface environment and one for the calculation of doses to individual inhabitants of that biosphere. The former is the result of a detailed review of the modelling requirements for such applications and is based on a comprehensive consideration of all features, events and processes (FEPs) relevant to Swiss biospheres, both in the present-day biosphere and in potential future biosphere states. Representations of the transport processes are derived from first principles. Mass balance for water and solid material fluxes is used to determine the rates of contaminant transfer between components of the biosphere system. The calculation of doses is based on existing representations of exposure pathways and draws on experience both from Switzerland and elsewhere.

Both sub-models are described in full and default datasets (i.e. those which are not strongly site specific) are included for the example calculations. The performance of *TAME* is compared and contrasted with that of earlier biosphere models used in Switzerland, demonstrating the correct working of the new model by limiting the scope to be similar to that of the older models, and illustrating the advantages of the new model when the full *TAME* dataset is used.
Zusammenfassung


Beide Teilmodelle werden vollständig beschrieben und Standard-Datensätze (d.h. diejenige Datensätze, die nicht stark standortspezifisch sind) werden in den Beispielrechnungen berücksichtigt. Um das richtige Funktionieren des neuen Modells prüfen zu können, werden die Resultate von TAME auch mit denen der früher in der Schweiz benutzten Biosphärenmodellen verglichen. Dazu musste der Rahmen von TAME den einfacheren, älteren Modellen angepasst werden. Der Vergleich illustriert die Vorteile des neuen Modells, falls der vollständige TAME-Datensatz benutzt wird.
Résumé


Ce document donne une description complète des représentations mathématiques de la biosphère dans *TAME*. Celles-ci sont basées sur l’examen détaillé de la structure conceptuelle sous-jacente du modèle considéré. Des cas-tests permettent, d’une part d’illustrer le comportement des modèles conceptuels et mathématiques, d’autre part de mettre en lumière les domaines où les simplifications utilisées pour définir la biosphère considérée doivent être révisées dans les applications futures. On montre que la valeur finale des doses est peu sensible aux hypothèses simplificatrices du modèle, faites pour décrire la biosphère dans les calculs des cas-tests.

*TAME* comprend deux sous-modèles distincts: l’un représente le transport des radionucléides dans l’environnement proche de la surface; l’autre calcule les doses reçues par les habitants de la biosphère. Le premier sous-modèle provient d’un examen détaillé des exigences de modélisation pour de telles applications. Il s’appuie sur la compréhension de toutes les caractéristiques, évènements et processus (FEPs) relatifs aux biosphères suisses, tant en ce qui concerne la biosphère actuelle, que les états possibles des futures biosphères. Le bilan de masse pour les flux des matériaux solides et d’eau sont utilisées pour déterminer des faux de transport entre les compartiments de la biosphère. Le calcul de la dose est basé sur des représentations existantes des voies d’exposition. Il tient compte de l’expérience à la fois en Suisse et à l’étranger.

Les deux sous-modèles sont entièrement décrits. Des bases de données par défaut (c’est-à-dire celles qui ne sont pas fortement liées au site) sont inclues pour les calculs des cas-tests. La comparaison des performances de *TAME* avec les résultats fournis par les modèles de biosphère utilisés précédemment en Suisse permet de démontrer le fonctionnement correct de ce nouveau modèle, dans un domaine de validité comparable à celui des anciens modèles, et d’illustrer ses avantages lorsque toute la base de données de *TAME* est utilisée.
TAME: The Terrestrial - Aquatic Model of the Environment:
Model Definition.

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1. Introduction: Biosphere models for waste disposal assessments

*TAME* - the *Terrestrial-Aquatic Model of the Environment* is designed for use in illustrative assessments of the radiological impact of release of radionuclides to the biosphere, specifically those arising from the disposal of radioactive waste in geological repositories. In this context the term biosphere is understood to encompass not only those parts of the environment used for food production and as a source of water supply (including near-surface aquifers) but also those which are implicated in the exposure of inhabitants to environmental concentrations of radionuclides which do not directly involve the food-chain. Also included in this definition are those intermediate media which play a rôle in the transport of contaminants, but with which potentially exposed individuals are not in direct contact.

The goal of biosphere model calculations in such assessments is the estimation of the potential radiological exposure of hypothetical individuals living at locations associated with sites of radioactive waste repositories. This is in accordance with the regulatory guidelines laid down by the *Hauptabteilung für die Sicherheit der Kernanlagen* (HSK/KSA, 1993) which require that the annual dose to representative individuals in the exposed population be not greater than 0.1 mSv y\(^{-1}\). The objective is thus given in terms of a potential effect on the health of the exposed individuals (for comparison, the range of exposure in Switzerland to natural background radiation is 1 - 10 mSv y\(^{-1}\)). The use of dose as the end-point of assessment calculations is also useful since it allows the releases of different radionuclides to be compared on an equal basis according to their health effects. N.B. in *TAME*, the definition of dose corresponds to the ICRP 26 definition of dose (ICRP, 1977)\(^1\).

\(^1\) The term *dose* (or *annual individual dose*, or *annual dose*) is used here as a short-hand way of referring to the potential effects on health due to the bodily absorption of radiation. According to HSK/KSA (1993), this quantity is defined as follows: *Dose* - Effective dose from internal and external exposure, which is the sum of dose equivalents weighted with tissue weighting factors for all organs and living tissues. The unit of dose is the sievert (Sv)

\[
E = \sum_{T} w_{T} H_{T}
\]

where \(E\) = effective dose, \(w_{T}\) = tissue weighting factor, contribution to total risk from tissue or organ \(T\), \(H_{T}\) = dose equivalent to organ \(T\), sum of the absorbed doses from all types of radiation weighted with the radiation weighting factors.

As used in performance assessments, the appropriate organ doses (\(H_{T}\) above) must take into account that intake over a specified time (i.e., the annual intake in the framework of *TAME*) can lead to dose arising over the lifetime of the individual. In Switzerland, radiological protection legislation recognises this and recommended values used for dose per unit intake (STRAVO 1994, page 100) are based on the *committed effective dose* assuming a 50 year adult life (ICRP, 1990). However, the basic definition of dose in the same legislation (STRAVO 1994, pages 57 - 58) is rather less precise. In the ICRP (1977) system (as used in to define the *TAME* database) the same quantity was named the *committed effective dose equivalent* and was calculated in the same way, but with slightly different numerical inputs. The important point is that the HSK and *TAME* usage is the same.
In earlier assessments in Switzerland (NAGRA, 1985; BÖHRINGER et al., 1986), biosphere modelling was carried out with a number of different biosphere models and using a variety of tools, some including the aquatic environment - i.e. rivers and lakes (BÖHRINGER et al., 1986; GROGAN et al., 1991) and others not (BAEYENS et al., 1991), but all including representations of the terrestrial components of the biosphere - soils and farmland, as well as a detailed network of exposure pathways. The methodology for representing the biosphere was thus well defined but there was no fixed framework. However, experience gained in the BIOMOVS studies (BERGSTROM, 1988; GROGAN, 1989; SMITH, 1989; JONES, 1990; ZEEVAERT, 1990) and elsewhere (NEA, 1993) demonstrated how generic, flexible models can be used to advantage in modelling reference biosphere scenarios. Hence the requirement for TAME was identified - a generic model including representations of the aquatic as well as terrestrial parts of the biosphere.

A Reference Biospheres approach is adopted because the repository near-field is designed to contain the radionuclides for long periods and the geological location is chosen to maximise the transit times along groundwater flow paths of those radionuclides which are released from the repository. The time between emplacement and release to the near-surface environment can therefore be very long. Consequently, the exact state of the biosphere at the time of release cannot be known with any degree of precision.

As used in the context of TAME, Reference Biospheres means that a set of possible biosphere states is defined such that the basis for the calculation of doses in each is reasonable given the available information and the calculated doses are not significantly underestimated for the situation being represented. With the aid of geo-historical records, the set of reference biosphere scenarios can broadly be based on the kinds of environments inhabited by humankind in the present-day. These human behaviour analogues are, to a large extent, determined by climatological and topographic features.

In order to provide the flexibility required for such diverse applications as the representation of periglacial tundra climates and arid hot conditions, it was realised that a detailed review of the Features, Events and Processes (FEPs) governing the transport and accumulation of radionuclides in the biosphere was necessary, as well as a review of the way in which the exposure pathways were modelled.

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2 N.B., this is not precisely the same usage of the term Reference Biospheres as in the BIOMOVS II Working Group on Reference Biospheres! The Reference Biospheres methodology from BIOMOVS II was developed in parallel, though somewhat after the development of TAME (BIOMOVS II, 1996a).
Figure 1-1: Schematic representation of the Reference Biospheres methodology. Development of a model proceeds from top-left to bottom-right. The current Reference Biospheres methodology enables the conceptual model to be defined. The mathematical models are defined by the parameterisation of the FEPs to be included in the conceptual model. In practice, there is considerable feedback, e.g. the use of a compartment model is influenced by previous work (e.g. BIOMOVS). In turn, this strongly influences the parameterisation of the FEPs.
The framework for the construction of biosphere models for waste disposal assessments can be used to illustrate how the \textit{TAME} model was constructed. The development is illustrated in Figure 1-1. This is the procedure used for definition of the model given in Section 2 of this report. As currently formulated, the \textit{Reference Biospheres} methodology enables a suitable conceptual model of the biosphere to be constructed following the definition of the \textit{Assessment Context} and the identification and screening of Features, Events and Processes (FEPs) in the biosphere\(^3\) to be modelled (top-left of Figure 1-1). The corresponding phase in the development of \textit{TAME} is given in Sections 2.1 and 2.2.

The parameterisation of the FEPs is a major undertaking. At the stage of model development in the middle of Figure 1-1, the measurable and observable quantities in the modelled biosphere must be translated into mathematical expressions which can be subsequently coded in computer programs. This is reported in Section 2.3.

Current practice in models for similar purposes (e.g., in BIOMOVS I and II) indicates that a combination of a compartmental approach using first order, donor-controlled, linear kinetics to model contaminant transport in the biosphere together with an equilibrium approach for estimating the exposures via multiple pathways is appropriate. The flexibility of \textit{TAME} arises from the comprehensive list of FEPs included in the model (drawn from the international biosphere FEP-list (BIOMOVS II, 1994)) and from the way in which data specific to the model scenario determine the relative importance of each FEP on a case-by-case basis. The way in which the FEPs are associated with their mathematical relationships also allows for improved transparency compared with earlier models.

In practice, the parameterisation of \textit{TAME} (bottom-right in Figure 1-1) relied on previous experience both in BIOMOVS and in Switzerland. However, a major feature of the \textit{TAME} parameterisation of the FEPs defining radionuclide transport is the use of carefully defined mass balance schemes for liquid (solute) and solid material transport (see Section 2.3.1). As explained in Section 2.3.3, the exposure pathway sub-model was taken from an existing model.

The working of \textit{TAME} is illustrated in a set of example calculations. These show how \textit{TAME} is applied to a generic Swiss biosphere (Section 3) as well as how, in general terms, the model behaves (Section 4). An Appendix also gives the results of verification calculations, in which the performance of \textit{TAME} was compared to earlier models used in Swiss safety assessments. This comparison demonstrated not only the correct implementation of the model, but also the benefits of the new model.

\(^3\) FEPs! - the use of the terminology can be confusing. As used here, \textit{Features} in the biosphere are such things as soils, rivers, sediments, crops, livestock, etc. - i.e. clearly defined elements of the biosphere, each with their own properties and behaviour. A \textit{Process} can be thought of as a mechanism which affects features in the biosphere (perhaps even alters them) and by which the movement or accumulation of radionuclides in the biosphere features occurs. By implication, a process is continuous and thus it is distinguished from an \textit{Event}, which is intermittent in nature. Thus, evapotranspiration is a process, whereas rainfall in the biosphere is a sequence of events and flooding would be an event. There is considerable overlap and events could be modelled as continuous processes by using time-averaged parameters. If the effect of an event were to be sufficiently disruptive, the modelled biosphere might be left in an altered state, i.e. an alternative \textit{Scenario} for which the original description is no longer valid.
Requirements for future work are discussed in Section 5. For example, the need for an improved definition of suitable generic critical groups is clear, the definition used here being taken from existing models which are not applicable to all potential future biosphere scenarios.

*TAME*, in its most generic form, is a set of mathematical relationships linking the features of the biosphere. In practice, each application of the model requires a thorough review of the representations of the FEPs. It has been applied to analyses of the hypothetical releases from both HLW and L/ILW Swiss disposal concepts. The detailed results of the safety assessments (NAGRA, 1994a; NAGRA, 1994b) and input data for the biosphere modelling are given elsewhere (KLOS *et al.*, 1995). Details in these cases are not precisely as described in the example calculations here. Mass balance schemes can only be set up on a case-by-case basis.
2. Model definition

2.1 Preliminary scenario analysis

As illustrated in Figure 1-1, the development of any model requires a context. For TAME this is set by the Swiss regulations regarding radiological protection in the case of waste disposal (HSK/KSA, 1993). Within this framework, the development proceeded with a preliminary analysis of the conditions relevant to Swiss reference biosphere descriptions, i.e. inland biospheres with a range of river types, from small streams in valley sides to large regional rivers, such as the Rhine. Additionally, lakes could potentially form part of the Swiss biosphere. Marine and coastal environments are, however, ruled out.

More data are available on the present-day temperate climate and, naturally, the present-day biosphere forms an important yardstick for biosphere calculations. However, the current interglacial climate conditions are expected to have a limited duration before a return to a colder and drier periglacial state. The potential effects of global warming must also be included in the conceptual model and database. A humid biosphere state, in which the mean annual temperature is higher, but with a corresponding increase in rainfall, as well as the possibility of an arid, hot environment in which rainfall is decreased compared to the present-day are relevant. Table 2-1 summarises the Swiss reference biospheres and lists some of their distinguishing features.

As mentioned in Chapter 1, the reference biospheres approach is used to identify calculational scenarios which will not significantly underestimate the radiological impact of potential releases. A common approach is to calculate doses on the basis of a group of critically exposed individuals as characterised by a closed, self-sustaining agricultural system. The use of such a system maximises the exposure of individuals by preventing any significant dilution with uncontaminated, external material, as well as restricting losses from the modelled system to those which can be expected to occur irrespective of the influence of humans, e.g. the transport of radionuclides downstream in solution or sorbed onto solid material transported in rivers or lakes.

Release from the geosphere following transport in subterranean water is envisaged as the primary route by which radionuclides from underground waste repositories would enter the biospheres represented by TAME. In Switzerland, the use of well water is of primary importance. Other release types are possible but these more often than not involve disturbances to the so-called normal evolution of the repository and far-field system. Biosphere model scenarios should also be able to deal with these situations. Another important feature of the potential releases from the geosphere is the long timescale over which the release of radionuclides to the biosphere can occur.

This naturally leads to a division of modelling tools between the dynamic modelling of FEPs with characteristic timescales greater than years - mainly physical transport processes between compartments representing soils and water in the biosphere (aquifers as well as surface water bodies) and biological entities in the reference biosphere (crops,
livestock and humans) for which accumulation and internal transport mechanisms are largely determined by the annual cycle. Thus, the compartment model for transport between soils and water bodies is modelled dynamically, whereas the accumulation processes in the biological systems employ an equilibrium (steady-state) approach.

The spatial extent of the modelled biosphere depends on the size of the region affected by the release from the geosphere and/or on the minimum area for which the criteria defining the critical group are valid. The former spatial scale is determined by the properties of the geosphere and the modelled area is the smallest consistent with the homogeneous distribution of radionuclides on the timescale of interest. The spatial extent in the latter case is determined by the assumption of critical group practices consistent with the conditions in the modelled biosphere, for example, the area required to support a small self-sustaining agricultural community. The larger of these two areas is generally chosen for the biosphere model.

Contaminants will leave the region of release from the geosphere to the biosphere by a number of transport processes and accumulation may occur in other parts of the biosphere. In most of the scenarios identified above, the principal means by which radionuclides are transported in the biosphere is in solution or suspension in surface water bodies. This leads to the identification of the single river (lake) section as the basic unit of the biosphere for modelling purposes. It may be necessary to link a number of such sections together, particularly if it is required that the ultimate fate of the contaminants be assessed.

The boundary between the geosphere and the biosphere is a matter of some debate. Of the models which participated in the BIOMOVS II Complementary Studies intercomparison exercise (BIOMOVS II, 1996b) some defined the near-surface aquifer as part of the biosphere system (as is done here), whereas others treated aquifers as part of the geosphere. The Complementary Studies clearly show that this assumption can have a significant impact on the form and magnitude of the projected radiological exposures as a function of time. However, within the Reference Biospheres working group (BIOMOVS II, 1996a), a consensus has emerged that the biosphere should indeed include a representation of any near-surface aquifers which are involved in exchanges with the surface hydrology - rivers, lakes, precipitation, etc., - with timescales for the exchanges of the order of one year or less. The development of TAME predates these discussions, but the aquifer is included in the biosphere representations described here.
Table 2-1: Summary of the Swiss reference biospheres listing distinguishing characteristics, assumptions and present-day analogues. The list is not exhaustive, but serves to illustrate the range of scenarios for which TAME is designed.

<table>
<thead>
<tr>
<th>characteristics</th>
<th>interglacial</th>
<th>periglacial</th>
<th>humid</th>
<th>arid</th>
</tr>
</thead>
<tbody>
<tr>
<td>climate: temperature precipitation evapotranspiration (ETP)</td>
<td>continental</td>
<td>permafrost lower</td>
<td>hot humid higher</td>
<td>hot dry higher</td>
</tr>
<tr>
<td></td>
<td>present day</td>
<td>lower</td>
<td>higher</td>
<td>lower</td>
</tr>
<tr>
<td></td>
<td>present day</td>
<td>lower</td>
<td>higher</td>
<td>lower</td>
</tr>
<tr>
<td></td>
<td>present day</td>
<td>lower</td>
<td>higher</td>
<td>lower</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>water fluxes*</th>
<th>ETP &lt; precipitation</th>
<th>permafrost: no infiltration</th>
<th>ETP &lt; precipitation</th>
<th>ETP &gt; precipitation, irrigation needed</th>
</tr>
</thead>
<tbody>
<tr>
<td>soil types</td>
<td>parabraunerde, pseudogley, silt - sand, organic</td>
<td>gley</td>
<td>silt / sand</td>
<td>silt / sand</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>solid material fluxes</th>
<th>bioturbation, dredging, water erosion, suspended sediment load</th>
<th>bioturbation, wind erosion, water erosion, suspended sediment load</th>
<th>bioturbation, dredging, wind erosion, water erosion, suspended sediment load</th>
</tr>
</thead>
<tbody>
<tr>
<td>present-day analogue</td>
<td>Switzerland</td>
<td>Northern Scandinavia</td>
<td>SE Asia</td>
</tr>
<tr>
<td>societal/ agricultural system</td>
<td>present-day, self-sustaining, herding</td>
<td>self-sustaining, paddy fields</td>
<td>present-day, self-sustaining</td>
</tr>
<tr>
<td>water resources</td>
<td>near-surface aquifer, surface water</td>
<td>surface water</td>
<td>near-surface aquifer, surface water</td>
</tr>
<tr>
<td>produce and exposure pathways</td>
<td>pasture, cereals, root veg., green veg., meat and animal products, poultry, fish, external γ-irradiation, airborne dust</td>
<td>lichens, shrubs, reindeer, external γ-irradiation, airborne dust</td>
<td>pasture, cereals, root veg., green veg., meat and animal products, poultry, fish, external γ-irradiation, airborne dust</td>
</tr>
</tbody>
</table>

* Irrigation can take place in any climatic state, but only in the arid state is it essential. A large irrigation excess would be required here to overcome the tendency for salt to accumulate in the rooting zone. In such a climate even the use of large amounts of irrigation would not be sufficient to prevent such a build-up indefinitely.
2.2 The *TAME* conceptual model

2.2.1 Features, Events and Processes (FEPs) for radionuclide transport in the near-surface environment

In the construction of *TAME*, the next stage was to identify the relevant FEPs in the defined biospheres (cf. Table 2-1). The present-day biosphere is used as a starting point for this procedure, because the database describing it is more complete and more readily available. Additional FEPs necessary for modelling alternative scenarios were then added as required.

Figures 2-1 and 2-2 illustrate the geomorphological components of a generic, agricultural biosphere section. If estimates for the radiological consequences of release are required for adjacent regions where the dilution is larger, additional (downstream) biosphere sections, each made up of the five-compartment structure, can be coupled together. The features shown here are a surface water body with bed sediments, a near-surface aquifer and different horizons of soil. Figure 2-1 lists the transport processes for water and solutes in the system, while Figure 2-2 lists the processes by which solid material moves.

The principal components of the biosphere are

- valley sides,
- aquifer and water table,
- surface water (with suspended and bed sediments),
- rooting zone soils,
- lower soil horizons,
- vegetation, livestock and anthropogenic factors.

These take part in the transport of contaminants in the biosphere via the processes identified in Figure 2-1 and Figure 2-2. In modelling the transport of contaminants it is important to recognise that transport can take place in solution and on solid material and a clear distinction must be made between contaminants in solution and those fixed onto solid materials (including transport mediated by flora and fauna). Within the scope of *TAME*, the environmental concentrations of contaminants are at trace levels and in these circumstances it is possible to model the equilibrium partition between solution and solid phase in a simple way, using the $k_d$ concept (see Appendices A and B).

In addition to transport with fluxes of materials, diffusion processes can also play a rôle between regions of varying concentrations. The applicability of compartment models is closely related to the modelling of diffusion-type processes, since the compartment model approach assumes that the contents of the boxes reach their equilibrium values instantaneously and that their distributions are thereafter homogeneous within the compartments. Diffusion processes can therefore be satisfactorily modelled provided that the assumptions for compartment models are valid (see Sections 2.3.2, 4.3 and Appendix C for a detailed discussion of the implementation and the consequences). The
question of validity is then clearly one of the temporal and spatial dimensions over which the compartment model assumptions are valid. The lower limit for the timescale of interest in TAME is the annual cycle. An investigation of the influence of complexity in models for soil systems has been undertaken by ELERT (1993) and in the wider context of models for waste disposal assessments by KLOS (1993).

In common with earlier biosphere models, the TAME representation assumes that the rooting zone soil is well mixed on timescales of one year and that the near-surface aquifer can be treated as a homogeneous unit. Similarly, the intermediate soil horizons between the aquifer and the rooting zone are treated as a single entity.

While this is not a completely satisfactory way of treating the near-surface aquifer, it has the benefit of being easily implemented and interfaced with the other parts of the biosphere model. As shown by ZEEVAERT (1990), a more accurate treatment of the advection-dispersion equation in aquifers is possible. The main practical consequence for the results of the biosphere model would be a slower approach to equilibrium in the aquifer, with higher localised concentrations at earlier times. The assumption of a single compartment for the biosphere aquifer is therefore non-conservative at earlier times, but at later times, as the contaminant concentrations in the aquifer approach uniformity, results from the TAME representation would not differ greatly from those from a more detailed representation.

In the aquatic environment a distinction is made between the surface water and the bed sediments as separate compartments (each with its own $k_d$ so that the sorption of a given radionuclide onto suspended sediment in the surface water could be different to that in the bed sediment). Events and processes leading to the transfer of bed sediment material to soils could be important if there are significant accumulations of contaminants in the sediments. This also implies that an explicit representation of aquatic sediments is to be preferred, in general terms, to one which assumes equilibrium between the bed sediments and the water column. This assumption is also more realistic as well as offering greater flexibility.

These five compartments form the model for the transport of contaminants within a single biosphere section. The TAME representation of a single biosphere section is shown in Figure 2-3. The names of the compartments used here have distinct meanings and they are used in the mathematical representation to identify compartments. Table 2-2 summarises the TAME nomenclature.

In the horizontal plane, spatial inhomogeneities are effectively smoothed out because the end-point of the calculations is an annual exposure based on the intakes from produce grown over an extended area. Similarly, individuals exposed to external irradiation receive doses from a wide area and, for a critical group made up of tens or hundreds of people, a relatively large area of land is required; this tends to act as an averaging mechanism in the calculation of the radiation exposures.

In moving to this simplified representation, care is taken to ensure that the radiological consequences to the critical group inhabiting the modelled region are not underestimated. This is achieved by imposing boundary conditions which maximise the environmental concentrations of the radionuclides in each of the biosphere sections. These are given in Table 2-3.
Principal water fluxes in the reference agricultural biosphere section. The components of the biosphere section involved in the waterborne transport of contaminants are illustrated. The corresponding solid material fluxes are shown in Figure 2-2.
solid flux | short description
--- | ---
a | External deposition on surface water
b | External deposition on soil surface
c | Bioturbation and water-mediated transport: rooting zone soil → lower soils
d | Bioturbation: lower soils → rooting zone soil
j | Flooding, dredging and irrigation: suspended solid material → soils
k | Regional erosion: external losses
m | Regional erosion: rooting zone soil → surface water
n | Erosion of banks: lower soils → surface water
p | Resuspension: aquatic bed sediments → surface water
q | Deposition: waterborne solid material → bed sediment
r | Suspended sediment throughput (inflow)
s | Bed sediment throughput (viscous drag - inflow)
t | Suspended sediment throughput (outflow)
u | Bed sediment throughput (viscous drag - outflow)
v | Lateral flow: valley sides → rooting zone soil
w | Lateral flow: valley sides → lower soils

Figure 2-2: Principal solid material fluxes in the reference agricultural biosphere section. The components of the biosphere section involved in the solid phase transport of contaminants are illustrated. The corresponding water fluxes are shown in Figure 2-1.
Table 2-2: Nomenclature of the *TAME* compartments. The symbolic names are used in the mathematical description of the model in the next section.

<table>
<thead>
<tr>
<th><em>TAME</em> Compartment</th>
<th>Symbol</th>
<th>Distinguishing Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Local aquifer</td>
<td>L</td>
<td>Near-surface geological media supporting groundwater flow associated with the surface water body.</td>
</tr>
<tr>
<td>Deep soil</td>
<td>D</td>
<td>Soil horizons between the aquifer and the rooting zone of crops</td>
</tr>
<tr>
<td>Top soil</td>
<td>T</td>
<td>Soil horizons containing the roots of crops</td>
</tr>
<tr>
<td>Surface water</td>
<td>W</td>
<td>Springs, streams, rivers, ponds, lakes, reservoirs</td>
</tr>
<tr>
<td>Aquatic bed sediment</td>
<td>S</td>
<td>Solid material forming the bed of the surface water body (when distinguished from the aquifer material)</td>
</tr>
<tr>
<td>Elsewhere</td>
<td>E</td>
<td>A compartment acting as a sink for the five-compartment biosphere section denoted by L, D, T, W and S.</td>
</tr>
</tbody>
</table>
Figure 2-3: General arrangement of the compartments in the Terrestrial - Aquatic Model of the Environment showing the principal compartments for a generic representation of the biosphere. The transport of contaminants between the five main compartments is modelled using mass balance considerations for water and solid material fluxes between all the compartments in the section. Each of the transfers shown here is made up of the combination of a number of features, events and processes. Additional sections of the biosphere can be modelled by inserting the five-compartment structure for downstream sections of the drainage system. The Elsewhere compartment acts as a sink for the system.
Table 2-3: Nomenclature simplifications and boundary conditions used in TAME, based on the conceptualisations shown in Figures 2-1 and 2-2. In addition to the five compartments identified here and in Figure 2-3, the TAME nomenclature makes use of Contaminated (C) and Uncontaminated (U) compartments for sources of water and solid material, as well as the Atmosphere (A) acting as the source of precipitation. Contaminants are recycled between these compartments and the exposure pathway sub-model (Figure 2-4).

<table>
<thead>
<tr>
<th>TAME simplification</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Near-surface aquifer treated as a single unit (local aquifer - L).</td>
<td>Transport in aquifers is simplified to the extent that contaminants entering the aquifer instantaneously have an equilibrium distribution. Possible underestimation of the time to reach equilibrium concentration. Magnitude of final concentration not affected. The assumption of a single compartment is therefore conservative.</td>
</tr>
<tr>
<td>Rooting zone soil treated as a single unit (top soil - T).</td>
<td>Bioturbation within this compartment, as well as anthropogenic factors such as ploughing, ensure that, annually, this compartment is well mixed. Seasonal variations are not apparent in model results. Annual averages are assumed to adequately represent the effect on model consequences. The assumption is therefore mildly conservative.</td>
</tr>
<tr>
<td>Intermediate soil horizons treated as a single unit (deep soil - D).</td>
<td>Average concentration in the material between the local aquifer and the top soil is represented. The time to reach this equilibrium state may be underestimated, leading to higher concentrations at the top of the deep soil column, and hence higher concentrations in the top soil at earlier times. Deep soil material may exhibit significantly different properties than the local aquifer or top soil. The assumption of a single compartment is therefore conservative.</td>
</tr>
<tr>
<td>Surface water body treated as a single unit (surface water - W).</td>
<td>Mixing times are much shorter than one year. The greater the throughput, the more valid the assumption. This may not be valid for large lakes.</td>
</tr>
<tr>
<td>Aquatic sediments treated as a single unit (aquatic bed sediment - S).</td>
<td>Aquatic sediments are distinguished from the local aquifer and soils because they do not necessarily have features in common. In rivers they are likely to rapidly come into equilibrium with the surface water. In aquatic environments with slower moving waters, stratification of the sediments would lead to longer times to steady-state. The assumption of a single compartment is therefore conservative.</td>
</tr>
<tr>
<td>Fixed dimensions for top soil, deep soil and local aquifer, with no lateral outflow boundary conditions.</td>
<td>In reality, contaminants would spread out in the biosphere, away from the point of entry into the biosphere section. One way of modelling this is to allow the volumes of the compartments to change in time. However, the representation adopted here is to prevent loss of contaminants by this process by maintaining the initial dimensions of the biosphere while at the same time imposing no-flow boundary conditions at all of the external faces of the compartments representing the aquifer and the soils. Only internal transport is allowed in the model compartments. This also aids the application of mass balance schemes. This assumption is conservative, and does not allow evolution of the system. Loss from the five-compartment representation is only allowed downstream in the surface water system. (For details of a specific application see Section 3.)</td>
</tr>
<tr>
<td>Closed agricultural system</td>
<td>All activity leaving the transport system and entering the exposure pathways system is assumed to be returned, when annual averages are considered. Waste products are implicitly recycled as a source of nutrients.</td>
</tr>
<tr>
<td>Outflow only allowed downstream.</td>
<td>Only in the direction of water flow in surface water is transport of contaminants out of the system allowed. This constrains activity lost from upstream (the Previous section - denoted by P) to enter downstream. The assumption is conservative. At the end of the set of sections used to model the biosphere, there is the so-called Elsewhere compartment (denoted by E). This acts as a sink for all contaminants leaving the system.</td>
</tr>
</tbody>
</table>
2.2.2 Features, Events and Processes (FEPs) describing the exposure pathways

Radiological doses arise from the exposure of individuals to the environmental concentrations of radionuclides determined by transport in the biosphere and uptake in the food-chain. The ICRP 26 definition of dose (ICRP, 1977), on which the Swiss regulatory guidelines are based, specifies the effective dose equivalent as the sum of the weighted 50-year committed dose equivalents in specific organs from the intake of activity into the body in one year, plus the sum of weighted dose equivalents from external irradiation in one year. Intakes into the body are classified according to the type of intake, either ingestion or inhalation. The exposure pathways are therefore defined in terms of

- the intake of contaminated foodstuffs and water
- the inhalation of airborne radionuclides
- the exposure to external irradiation from contaminated environmental media.

The Swiss regulations specify that the subject individual should have normal behavioural habits, as defined by the group of individual inhabiting the reference biosphere.

As with the transport model, some assumptions can be made which ensure that representative individuals will not have their exposures underestimated. As already mentioned, this is achieved by assuming that all food consumed is grown locally (ensuring no dilution of intake of contaminants). Furthermore, the individual dose from environmental factors (leading to inhalation and external irradiation) is maximised by assuming that the individual resides in the region of highest concentration during the entire year. The choice of the closed agricultural community similarly justifies the use of a relatively high value for calorific intake, one which is commensurate with the strenuous working conditions of agricultural workers.

In order to simplify the model, the number of different foodstuffs consumed can be generalised to the following

- Consumption pathways:
  - drinking-water,
  - freshwater fish, meat, milk and dairy products, eggs,
  - grain, green vegetables, root vegetables
- Environmental exposures:
  - external γ-irradiation, dust inhalation

The database for generic foodstuff accumulations is sparse and often contradictory. Best estimate parameter values for a small number of the major pathways are therefore more reliable. For a typical lifestyle in the TAME biosphere representations, the only significant contribution to external dose comes from the γ-emitting radionuclides in the environment. Direct external β-doses are not calculated because of the limited range in air.
Figure 2-4: Schematic representation of the TAME exposure pathways indicating the relationship between the compartments in the dynamic part of the model (Figure 2-3) which are involved in the receipt of dose via ingestion or inhalation of contaminated material, or by direct γ-irradiation. Over the yearly cycle, the exposure pathway sub-system is assumed to be in a state of dynamic equilibrium with the biosphere transport model.
As the exposure pathways and transport sub-models are separate in TAME, there is implicit recycling of the radionuclides removed from the TAME compartments. This is conservative but it is consistent with the assumption of a closed critical group. Included in the assumption are the reapplication of animal and human wastes, and vegetable matter as fertilisers as well as the burial of the dead.

Figure 2-4 shows the network of exposure pathways considered in TAME. The structure is complex and each of the connections illustrated in the figure can, in principle, be made up of a number of contributing FEPs. Contaminant concentrations in crops and other vegetation arise from

- root uptake,
- irrigation and foliar absorption,
- soil particles on external surfaces.

Similarly, concentrations of contaminants in animal tissues and other products arise from

- drinking-water consumption,
- intake of contaminated foodstuffs,
- direct intake of contaminated soils.

2.2.3 Choice of a compartment model approach

Overall, the acceptability of the shortcomings of the compartment model approach to biosphere transport calculations rests on two factors:

- The object of the model is not to predict actual distribution of radionuclides in the surface environment but to estimate radiological impact to a typical individual dwelling locally in a representative community. In this context, actual concentration and its variability is not an issue since internal doses will result from intake of a range of food products obtained from a sizeable area and, similarly, external exposure and inhalation will be related to average conditions experienced by individuals who will move about the local environment. Hence, human diet and habits can be seen to act as an averaging mechanism.

- The neglect of temporal variability and change is a practical constraint imposed by the observation that the processes of temporal change in the biosphere operate on such a range of timescales (daily, seasonal, annual and longer term) that it would be impossible to represent them all explicitly in a

1 Inhalation by animals is not considered as an intake mechanism for subsequent human consumption. The amounts of activity are likely to be much smaller than those involved in the consumption pathways and would most likely accumulate in the lungs, which are not assumed to be consumed. This is a non-conservative assumption.
single assessment model. Therefore, the approach taken is to define constant transfer coefficients that take account implicitly of the long-term average effect of shorter-term periodic events, e.g. seasonality and episodic events such as flooding, that it is not possible to take into account in any other way.

These assumptions are justified by the end-point of the calculations, i.e. dose to a typical individual living in a representative community. The quantity which best describes the radiological impact on such an individual is the lifetime dose; however most factors for the dose per unit exposure have been defined for practical operational procedures in present-day medical, industrial or research environments. From a health and safety at work perspective, they are best expressed in terms of annual doses.

2.3 The TAME mathematical representation

2.3.1 The transport equation for compartment models

Denoting by $N_i$ [Bq] the quantity of radioactivity of radionuclide $N$ in the $i$th compartment of a network of compartments (such as that shown in Figure 2-3), the time variation of this amount is given by

$$\frac{dN_i}{dt} = \left( \sum_{j \neq i} \lambda_{ji} N_j + \lambda_N M_i + S_i(t) \right) - \left( \sum_{j \neq i} \lambda_{ij} N_i + \lambda_N N_i \right) \text{[Bq y}^{-1}]$$.

(1)

This is the donor-controlled, first-order linear ordinary differential equation for transport in compartment models. The following terms are also defined:

- $N_i$ [Bq] is the amount of radionuclide $N$ in biosphere compartment $i$
- $N_j$ [Bq] is the amount of radionuclide $N$ in biosphere compartment $j$
- $M_i$ [Bq] is the amount of radionuclide $M$ in biosphere compartment $i$ ($M$ is the precursor radionuclide of $N$ in a decay chain)
- $S_i(t)$ [Bq y$^{-1}$] is an external source term of radionuclide $N$ to compartment $i$
- $\lambda_N$ [y$^{-1}$] is the decay constant for radionuclide $N$
- $\lambda_{ji}$ [y$^{-1}$] is a set of transfer coefficients inputs to compartment $i$ from the other $j$ ($\neq i$) compartments in the system

and $\lambda_{ij}$ [y$^{-1}$] is the set of transfer coefficients representing the loss terms of $N$ from compartment $i$ to the other $j$ ($\neq i$) compartments of the system - including external losses.

Note that the first bracket on the right-hand side of Equation (1) represents inputs to the systems and the second bracket represents losses.
The intercompartment transfer coefficients, \((\lambda_{ij})\) are the mathematical representations for the FEPs described in the preceding section. Use is made here of the linearity of the compartment model approach in that the transfer processes acting between two compartments \(i\) and \(j\) can be described as a linear sum of all processes \((k)\) acting between the compartments:

\[
\lambda_{ij} = \sum_{FEPs,k} \lambda_{ij}^{(k)} [y^{-1}].
\]

They must therefore be defined in terms of the fluxes of water and solid material and the other processes identified above.

An important result of the first phase of BIOMOVS was the development of this approach, which began with the use of simple numerical values for the \(\lambda_{ij}\) and ended with the identification of site specific factors affecting the \(\lambda_{ij}\).

\[\text{2.3.2 Generic representation of the intercompartment transfers in TAME}\]

The parameterisation of the \(\lambda_{ij}\) allows the transport characteristics of the model representation of the biosphere to be determined in terms of properties of the site. The parameterisation of the transport FEPs which follows is generic because the representation of the FEPs is valid for a wide variety of biosphere types; only the numerical values of the \(\lambda_{ij}\) will change, and these changes will be a result of the differences in site characteristics.

For each FEP \(k\), represented in the model, the transfer coefficient between two compartments is defined as

\[
\lambda_{ij}^{(k)} = \frac{\text{amount moved from } i \text{ to } j \text{ in unit time by process } k}{\text{current compartment content}} = \frac{1}{N_i} \left( \frac{dN_{ij}}{dt} \right)^{(k)} [y^{-1}].
\]

These transfers deal with the movement of contaminants and these must be related to the FEPs identified in Section 2.2.1 which describe the fluxes of water and solid material. Additionally the dynamic exchange of contaminants by diffusion must also be considered. The compartment solid - liquid distribution coefficient \((k_d)\) is used to determine the amount of contaminant in solution compared to the amount on the solid material in the compartment. The rôle of the \(k_d\) in compartment models is described in detail in Appendix A. Appendix B describes the derivation of the \(\lambda_{ij}\) from compartment characteristics using Equation (3). Appendix D describes the parameterisation of the various processes involved in the movement of solid material in the biosphere and Appendix C describes the compartment model representation of diffusion processes.

For reference, the basic form of the TAME transfer coefficients is as follows:
The transfer coefficient for contaminants moved by advective water fluxes is given by

\[
\lambda_{ij}^{\text{advective}} = \frac{1}{\theta_i + (1-\epsilon_i)\rho_i k_i} \frac{F_{ij}}{A_i l_i} \quad [\text{y}^{-1}],
\]  

(4)

and the transfer coefficient for solid-flux driven transport is

\[
\lambda_{ij}^{\text{solid}} = \frac{1}{\theta_i + (1-\epsilon_i)\rho_i k_i} \frac{k_i M_{ij}}{A_i l_i} \quad [\text{y}^{-1}].
\]

(5)

The diffusive transfer from \( i \) to \( j \) is given by a similar form to that of the advective fluxes

\[
\lambda_{ij}^{\text{diffusion}} = \frac{D_{ij}}{\theta_i + (1-\epsilon_i)\rho_i k_i} \quad [\text{y}^{-1}],
\]

(6)

with a corresponding return transfer from \( j \) to \( i \).

The general form for the transfer of contaminants from \( i \) to \( j \) is therefore obtained by combining these expressions, according to Equation (2):

\[
\lambda_{ij} = \frac{1}{\theta_i + (1-\epsilon_i)\rho_i k_i} \left( \frac{F_{ij} + k_i M_{ij}}{A_i l_i} + D_{ij} \right) \quad [\text{y}^{-1}]
\]

(7)

In these representations, the following parameters are used:

- \( F_{ij} \) [\( m^3 \text{ y}^{-1} \)] is the water flux from compartment \( i \) to \( j \) (Appendices B and D)
- \( M_{ij} \) [\( kg \text{ y}^{-1} \)] is the solid material flux from compartment \( i \) to \( j \) (Appendices B and D)
- \( A_i \) [\( m^2 \)] is the area of the compartment. Together with
- \( l_i \) [\( m \)] the thickness of the compartment, these define the physical volume of the compartment (Appendices B and D)
- \( \epsilon_i \) [-] is the porosity of the compartment material (see Appendices A, B and D)
- \( \theta_i \) [-] is the volumetric moisture content (see Appendices A, B and D)
- \( \rho_i \) [\( kg \text{ m}^{-3} \)] is the density of the solid material in the compartment (Appendices A, B and D)
- \( k_i \) [(\( Bq \text{ kg}^{-1} \))] is the solid - liquid distribution coefficient for the contaminant in the compartment (i.e. the compartmental \( k_d \)). (Appendices A and D)

2 The \( k_d \) value for sorption on mobile and immobile solids may not be the same. As implemented here, TAME assumes that the values are the same. Appendix A looks at the consequences of this assumption.
$D_{ij} \ [y^{-1}]$ is the effective diffusion rate for contaminants in solution moving between compartments $i$ and $j$ (Appendix C).

Equations (4) - (7) form the basis for the transport modelling in TAME at their most generic level. Further parameterisation is possible, but only on a site- and case-specific basis.

Therefore it can be seen that TAME employs a mass balance scheme to define all intercompartment water ($F_{ij} \ [m^3 \ y^{-1}]$) and solid fluxes ($M_{ij} \ [kg \ y^{-1}]$) and these must be related to measurable or observable quantities in, or relevant to, the biosphere to be modelled. Table 2-4 summarises the generic parameters used to characterise the TAME compartments and the contaminant fluxes. The following examples are used to illustrate how this is done in practice using the mass balance scheme defined in Sections 3.2.2 and 3.2.3 for the example calculations discussed in Section 4 to illustrate the behaviour of TAME.

The water flux between the TAME top soil ($T$) and deep soil ($D$) compartments provides an example:

$$F_{TD} = (d_{AT} - d_{TA} + d_{LT} + d_{WT}) A_f \ [m^3 \ y^{-1}],$$

where $d_{AT} \ [m \ y^{-1}]$ is the annual precipitation (from the atmosphere box - $A$ - to the top soil), $d_{TA} \ [m \ y^{-1}]$ is the loss of water from the top soil to the atmosphere (by evapotranspiration - see Figure 2-1), $d_{LT} \ [m \ y^{-1}]$ is the annual amount of irrigation applied to the top soil which is abstracted from a well in the local aquifer ($L$) and $d_{WT} \ [m \ y^{-1}]$ is the corresponding amount of irrigation taken from the surface water ($W$). The surface area of land in the biosphere section is $A_f \ [m^2]$. N.B. in the transport model, it is assumed that water falling on the crops in the biosphere flows directly to the top soil - effectively with no delay. The interaction of the crops with contaminants in the irrigation water is modelled separately in the exposure pathway sub-model.

In a similar way, the erosion and bioturbation processes contribute to the transport of solid material from the deep soil to the top soil according to

$$M_{DT}^{(bioturbation)} + M_{DT}^{(erosion)} = m_D w_D A_f + m_e A_f \ [kg \ y^{-1}],$$

where $m_D \ [kg \ m^{-2}]$ is the biomass of deep-burrowing fauna (travelling between the deep soil and the top soil)

$w_D \ [y^{-1}]$ represents the number of cycles carried out by the fauna in a single year, each time displacing an amount of soil equal to body mass

$m_e \ [kg \ m^{-2} \ y^{-1}]$ is the regional erosion rate (decoupled from the water fluxes because regional erosion rates can be independently determined)

and $A_f \ [m^2]$ is the surface area of the soils, as before.

The term $D_{ij} \ [y^{-1}]$ is used here to denote a diffusive transfer rate in water between compartments $i$ and $j$. The denominator in Equation (6) includes the effects of sorption in the source compartment. Extreme caution must be applied in modelling diffusive
processes in compartment models. However approximations can be made which allow the diffusive fluxes to be included. The NRPB model MiniBIOS (MARTIN et al., 1991) approximates the diffusive term as

\[ D_q = \frac{1}{l_i \min(l_j, l_i)} \cdot \frac{D_0}{T_i} \, [y^{-1}], \]  

(10)

which includes some of the compartment properties - thickness of donor and receptor compartments \((l_i, l_j \, [\text{m}])\), a tortuosity factor \((T_i \, [-])\) and the diffusion coefficient for the radionuclide in solution \((D_0 \, [\text{m}^2 \, \text{y}^{-1}])\). In this form, diffusion has been shown to be an important process for low \(k_d\) radionuclides at early times after the commencement of release to the biosphere from the geosphere (NEA, 1993) and so it was felt that it would not be wise to neglect diffusion in the generic formulation of TAME. This form is implemented in the version of TAME used in the example calculations (see Section 4). However these results indicate that this expression is not satisfactory and a more appropriate alternative is therefore developed in Appendix C.

The complete list of transfer coefficients used in a generic version of TAME for a single section of five compartments plus a downstream sink is given in Appendix D. Table 2-5 shows which processes \((F_{ij}, M_{ij} \) and \(D_{ij}\)) are included in the derivation of each of the intercompartmental transfer coefficients, \(\lambda_{ij}\).

In the derivation of the TAME transfer coefficients, emphasis has been placed on the use of annual averages for parameter values. Longer timescale variations are not considered, either for the flux parameters or for the compartment volumes. The former feature of the model representation fits in with the reference biospheres approach, in which alternative parameter sets are used to describe the model biosphere at different stages of its evolution - alternative climate states for example. This does mean that the timescale for which any given biosphere representation is valid is limited by major changes to the parameters used to define the transfer coefficients in Equation (7). Indeed, one function of the dynamic approach used in biosphere modelling is to identify the characteristic timescale for a particular biosphere representation - which can be defined as the time for all components of the biosphere to come into equilibrium in response to a constant input of activity.
Table 2-4: Summary of the parameters used to characterise the dynamic transport model in TAME.

<table>
<thead>
<tr>
<th>category</th>
<th>parameter</th>
<th>symbol</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Top soil</td>
<td>compartment thickness</td>
<td>$l_T$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>suspended solids in porewater</td>
<td>$\alpha_T$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment solid material density</td>
<td>$\rho_T$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment porosity</td>
<td>$\varepsilon_T$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>volumetric moisture content</td>
<td>$\theta_T$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>compartment tortuosity</td>
<td>$T_T$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>nuclide solid - liquid distribution coefficient</td>
<td>$k_T$</td>
<td>(Bq kg^{-1})(Bq m^{-3})^{-1}</td>
</tr>
<tr>
<td>Deep Soil</td>
<td>compartment thickness</td>
<td>$l_D$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>suspended solids in porewater</td>
<td>$\alpha_D$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment solid material density</td>
<td>$\rho_D$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment porosity</td>
<td>$\varepsilon_D$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>volumetric moisture content</td>
<td>$\theta_D$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>compartment tortuosity</td>
<td>$T_D$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>biomass involved in bioturbation</td>
<td>$m_D$</td>
<td>kg m^{-2}</td>
</tr>
<tr>
<td></td>
<td>biomass activity</td>
<td>$W_D$</td>
<td>y^{-1}</td>
</tr>
<tr>
<td></td>
<td>nuclide solid - liquid distribution coefficient</td>
<td>$k_D$</td>
<td>(Bq kg^{-1})(Bq m^{-3})^{-1}</td>
</tr>
<tr>
<td>Local Aquifer</td>
<td>compartment thickness</td>
<td>$l_L$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>suspended solids in porewater</td>
<td>$\alpha_L$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment solid material density</td>
<td>$\rho_L$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment porosity</td>
<td>$\varepsilon_L$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>volumetric moisture content</td>
<td>$\theta_L$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>compartment tortuosity</td>
<td>$T_L$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>nuclide solid - liquid distribution coefficient</td>
<td>$k_L$</td>
<td>(Bq kg^{-1})(Bq m^{-3})^{-1}</td>
</tr>
<tr>
<td>Surface Water</td>
<td>water depth</td>
<td>$d_w$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>compartment width</td>
<td>$w_w$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>compartment length</td>
<td>$l_w$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>suspended sediment load</td>
<td>$\alpha_w$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment solid material density</td>
<td>$\rho_w$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>nuclide solid - liquid distribution coefficient</td>
<td>$k_w$</td>
<td>(Bq kg^{-1})(Bq m^{-3})^{-1}</td>
</tr>
<tr>
<td>Aquatic Sediment</td>
<td>compartment thickness</td>
<td>$l_s$</td>
<td>m</td>
</tr>
<tr>
<td></td>
<td>compartment solid material density</td>
<td>$\rho_s$</td>
<td>kg m^{-3}</td>
</tr>
<tr>
<td></td>
<td>compartment porosity</td>
<td>$\varepsilon_s$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>compartment volumetric moisture content</td>
<td>$\theta_s$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>compartment tortuosity</td>
<td>$T_s$</td>
<td>-</td>
</tr>
</tbody>
</table>
Table 2-4: Summary of the parameters used to characterise the dynamic transport model in TAME (continued).

† Corresponds to the transfer of a mass of bed sediment to unit area of farmed land.

<table>
<thead>
<tr>
<th>category</th>
<th>parameter</th>
<th>symbol</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Miscelloaneous</td>
<td>area of agricultural land</td>
<td>$A_f$</td>
<td>m$^2$</td>
</tr>
<tr>
<td></td>
<td>diffusion constant for ions in pure water</td>
<td>$D_0$</td>
<td>m$^2$ y$^{-1}$</td>
</tr>
<tr>
<td>Water Fluxes</td>
<td>regional rainfall</td>
<td>$d_{AT}$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>regional evapotranspiration</td>
<td>$d_{TA}$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>deep soil - top soil (capillary rise)</td>
<td>$d_{DT}$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>local aquifer - deep soil (capillary rise)</td>
<td>$d_{LD}$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>irrigation with surface water</td>
<td>$d_{WT}$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>irrigation with aquifer water</td>
<td>$d_{LT}$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>inflow to aquifer (contaminated)</td>
<td>$F_{CL}$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>inflow to aquifer (uncontaminated)</td>
<td>$F_{UL}$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>inflow from previous water body</td>
<td>$F_{PW}$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>outflow to the elsewhere compartment</td>
<td>$F_{WE}$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Solid Material Fluxes</td>
<td>regional erosion rate</td>
<td>$m_e$</td>
<td>kg m$^{-2}$ y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>annual exchange with water column</td>
<td>$\kappa_S$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>annual transfer to agricultural land†</td>
<td>$m_S$</td>
<td>kg m$^{-2}$ y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>deep soil - top soil</td>
<td>$M_{DT}$</td>
<td>kg y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>local aquifer - deep soil</td>
<td>$M_{LD}$</td>
<td>kg y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>surface water - top soil</td>
<td>$M_{LT}$</td>
<td>kg y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>inflow with surface water</td>
<td>$M_{WT}$</td>
<td>kg y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>loss from surface water (river water flow)</td>
<td>$M_{PW}$</td>
<td>kg y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>$M_{WE}$</td>
<td>kg y$^{-1}$</td>
</tr>
</tbody>
</table>
Table 2-5: Summary of the transport mechanisms in the TAME representation of a single biosphere section. The names of the compartments are given by the letter used by the TAME codes to identify them (see Table 2-2). Not all these processes are included in every scenario. The site-specific values for each of these fluxes determine the transport, but the model allows for these mechanisms in each case:

- $F_{ij}$, water flux-driven transport between compartments $i$ and $j$,
- $M_{ij}$, solid material flux-driven transport,
- $D_{ij}$, diffusive exchanges - assumed to be non-zero only for contiguous, saturated or partially saturated compartments.

<table>
<thead>
<tr>
<th>from \ to</th>
<th>L</th>
<th>D</th>
<th>T</th>
<th>W</th>
<th>S</th>
<th>E</th>
</tr>
</thead>
<tbody>
<tr>
<td>L</td>
<td>-</td>
<td>$F_{LD,LD}$, $M_{LD}$, $D_{LD}$</td>
<td>$F_{LT,LT}$, $M_{LT}$</td>
<td>$F_{LS,LS}$, $D_{LS}$</td>
<td>$F_{LE,LE}$, $M_{LE}$</td>
<td></td>
</tr>
<tr>
<td>D</td>
<td>$F_{DL,DL}$, $M_{DL}$, $D_{DL}$</td>
<td>-</td>
<td>$F_{DT,DT}$, $M_{DT}$</td>
<td>$F_{DW,DW}$, $M_{DW}$</td>
<td>$F_{DS,DS}$, $M_{DS}$</td>
<td>$F_{DE,DE}$, $M_{DE}$</td>
</tr>
<tr>
<td>T</td>
<td>$F_{TL,TL}$, $M_{TL}$, $D_{TL}$</td>
<td>$F_{TD,TD}$, $M_{TD}$</td>
<td>-</td>
<td>$F_{TW,TW}$</td>
<td>$F_{TS,TS}$</td>
<td>$F_{TE,TE}$, $M_{TE}$</td>
</tr>
<tr>
<td>W</td>
<td>$F_{WL,WL}$, $M_{WL}$</td>
<td>$F_{WD,WD}$, $M_{WD}$</td>
<td>$F_{WT,WT}$</td>
<td>-</td>
<td>$F_{WS,WS}$</td>
<td>$F_{WE,WE}$</td>
</tr>
<tr>
<td>S</td>
<td>$F_{SL,SL}$, $M_{SL}$, $D_{SL}$</td>
<td>$F_{SD,SD}$, $M_{SD}$</td>
<td>$F_{ST,ST}$</td>
<td>$F_{SW,SW}$</td>
<td>-</td>
<td>$F_{SE,SE}$</td>
</tr>
</tbody>
</table>
A related problem is that of the time evolution of the physical dimensions of the compartments. The component of Equation (7) dealing with the erosion of the top soil to the surface water is given by

\[
\lambda_{TW}^{(erosion)} = \frac{1}{\theta_T + (1 - \epsilon_T) \rho_T k_T l_T} \cdot \frac{k_T m_w}{l_T} \quad [y^{-1}],
\]

however, erosion process means a loss of material from the compartment with a corresponding loss of mass and volume and hence \( l_T \) could vary in time, with unrealistic consequences in the limit as \( l_T \to 0 \). Maintaining fixed boundaries in the compartment solves this problem, with the consequence that erosion of the surface material also produces a net upward transport of contaminant from the deeper compartments:

\[
\lambda_{DT}^{(erosion)} = \frac{1}{\theta_D + (1 - \epsilon_D) \rho_D k_D l_D} \cdot \frac{k_D m_w}{l_D} \quad [y^{-1}],
\]

with a similar component in the expression for \( \lambda_{LD} \), the transfer coefficient from the aquifer to the deep soil. Mass balance considerations mean that there may be an inflow to the biosphere system from the geosphere, or vice versa if the region is one of net deposition of material.

### 2.3.3 Generic representation of annual individual dose in TAME

The solution of Equation (1) leads to expressions for the inventories of the radionuclide \( N \) in compartment \( i \) as a function of time \((N_i(t) \text{ [Bq]})\). In an analogous way to the combination of FEPs in the dynamic parts of the TAME representation, these inventories can be used to give the annual individual dose from exposure to \( N \) in each of the \( i \) compartments in the system which contribute to the exposures via pathway \( p \). The dose from this pathway \( p \) is denoted by \( D_p^{(N)}(t) \text{ [Sv y}^{-1}]\):

\[
D_p^{(N)}(t) = \sum_{i, \text{exp}} E_p H_{exp}^{(N)} P_{p,i} N_i(t) \text{ [Sv y}^{-1}],
\]

where \( N_i(t) \text{ [Bq]} \) provides the only time dependence in the expression for dose.

\( P_{p,i} \) is a processing factor which transforms \( N_i(t) \) into a concentration in pathway \( p \). As seen below, these factors can require detailed parameterisation, for example when the accumulation in meat or milk for human consumption is being modelled.

\( E_p \) is an exposure factor for the pathway which can represent the consumption rate of foodstuffs or occupancy of the modelled region, etc.
and \( H_{\text{exp}}^{(N)} \) [Sv Bq\(^{-1}\)] is the dose per unit intake for radionuclide \( N \) and is used to convert the exposure to environmental concentrations of radionuclides to the corresponding dose.

Unlike the FEPs governing the movement of radionuclides in the biosphere, which are largely mediated by mass balance considerations, the FEPs describing the doses potentially received by individual humans are likely to be, in extreme cases, very individual specific. This being the case, it is necessary to provide as comprehensive a list as possible in the TAME exposure pathway sub-model. The equivalent of mass balance in the exposure pathway sub-models is the assumption that the modelled individual has a given dietary intake requirement, but with preferences for individual food types being averaged over a representative group of individuals of some kind. In this way, consumption and exposure rates are more likely to be specific to a given lifestyle and region and thus broadly applicable over a comparatively large spatial (and perhaps temporal) extent. Similarly, the factors influencing uptake and accumulation in foodstuffs are expected to be valid over a larger spatial extent than, for example, the mass balance schemes.

The existing Swiss biosphere representations were not felt to be sufficiently comprehensive in their representations of the exposure pathway FEPs and the methodology used by the United Kingdom National Radiological Protection Board in the MiniBIOS model (MARTIN et al., 1991) has therefore been adopted. This is a well established model which has seen many years of application in a variety of situations (e.g. NEA, 1993) and is believed to be the equal of any contemporary model for the evaluation of potential exposures to a critical group consisting of a small, subsistence agricultural community (see the comparisons in BIOMOVS II, 1996b).

A detailed description of the modelling of all the TAME exposure pathways is given in Appendix E. Some modifications to the formulation used by the NRPB have been implemented. These concern the derivation of concentration of radionuclides in the water phases in the TAME compartments (see Appendix A). Additionally, many of the parameter values used in the calculation of exposures have either been taken directly from the original Swiss database (NAGRA, 1985) or, where processes are now modelled in a different way, have been modified to fit the representation in the same model. The aim was to reproduce, as far as possible, the results of the earlier Swiss model, but to include the additional detail of the MiniBIOS model. Appendix F details the comparisons with the earlier models by selecting parameter values which effectively switch off the newly included FEPs. (A further illustration of the TAME representation with all FEPs switched on is also given.)

The default database for the TAME exposure pathways is therefore something of a hybrid, which, while not being entirely satisfactory, does not give contradictory results when compared with previous assessments. Other databases are available (e.g. IAEA, 1994) but the data are not always entirely consistent with waste disposal applications, nor is the range of radionuclides for which data are available broad enough to cover all eventualities in waste disposal.

Analysis of the exposure pathways models in the participating codes in Complementary Studies (BIOMOVS II, 1996b) indicates that there is a much more divergent
mathematical approach to modelling the exposure pathways than in the case of the transport process modelling, although there is considerable convergence in the FEPs in the exposure pathway models for the subsistence community. Clearly more convergence in the representations is desirable at an international level. Similarly, the close inter-model agreement in the calculational mid-points in the exercise shows that a great deal of the variability in model results seen in the earlier BIOMOVS exercise (see, for example, GROGAN, 1989) arose from major differences between assumptions, model representations or databases. A review of suitable databases for waste disposal applications is required.

2.3.4 Examples of the representations of FEPs in the TAME exposure pathway sub-model

Appendix E gives a full description of the mathematical representation of all the FEPs in the TAME exposure pathway sub-model. The following examples give an indication of the detail required to give a comprehensive representation of the FEPs. As set out in Equation (13), the expressions for the doses are given in terms of the compartment contents \( N_i \) [Bq] which are given by the solution of the differential equation in Equation (1). These expressions show the need for clear identification and categorisation of the FEPs contributing to the potential exposures.

Dose from the consumption of contaminated surface water:

The parameters used in evaluation of the dose from consumption of contaminated drinking-water from both the surface water and aquifer compartments are given in Table 2-6. Using this nomenclature Equation (13) translates into

\[
D_{\text{wat}} = H_{\text{ing}} I_{\text{wat}} (1 - f_{\text{well}}) \left(1 + \frac{(1 - f_{\text{filter}}) \alpha_w k_w}{1 + \alpha_w k_w}\right) \frac{N_w}{I_w w_w d_w} \text{[Sv y}^{-1}] \tag{14}
\]

for the doses received from consumption of contaminated surface water. Here the dose per unit intake is clearly defined as the value appropriate for the ingestion of the radionuclide. The intake rate \( I_{\text{wat}} (1 - f_{\text{well}}) \) is likewise clear, but it involves the fraction of the total drinking-water intake taken from the well, \( f_{\text{well}} \), which is an alternative source of water. As given in Appendix A, the concentration in the river water including the radionuclides sorbed onto suspended solid material is \( \frac{N_w}{I_w w_w d_w} \), but the concentration in the consumed drinking-water is reduced if the particulates are first filtered. The fraction of the drinking-water which is thus processed is \( f_{\text{filter}} \).
**Dose from the consumption of contaminated well water:**

The dose from the well water pathway is given by

\[ D_{\text{well}} = H_{\text{ing}} I_{\text{wat}} f_{\text{well}} \frac{1+ (1- f_{\text{filter}}) \alpha_L k_L}{\theta_L + (1- \epsilon_L) \rho_L k_L} N_L l_L A_f \]  

\[ \text{[Sv y}^{-1}] \]  

(15)

Again the filtered fraction is evident and the porewater concentration is used.

**Dose from the consumption of contaminated meat products**

The drinking-water doses are among the most straightforward of the exposure pathway representations, but it can be seen how the expressions are complicated by multiple, parallel pathways from the aquifer and the surface water compartments. In the case of the livestock pathways, the parallel nature of the exposure pathways produces greater complexity.

The dose to the human consumer of meat is given by

\[ D_{\text{meat}} = H_{\text{ing}} I_{\text{meat}} K_{\text{meat}} \left\{ I_{\text{livestock water}} + I_{\text{livestock water-irrigation-pasture}} + I_{\text{livestock soil-pasture}} + I_{\text{livestock soil}} \right\} \]  

\[ \text{[Sv y}^{-1}] \]  

(16)

The dose per unit intake is again the appropriate value for ingestion and the consumption rate of meat is clear. The remaining terms define the concentration in the consumed meat:

\[ C_{\text{meat}} = K_{\text{meat}} \left\{ I_{\text{livestock water}} + I_{\text{livestock water-irrigation-pasture}} + I_{\text{livestock soil-pasture}} + I_{\text{livestock soil}} \right\} \]  

\[ \text{[Bq kg}^{-1}] \]  

(17)

The concentration in the consumed animal tissue therefore depends on the biochemical properties of the element and the intake by the animal. This occurs via four intake pathways:

- water consumption by the animal
- consumption of pasture contaminated by the foliar interception of irrigation water
- consumption of pasture contaminated by root uptake
- intake of soil on the surface of the pasture or directly with the pasture.

3 In this representation no distinction is made between the different distribution factors for different tissues that might be consumed in reality and only one animal-specific parameter is used - \( K_{\text{meat}} \) [Bq kg\(^{-1}\) (Bq day\(^{-1}\))]\(^{3}\)]. In the TAME database the values correspond to muscle. Other meat products could be consumed, e.g. offal. This representation therefore corresponds to the assumption that only muscle is consumed (see Section 3.4 for the values used). This could be non-conservative since some radionuclides can accumulate in other organs, e.g. the liver. Similarly, in the subsistence community represented here it is likely that all products would be utilised.
Table 2-6: Parameters used to characterise the dose from the drinking-water pathways. In these expressions, the subscripts W and L refer to the surface water and local aquifer.

<table>
<thead>
<tr>
<th>parameter</th>
<th>units</th>
<th>description</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \alpha_W )</td>
<td>[kg m(^{-3})]</td>
<td>suspended solid load in the surface water</td>
</tr>
<tr>
<td>( \alpha_L )</td>
<td>[kg m(^{-3})]</td>
<td>suspended solid load in the aquifer porewater</td>
</tr>
<tr>
<td>( k_W )</td>
<td>[(Bq kg(^{-1})) (Bq m(^{-3}))(^{-1})]</td>
<td>nuclide specific solid - liquid distribution coefficient in the water compartment</td>
</tr>
<tr>
<td>( k_L )</td>
<td>[(Bq kg(^{-1})) (Bq m(^{-3}))(^{-1})]</td>
<td>nuclide specific solid - liquid distribution coefficient in the local aquifer</td>
</tr>
<tr>
<td>( \theta_L ) and ( \varepsilon_L )</td>
<td>[-]</td>
<td>volumetric moisture content and porosity of the local aquifer</td>
</tr>
<tr>
<td>( f_{\text{filter}} )</td>
<td>[-]</td>
<td>fraction between 0 and 1 which determines the amount of suspended solid material filtered out of the water before consumption</td>
</tr>
<tr>
<td>( f_{\text{well}} )</td>
<td>[-]</td>
<td>fraction of drinking-water obtained from a well in the local aquifer. This is used to partition ( I_{\text{wat}} )</td>
</tr>
<tr>
<td>( I_{\text{wat}} )</td>
<td>[m(^3) y(^{-1})]</td>
<td>exposure factor, the total annual consumption of drinking-water via each of the two pathways</td>
</tr>
<tr>
<td>( H_{\text{ing}} )</td>
<td>[Sv Bq(^{-1})]</td>
<td>dose per unit intake on ingestion</td>
</tr>
</tbody>
</table>
These intakes can be written in terms of the compartment inventories as follows:

**Livestock intake via drinking-water:**

\[
I_{\text{livestock}}^{\text{water}} = I_{\text{w}} \left[ f_{A} \frac{1}{\theta_{L} + (1 - \epsilon_{L}) \rho_{L} k_{L}} \left( N_{L} + (1 - f_{A}) \frac{N_{W}}{I_{w} d_{w} w_{w}} \right) \right] \text{[Bq day}^{-1}] \tag{18}
\]

where \( I_{\text{w}} \) [m\(^3\) day\(^{-1}\)] is the daily water consumption by the animal

and \( f_{A} \) [-] is the fraction of that water obtained from the well

**Livestock consumption of pasture contaminated by irrigation\(^4\):**

\[
I_{\text{livestock}}^{\text{water, irrigation-pasture}} = Z I_{pc} \left[ 1 - \frac{1}{Y_{p}(W_{p} + H_{pc})} \frac{1}{\theta_{c} + (1 - \epsilon_{c}) \rho_{c} k_{c}} \frac{N_{c}}{I_{c} A_{c}} F_{LT} + \frac{N_{w}}{I_{w} d_{w} w_{w}} F_{WT} \right] \text{[Bq day}^{-1}] \tag{19}
\]

The intake of pasture is defined by the parameters

- \( Z \) [-] ratio by weight of fresh pasture to hay

- \( I_{pc} \) [kg day\(^{-1}\)] daily consumption of dry fodder by the animal

which give the equivalent weight of wet pasture consumed daily.

The amount of radionuclide retained by the crop depends on the irrigation interception factor and the removal from the crop as a result of weathering and cropping. These factors are given in the first square brackets, using

- \( \mu_{p} \) [m\(^2\) kg\(^{-1}\)] pasture irrigation mass-interception factor

- \( Y_{p} \) [kg m\(^{-2}\)] yield of pasture

- \( W_{p} \) [y\(^{-1}\)] weathering rate for the radionuclide on pasture

and \( H_{pc} \) [y\(^{-1}\)] harvesting rate for the crop as a result of grazing by cattle and for cattle consumption as hay - see Appendix E.

The second square bracket in Equation (19) deals with the radionuclide concentration in irrigation water (including sorption on suspended solids) and the amount of irrigation

---

\(^4\) Although widely used (LAWSON & SMITH, 1984; MARTIN et al., 1991) and discussed (GROGAN, 1989), the generic TAME expression for the interception of contaminated irrigation water by plants in Equation (19) is not the only one possible. The origin of the expression is unfortunately not clear in the original references. BERGSTRÖM (1996) has pointed out that the expression in the first square bracket in Equation (19) is an approximation valid for long half-life radionuclides with low loss rates and that the time averaging implicit here is an annual average, whereas irrigation is not a continuous process and the averaging should take the individual irrigation periods into account (with duration \( \leq 30 \) days). A number of different possibilities for this process are discussed by BIOMOVS II (1996b). The modelling of this process is under review, but this MiniBIOS formulation was used in the example calculations.
Livestock consumption of pasture contaminated by root uptake:

\[
I_{\text{soil - pasture}}^{\text{livestock}} = I_{pc} ZK_p \frac{1}{(1 - \epsilon_T) \rho_T} \frac{N_T}{l_T A_f} \quad \text{[Bq day}^{-1}] \tag{20}
\]

Here, the concentration in the pasture is derived from the concentration in the top soil using \(K_p\) \([(\text{Bq kg}^{-1} \text{ crop, fresh weight})(\text{Bq kg}^{-1} \text{ soil, dry weight})^{-1}]\), the soil - pasture transfer factor.

Livestock ingestion of soil during grazing:

\[
I_{\text{soil}}^{\text{livestock}} = S_{pc} ZI_{pc} \frac{1}{(1 - \epsilon_T) \rho_T + \epsilon_T \rho_W} \frac{N_T}{l_T A_f} \quad \text{[Bq day}^{-1}] \tag{21}
\]

In this form, \(S_{pc}\) is the fraction of the weight of pasture made up of wet soil.

2.4 The TAME representations in context

Results from the BIOMOVS II working groups on Reference Biospheres and Complementary Studies indicate that there is a general consensus in the international modelling community as to what features, events and processes should be included in biosphere models for waste disposal applications. Work on TAME was started before BIOMOVS II and it is encouraging to note that many of the FEPs included in TAME also appear in the Reference Biospheres FEP list (BIOMOVS II, 1996a) - for example the need to represent near-surface hydrology in the biosphere water cycle has been recognised.

At a more detailed level, the transport processes in the biosphere are modelled in very similar ways in most of the models used in the Complementary Studies exercise (BIOMOVS II, 1996b). This convergence probably reflects the work carried out in the first BIOMOVS project which looked at many similar and related processes. The work of the Complementary Studies working group was intended to develop from this base. Comparison of the exposure pathway sub-models has been a feature of the work and this has revealed that, although there are common FEPs in the models, the mathematical representation of them varies considerably from model to model (particularly for the modelling of the irrigation process). In this respect, the TAME representation is no better and no worse than any of the others. It is hoped that a consensus on the modelling can emerge for the exposure pathway modelling in a similar way to what happened with the modelling of the transport sub-models after the first phase of BIOMOVS. Further work
would then be required to identify suitable parameter values for future developments. As has already been mentioned, the current Swiss database for dose evaluations is not entirely satisfactory but it has much in common with other similar databases internationally.

The examples given here serve to illustrate just how parameter-intensive biosphere modelling can be, particularly with respect to the exposure pathway sub-models. It is well to bear in mind, however, that the TAME exposure pathway sub-model is derived from models for assessing the consequences of nuclear accidents and is considerably simplified compared with its progenitors.
3. Dataset for the example calculations

3.1 Generic data

Following the definition of the TAME conceptual and mathematical models in the previous Section, the remainder of this report deals with how the model behaves. This section describes a generic dataset for TAME and results from the model using it are discussed in Section 4, with reference to accumulation in the transport model and effect on the calculated doses. A Section on the uncertainty in the model results introduced by the assumptions used to set up this representation of a generic Swiss biosphere is also included.

This dataset has been compiled for test purposes but relies on existing databases from earlier Swiss biosphere model implementations. Results from the example calculations are representative of the results to be expected from similar biosphere scenarios but the numerical values should only be used in the context of this report for

1. the verification of the correct implementation of TAME and

2. the quantification of the differences between the results from TAME and earlier models used in Swiss safety assessments (See Appendix F).

This Section provides an overview of the biosphere example used here. Section 3.2 illustrates how the mass balance schemes are set up in the TAME transport calculations and Section 3.3 gives the data used to represent the behavioural aspects of a typical individual from the critical group formed by a closed, self-sustaining agricultural community (see also Appendix E for the distribution of the food consumption rates). Section 3.4 discusses the nuclide- and element-specific data used in the calculations.

The model biosphere used to generate the test results is based on a valley in Central Switzerland. The physical characteristics for compartments in this biosphere are shown in Table 3-1, the assumptions for human behaviour are given in Table 3-2 and the parameters characterising the agricultural products in the model are given in Table 3-3. Figure 3-1 illustrates the physical dimensions of the model compartments. Numerical values used in the example can be regarded as a default dataset for biosphere modelling purposes, because the data are only weakly location-specific, because the model is known not to be sensitive to variations in the parameter values, or because there is an international consensus as to what the appropriate numerical values should be. Many of the parameters have broad applicability - the porosities of the soils, for example, are representative of most regions likely to be considered. Others are taken from the existing Swiss biosphere database compiled for Projekt Gewöhr (NAGRA, 1985). Some of the modelling assumptions and corresponding parameter values are simplistic in nature and represent a first attempt to represent FEPs known to be relevant in certain cases, but which have not previously been implemented in a Swiss biosphere waste disposal model. The effects of these assumptions are discussed with a view to improving the representations in future assessments.
The NRPB model on which the TAME dose model is based includes a database of parameter values suitable for conditions in the United Kingdom. These are not necessarily applicable to Swiss assessments. For this reason the Gewähr values have been translated for the TAME model. This means that, where there is a commonality of modelled FEPs, the TAME database is consistent with that used in earlier Swiss assessments, otherwise the values from the NRPB model are used (MARTIN et al., 1991). This also provides a means of comparing the behaviour of TAME with existing Swiss biosphere assessment models, both to ensure correct implementation of the mathematical model in TAME and to illustrate the enhancements over existing models which accrue from the inclusion of a more broadly based set of FEPs. Appendix F illustrates the results from an intercomparison of TAME with BIOSPH (BÖHRINGER et al., 1986), firstly using a common dataset (i.e. all new FEPs switched off) and secondly with the new FEPs switched on.

To illustrate the behaviour of the model, the release to the local aquifer at a constant rate of $10^6$ Bq y$^{-1}$ for $10^4$ years of each of $^{36}$Cl, $^{135}$Cs and $^{237}$Np is considered. $^{36}$Cl is chosen because it is known to sorb very poorly and so is assumed to have zero $k_d$ in all TAME compartments and therefore to move with the water fluxes. $^{135}$Cs is employed because it has a moderate $k_d$ and because it has been prominent in recent Swiss assessments (NAGRA 1994b) and $^{237}$Np is chosen as an example of a decay chain with relatively high $k_d$s. The decay products of $^{237}$Np are $^{233}$U and $^{229}$Th. These grow in in the biosphere as a result of radioactive decay.

This use of a top-hat function source term allows various important features of the biosphere model to be investigated. In particular, $10^4$ years is long enough for the radionuclide distributions in the TAME compartments to approach steady-state and is comparable with the timescale for significant change in the biosphere, for example the next ice age is likely to occur between $10^4$ and $10^5$ from the present (KLEMENZ, 1993). Similarly, a release time of $10^4$ years is not overlong in comparison with the characteristic timescales of the peak releases from geological radioactive waste.
repositories. The sharp cut-off at the end of the source can be used to illustrate the retention of radionuclides in the biosphere.

It should be noted that the radionuclides, source term and biosphere characteristics given here represent only a small fraction of the parameter space which TAME is able to model. The results presented here are intended to give an overview of the behaviour of the model, based on an example biosphere with the mass balance schemes for water and solid material fluxes which are discussed in the next section. Similarly, the dietary preferences of inhabitants of the modelled region are also given below. The intention is not to discuss the minutiae of the model's performance in this document - details are given in the biosphere modelling reports for specific safety assessment applications (NAGRA, 1994a; NAGRA, 1994b) or in other reports detailing aspects of model behaviour of particular interest. Further details can also be found in the reports of the BIOMOVS II Working Group on Complementary Studies (BIOMOVS II, 1996b).

3.2 Definition of the mass balance scheme

3.2.1 The basis for defining mass balance in TAME

An important feature of TAME is that transport of radionuclides in the compartment model is wholly derived from mass balance considerations for both water and solid material in the system. The relationship between the radionuclide transfer coefficients and the water and solid fluxes is derived from first principles in Appendices A and B. The mass balance schemes for the example calculations given here are shown in Figure 3-2, with the corresponding numerical data given in Table 3-4 and Table 3-5 for water and solid fluxes respectively. In general, the balance of the water fluxes is most important since this determines some of the fluxes of solid material, for example suspended solid material in the surface water or in soils. As can be seen from the numerical values in the tables, the overall mass balance scheme must be written with high numerical precision since there can be many orders of magnitude difference between some of the components (though only two significant figures are given here). The mass balance calculations in TAME are therefore carried out in double precision in the codes (compare, for example, the values for $M_{LT}$ and $M_{WE}$).

In setting up the mass balance schemes for this example biosphere, all of the processes discussed in Section 2.2.1 (and in Appendix F) have been included, albeit in a generic way since the biosphere described by the TAME dataset here is not precisely based on the detailed description of a specific site (although it is representative of a valley in Central Switzerland). The generic nature of the biosphere has consequences for the way in which the boundary conditions for the model compartments are interpreted and these are discussed in the following two sections.
Table 3-1: Characteristics of the TAME compartments in the example biosphere. Data are based on a valley in Central Switzerland (KLOS et al, 1995) except as indicated below.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>solid material in suspension in water</td>
<td>( \alpha_T )</td>
<td>( 1.0 \times 10^{-3} )</td>
<td>kg m(^{-3} )</td>
</tr>
<tr>
<td>porosity</td>
<td>( \varepsilon_T )</td>
<td>0.4</td>
<td>-</td>
</tr>
<tr>
<td>thickness</td>
<td>( l_T )</td>
<td>0.25</td>
<td>m</td>
</tr>
<tr>
<td>volumetric moisture content</td>
<td>( \theta_T )</td>
<td>0.3</td>
<td>-</td>
</tr>
<tr>
<td>compartment tortuosity(^C)</td>
<td>( T_T )</td>
<td>3.9</td>
<td>-</td>
</tr>
<tr>
<td>Top Soil</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>solid material in suspension in water</td>
<td>( \alpha_D )</td>
<td>( 1.0 \times 10^{-3} )</td>
<td>kg m(^{-3} )</td>
</tr>
<tr>
<td>porosity</td>
<td>( \varepsilon_D )</td>
<td>0.4</td>
<td>-</td>
</tr>
<tr>
<td>thickness</td>
<td>( l_D )</td>
<td>0.75</td>
<td>m</td>
</tr>
<tr>
<td>biomass involved in bioturbative transport(^D)</td>
<td>( m_D )</td>
<td>0.1</td>
<td>kg m(^{-2} )</td>
</tr>
<tr>
<td>biomass activity(^D)</td>
<td>( w_D )</td>
<td>20</td>
<td>y(^{-1} )</td>
</tr>
<tr>
<td>volumetric moisture content</td>
<td>( \theta_D )</td>
<td>0.3</td>
<td>-</td>
</tr>
<tr>
<td>compartment tortuosity(^C)</td>
<td>( T_D )</td>
<td>3.9</td>
<td>-</td>
</tr>
<tr>
<td>Deep Soil</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>solid material in suspension in water</td>
<td>( \alpha_L )</td>
<td>( 1.0 \times 10^{-3} )</td>
<td>kg m(^{-3} )</td>
</tr>
<tr>
<td>porosity</td>
<td>( \varepsilon_L )</td>
<td>0.2</td>
<td>-</td>
</tr>
<tr>
<td>thickness</td>
<td>( l_L )</td>
<td>15.0</td>
<td>m</td>
</tr>
<tr>
<td>volumetric moisture content</td>
<td>( \theta_L )</td>
<td>0.2</td>
<td>-</td>
</tr>
<tr>
<td>compartment tortuosity(^C)</td>
<td>( T_L )</td>
<td>8.6</td>
<td>-</td>
</tr>
<tr>
<td>Local Aquifer</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>suspended sediment load in river(^a)</td>
<td>( \alpha_W )</td>
<td>0.1</td>
<td>kg m(^{-3} )</td>
</tr>
<tr>
<td>depth of river</td>
<td>( d_W )</td>
<td>0.5</td>
<td>m</td>
</tr>
<tr>
<td>width of river</td>
<td>( w_W )</td>
<td>10</td>
<td>m</td>
</tr>
<tr>
<td>length of river section</td>
<td>( l_W )</td>
<td>( 1.0 \times 10^{3} )</td>
<td>m</td>
</tr>
<tr>
<td>Surface Water</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>River Bed Sediment</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>porosity</td>
<td>( \varepsilon_S )</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>annual exchange with water column(^B)</td>
<td>( \kappa_S )</td>
<td>1</td>
<td>y(^{-1} )</td>
</tr>
<tr>
<td>thickness of sediment</td>
<td>( l_S )</td>
<td>0.1</td>
<td>m</td>
</tr>
<tr>
<td>transfer of sediment to river banks(^H)</td>
<td>( s_S )</td>
<td>0.1</td>
<td>kg m(^{-2} ) y(^{-1} )</td>
</tr>
<tr>
<td>volumetric moisture content</td>
<td>( \theta_S )</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>compartment tortuosity(^C)</td>
<td>( T_S )</td>
<td>2.9</td>
<td>-</td>
</tr>
<tr>
<td>Miscellaneous</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>recipient compartment</td>
<td>-</td>
<td>Local Aquifer</td>
<td></td>
</tr>
<tr>
<td>area of agricultural land</td>
<td>( A_f )</td>
<td>( 1 \times 10^{6} )</td>
<td>m(^2 )</td>
</tr>
<tr>
<td>diffusion constant in pure water</td>
<td>( D_0 )</td>
<td>( 3.8 \times 10^{-2} )</td>
<td>m(^2 ) y(^{-1} )</td>
</tr>
<tr>
<td>annual average evapotranspiration</td>
<td>( d_{T_A} )</td>
<td>0.5</td>
<td>m(^3 ) m(^{-2} ) y(^{-1} )</td>
</tr>
<tr>
<td>annual average precipitation</td>
<td>( d_{A_T} )</td>
<td>1.5</td>
<td>m(^3 ) m(^{-2} ) y(^{-1} )</td>
</tr>
<tr>
<td>annual average erosion</td>
<td>( m_e )</td>
<td>0.1</td>
<td>kg m(^{-2} ) y(^{-1} )</td>
</tr>
<tr>
<td>density of water</td>
<td>( \rho_W )</td>
<td>( 1.0 \times 10^{3} )</td>
<td>kg m(^{-3} )</td>
</tr>
<tr>
<td>dry density of solid material in i = T, D, L, S</td>
<td>( \rho_i )</td>
<td>( 2.65 \times 10^{3} )</td>
<td>kg m(^{-3} )</td>
</tr>
</tbody>
</table>

Notes:

\* This parameter is derived from mass balance considerations taking into account all input and output fluxes of solid material.

\† In the scheme for mass balance, this transfer is set to be numerically equal to the input of eroded soil to the river water. This value represents the mass of sediment transferred to the area of top soil.

C See Appendix C

D See Appendix D
Table 3-2: Parameter values characterising human behaviour and practices in the example biosphere. Most data values used in this model representation are taken from the Projekt Gewihr database (NAGRA, 1985). Those parameters derived from the existing database are shown, as are those newly introduced to the Swiss biosphere model.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Basic human requirements and behaviour</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>annual average food energy intake(^{(1)})</td>
<td>(E_o)</td>
<td>(4.6 \times 10^6)</td>
<td>kJ y(^{-1})</td>
</tr>
<tr>
<td>annual average breathing rate(^{(1)})</td>
<td>(l_{air})</td>
<td>(8.4 \times 10^3)</td>
<td>m(^3) y(^{-1})</td>
</tr>
<tr>
<td>annual average fluid intake(^{(1)})</td>
<td>(l_{wat})</td>
<td>1.0</td>
<td>m(^3) y(^{-1})</td>
</tr>
<tr>
<td>fraction of drinking-water obtained from well(^{(2)})</td>
<td>(f_{well})</td>
<td>1.0</td>
<td>-</td>
</tr>
<tr>
<td>occupancy factor at normal airborne dust conc.(^{(3)})</td>
<td>(O_r)</td>
<td>0.966</td>
<td>y y(^{-1})</td>
</tr>
<tr>
<td>occupancy factor at high airborne dust conc.(^{(3)})</td>
<td>(O_f)</td>
<td>0.034</td>
<td>y y(^{-1})</td>
</tr>
<tr>
<td><strong>Food energy contents</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>eggs</td>
<td>(n_{eggs})</td>
<td>(3.49 \times 10^2)</td>
<td>kJ egg(^{-1})</td>
</tr>
<tr>
<td>milk</td>
<td>(n_{milk})</td>
<td>(2.81 \times 10^6)</td>
<td>kJ m(^{-3})</td>
</tr>
<tr>
<td>meat</td>
<td>(n_{meat})</td>
<td>(7.31 \times 10^3)</td>
<td>kJ kg(^{-1})</td>
</tr>
<tr>
<td>fish(^{(4)})</td>
<td>(n_{fish})</td>
<td>(7.31 \times 10^3)</td>
<td>kJ kg(^{-1})</td>
</tr>
<tr>
<td>root vegetables</td>
<td>(n_{rv})</td>
<td>(2.86 \times 10^3)</td>
<td>kJ kg(^{-1})</td>
</tr>
<tr>
<td>grain</td>
<td>(n_{gr})</td>
<td>(1.51 \times 10^4)</td>
<td>kJ kg(^{-1})</td>
</tr>
<tr>
<td>green vegetables</td>
<td>(n_{gv})</td>
<td>(8.40 \times 10^2)</td>
<td>kJ kg(^{-1})</td>
</tr>
<tr>
<td><strong>Dietary preferences</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>fractional intake of milk</td>
<td>(f_{milk})</td>
<td>0.203</td>
<td>-</td>
</tr>
<tr>
<td>fractional consumption rate of eggs</td>
<td>(f_{egg})</td>
<td>0.0152</td>
<td>-</td>
</tr>
<tr>
<td>fractional consumption rate of fish</td>
<td>(f_{fish})</td>
<td>(3.18 \times 10^{-3})</td>
<td>-</td>
</tr>
<tr>
<td>fractional consumption rate of vegetables</td>
<td>(f_{veg})</td>
<td>0.810</td>
<td>-</td>
</tr>
<tr>
<td>of which fraction of (f_{veg})</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>from grain</td>
<td>(p_{gr})</td>
<td>0.755</td>
<td>-</td>
</tr>
<tr>
<td>from green vegetables</td>
<td>(p_{gv})</td>
<td>0.0177</td>
<td>-</td>
</tr>
<tr>
<td>from root vegetables</td>
<td>(p_{rv})</td>
<td>0.227</td>
<td>-</td>
</tr>
<tr>
<td><strong>Environmental factors</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>normal airborne dust loading(^{(3)})</td>
<td>(a_r)</td>
<td>(5.0 \times 10^{-8})</td>
<td>kg m(^{-3})</td>
</tr>
<tr>
<td>high airborne dust loading(^{(3)})</td>
<td>(a_f)</td>
<td>(1.0 \times 10^{-5})</td>
<td>kg m(^{-3})</td>
</tr>
</tbody>
</table>

Notes:

1. Inhalation doses in TAME use the annual adult breathing rate taken from ICRP Reference Man (ICRP, 1974). Data for the fluid intake and the food energy requirement of the reference individual in the biosphere are based on the Projekt Gewihr values (NAGRA, 1985).

2. Drinking-water for human consumption could be obtained from the river. In this example it is assumed (pessimistically) that the source is a well in the local aquifer (i.e. the primary compartment receiving the input of radionuclides from outside the biosphere system - see Table 3-1). Drinking-water is not assumed to be filtered \((\text{filter} = 0)\) in Equations (14) and (15)). Irrigation water is obtained in equal quantities from both the well and surface water (Table 3-1).

3. High airborne dust levels are associated with occupational activities such as ploughing. In deriving this value, 12 days per year is assumed for occupancy at the increased dust levels, the remainder of the time is assumed to be spent at the lower dust levels found under normal conditions. These default dust loadings are taken from LINSLEY (1978).

4. Assumed to be equal to that of meat from cattle in this example. It is likely to be an overestimation by a factor of around 2. However, as fish is a relatively small part of the diet as modelled here, any uncertainty in the value is insignificant.
Table 3-3: Parameter values characterising the usage of biosphere resources livestock and crops in the example biosphere. Most data values used in the model representation are taken from the Projekt Gewässer database (NAGRA, 1985). Those parameters derived from the existing database are shown, as are the parameters newly introduced to the Swiss biosphere model.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>cattle stocking density</td>
<td>$N_c$</td>
<td>200</td>
<td>cows km$^{-2}$</td>
</tr>
<tr>
<td>daily water consumption</td>
<td>$I_{wc}$</td>
<td>0.03</td>
<td>m$^3$ day$^{-1}$</td>
</tr>
<tr>
<td>daily pasture consumption</td>
<td>$I_{pc}$</td>
<td>20</td>
<td>kg day$^{-1}$</td>
</tr>
<tr>
<td>fraction of drinking-water obtained from well$^{(1)}$</td>
<td>$f_A$</td>
<td>1.0</td>
<td></td>
</tr>
<tr>
<td>daily water consumption</td>
<td>$I_{wp}$</td>
<td>0.2</td>
<td>litre day$^{-1}$</td>
</tr>
<tr>
<td>daily grain consumption</td>
<td>$I_{gp}$</td>
<td>0.07</td>
<td>kg day$^{-1}$</td>
</tr>
<tr>
<td>fraction of water obtained from well</td>
<td>$f_p$</td>
<td>1.0</td>
<td></td>
</tr>
<tr>
<td>grain</td>
<td>$Y_{gr}$</td>
<td>0.4</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td>green vegetables</td>
<td>$Y_{gv}$</td>
<td>3.0</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td>root vegetables</td>
<td>$Y_{rv}$</td>
<td>3.5</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td>pasture (dry weight)</td>
<td>$Y_p$</td>
<td>1.66</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td>grain</td>
<td>$\mu_{gr}$</td>
<td>1.01</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td>green vegetables</td>
<td>$\mu_{gv}$</td>
<td>0.13</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td>root vegetables</td>
<td>$\mu_{rv}$</td>
<td>0.11</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td>pasture</td>
<td>$\mu_p$</td>
<td>0.24</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td>grain (human)</td>
<td>$S_{gr}$</td>
<td>9.0×10$^{-5}$</td>
<td>kg kg$^{-1}$</td>
</tr>
<tr>
<td>green vegetables (human)</td>
<td>$S_{gv}$</td>
<td>2.0×10$^{-4}$</td>
<td>kg kg$^{-1}$</td>
</tr>
<tr>
<td>root vegetables (human)</td>
<td>$S_{rv}$</td>
<td>0.0</td>
<td>kg kg$^{-1}$</td>
</tr>
<tr>
<td>pasture (wet soil / dry pasture - cattle)</td>
<td>$S_{pc}$</td>
<td>1.0×10$^{-2}$</td>
<td>kg kg$^{-1}$</td>
</tr>
</tbody>
</table>

Notes

1. Drinking-water for livestock consumption could be obtained from river water. In this example it is assumed (pessimistically) that the source is a well in the local aquifer (i.e. the compartment receiving the input of radionuclides from outside the biosphere system - see Table 3-1). Water for consumption by livestock is not assumed to be filtered. Irrigation water is obtained in equal quantities from both the well and the surface water (Table 3-1).

2. Yields here refer to the annual weight of produce of each crop type. Harvesting rates are once per year for all crops for human consumption. The cropping rate of pasture is determined by the cattle stocking density and the consumption rate, as well as the pasture growth rate. The ratio of wet-to-dry weight for pasture is 5:1.
3.2.2 Water fluxes

In the valley bottom biosphere represented by this dataset, the principal inputs to the system are the water throughflow in the river \(F_{PW} \text{[m}^3 \text{y}^{-1}]\), the effective precipitation (rainfall less evapotranspiration: \(d_{AT} - d_{TA} \text{[(m}^3 \text{y}^{-1}) \text{m}^{-2}]\)) and the groundwater influx from the implied geosphere \(F_{UL} \text{[m}^3 \text{y}^{-1}]\). Fluxes to the soils from the valley sides are not included in the dataset and it is implicitly assumed that all groundwater in the valley flows through the aquifer. The scheme is illustrated in Figure 3-2 (a) and the numerical values used are given in Table 3-4.

Irrigation is also often required to guarantee crop yields during the drier periods of the year and the irrigation water is taken from both the river and the aquifer \(d_{WT} \text{[(m}^3 \text{y}^{-1}) \text{m}^{-2}]\) and \(d_{LT} \text{[(m}^3 \text{y}^{-1}) \text{m}^{-2}]\). This is applied to the top soil\(^1\). For modelling purposes, it is assumed that irrigation water is taken in equal quantities from both sources.

Other water fluxes play an important rôle in determining the movement of dissolved radionuclides in the system. Values for capillary rise in soils \((d_{LD} \text{ and } d_{DT} \text{[m} \text{y}^{-1}])\) are taken from GROGAN \textit{et al.} (1991). These are believed to be reasonably representative for most locations in Switzerland. The volumetric moisture content is assigned the annual average values for each of the compartments. To distinguish the source of radionuclides entering the biosphere in contaminated water fluxes from other external sources, the flux of contaminated water entering the local aquifer \(F_{CL} \text{[m}^3 \text{y}^{-1}]\) is identified and is arbitrarily set to be equivalent to 1% of the total flux of water into the aquifer from higher up the valley\(^2\). Furthermore, 20% of the total water flux in the aquifer is assumed to exfiltrate to the river, with the remainder flowing downstream \(F_{LE} \text{[m}^3 \text{y}^{-1}]\) and out of the modelled section. The precise quantity is determined by mass balance, as are \(F_{TD}, F_{DL}, \text{ and } F_{LE}\), denoted by the dashed lines in Figure 3-2(a).

The bed sediment is treated simplistically in this representation. It is assumed that there is rapid mixing of the sediment \textit{solids} and the river water (see Section 3.2.3) and it is

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\(^1\) In the TAME dose model, radionuclides in irrigation water are assumed to be intercepted by plant foliage. In reality the intercepted radionuclides would not be part of the water balance in the TAME transport model. However, it is conservatively (and implicitly) assumed that the water and radionuclides are instantaneously transported from the plants to the top soil (since transport assumes direct deposition on soil during irrigation). This consequence of assuming a closed system in which all water and solid material wastes are recycled is reasonable on an annual average. However, this decoupling of the transport and dose models means that the plant concentration is overestimated because the plant concentration due to irrigation supplements the plant concentration due to root uptake and there is effectively double counting of the radionuclides in the soil because some of these radionuclides are still in the plants at the time at which the root uptake is calculated. The effect is not likely to be large.

\(^2\) This is not a significant assumption - it is the release from the geosphere [Bq y\(^{-1}\)] which is important since this determines the amount of radionuclide in the modelled biosphere system. The inflowing \textit{contaminated} water flux allows a concentration in the groundwater in the associated geosphere to be calculated. By employing a contaminated flux, the concentration in the water discharged from the geosphere can be matched to that entering the biosphere.
assumed that the main interaction of the sediment with the other parts of the system is via the movement of solid material. Consequently, river water - bed sediment porewater interactions were neglected, as was exfiltration from the aquifer to the bed sediment - release was assumed to be directly from the aquifer to the river. These assumptions effectively isolate the bed sediment from the rest of the water flux balance scheme but not from the diffusive interactions which are assumed to occur between all contiguous water-filled compartments in TAME. The influence of diffusion on the calculated doses in TAME is discussed in Section 4.6.

3.2.3 Solid material fluxes

At first sight, Figure 3-2 (b) shows that there are many more processes requiring numerical values in the solid flux scheme than in the case of the water fluxes. This is not the case because, as described in Appendix D, many of the solid material fluxes are derived from the transport of solid material in the water fluxes. Thus estimation of the suspended solid load in the water in each of the compartments is required (αi in Table 3-1) together with the corresponding water fluxes in Table 3-4. The values for the solid material fluxes are given in Table 3-5.

Additional inputs to the mass balance system are the local erosion rate, the parameters for bioturbation and estimates of the parameters associated with sedimentation in the river. In this database, the erosion rate of 0.1 kg m\(^{-2}\) y\(^{-1}\) is taken from KLEMENZ (1993). This value is used to give an estimate of erosion in the model area and therefore assumes no deposition from outside the model region. The two parameters for bioturbation are set to the default values discussed in Appendix D.

It is assumed that a balance exists between the material eroded from the soil to the surface water \(M_{TW}\) and the material transported from the bed sediment to the soil \(M_{ST}\) as a result of human action or by flooding. Additionally, the assumption is made that material in the bed sediment is exchanged once per year with the water column. This is not unreasonable for a fast-flowing alpine river, in which high water fluxes during snowmelt are expected to remove all the bed sediment. Bed sediment would then reform during more placid flow regimes. This is the main reason for neglecting the water-flux interactions of the bed sediment. For larger rivers and lakes, this assumption would not be valid. The rôle played by this modelling assumption is investigated by setting the turnover rate \(\kappa_{SW} [y^{-1}]\) to zero and the results are discussed in Section 4.6.

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As is discussed in Section 4.3, these simplifying assumptions had unforeseen consequences, from which it is concluded that, in a system with a relatively high degree of internal detail, the same level of detail should be applied to each of the FEPs.
The TAME mass balance schemes for the example biosphere calculations. The corresponding numerical values are given in Table 3-4 and Table 3-5. The dashed lines indicate parameters determined by mass balance. Diffusive fluxes between contiguous saturated and partially saturated compartments are not shown. Note the simplifying assumption which isolates the bed sediment in the water flux balance. Bed sediment interactions are assumed to be dominated by solid material transport.
Table 3-4: Mass balance for the water fluxes in the example biosphere. Numerical values are given to two significant figures. All data are representative of a valley bottom biosphere in Central Switzerland (see KLOS et al., 1995). Additional sources of data are noted below.

<table>
<thead>
<tr>
<th>Water flux</th>
<th>Symbol</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Regional rainfall</td>
<td>$d_{TA}$</td>
<td>1.5</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td>Regional evapotranspiration</td>
<td>$d_{AT}$</td>
<td>0.5</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td>Deep soil → top soil</td>
<td>$d_{DT}$</td>
<td>0.1</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td>Local aquifer → deep soil</td>
<td>$d_{LD}$</td>
<td>0.1</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td>Irrigation from local aquifer</td>
<td>$d_{LT}$</td>
<td>0.15</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td>Irrigation from river water</td>
<td>$d_{WR}$</td>
<td>0.15</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td>Inflow to local aquifer (uncontaminated)</td>
<td>$F_{UL}$</td>
<td>$3.2 \times 10^6$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Inflow to local aquifer (contaminated)</td>
<td>$F_{CL}$</td>
<td>$3.2 \times 10^4$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Exfiltration, local aquifer to river water</td>
<td>$F_{LW}$</td>
<td>$6.4 \times 10^5$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Flow into surface water</td>
<td>$F_{PW}$</td>
<td>$3.0 \times 10^8$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Top soil → deep soil</td>
<td>$F_{TD}$</td>
<td>$1.4 \times 10^6$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Deep soil → local aquifer</td>
<td>$F_{DL}$</td>
<td>$1.4 \times 10^6$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Local aquifer downstream flow</td>
<td>$F_{LE}$</td>
<td>$3.7 \times 10^6$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Surface water downstream flow</td>
<td>$F_{WE}$</td>
<td>$3.0 \times 10^8$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
</tbody>
</table>

Notes:
1. The corresponding volumetric fluxes are obtained by multiplying by the area of farmland $F_i = d_i A_f$.
3. The volume of contaminated water (from the assumed geosphere) entering the system is assumed to be 1% of the total flow in the aquifer.
4. The downstream flow in the aquifer is assumed to be 20% of total flow in the aquifer - the remaining 80% exfiltrates to the river.
Table 3-5: Mass balance for the solid material fluxes in the example biosphere. The symbols used for the parameters are given in Table 3-1. Numerical values are only given to two significant figures. Note that many of these fluxes are based on transport of solid material suspended in the water fluxes defined in Table 3-4. All data are taken from KLOS et al. (1995).

<table>
<thead>
<tr>
<th>Solid Material Flux kg y⁻¹</th>
<th>Symbol</th>
<th>Derivation</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Top soil → surface water</td>
<td>( M_{TW} )</td>
<td>( m_s A_f + \alpha_F F_{TW} )</td>
<td>( 1.0 \times 10^5 )</td>
</tr>
<tr>
<td>Deep soil → top soil(0)</td>
<td>( M_{DT} )</td>
<td>( w_D m_D + m_s A_f + \alpha_D F_{DT} )</td>
<td>( 2.1 \times 10^6 )</td>
</tr>
<tr>
<td>Local aquifer → deep soil</td>
<td>( M_{LD} )</td>
<td>( m_s A_f + \alpha_L F_{LD} )</td>
<td>( 1.0 \times 10^5 )</td>
</tr>
<tr>
<td>Previous river compartment → river water</td>
<td>( M_{PW} )</td>
<td>( \kappa_F F_{PW} )</td>
<td>( 3.0 \times 10^7 )</td>
</tr>
<tr>
<td>Local aquifer → top soil irrigation</td>
<td>( M_{LT} )</td>
<td>( \alpha_L F_{LT} )</td>
<td>( 1.5 \times 10^5 )</td>
</tr>
<tr>
<td>River water → top soil irrigation</td>
<td>( M_{WT} )</td>
<td>( \alpha_W F_{WT} )</td>
<td>( 1.5 \times 10^4 )</td>
</tr>
<tr>
<td>Bed sediment → top soil(0)</td>
<td>( M_{ST} )</td>
<td>( M_{TW} )</td>
<td>( 1.5 \times 10^5 )</td>
</tr>
<tr>
<td>Bed sediment → river water(0)</td>
<td>( M_{SW} )</td>
<td>( \kappa_s w_{SW} w_d s (1 - \varepsilon_s) \rho_s )</td>
<td>( 1.3 \times 10^6 )</td>
</tr>
<tr>
<td>Local aquifer → river water</td>
<td>( M_{LW} )</td>
<td>( \alpha_L F_{LW} )</td>
<td>( 6.4 \times 10^2 )</td>
</tr>
<tr>
<td>Downstream flow in local aquifer</td>
<td>( M_{LE} )</td>
<td>( \alpha_L F_{LE} )</td>
<td>( 3.7 \times 10^3 )</td>
</tr>
<tr>
<td>Input from the geosphere</td>
<td>( M_{GL} )</td>
<td>( m_s A_f )</td>
<td>( 1.0 \times 10^5 )</td>
</tr>
<tr>
<td>Top soil → deep soil</td>
<td>( M_{TD} )</td>
<td>derived from mass balance</td>
<td>( 2.1 \times 10^6 )</td>
</tr>
<tr>
<td>Deep soil → local aquifer</td>
<td>( M_{DL} )</td>
<td>derived from mass balance</td>
<td>( 1.2 \times 10^5 )</td>
</tr>
<tr>
<td>Local aquifer → geosphere</td>
<td>( M_{LG} )</td>
<td>derived from mass balance</td>
<td>( 1.1 \times 10^5 )</td>
</tr>
<tr>
<td>Surface water → bed sediment</td>
<td>( M_{WS} )</td>
<td>derived from mass balance</td>
<td>( 1.4 \times 10^6 )</td>
</tr>
<tr>
<td>Surface water downstream flow</td>
<td>( M_{WE} )</td>
<td>derived from mass balance</td>
<td>( 3.0 \times 10^7 )</td>
</tr>
</tbody>
</table>

Notes:

† This flux is defined to be equal to the amount removed from the top soil; see text for details.

D See Appendix D
The transfer of solid material from the top soil to the deep soil (\(M_{TD}\)), the deep soil to the local aquifer (\(M_{DL}\)), the surface water to bed sediment (\(M_{WS}\)) and the transport of solid material downstream (\(M_{WE}\)) are all calculated by mass balance.

One further flux is also calculated, which is a direct consequence of the boundary conditions applied to the modelled region. The system of compartments is assumed to have a constant volume. As discussed earlier, the only way in which the volume of a compartment can be maintained constant and be eroded at the same time is for there to be an effective transfer of material from the supporting compartment. This accounts for the erosion terms in the derivation of \(M_{TW}\), \(M_{DT}\), \(M_{LD}\) and \(M_{GL}\). This latter quantity is the effective flux of solid material from the implied geosphere to the local aquifer as a result of the need to keep the compartment volumes constant. The final flux calculated here is the net flux from the local aquifer to the geosphere (as a result of deposition of solid material on the top soil). Note that the mass balance calculations reveal this region to be one of net deposition since \(M_{LG} > M_{GL}\). Potentially this passive transfer could be very important, since the highly sorbing radionuclides would accumulate in the deeper parts of the system. The effect of setting this component of \(\lambda_{ij}\), \(\lambda_{ij}\) and \(\lambda_{ij}\) equal to zero is discussed in Section 4.6.

This interaction with the implied geosphere is not assumed to be involved directly with the input of radionuclides to the biosphere - in practice the release fluxes are determined by the output from the geosphere models. It should also be noted that there could be external deposition on the top soil either by wind deposition or by erosion of the valley sides. These processes are not included here.

### 3.3 Definition of diet, preferences and agricultural practices

The diet of the typical adult inhabitant\(^4\) of the biosphere is based on a food energy intake of around 3000 kcal day\(^{-1}\), corresponding to 4.6×10\(^6\) kJ y\(^{-1}\). This figure and the dietary

\(^4\) Doses to adults are calculated because of the requirement in the Swiss Regulatory Guidelines for exposures arising from radioactive waste repositories that doses to an average individual in the population group be calculated (HSK/KSA, 1993). While it is true that the radiosensitivity of children can be significantly greater than that of adults, the approach is taken that the average individual is better defined as an adult than as a child. Principally, this is because the nature of the releases to the biosphere in the situations for which TAME is intended are of long duration and the exposures are likely to be at the calculated levels throughout the lifetime of the affected individual. In this sense the calculated value can be thought of as an annual average of the lifetime dose. The adoption of the dose to the average adult as the yardstick for radiological impact is therefore potentially mildly non-conservative since the definition of the average individual should be weighted to include the dose per unit intake values for children. Conversely, the intake rates based on diet are lower for children. The overall effect of using diet and habits of typical adults is therefore not likely to significantly misrepresent the radiological impact of potential releases from the repository.
preferences shown in Table 3-2 are based on the data used in *Projekt Gewähr* (NAGRA, 1985) and these are in broad agreement with the typical present-day Swiss diet.

In calculating the inhalation doses, the breathing rate is required. The value used here is taken from ICRP's Reference Man (ICRP, 1974). Intake of activity by animals by inhalation is not considered.

In *TAME*, the consumption rates of the seven foodstuffs (including drinking-water) are determined by specifying the fraction of the annual food energy intake derived from each source (Table 3-2). The true consumption rates for each of the foodstuffs are calculated by the code at run-time, using the relationships given in Appendix E. The fractional consumption rates used in this example calculation correspond to the consumption rates defined for *Projekt Gewähr*. In Table 3-2, the fractional consumption rates are given with sufficient precision to duplicate the *Gewähr* values exactly, with the exception that in this case it is assumed (for example purposes) that 2 kg of fish is consumed with a corresponding decrease in the quantity of meat. The numerical results for diet are given in Appendix E and show that the numerical values are equivalent to the previous formulation.

In this representation of the biosphere, well water is used as the exclusive source of drinking-water for both humans and livestock. This feature is denoted in Tables 3-2 and 3-3 by the factors \( f_{\text{well}} \) for humans, \( f_A \) for cattle and \( f_p \) for poultry.

The data values for the consumption rates of livestock are taken from *Projekt Gewähr*. For cattle, the values are typical of Swiss animals in that mixed herds of dairy and beef cattle are common. The values for soil consumed during feeding are likewise taken from the *Gewähr* database.

Crop yields are taken directly from *Projekt Gewähr* but the new parameterisation of the irrigation process - taken from the NRPB approach - means that the more directly measurable irrigation mass interception factor is now used in *TAME*. Alternative numerical values are available, but the numbers used here are derived from the corresponding *Gewähr* values.

### 3.4 Radionuclide- and element-specific data

These parameters can be split into three classes: chemical retardation in the transport sub-model, uptake and accumulation in the exposure pathways sub-model and dosimetric data.

The \( k_d \) values used here are based on the values derived using the methodology from TITS *et al.* (1996); the radionuclide-specific data for the examples are given in Table 3-6 and the dosimetric data are given in Table 3-7. The adoption of the NRPB exposure pathway model means that additional, weakly element-dependent parameters are also included here. These are shown in Table 3-8.
Table 3-6: Solid-liquid equilibrium distribution coefficients for Kristallin-I biosphere modelling. Data taken from TITS et al. (1996).

<table>
<thead>
<tr>
<th>Element</th>
<th>Coarse</th>
<th>Fine</th>
<th>Local aquifer (coarse)</th>
<th>Deep soil (coarse)</th>
<th>Top soil (coarse)</th>
<th>Surface water (fine)</th>
<th>Aquatic sediment (coarse)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cl</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Cs</td>
<td>0.1</td>
<td>1.0</td>
<td>0.1</td>
<td>0.1</td>
<td>1.0</td>
<td>0.1</td>
<td>0.1</td>
</tr>
<tr>
<td>Np</td>
<td>1.0</td>
<td>10</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>10.0</td>
<td>1.0</td>
</tr>
<tr>
<td>U</td>
<td>1.0</td>
<td>10</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>10.0</td>
<td>1.0</td>
</tr>
<tr>
<td>Th</td>
<td>1.0</td>
<td>10</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>10.0</td>
<td>1.0</td>
</tr>
</tbody>
</table>

Table 3-7: Radionuclide and dosimetric data used in the example calculations.

<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Half-life [years]</th>
<th>Dose per unit intake on ingestion [Sv Bq(^{-1})] NRPB (1987)</th>
<th>Dose per unit intake on inhalation [Sv Bq(^{-1})] NRPB (1987)</th>
<th>γ-ray exposure factor [Sv (Bq m(^{-3}))(^{-1}) year(^{-1})] SVENSSON (1979)</th>
</tr>
</thead>
<tbody>
<tr>
<td>(^{36})Cl</td>
<td>3.01×10(^5)</td>
<td>8.2×10(^{-10})</td>
<td>5.5×10(^{-9})</td>
<td>-</td>
</tr>
<tr>
<td>(^{135})Cs</td>
<td>2.30×10(^6)</td>
<td>1.7×10(^{-9})</td>
<td>1.1×10(^{-9})</td>
<td>-</td>
</tr>
<tr>
<td>(^{237})Np</td>
<td>2.14×10(^6)</td>
<td>1.1×10(^{-6})</td>
<td>1.3×10(^{-4})</td>
<td>1.6×10(^{-11})</td>
</tr>
<tr>
<td>(^{233})U</td>
<td>1.59×10(^5)</td>
<td>7.1×10(^{-8})</td>
<td>3.6×10(^{-5})</td>
<td>5.3×10(^{-13})</td>
</tr>
<tr>
<td>(^{229})Th</td>
<td>7.34×10(^3)</td>
<td>1.1×10(^{-6})</td>
<td>4.7×10(^{-4})</td>
<td>1.8×10(^{-11})</td>
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</tbody>
</table>
Table 3-8: Radionuclide- and element-specific data for the example biosphere calculations.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Cl</th>
<th>Cs</th>
<th>Np</th>
<th>U</th>
<th>Th</th>
</tr>
</thead>
<tbody>
<tr>
<td>Concentration factor for fish (Bq kg⁻¹)(Bq m⁻³)⁻¹</td>
<td>1.0×10⁻² 1.0 1.0×10⁻² 2.0×10⁻³</td>
<td>3.0×10⁻²</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distribution factor for meat (Bq kg⁻¹)(Bq day⁻¹)⁻¹</td>
<td>8.0×10⁻² 2.6×10⁻² 2.0×10⁻⁴</td>
<td>2.0×10⁻⁴ 2.0×10⁻⁴</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distribution factor for milk (Bq kg⁻¹)(Bq day⁻¹)⁻¹</td>
<td>5.0×10⁻² 7.1×10⁻³ 5.0×10⁻⁶</td>
<td>3.7×10⁻⁴ 5.0×10⁻⁶</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distribution factor for eggs (Bq egg⁻¹)(Bq day⁻¹)⁻¹</td>
<td>1.0 2.5×10⁻² 4.4×10⁻⁴</td>
<td>5.1×10⁻² 5.0×10⁻²</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transfer factor for grain (Bq kg⁻¹ fresh weight) (Bq kg⁻¹ dry soil)⁻¹</td>
<td>4.5×10¹ 1.3×10⁻² 1.5×10⁻²</td>
<td>1.3×10⁻² 7.1×10⁻⁴</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food processing retention factor for grain †</td>
<td>0.15 0.15 0.10 0.15 0.15</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weathering rate for grain †</td>
<td>y⁻¹ 8.4 8.4 5.06 8.4 8.4</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transfer factor for green veg. (Bq kg⁻¹ fresh weight) (Bq kg⁻¹ dry soil)⁻¹</td>
<td>5.0 1.3×10⁻² 2.7×10⁻³</td>
<td>3.8×10⁻⁴ 3.8×10⁻⁴</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food processing retention factor for green veg. ‡</td>
<td>- 0.5 0.5 0.5 0.5 0.5</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weathering rate for green veg. ‡</td>
<td>y⁻¹ 18.1 18.1 5.06 18.1 18.1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transfer factor for root veg. (Bq kg⁻¹ fresh weight) (Bq kg⁻¹ dry soil)⁻¹</td>
<td>7.5 8.0×10⁻³ 6.0×10⁻²</td>
<td>5.7×10⁻⁴ 5.7×10⁻⁴</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food processing retention factor for root veg. ‡</td>
<td>- 0.5 0.5 0.5 0.5 0.5</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weathering rate for root veg. ‡</td>
<td>y⁻¹ 18.1 18.1 18.1 18.1 18.1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Translocation factor for root veg.</td>
<td>y⁻¹ 2.02 2.02 0.0 0.0 0.18</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transfer factor for pasture (Bq kg⁻¹ fresh weight) (Bq kg⁻¹ dry soil)⁻¹</td>
<td>1.3×10¹ 2.0×10⁻² 2.4×10⁻³</td>
<td>9.5×10⁻⁴ 9.5×10⁻⁴</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weathering rate for pasture ‡</td>
<td>y⁻¹ 18.1 18.1 18.1 18.1 18.1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: All data are taken from Projekt Gewähr (NAGRA, 1985), with the following exceptions
† GROGAN (1985),
‡ SIMMONDS & CRICK (1982).
4. Model results

4.1 Introduction

The main purpose of the TAME calculations is to provide illustrative assessments of the radiological impact of releases to the modelled biosphere on hypothetical individuals. As discussed in Section 1, the end-point of the calculations is the maximum annual individual dose \([Sv \text{ y}^{-1}]\) received by the members of the critical group (see also Section 3.3) at any time after the start of the release from the geosphere. However, as is clear from Section 2, the formulation of TAME means that there are manifold intermediate quantities in the calculations. Some of these - the ones most useful in understanding how the maximum annual individual dose is calculated - are discussed below. They include

- The TAME intercompartmental transfer coefficients. These determine the temporal characteristic of the response of the modelled system as well as the distribution of the radionuclides in the biosphere compartments. In their turn, they are sensitive to input parameters, such as material fluxes, compartmental \(k_{dS}\) and other physical, chemical and biological properties of the biosphere compartments. This sensitivity can be seen in an analysis of the factors affecting the annual individual doses\(^1\). See Section 4.2.

- The compartmental inventories (Section 4.3) are the end-points of the solution of Equation (1). They indicate the distribution of radionuclides in the biosphere as a result of the transfer coefficients and the source of radionuclides and form the basis for further calculations in the exposure pathway sub-model. An analysis of the compartmental inventories provides a detailed understanding of the PEPs in the model.

- The foodstuff concentrations (Section 4.4) are an intermediate stage in the calculation of the total annual individual dose. They are discussed here because of the way they demonstrate the effects of the multiple pathways approach taken in TAME and because they show the intake rates of radionuclides by the food consumption pathway.

- The ultimate end-points are the annual individual doses from all of the exposure pathways modelled in TAME (Section 4.5). The results from TAME are presented and the time evolution of the doses is discussed in terms of the contributing pathways.

TAME calculates the time evolution of the output quantities even though this information is not a necessary part of the assessment requirements. There are many reasons for this. An understanding of the dynamics of the modelled system enables identification of

\(^1\) Playing a corresponding rôle in the exposure pathway sub-model are the pathway exposure rates. The intake rates of the different food consumption pathways are taken from the JISKRA (1985) database used for Projekt Gewähr, albeit using a revised methodology. The methodology and the intake rates are presented in Appendix E. The methodology allows for variability in human consumption habits to be modelled but the sensitivity of annual individual dose to the consumption rates is small (KLOS et al., 1990; NEA, 1993).
those relatively fast-acting processes which will have an effect before there is a significant change in the biosphere. Similarly, the longer-acting processes are identified. As is argued in Section 4.2, the temporal dynamics of the system are determined by the FEPs included in the analysis and the important FEPs must be identified to present a defensible safety assessment. The evaluation of the dynamics of the system is a relatively trivial step in the analysis of the system as defined by the analysis of the FEPs.

4.2 The TAME transfer coefficient rate constants

The data discussed in the previous sections enable TAME to calculate the transfer coefficients for the biosphere system. These are then used to calculate the inventories in the compartments as a function of time and hence the concentrations in the food-chains. The behaviour and habit data are then used to define the exposure rates for the various pathways and these are subsequently used to calculate doses.

The calculated, time-independent intercompartmental transfer coefficients define the dynamics of transport in the modelled biosphere. The numerical values in these calculations range from $4 \times 10^{-6}$ y$^{-1}$ for the transfer of highly sorbed radionuclides from the deep soil to the top soil, to $6 \times 10^{4}$ y$^{-1}$, which characterises the downstream flow in the river for all radionuclides. Table 4-1 shows the half-times corresponding to these transfers. The half-times, $\tau_{ij}$ [y], defined by

$$
\tau_{ij} = \frac{\ln 2}{\lambda_j} \quad [y]. \quad (22)
$$

show a similarly broad range of characteristic timescales, with the influence of retention in the compartments (modelled by the compartmental $k_d$s) being clearly seen.

This influence can be seen in several of the other transfers and particularly in the rôle played by the transport of radionuclides on solid material. For example, there are transfers from sediment to top soil and to water and transfers from water to sediment and from top soil to water, all of which are driven by solid material fluxes (i.e. no water fluxes are associated with these transfers in this representation and the transfers only involve the radionuclides sorbed onto the solids - see Appendix D). As $^{36}$Cl has zero $k_d$ in all compartments, the corresponding transfer rates for these mechanisms are all zero (corresponding to infinite half-times). In going from the zero top soil $k_d$ of $^{36}$Cl to $^{135}$Cs ($0.1 \text{ m}^3 \text{kg}^{-1}$) and the $^{237}$Np chain ($1.0 \text{ m}^3 \text{kg}^{-1}$), the corresponding half-times for the top soil to deep soil transfer are $3.7 \times 10^{-2}$ y for $^{36}$Cl and 17 y for $^{135}$Cs, rising to 78 y for the $^{237}$Np chain. Similar results are seen for the transfer from deep soil up to the top soil. For $^{36}$Cl the half-time is 1.4 y and this rises to 258 years for $^{135}$Cs and 375 years for the $^{237}$Np chain. In this case the transport on the solid material involved in erosion and bioturbation is responsible for the transfers - albeit with long timescales. Table 4-2 uses the example of the transfer from deep soil to top soil to illustrate how the three...
processes advective transport, solid material transport and diffusive transport determine
the transfer coefficient between the two soil compartments.

For $^{36}\text{Cl}$ with $k_d = 0$, transport does not take place on the solid material and only the ad-vective and diffusive fluxes are involved, with $\lambda_{\text{advective}}^{\text{advection}}$ being around 8 times greater
than $\lambda_{\text{diffusion}}^{\text{diffusion}}$. The magnitudes of the diffusive and advective transfer coefficients are
similar for the radionuclides with non-zero $k_d$s. However, the absolute magnitudes of
these transfers is much smaller because a large proportion of the radionuclides in the compartments are bound to the fixed solid material in the compartments (99.81% for $^{135}\text{Cs}$, 99.98% for the $^{237}\text{Np}$ chain), and the effect of the transfer in solution is greatly
diminished. In this case, the relative magnitude of the transfer of solid material
dominates the transfer and the rates for $^{135}\text{Cs}$ and the $^{237}\text{Np}$ chain are equal (to three
significant figures), so that for $^{135}\text{Cs}$ the solid material transfer proceeds at twice the rate
of the advective/diffusive transfers; for $^{237}\text{Np}$ chain members the ratio is 21.

The results are useful in indicating how the detailed representation of FEPs in TAME
expresses the different timescales in the biosphere. In radioactive waste disposal
assessments, the timescales are long because the release to the biosphere from the
geosphere can continue at or near peak release rates over long periods. Likewise, the
half-lives of the radionuclides of interest are long. These results clearly indicate that,
over long periods, the transport of radionuclides on solid materials is an important
process for evaluating the radiological exposures arising from the build-up and retention
of radionuclides in the biosphere. Only for assessments over much shorter times and for
contaminants with relatively short residence times (typically a few hundred years) is it
possible to neglect the effects of transport on solid material.
Table 4-1: Transport half-times in the TAME example calculations. These are derived from the transfer coefficients using Equation (7) and show a broad range of values from $10^{-5}$ y to $2.5 \times 10^6$ y. (The $\infty$ in the values for $^{36}$Cl corresponds to zero transfer of the radionuclide between these compartments, i.e. $\lambda_{ij} = 0$). The values for the members of the $^{237}$Np chain are equal because the $k_{dS}$ of the elements (see Table 3-6) are the same and all other factors are non-nuclide-specific. The shaded regions denote transfers which are not represented in this generic TAME representation.

<table>
<thead>
<tr>
<th>From \ to</th>
<th>$^{36}$Cl</th>
<th>Local</th>
<th>Deep soil</th>
<th>Top soil</th>
<th>Water</th>
<th>Sediment</th>
<th>Elsewhere</th>
</tr>
</thead>
<tbody>
<tr>
<td>Local</td>
<td>-</td>
<td>2.1 x $10^1$</td>
<td>1.4 x $10^2$</td>
<td>3.3</td>
<td>2.3 x $10^2$</td>
<td>5.5 x $10^4$</td>
<td></td>
</tr>
<tr>
<td>Deep soil</td>
<td>1.1 x $10^1$</td>
<td>-</td>
<td>1.4</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Top soil</td>
<td>3.7 x $10^2$</td>
<td>-</td>
<td>-</td>
<td>$\infty$</td>
<td>$\infty$</td>
<td>$\infty$</td>
<td></td>
</tr>
<tr>
<td>Water</td>
<td>2.3 x $10^2$</td>
<td>-</td>
<td>-</td>
<td>$\infty$</td>
<td>1.2 x $10^5$</td>
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<tr>
<td>Sediment</td>
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<table>
<thead>
<tr>
<th>From \ to</th>
<th>$^{135}$Cs</th>
<th>Local</th>
<th>Deep soil</th>
<th>Top soil</th>
<th>Water</th>
<th>Sediment</th>
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</thead>
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<td>3.4 x $10^3$</td>
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</tr>
<tr>
<td>Deep soil</td>
<td>5.9 x $10^1$</td>
<td>-</td>
<td>2.6 x $10^2$</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Top soil</td>
<td>1.7 x $10^1$</td>
<td>-</td>
<td>2.8 x $10^3$</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Water</td>
<td>2.3 x $10^2$</td>
<td>-</td>
<td>-</td>
<td>2.7 x $10^3$</td>
<td>1.2 x $10^5$</td>
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</tr>
<tr>
<td>Sediment</td>
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<td>9.2</td>
<td>7.0 x $10^{-1}$</td>
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<table>
<thead>
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<th>From \ to</th>
<th>$^{237}$Np</th>
<th>Local</th>
<th>Deep soil</th>
<th>Top soil</th>
<th>Water</th>
<th>Sediment</th>
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<td>1.5 x $10^5$</td>
<td>3.5 x $10^4$</td>
<td>2.5 x $10^6$</td>
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<tr>
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<td>3.7 x $10^2$</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Top soil</td>
<td>7.8 x $10^1$</td>
<td>-</td>
<td>2.8 x $10^3$</td>
<td>-</td>
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<td>-</td>
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</tr>
<tr>
<td>Water</td>
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<td>-</td>
<td>4.8 x $10^4$</td>
<td>1.2 x $10^5$</td>
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<td></td>
</tr>
<tr>
<td>Sediment</td>
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<td>9.2</td>
<td>6.9 x $10^{-1}$</td>
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<table>
<thead>
<tr>
<th>From \ to</th>
<th>$^{233}$U</th>
<th>Local</th>
<th>Deep soil</th>
<th>Top soil</th>
<th>Water</th>
<th>Sediment</th>
<th>Elsewhere</th>
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<td>1.5 x $10^5$</td>
<td>3.5 x $10^4$</td>
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<tr>
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<td>3.7 x $10^2$</td>
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<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
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<td>7.8 x $10^1$</td>
<td>-</td>
<td>2.8 x $10^3$</td>
<td>-</td>
<td>-</td>
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<td></td>
</tr>
<tr>
<td>Top soil</td>
<td>2.3 x $10^2$</td>
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<td>-</td>
<td>4.8 x $10^4$</td>
<td>1.2 x $10^5$</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Water</td>
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<td>9.2</td>
<td>6.9 x $10^{-1}$</td>
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<td>-</td>
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</tbody>
</table>
Table 4-2: FEPs contributing to the transfer of radionuclides from the deep soil to the top soil. The model data for the biosphere calculations are used with the Equations in Section 2.3.2 to give the TAME transfer coefficients. To three significant figures, the transfer of $^{135}$Cs and the $^{237}$Np chain on solid material proceed at the same rate. The major influence on the total transfer coefficient between the deep and top soils is then the reduced advective transfer in the case of the more highly sorbing $^{237}$Np chain members.

<table>
<thead>
<tr>
<th>$k_d$</th>
<th>$^{36}$Cl</th>
<th>$^{135}$Cs</th>
<th>$^{237}$Np chain</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(Bq kg$^{-1}$) (Bq m$^{-3}$)$^{-1}$</td>
<td>0</td>
<td>0.1</td>
</tr>
<tr>
<td>$\lambda_{DT}^{(\text{water fluxes})}$</td>
<td>$y^{-1}$</td>
<td>$4.44 \times 10^{-1}$</td>
<td>$8.37 \times 10^{4}$</td>
</tr>
<tr>
<td>$\lambda_{DT}^{(\text{solid material})}$</td>
<td>$y^{-1}$</td>
<td>0</td>
<td>$1.76 \times 10^{3}$</td>
</tr>
<tr>
<td>$\lambda_{DT}^{(\text{diffusion})}$</td>
<td>$y^{-1}$</td>
<td>$5.20 \times 10^{-2}$</td>
<td>$9.79 \times 10^{5}$</td>
</tr>
<tr>
<td>$\lambda_{DT} = \sum_{FEP,k} \lambda_{DT}^{(k)}$</td>
<td>$y^{-1}$</td>
<td>$4.96 \times 10^{-1}$</td>
<td>$2.69 \times 10^{3}$</td>
</tr>
</tbody>
</table>
4.3 Compartmental inventories

Equation (1) has been solved for the set of transfer coefficients discussed in Section 4.2 and a top-hat function release of $10^6$ Bq yr$^{-1}$ for $10^4$ years of $^{36}$Cl, $^{135}$Cs and $^{237}$Np. The results for the compartment inventories and compartment total concentration$^2$ of the radionuclides in the biosphere are plotted as a function of time in Figure 4-1, Figure 4-2 and Figures 4-4 to 4-6. The two sets of plots show that similar inventories can give rise to very different concentrations and vice versa since the concentrations give information about the storage capacity of the compartments. These concentrations are related to the concentrations in water used for drinking purposes (see Appendices A and E), but are not used directly to calculate doses. They are used here to illustrate features of the TAME transport model.

In developing TAME, emphasis has been placed on the processes influencing radionuclide transport, particularly transport in water fluxes, solid material fluxes and by diffusion. The representation of $^{36}$Cl here as non-sorbing is useful in that it allows the model's response to be seen for radionuclides which are only transported with the water fluxes (i.e. the solid material transport plays no rôle for this radionuclide). This can then be contrasted with the other radionuclides in this example, which have increasingly higher sorption on solid material$^3$.

In the case of $^{36}$Cl (Figure 4-1), the response of the model to the input top-hat function of release flux is very close to a top-hat function. There is a very sharp drop in the compartment concentrations at the end of the source term, but there is a slight lag ($\leq 3$ years) before the compartment contents come into equilibrium (defined as the time taken from the start of the release until steady-state is reached, for which $\frac{dN_i}{dt} = 0 \ \forall \ i$). $^{36}$Cl has zero $k_d$ in all compartments and this behaviour is typical of the model's response in such cases. Most of the released radionuclide is lost rapidly downstream. The aquifer has the highest concentration because it is the receiving compartment and also because it has only a limited interaction with the other compartments, compared to the downstream flow of water in it, i.e. there is no retention in the modelled biosphere, although there is some recycling. The top soil and deep soil have similar concentrations,

$^2$ The inventories arise from the direct integration of Equation (1) and correspond to the total quantity of radioactivity in the physical volume of the compartment. The total concentrations in the compartments are given by the quotient of total content and total physical volume (see Appendix E, Equation (E.2)) - an alternative would be the porewater concentration, which involves partitioning between the liquid and solid material in the compartment.

$^3$ The half-lives of all the radionuclides having external sources in this biosphere representation ($^{129}$I and $^{237}$Np) are more than ten times longer than the duration of the source (see Table 3-7). The effect of radioactive decay is therefore negligible in this analysis and very little of the $^{237}$Np daughters ($^{233}$U and $^{229}$Th) appear in the biosphere. In a real situation, it would be impossible for there to be a release of $^{237}$Np without the release of some of the daughters, because the transit time through the geosphere and the containment time in the repository are extremely unlikely to be insignificant - a repository with such behaviour would not be built in such a host rock! As with the rest of this example, this release function is illustrative of the response of the biosphere model to the stimulus of different radionuclides with different properties and behaviours.
although leaching of this very mobile element from the top soil means that the top soil concentration is always lower than that in the deep soil.

The results for the inventory of $^{36}$Cl in the sediment compartment are unrealistically high, indicating a problem with the generic representations used here. The origins of this result are found in the modelling assumptions for the river bed sediments (see Sections 3.2.2 and 3.2.3), where it is assumed that

1. the primary interaction of the radionuclide content of the aquatic bed sediment with the other parts of the biosphere would be by solid material transport, so that transport in water fluxes could be neglected,

2. the bed sediment turnover in a fast moving river would be rapid (N.B. the value used here is once per year), and

3. diffusion should be modelled between contiguous water-filled compartments.

The last assumption means that the bed sediment compartment is, in fact, not isolated from the porewater content of the aquifer, and that the porosity in the sediment is effectively and unphysically modelled as being filled with stagnant water into which activity in the aquifer porewater can diffuse. However, this is not the full story and in these circumstances the porewater concentrations in the local aquifer and the sediment (defined in the case of $k_d = 0$ by $N_i/\left(V_i^n\right)$) should be equal. That they are not indicates that the representation of diffusion (Equation (10)) is incorrect. This rather serendipitous situation where the bed sediment porewater is isolated from the rest of the system suggests a test for an improved representation of the diffusion process. This is discussed in Appendix C and leads to a revised form of the diffusive transfer coefficient. Results using this new form are compared to results obtained from a numerical solution of the advection-dispersion equation.

This situation provides a very important lesson for biosphere modelling. In this example, a great deal of care has been exercised in the setting up of mass balance schemes for water and solid material fluxes in the soil system. Due to less precise knowledge, the attention to detail in the modelling of the aquatic processes was not as great and the simplistic assumptions listed above were made on the expectation that they would be of little consequence. While this is true for the overall dose arising from this example (see Section 4.6), this analysis of the intermediate points of the calculation indicates that more attention to detail might be required for future modelling of the aquatic system. More generally, it is difficult to decide upon the appropriate level of detail to be included in the modelling of biosphere sub-systems if no detailed assessment of the sensitivity of model results has been carried out. As indicated in Figure 1-1, the design of suitable, site-specific models is an iterative process. Section 4.6 looks at the effects of the modelling assumptions used in these example calculations on maximum annual individual dose.

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4 The new format will be implemented from TAME 3b, in the Autumn of 1996. See Appendix H.
Figure 4-1: Results of the TAME transport calculations for $^{36}$Cl. For this radio-nuclide, with zero $k_d$ in all compartments, there is no retention in the biosphere and all the released activity is lost to the downstream compartment. N.B. the falling-off of the inventories before the end of the source is an artefact of the solving routines. These have subsequently been replaced and the situation corrected - see Appendix H.
Results of the TAME transport calculations for $^{137}$Cs. For this moderately strongly sorbing nuclide, 8% of the total released amount is retained in the modelled biosphere at the end of the release period. 92% is lost downstream.
$^{135}\text{Cs}$ (Figure 4-2) behaves in a noticeably different way to $^{36}\text{Cl}$. There is a gradual build-up in each of the compartments until plateaux are reached at around $5\times10^3$ years. Quite rapid decay of the compartment contents follows at the end of the source term, although the rate of loss is less steep than in the case of $^{36}\text{Cl}$. This behaviour can be understood in terms of the $k_d$s in the compartments where retention and transport on solid materials dominate the transport processes.

Figure 4-3 shows the ratios of the inventories of $^{135}\text{Cs}$ to $^{36}\text{Cl}$ as a function of time. This illustrates the way in which compartment $k_d$ influences the response of the model. At the early times after the start of release, the ratio $N(^{135}\text{Cs})/N(^{36}\text{Cl})$ is much less than unity (except in the local aquifer which is the receptor compartment). This is because the $k_d$ in the local aquifer is acting to retard the transport of $^{135}\text{Cs}$ compared to non-sorbing $^{36}\text{Cl}$. At later times, the $^{135}\text{Cs}$ that has been transferred into the other compartments is retained there and $^{135}\text{Cs}$ accumulates in large quantities compared to $^{36}\text{Cl}$. In the local aquifer, which receives the external source term, the ratio of Cs to Cl is always higher because of the immediate retention in the aquifer. In the water the ratio tends to unity as the retarded $^{135}\text{Cs}$ eventually enters the water compartment. Radionuclides in the river water move with the flow velocity of the water irrespective of whether they are in solution or are sorbed onto the suspended solids in the water column and hence the water and suspended solid $k_d$s do not play a role in determining the ratios in the water.

In Figure 4-2, the curves for the aquifer, surface water and sediment are all approximately parallel for $^{135}\text{Cs}$. The rate of increase for the soil contents is much higher and this arises as a result of the bioturbative mixing between the soil layers (mediated by solid material transport) and the retention of the $^{135}\text{Cs}$ in the soils. Initially the concentration in the top soil is higher as it is retained and recirculated into the deep soil. After around 100 y, the effect of erosion begins to dominate and the amount in the top soil falls below that in the deep soil.

In comparison to $^{36}\text{Cl}$, high bed sediment concentrations are not seen in this case because of the non-zero $k_d$ for $^{135}\text{Cs}$ in that compartment. In this case the assumption that movement of solid material would dominate transport from the bed sediment is justified. Activity sorbed onto the bed sediment is transferred to the top soil (half-time is 9.4 years in Table 4-1), as well as to the water (0.7 years). Here the interaction with the aquifer (again only by diffusion) takes $1.4\times10^5$ years (cf. $5.3\times10^{-1}$ years for $^{36}\text{Cl}$).

The results for the $^{237}\text{Np}$ chain (Figures 4-4, 4-6 and 4-7) confirm the trends seen in going from $^{36}\text{Cl}$ to $^{135}\text{Cs}$. Higher $k_d$s mean greater retention in the compartments in the approach to steady-state. In the case of $^{237}\text{Np}$, $k_d$s a factor of ten higher than for $^{135}\text{Cs}$ mean that equilibrium is not, in fact, reached before the end of the source term at $10^4$ years, and also that the loss from the biosphere system at the end of the source term is more gradual than for lower $k_d$ radionuclides. Retention in the aquifer means that more than half of the $^{237}\text{Np}$ released remains in the aquifer, with much smaller fractions in the other compartments. Nevertheless, the concentration in the soils is within a factor of ten of the aquifer concentration and the concentrations in the deep and top soils are more nearly similar than in the case of $^{135}\text{Cs}$. Otherwise, the analysis of the inventories of $^{135}\text{Cs}$ in the soils and the aquatic compartments is also valid for $^{237}\text{Np}$. 
Figure 4-3: Illustration of the difference between retention (accumulation) and retardation (inhibited transport). The plot shows the ratios of the compartment inventories of $^{135}$Cs to $^{36}$Cl as a function of time. To the lower-left, the compartmental $k_d$s retard the $^{135}$Cs, whereas the non-sorbing $^{36}$Cl moves more rapidly. To the upper-right, the effects of the accumulation of the $^{135}$Cs as a result of retention are seen. The upturn between $8 \times 10^3$ years and $10^4$ years is caused by a problem the integration routines have in dealing with sharp edges. This has now been corrected - see Appendix H.
Results of the TAME transport calculations for $^{237}$Np. For this strongly sorbing nuclide, 60% of the total released amount is retained in the modelled biosphere at the end of the release period, with only 40% lost downstream.
Results of the TAME transport calculations for $^{233}$U. There is no external source of this radionuclide to the modelled biosphere; it grows in as a result of the decay of its parent, $^{237}$Np, which is released.
Figure 4-6: Results of the TAME transport calculations for $^{229}$Th. There is no external source of this radionuclide to the modelled biosphere; it grows in as a result of the decay of its parent, $^{233}$U, which in turn has grown in as a result of the decay of $^{237}$Np, which is released.
Because all the members of the $^{237}$Np chain have the same compartmental $k_d$s, the results for $^{233}$U and $^{229}$Th serve to illustrate how ingrowth of decay products is represented in TAME. The relatively high $k_d$s in the soils ($1 \text{ (Bq kg}^{-1})(\text{Bq m}^{-3} \text{ r}^{-1})$ again lead to similar concentrations in the two soil compartments. The decay of the large quantities of $^{237}$Np in the aquifer means that the largest inventories and the highest concentrations of the chain daughters are also found in the local aquifer. Figure 4-7 illustrates the approach to secular equilibrium as a function of time.

For illustrative purposes, these calculations have assumed that the daughters are not present in the source term and so they start off far from secular equilibrium. The plots show that, up until the end of the $^{237}$Np input at $10^4$ years, the source term prevents the $^{233}$U reaching equilibrium with its parent, with the ratio of the aquifer inventories being lower than for the soils because the aquifer is the receptor (and similarly for the water and the sediments because their inventories are parallel to the aquifer content). At the end of the source term, the biosphere compartments move more quickly into secular equilibrium. However, even at the end of the calculations carried out here, the $^{233}$U had not fully reached secular equilibrium with $^{237}$Np. Similar comments apply to the ratio of $^{229}$Th to $^{233}$U. The effect of the source term is less obvious here because it is only felt indirectly through the $^{233}$U inventory. These two radionuclides do eventually come into secular equilibrium at around $7\times10^4$ years, or $6\times10^4$ years after the end of the source.

In these example calculations, chosen to include a representation of the typical Swiss biosphere with relevant radionuclides and release of the radionuclides to a part of the biosphere relevant to performance assessment applications, a number of important conclusions can be drawn.

The role of $k_d$ both in terms of retardation and retention is highlighted by the examples presented here. Storage in the deeper regions of the terrestrial compartments in TAME illustrates the potential for subsequent erosion to bring contaminants to the near-surface environment (although in this example the erosion of the top soil is deliberately balanced by assumptions about the management of the river system and the transfer of radionuclides to the top soil via irrigation is correspondingly more important). In this case, with the local aquifer being the TAME receptor compartment, significant quantities of the strongly and moderately sorbing radionuclides remain in the aquifer. Also of importance is the result that, for low to moderately sorbing radionuclides, most of the radioactivity released is subsequently lost downstream. Only the most highly sorbing radionuclides remain predominantly in the model region.

In the kinds of assessment for which TAME is intended - i.e. release durations from hundreds up to millions of years - the time dependence of the release to the biosphere has a very strong influence on the distribution of radionuclides in the biosphere. This is particularly so for low $k_d$ radionuclides (e.g. $^{36}$Cl). For more strongly sorbing radionuclides, the time required to reach steady-state in the biosphere can be long and the removal of contaminants retained in the system can also take place on a long timescale. In many cases not requiring combinations of extreme parameter variations, the time to steady-state can be significantly longer than the timescales on which major changes to the biosphere conditions could be expected (typically from a few hundred years for the effects of global warming to $10^4$ years before the return of glacial conditions).
Figure 4-7: Illustration of the approach to secular equilibrium in the biosphere. Ratios of the compartmental inventories of the daughter nuclide to the immediate precursor in the $^{237}$Np decay chain are shown. N.B. there is very little difference in the ratios between the different compartments, making the individual plots difficult to resolve.
This example therefore illustrates the importance of the dynamic modelling of the compartment inventories. The time taken to reach steady-state is strongly affected by $k_d$, half-life and source term, so that an approach which assumed instantaneous equilibrium in the biosphere (i.e. solving Equation (1) for $\frac{dN_i}{dt} = 0$) might lead to unnecessary over-estimations of the radionuclide concentrations in the biosphere. Such an approach is more defensible for long release timescales and for long physical decay half-lives. However, the problem of the validity of assuming constant biosphere conditions in such circumstances remains. In any case, it is necessary to adequately derive the transfer matrix in Equation (1), i.e. the transfer coefficients $\lambda_{ij}$; irrespective of whether the steady-state or the full time-dependent solution is used, the derivation of the $\lambda_{ij}$ requires a detailed and comprehensive understanding of the features, events and processes operating in the biosphere. (This observation is particularly relevant given the relatively simple way in which the interactions of the aquatic sediment with the other compartments are modelled in the example calculations - see Appendix D.)

It is a misapprehension to believe that biosphere modelling can be simplified by taking the steady-state solution - the resources required to perform the integration of Equation (1) are trivial on modern computers, especially compared with the effort required to adequately characterise the biosphere processes. This effort is required anyway even if steady-state calculations are to be performed. In assessments where the biosphere conditions are assumed to be constant during the release from the geosphere, it is also useful to estimate the time to steady-state in the modelled biosphere.

In a similar way, it can be seen from this example that radionuclide chains should be modelled carefully, particularly when the chain members have significantly different properties. The assumption of secular equilibrium for the decay chain members is not justified unless the source terms are also in equilibrium. Where details of the release of decay chains are available, it is better to release the daughters into the model and let the model perform the calculations, even when the $k_d$s and other transfer factors are equal, because the dose per unit intake of each of the radionuclides may not be equal.

The example calculations here have focused on radionuclides with relatively long half-lives. In principle, the same calculations could be carried out for more rapidly decaying radionuclides (for example, $^{137}$Cs). The selection of a suitable source term then becomes important. The relatively long-duration top-hat function used here is not applicable because releases of short half-life nuclides from the geosphere would certainly not have such forms and, although steady-state would be reached earlier than for comparable radionuclides with long half-lives, the shape of the source term would then have an increasingly important influence on the biosphere model response as the half-life becomes shorter. For the reasons mentioned above, the full time-dependent biosphere calculations are to be preferred.

It could be argued that, as significant computing power is available and cheap, detailed two- or perhaps three-dimensional representations of the biosphere are possible, full time-dependence and evolution of biosphere should be included in assessment calculations. However, the framework on which TAME is based (originating from the Swiss regulatory guidelines for radiological protection) and the uncertainty in TAME-like databases caution against such a power-hungry approach. More effort should be
applied to understanding sources and consequences of model and data uncertainty before more complex models are considered. Indeed, it is likely that more detailed models can never be justified since realistic simulations of environmental conditions and human behaviour will be never be wholly convincing and because the radiological protection criteria do not warrant such an increase in complexity.

4.4 Concentrations of radionuclides in foodstuffs

In calculating the annual individual doses arising from the distribution of radionuclides in the environment, TAME evaluates the concentration of radionuclides in various foodstuffs. Figures 4-8 to 4-10 show how these vary in time for $^{36}$Cl, $^{135}$Cs and $^{237}$Np.

As with the results for the radionuclide inventories in the compartments, the purpose of these calculations is to illustrate the behaviour of the model, not to discuss the characteristics of the particular radionuclides or the quality of the database used. However, one feature of these results that is immediately apparent is that, for the radionuclides released to the system ($^{36}$Cl, $^{135}$Cs, $^{237}$Np), the range of peak concentrations in food is very similar. These are shown in Table 4-3.

The dynamic ranges of the foodstuff concentrations (over radionuclides) are shown and these are compared to the dynamic ranges of the concentration, distribution and transfer factors given in Table 3-8. For the concentrations in crops and pasture, the dynamic ranges are very different. This indicates that a non-nuclide-specific process is responsible for accumulation in the plants and that the soil-crop transfer factors play a very limited role in determining the concentrations in the vegetable pathways here.

The results for meat, milk and eggs are, however, all very close to the values expected if the distribution factors are compared. This indicates that a single process is responsible for accumulation in these food types. The reason is not difficult to see - because all the crop concentrations are within an order of magnitude, the main process involved in accumulating the radionuclide in these foods is the distribution in the animal tissue, using the distribution factors in Table 3-8. For the concentration in fish, the agreement is again not exact; this is surprising, at first, since the river water concentrations are similar and the calculation of the fish concentration is one of the simplest (see Appendix E, Equation E.6).
Foodstuff concentrations:

- **meat** (Bq kg⁻¹)  
- **fish** (Bq kg⁻¹)  
- **pasture** (Bq kg⁻¹)  
- **root vegetables** (Bq kg⁻¹)

- **milk** (Bq kg⁻¹)  
- **eggs** (Bq egg⁻¹)  
- **grain** (Bq kg⁻¹)  
- **green vegetables** (Bq kg⁻¹)

**Figure 4-8:** Foodstuff concentrations derived from the compartmental inventories as a function of time for ³⁶Cl.

Foodstuff concentrations:

- **meat** (Bq kg⁻¹)  
- **fish** (Bq kg⁻¹)  
- **pasture** (Bq kg⁻¹)  
- **root vegetables** (Bq kg⁻¹)

- **milk** (Bq kg⁻¹)  
- **eggs** (Bq egg⁻¹)  
- **grain** (Bq kg⁻¹)  
- **green vegetables** (Bq kg⁻¹)

**Figure 4-9:** Foodstuff concentrations derived from the compartmental inventories as a function of time for ¹³⁵Cs.
Figure 4-10: Foodstuff concentrations derived from the compartmental inventories as a function of time for $^{237}$Np.

Table 4-3: Maximum foodstuff concentrations for the radionuclides released into the example biosphere, illustrating the dynamic range of the calculated values (results) compared to the dynamic range of the concentration, distribution and transfer factors in the various foodstuffs (the input data in Table 3-8). The foodstuff with the maximum concentration for each of the radionuclides is indicated in bold type. Dynamic range is defined by

\[
dyn.\text{range} = \log \left( \frac{\text{maximum of the quantity for the nuclide}}{\text{minimum of the quantity for the nuclide}} \right)
\]

<table>
<thead>
<tr>
<th>foodstuff concentration</th>
<th>units</th>
<th>$^{36}$Cl</th>
<th>$^{137}$Cs</th>
<th>$^{237}$Np</th>
<th>dynamic range of food conc.</th>
<th>dynamic range of factors in Table 3-8</th>
</tr>
</thead>
<tbody>
<tr>
<td>green vegetables</td>
<td>Bq kg$^{-1}$</td>
<td>1.2×10$^{-4}$</td>
<td>1.4×10$^{-4}$</td>
<td>1.1×10$^{-3}$</td>
<td>0.1</td>
<td>3.0</td>
</tr>
<tr>
<td>pasture</td>
<td>Bq kg$^{-1}$</td>
<td>3.7×10$^{-4}$</td>
<td>3.6×10$^{-4}$</td>
<td>2.9×10$^{-3}$</td>
<td>0.1</td>
<td>3.7</td>
</tr>
<tr>
<td>grain</td>
<td>Bq kg$^{-1}$</td>
<td>6.9×10$^{-4}$</td>
<td>1.5×10$^{-3}$</td>
<td>4.7×10$^{-3}$</td>
<td>0.5</td>
<td>3.5</td>
</tr>
<tr>
<td>root vegetables</td>
<td>Bq kg$^{-1}$</td>
<td>4.2×10$^{-4}$</td>
<td>4.0×10$^{-4}$</td>
<td>1.8×10$^{-3}$</td>
<td>0.7</td>
<td>3.0</td>
</tr>
<tr>
<td>fish</td>
<td>Bq kg$^{-1}$</td>
<td>4.9×10$^{-6}$</td>
<td>4.9×10$^{-4}$</td>
<td>3.7×10$^{-6}$</td>
<td>2.1</td>
<td>2.0</td>
</tr>
<tr>
<td>meat</td>
<td>Bq kg$^{-1}$</td>
<td>3.0×10$^{-3}$</td>
<td>9.9×10$^{-4}$</td>
<td>1.0×10$^{-3}$</td>
<td>2.5</td>
<td>2.6</td>
</tr>
<tr>
<td>eggs</td>
<td>Bq egg$^{-1}$</td>
<td>2.7×10$^{-4}$</td>
<td>6.4×10$^{-6}$</td>
<td>4.1×10$^{-8}$</td>
<td>3.8</td>
<td>3.4</td>
</tr>
<tr>
<td>milk</td>
<td>Bq kg$^{-1}$</td>
<td>1.9×10$^{-3}$</td>
<td>2.7×10$^{-4}$</td>
<td>2.6×10$^{-7}$</td>
<td>3.9</td>
<td>4.0</td>
</tr>
</tbody>
</table>
The explanation is that the activity in the consumed fish flesh is assumed to be in equilibrium with the activity in solution in the river water and does not include the activity on suspended solids. Hence, for $^{36}\text{Cl}$ with zero $k_d$ in the river water, the water-to-fish concentration factor explains the relative concentration in the $^{36}\text{Cl}$ and $^{135}\text{Cs}$. For $^{237}\text{Np}$, with a higher $k_d$ in the river water than $^{135}\text{Cs}$, (by a factor of 10), the effect of the concentration on suspended solids becomes apparent.

There are other points to note here - for example, the results for $^{237}\text{Np}$ (Figure 4-10) show that the distribution of concentrations can vary as a function of time, with different pathways having different gradients. The contributions of the different mechanisms are discussed in the following section for $^{237}\text{Np}$ in the context of the contributions to the dose from the different pathways.

The foodstuff concentrations of the $^{237}\text{Np}$ chain daughters are much lower than in the case of the parent and this is to be expected from Figures 4-4 to 4-6. Results for the daughters (not plotted here) indicate that the chain daughters behave similarly to each other in the food-chain, both being most highly concentrated in grain. It should, however, be remembered that it is difficult to make comparisons of radiological impact on the basis of concentrations. The use of dose as the end-point of the calculations makes comparisons possible since dose-per-unit intake weights the different radionuclides according to their health effects.

### 4.5 Annual individual dose

The end-point of the TAME calculations is annual individual dose over the eleven exposure pathways outlined in Appendix E. In terms of compliance with the regulatory guidelines, the model output in Table 4-4 summarises the maximum dose over all pathways (in this case $6.84 \times 10^{-7}$ Sv y$^{-1}$, arising at $10^4$ years - the end of release) and this is all that is required. Obviously, the model calculates more than just this, and the summary lists the contribution to this total dose by nuclide (where appropriate) and by pathway. For chains, the summary information is calculated for each of the chain members as well as for the sum over that chain, because the time of maximum dose from the individual chain members need not be the same as for the whole chain and because different pathways are important for different radionuclides. In this case, however, the total dose is dominated by $^{237}\text{Np}$ (99.8%), with the root vegetable pathway contributing 66% of the dose, 20% from drinking-water (well) and 11% from the grain consumption pathway.

The individual maxima for the five example radionuclides are shown in Table 4-5. Peak doses arise before $10^4$ years for $^{36}\text{Cl}$ and $^{135}\text{Cs}$, at $10^4$ years for $^{237}\text{Np}$ and after $10^4$ years for $^{233}\text{U}$ and $^{229}\text{Th}$, showing the importance of ingrowth of the $^{237}\text{Np}$ retained in the aquifer. Note also that the dose from $^{229}\text{Th}$ is greater than from $^{233}\text{U}$ and that here the dust inhalation pathway makes a significant contribution to the total dose.
Table 4-4:  \textit{TAME} radiological consequence summary at the time of maximum annual individual dose from the members of the $^{237}$Np chain in the example calculations. (Taken directly from the \textit{TAME} summary file).

<table>
<thead>
<tr>
<th>Doses calculated for:</th>
<th>np237</th>
</tr>
</thead>
<tbody>
<tr>
<td>Release Scenario:</td>
<td>test.4</td>
</tr>
<tr>
<td>Parent nuclide:</td>
<td>np237</td>
</tr>
<tr>
<td>Sun over:</td>
<td>3 chain members.</td>
</tr>
</tbody>
</table>

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Maximum annual individual dose: $6.840\times10^{-7}$ Sv/y
$6.840\times10^{-4}$ mSv/y

Arising at: 1.0000E+04 y

In river section: Root vegetables

Contribution to peak:

\begin{itemize}
\item np237 99.80%
\item u233 0.05%
\item th229 0.15%
\end{itemize}

12 Exposure pathways:

\begin{itemize}
\item Drinking-water (surface)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\item Drinking-water (well)
\end{itemize}

Table 4-5:  Results for maximum dose for the example radionuclides, showing magnitude, time of occurrence and the three most significant pathways.

<table>
<thead>
<tr>
<th>radionuclide</th>
<th>maximum dose [Sv y$^{-1}$]</th>
<th>time of maximum [y]</th>
<th>1st pathway</th>
<th>2nd pathway</th>
<th>3rd pathway</th>
</tr>
</thead>
<tbody>
<tr>
<td>$^{36}$Cl</td>
<td>$1.1\times10^{-9}$</td>
<td>20</td>
<td>Milk</td>
<td>46.7</td>
<td>Meat</td>
</tr>
<tr>
<td>$^{135}$Cs</td>
<td>$1.12\times10^{-9}$</td>
<td>$8\times10^3$</td>
<td>Grain</td>
<td>33.1</td>
<td>Root veg.</td>
</tr>
<tr>
<td>$^{237}$Np</td>
<td>$6.8\times10^{-7}$</td>
<td>$10^4$</td>
<td>Root veg.</td>
<td>66.0</td>
<td>Drinking (well)</td>
</tr>
<tr>
<td>$^{233}$U</td>
<td>$3.7\times10^{-10}$</td>
<td>$1.3\times10^4$</td>
<td>Drinking (well)</td>
<td>46.0</td>
<td>Grain</td>
</tr>
<tr>
<td>$^{226}$Th</td>
<td>$1.8\times10^{-9}$</td>
<td>$2\times10^4$</td>
<td>Drinking (well)</td>
<td>48.0</td>
<td>Grain</td>
</tr>
</tbody>
</table>
The time evolution of the doses for $^{36}$Cl (Figure 4-11) is, as expected from the results for the compartment inventories and the foodstuff concentrations, close to the form of a top-hat function. The ranking of the contributing pathways remains constant in time, with milk consumption giving rise to the largest dose, followed by meat and drinking-water from the aquifer. A comparison of Figures 4-13 to 4-18 shows how the consumption rate data influence the doses by pathway in relation to the foodstuff concentrations. The concentration of $^{36}$Cl in fish is relatively low and, because the consumption rate of fish (2 kg y$^{-1}$ in Table E.3) is also low, the result is an insignificant contribution to the total dose from this pathway. This again illustrates the use of dose as an end-point, allowing meaningful comparisons across pathways to be made.

Figure 4-12, for $^{135}$Cs, shows that all pathways are mutually parallel with the exception of the dose from dust inhalation. As with $^{36}$Cl, the ranking of pathways remains constant in time, in the order grain, drinking-water, root vegetables and meat. The other pathways make minor contributions (cf. Table 4-5). The non-parallel response of the doses from dust inhalation and grain consumption can be understood in terms of the processes contributing to each of the doses. This is investigated more fully below.

The plots for the $^{237}$Np chain members show similar relationships between the pathways. In the case of $^{237}$Np the effect is more striking than in the case of $^{135}$Cs in that both dust inhalation and external $\gamma$-irradiation have much steeper gradients than the other pathways (Figure 4-13) and several of the foodstuff pathways have a distinct change in gradient as a function of time, leading to a different ranking of the pathways as a function of time. In particular, for $^{237}$Np, the dose from drinking-water dominates at early times, with the contribution to the total dose from root vegetables second in importance. At the time of the peak dose, the root vegetables pathway dominates (see Table 4-4).

As can be seen more clearly in Figure 4-10, the concentration in green vegetables starts parallel to the concentration in pasture but the rate of increase picks up noticeably after around ten years, until (in Figure 4-13) the curve for root vegetables is more closely parallel to the doses from dust inhalation and external $\gamma$-irradiation. Similar effects are seen for grain, green vegetables and meat and also in the case of the chain daughters, but the effect is more pronounced for $^{237}$Np. The contributions to concentrations, the human exposures and the doses are given in Table 4-6 for $^{237}$Np at $t = 4$ years and $t = 10^4$ years; this shows how the different processes modelled in $TAME$ combine. The effects seen here are most significant for the sorbing radionuclides and the changes in pathway ranking with time are particularly informative for $^{237}$Np since the biosphere compartment inventories (and hence doses) do not reach equilibrium before the end of the source term at $10^4$ years.

---

5 It should be remembered that, in this biosphere, the parameter $f_{well}$ is set to 1 in Table 3-2, indicating that water for human consumption is obtained only from the well in the local aquifer.

6 N.B. because the source terms are artificial, it is not reasonable to make comparisons of the radiological impacts across the radionuclides in this example, but the use of dose does allow this in applications where true source terms are estimated. As a consequence of this, the results for the doses summed over all the chain members are virtually indistinguishable from the results for $^{237}$Np alone.
Figure 4-11: Annual individual dose as a function of time for $^{36}$Cl in the example biosphere.

Figure 4-12: Annual individual dose as a function of time for $^{137}$Cs in the example biosphere.
Figure 4-13: Annual individual dose as a function of time for $^{237}$Np in the example biosphere. Doses over the $^{237}$Np chain show the same response because of the low doses from the daughters.

Figure 4-14: Annual individual dose as a function of time for $^{233}$U in the example biosphere.
From this table the reason for the variation in the ranking of the pathways as a function of time (in Figure 4-13) is clearly seen. The dust inhalation pathway depends only on the concentration of radionuclides in the top soil and the gradient of the doses from this pathway is higher than the other pathways up to the end of the source. There is some levelling off but this is associated with the approach to steady-state in the compartment inventories (compare the more rapid equilibration of the soil-dominated pathways for $^{135}\text{Cs}$). Similarly, the drinking-water dose from the aquifer shows a constant gradient up to the end of the source because of the release of the $^{237}\text{Np}$ parent to the aquifer.

The behaviour of the root vegetables and meat consumption pathways can be explained in that, at early times, irrigation provides the quickest route from the aquifer to the crops. However, sorption in the soils dominates at later times and the gradient of the root vegetables curve, having started off parallel to the curve of the local aquifer inventory, tends to the gradient of the top soil inventory as the concentration in the soils increases (at times greater than 400 years) and the root uptake mechanism takes over. The effect is less pronounced for $^{135}\text{Cs}$ because of its overall lower $k_d$s and it is entirely absent for $^{36}\text{Cl}$ which has $k_d = 0$.

---

*Remembering that, in this example, irrigation is taken in equal quantities from the aquifer and from the surface water. Note also that, arising from the earlier Swiss biosphere databases, the soil contamination on the external surfaces of the root vegetables is assumed to be removed during food preparation in this example, $f_w = 0$.***
Table 4-6: Contributions to the pathway concentration, annual intake by exposed individuals and annual individual dose from $^{237}$Np for selected exposure pathways at two different times (to two significant figures). Maximum contributions from parallel pathways are shown in bold type.

<table>
<thead>
<tr>
<th>pathway</th>
<th>quantity</th>
<th>time</th>
<th>processes and $TAME$ compartments</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Σ (processes)</td>
</tr>
<tr>
<td>root vegetables</td>
<td>conc. Bq kg$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$5.2\times10^{-8}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>intake Bq y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$1.2\times10^{-5}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>annual dose Sv y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$1.3\times10^{-11}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>drinking-water</td>
<td>conc. Bq kg$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$1.4\times10^{-4}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>intake Bq y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$1.0\times10^{-4}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>annual dose Sv y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$1.1\times10^{-10}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>dust inhalation</td>
<td>conc. Bq kg$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$3.7\times10^{-14}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>intake Bq y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$3.1\times10^{-10}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>annual dose Sv y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$4.0\times10^{-14}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>meat</td>
<td>conc. Bq kg$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$3.3\times10^{-9}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>intake Bq y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$3.1\times10^{-7}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>annual dose Sv y$^{-1}$</td>
<td>4 y $10^4$ y</td>
<td>$3.4\times10^{-13}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: The table is organized to show the contributions to the pathway concentration, annual intake by exposed individuals, and annual individual dose from $^{237}$Np for different pathways at two different times, with maximum contributions from parallel pathways highlighted in bold.
In the meat pathway, the sorption of $^{237}$Np in soils is so important that the dominating mechanism for the dose from meat consumption in this example at later times is intake of soil particulates on pasture during grazing. This takes over from the irrigation mechanism (interception of spray irrigation is much more efficient for pasture than for root vegetables - compare the values for $\mu_r$ and $\mu_p$ in Table 3-3). The contribution of root uptake is less than a third of this.

Put another way - the dust inhalation pathway could come to dominate given the higher gradients associated with the doses from the accumulation in the soils. This would arise if the daughters of $^{237}$Np were released in significant quantities into the modelled biosphere. In this example, the dose from residence in the biosphere at normal, background dust levels gives the maximum contribution to the inhalation dose. Thus, the parameter describing inhalation of airborne dust at increased levels could come to dominate the radiation exposures in situations where accumulation takes place over long periods of time.

It should again be stressed that these results are not necessarily representative of real situations. As discussed in Section 4.3, the rôle played by the source term, particularly for the more strongly sorbing radionuclides, is very important. However, a point arising in this analysis which is observed elsewhere (NAGRA, 1994b) is that the drinking-water dose, even when taken from a well in the aquifer (which has the highest porewater concentration of all the TAME compartments by virtue of its being the compartment receiving the release from the geosphere), is not the highest dose when all the pathway doses are evaluated. KLOS & VAN DORP (1995) deal with this feature in greater detail in the context of the exploitation of a mineral water source compared with the same flux released to a biosphere modelled using the full TAME representation similar to that in the example given here.

4.6 Conceptual uncertainty

As discussed in Sections 2 and 3, some of the assumptions on which the conceptual model are based are treated in a very simplified form in the representation of the biosphere used in these example calculations. In particular these were

1. the representation of diffusion,

2. the assumption that the interactions of the aquatic bed sediment with the other parts of the system were dominated by the movement of solid material,

---

8 The effect is small in the example given here but, in practice, it can be more significant (KLOS & VAN DORP, 1995). The consumption of well water for the chain daughters $^{235}$U and $^{229}$Th does lead to the highest doses in this example. This is because of the source term assumed for $^{237}$Np. The daughters present in the system grow in from $^{237}$Np and they are predominantly in the aquifer because the parent is strongly sorbed there. The timescale chosen for the source term in these calculations does not allow time for a significant quantity of the daughters to grow in in the system.
3. the turnover time from the bed sediment to the water column was once per year, and
4. the assumption of constant volumes in the soil and aquifer compartments required an
   upward transfer from each deeper compartment.

It should be stressed that these are uncertainties in the conceptual model of the example
biosphere and not parameter uncertainties\(^9\). They arise because of simplifications of the
modelled system, or as a consequence of the modelling approach adopted.

Some idea of the influence of these assumptions on the end-point of the calculations can
be gained by altering the representations and/or parameter values relative to the
reference case and carrying out calculations accordingly. The results for the maximum
dose over all pathways and the time of occurrence of the maximum are shown in Table
4-7.

Turning off diffusion altogether in all compartments has a very small effect on the
annual individual dose. Doses from \(^{36}\)Cl decrease by 0.05\% and from \(^{237}\)Np they are
lower by 0.1\%. The only effect on the time to maximum dose is that the doses from \(^{36}\)Cl
arise 4 years earlier. It can be concluded that, in this example, diffusion plays an
insignificant rôle in determining the maximum dose.

A revised mass balance scheme in which the exfiltrating water flux from the aquifer to
the river water in the reference case is assumed to first pass through the river bed
sediment produces much greater changes, although the greatest change in any of the
doses is only 4.25\% in the case of \(^{237}\)Np. This can be understood in terms of the greater
accumulation of \(^{237}\)Np in the bed sediment leading to larger quantities of activity being
transferred to the top soil. This alternative has less than 1\% effect on the dose from \(^{36}\)Cl
and \(^{135}\)Cs. Again only the time to maximum is affected for \(^{36}\)Cl. Given the logarithmic
timescale used in these calculations, the effect on the time to maximum dose is likely to
be less than a few years in all cases.

The other two alternatives concern solid material transport and therefore do not
influence \(^{36}\)Cl, with \(k_d = 0\) in all compartments. Nor is there any effect on the time to
maximum dose for any of the radionuclides. When the turnover of the bed sediment
with the river water is switched off (\(k_{sw} = 0\)), the major effect is on \(^{237}\)Np (with its
higher compartmental \(k_d\) values). The total dose increases by just over one percent. For
\(^{135}\)Cs, the change is much less. The reason for the change is the rapid loss of activity in
the water column. In this case more activity is available for transfer to the top soil rather
than for rapid loss from the model system.

The largest of all the changes is seen with the assumption of upward transport of solid
material to compensate for the removal of the surface by erosion. In the dataset used
here to investigate the effect, the mass fluxes

\(^9\) As has been pointed out before, the purpose of this report is to illustrate the conceptual model and the
mathematical basis of it - parametric uncertainty is not covered here, but the uncertainty in the
conceptual model is.
Table 4-7: Assessment of the conceptual uncertainty in the example calculations. Absolute values for dose over all pathways and time to maximum dose are given alongside the percentage change caused by the alternative representation relative to the reference case:

\[
\% \text{ change} = 100 \times \frac{v_{alt} - v_{ref}}{v_{alt} + v_{ref}}.
\]

Shaded entries indicate no change relative to the reference case.

### Maximum dose over all pathways [Sv y\(^{-1}\)]

<table>
<thead>
<tr>
<th></th>
<th>(^{36})Cl</th>
<th>(^{135})Cs</th>
<th>(^{237})Np</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Reference Case</strong></td>
<td>1.1×10(^9)</td>
<td>1.2×10(^9)</td>
<td>6.8×10(^{-7})</td>
</tr>
<tr>
<td>No diffusion: (D_0 = 0)</td>
<td>1.1×10(^9)</td>
<td>1.2×10(^9)</td>
<td>6.8×10(^{-7})</td>
</tr>
<tr>
<td>Aquifer exfiltration through bed sediment</td>
<td>1.1×10(^9)</td>
<td>1.2×10(^9)</td>
<td>7.4×10(^{-7})</td>
</tr>
<tr>
<td>No bed sediment interaction with water column: (\kappa_{sw} = 0)</td>
<td>1.1×10(^9)</td>
<td>1.2×10(^9)</td>
<td>7.0×10(^{-7})</td>
</tr>
<tr>
<td>No erosive transfer from deeper compartments</td>
<td>1.1×10(^9)</td>
<td>1.2×10(^9)</td>
<td>5.6×10(^{-7})</td>
</tr>
</tbody>
</table>

### Time to maximum dose [y]

<table>
<thead>
<tr>
<th></th>
<th>(^{36})Cl</th>
<th>(^{135})Cs</th>
<th>(^{237})Np</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Reference Case</strong></td>
<td>20</td>
<td>7.9×10(^3)</td>
<td>10(^4)</td>
</tr>
<tr>
<td>No diffusion: (D_0 = 0)</td>
<td>16</td>
<td>7.9×10(^3)</td>
<td>10(^4)</td>
</tr>
<tr>
<td>Aquifer exfiltration through bed sediment</td>
<td>16</td>
<td>7.9×10(^3)</td>
<td>10(^4)</td>
</tr>
<tr>
<td>No bed sediment interaction with water column: (\kappa_{sw} = 0)</td>
<td>20</td>
<td>7.9×10(^3)</td>
<td>10(^4)</td>
</tr>
<tr>
<td>No erosive transfer from deeper compartments</td>
<td>20</td>
<td>7.9×10(^3)</td>
<td>10(^4)</td>
</tr>
</tbody>
</table>
\[ M_{DP}^{(erosion)} = M_{LD}^{(erosion)} = m_e A_f \ [\text{kg y}^{-1}], \]  

are not included, so that these terms do not move contaminated solid material up from the aquifer through the soil profile. The consequence is that less activity is transported to the top soil and the total dose thus decreases. The effect is insignificant for \(^{135}\text{Cs}\), but is around 10\% for \(^{237}\text{Np}\). As modelled, the inclusion of this process is conservative. In contrast with these results, NEA (1993) found that this process did not have a strong influence on the maximum dose. It is apparent that the use of a detailed mass balance approach in TAME is the reason why the effect is seen here, when it was not for less detailed, earlier models.

This brief investigation of the uncertainty associated with the conceptual formulation of the generic TAME representation and with these example calculations justifies the assumptions made - the results are robust with respect to maximum dose and time of occurrence.

It may seem that diffusion is unnecessary in these calculations\(^{10}\) and this cannot be disputed, but without further investigation of a broader spectrum of model situations it would not be wise to remove it entirely. The simplification of the interaction of the bed sediment water content with the rest of the water flux scheme is slightly non-conservative but the inclusion of a balance for erosion, introduced to make the assumption of constant transfer coefficients consistent, is conservative. This represents a largely ignored and potentially important part of the geosphere-biosphere interface, namely the reservoir of radionuclides sorbed onto solid material at depth below the modelled biosphere.

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\(^{10}\) In fact diffusion was not included in the processes modelled in the representations of the Wellenberg biospheres (NAGRA, 1994a) because the system was very similar to the case discussed here.
5. Requirements for further work

This report deals with the TAME conceptual model and shows how its mathematical representation has been developed. The example calculations show how a generic implementation works in practice.

The conceptual framework of the model, outlined in Section 2.2, is based on a thorough review of the features, events and processes relevant to the kinds of biospheres into which releases might be expected from underground repositories for radioactive waste in Switzerland, although there are clearly identified areas where the generic formulation should be improved.

Of key importance in the modelling is an understanding of the FEPs not included in the system; however, a clear identification of what has been ruled out of a given application is only possible on a site-specific basis. In the example calculations discussed here, which are only intended to be used to identify how the model behaves, it is not relevant to discuss what features of the modelled system have not been included.

The appropriate FEPs for this kind of modelling have recently been reviewed internationally - particularly in the BIOMOVS II working group on Reference Biospheres (BIOMOVS, 1996a). It is encouraging to note that no significant omissions from the TAME FEP list have been identified. This has been further illustrated, on a more practical level, by a comparison with the biosphere models participating in the BIOMOVS II Complementary Studies exercise (BIOMOVS II, 1996b).

The mathematical representation in the generic TAME transport sub-model has been derived from first principles in Appendices A, B and C. This is a major advantage for TAME since the basis for the implementations is clearly described. The exposure pathway sub-model has been adopted from an earlier well-established model. However, a fundamental review of this part of TAME is required.

The mathematical description of the FEPs in the transport and exposure pathway sub-models includes a large number of parameters and this places a heavy emphasis on data collection for specific applications. In the dataset used in the example calculations, use has been made of characteristics of a typical valley in Central Switzerland. Many of these data need only be revised if significant changes occur in the locations of the potential release from the geosphere, which would require a different approach for the geosphere-biosphere interface. It is important to note that the data collected in the field and in experiments normally cannot be applied directly to biosphere models. Varying degrees of preprocessing are necessary before they can be turned into a useable modelling database.

Of the parameters which are only weakly site-specific, the database for sorption has been recently revised (TITS et al., 1996). However most other non-site-specific data were compiled around 1985 and are in need of critical review, particularly the radionuclide specific parameters in the exposure pathway sub-models. A specific problem is that it is necessary to perform calculations for a broad spectrum of radionuclides; in many cases the available data are scarce and of questionable reliability.
Another consequence of the parameterisation given here is that not all the processes included in the model are important in all applications, e.g. diffusion in the situation described in the example calculations. Potentially some may not be important at all but at present it has not been demonstrated that they can be screened out generally. It must be remembered that the example calculations given here cover only a very limited part of the full TAME envelope.

Effort is therefore required on several fronts:

- **Classification of the model parameters as sensitive and non-sensitive** - i.e. those parameters having a significant effect on the results of the calculations and those which, although required for the accurate parameterisation of the model, do not have a significant rôle.

- **Identification of important and non-important FEPs.** The former must be retained in the model and, if necessary, their representation improved, while the latter could, potentially be deleted from the model with no loss of applicability. In the waste disposal context in Switzerland, and in many other countries, the regulations specify maximum dose (or risk expressed in an equivalent way). This suggests maximum dose over all pathways as the suitable end-point for calculating sensitivity to FEPs and parameter values. As discussed at the end of Section 4, some of the less firmly based modelling assumptions are shown to be robust in terms of the annual individual dose over all pathways.

- **Improvement in the representation of sensitive processes and better parameterisation of processes currently treated in an overly simplistic manner.** For example, the aquatic bed sediment processes are known to be simplistically represented in the example given here and, although the maximum dose is not sensitive to these assumptions, it could be argued that a better generic representation is desirable for internal consistency. It could be concluded that all parts of the model should be treated in similar detail to avoid problems with inconsistent assumptions however, this may not always be necessary. Where simplifying assumptions are used (as they must be in assessments models) some idea of the robustness of the assumptions should be given - this is analogous to the quantification of parametric uncertainty.

- **The problem identified with the generic representation of diffusive transfers is of a different kind.** As discussed in Appendix C, there was a problem with the mathematical representation of a FEP known to be active and to be potentially important for doses arising at early times for some classes of radionuclide in some biosphere systems. The generic representation of the diffusion process must be modified and implemented as shown in Appendix C.

- **The mechanisms by which radionuclides can be transported to the top soil, where they can be incorporated in crops and in livestock, are very important.** This means the parameterisation of the irrigation process and the treatment of movements in the groundwater table, as well as the implications for the representation of mass balance in the modelled system. Some of these details are already under discussion in the Complementary Studies exercise. Others are likely to emerge as detailed intercomparisons of the participating models are carried out and as feedback from the Reference Biospheres Working Group is included.
Verication of subsistence agriculture as the most suitable representation of the critical group is also required. This may involve a review of the non-agricultural pathways.

Improved and expanded generic databases are required. Site-specific data should be used when available and where relevant. However, it should be recognised that site characteristics of the present day may have limited applicability into the farther future - representative biosphere states based on suitable present day analogues should be considered.

In short, there are many potential improvements to the way in which contemporary biosphere models are implemented. An established international consensus is the best way to proceed.

In carrying out this review, there is a considerable role for parameteric sensitivity analyses using probabilistic analysis techniques. The existing parameterisation of TAME makes this an attractive and practical option for work and the potential for new parameterisation and revised parameterisation means that the work is likely to be iterative, though not indefinite.
6. Summary and Conclusions

*TAME*, the *Terrestrial - Aquatic Model of the Environment*, is a general purpose model of biosphere transport, accumulation and exposure pathways for use in radioactive waste disposal assessments. In developing *TAME* the aim has been to include those features and processes which determine the radiological consequences to humans and to assure that these will not be underestimated.

It should be remembered that no biosphere model can ever provide a *prediction* of the doses arising in the far future - there is too much uncertainty in the evolution of the environment and human behaviour. In the context of safety assessment of waste disposal options, however, the results provide an indication of the potential radiological impact of release to the biosphere. The end-product can then be compared with regulatory guidelines. Additionally, the use of dose as an end-point provides a useful yardstick for comparing the consequences of release of different radionuclides.

The conceptual framework of the model, outlined in Section 2.2, is based on a thorough review of the features, events and processes relevant to the kinds of biospheres into which releases from underground repositories for radioactive waste might be expected. Furthermore, a comparison with biosphere models for the assessment of radioactive waste disposal in the Working Groups *Reference Biospheres* and *Complementary Studies* of the international study BIOMOVS II shows that *TAME* includes all relevant FEPs (BIOMOVS II, 1996a; BIOMOVS II, 1996b). This is an important step in building confidence in the model.

The transition from the *TAME* conceptual model to the mathematical model includes a detailed representation of all the FEPs in the conceptual model and, in particular, attention is paid to questions of mass balance in the modelled system. This too is important in building confidence. The model includes processes which may appear insignificant (e.g. some of the solid material fluxes in Table 3-5), but they must be included in the model for consistency. In other modelling scenarios, it is not possible to say that these processes will invariably have a small influence on the consequences being evaluated. In addition to the overall transparency that this imparts to the model it also allows great flexibility. In summary, the *TAME* represents the state of the art of biosphere modelling for radioactive waste disposal.

The *TAME* conceptual model leads to a mathematical description (Section 2.3) which comprises two parts:

- biosphere transport in the environment leading to the distribution of radionuclides in the near-surface environment as a function of time,
- uptake and accumulation in the food-chain, and calculation of the corresponding doses to individual inhabitants in the modelled region.

The first part evaluates the contents of the biosphere as a function of time by solving a first order, ordinary differential equation which describes the fractional fluxes of contaminants between the elements of the biosphere (represented as compartments). The transfer coefficients in this transport equation are based on detailed consideration of the
water and solid material fluxes in the biosphere; these, in turn, are derived from a
detailed consideration of the site-specific properties in the modelled biosphere.

In the second part, accumulation and doses are calculated from the environmental dis-
tribution of radionuclides on the basis of a set of time-independent (that is, steady-state)
concentration and dose factors.

A model requires three components - a conceptual model, a set of mathematical
representations of those concepts and data to fit both the concepts and the
representation. Sections 3 and 4 of this report provide an illustration of some of the
properties of a generic implementation of TAME using data from the Swiss biosphere
database for a generic site in Central Switzerland. The radionuclides chosen for
illustrative purposes (\(^{36}\)Cl, \(^{135}\)Cs and the \(^{237}\)Np chain) are of relevance in the Swiss
waste disposal programme and show the model's behaviour for a constant source term of
radionuclide release into the biosphere for \(10^4\) years.

The detailed implementation of TAME discussed here is not precisely the same as those
used in recent assessments (NAGRA, 1994a, NAGRA, 1994b), but the conceptual
models are the same in all cases and the mathematical representations of radionuclide
transport are precisely those given in Equation (7) on page 21. The main differences
between these two applications mainly relate to the data characterising the sites. In
particular, the mass balance schemes on which radionuclide transport at the two sites are
based, also differ from each other. The exposure pathway sub-model is the same as
described here, although the databases show minor differences.

Summarising, the radionuclide concentrations in the different biosphere compartments
are determined by:

- dilution in water (size of the receiving aquifer) and dilution generally in the modelled
  biosphere (size of the biosphere)
- fluxes of water (radionuclides moving in solution) between the compartments
- fluxes of solid material between the compartments, in particular bioturbation and
  erosion
- sorption which determines the amount of radioactivity accumulated in the
  compartments

Transfers of radionuclides to the food-chains are determined by:

- the source of water for consumption by humans and cattle and for irrigation,
- whether irrigation is applied,
- the crop types represented.

Comparisons within the international study BIOMOVS II give confidence that the
model is performing as required, although this comparison is not a validation in the
strict sense. Proper verification has also been carried out by comparison with results of
the BIOSPH model (BÖHRINGER et al., 1986).
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Appendices
A. The rôle of $k_d$ in compartment models

A.1 Definition of quantities

Conceptually, the contents of TAME can be thought of in the terms shown in Figure A-1. Here the immobile solid component is denoted by $S$, the mobile solids by $M$, the liquid (i.e. the solute in the compartment) by $liq$ and the gas by $air$.

The parameters used in TAME are based on quantities which are ultimately determined by measurements. These include the total volume ($V_{tot}$ [m$^3$]), porosity ($\varepsilon$ [-]), volumetric moisture content ($\theta$ [-]), suspended solid load ($\alpha$ [kg m$^{-3}$]) and the density of the solid material ($\rho$ [kg m$^{-3}$]). The physical dimensions of the compartment are the surface area ($A$ [m$^2$]) and the depth ($l$ [m]).

The total volume of the compartment is given by

$$V_{tot} = l.A = V_S + V_M + V_{liq} + V_{air}, \quad [m^3] \quad (A.1)$$

so that the porosity is defined by
\[ \varepsilon = \frac{V_{\text{liq}} + V_{\text{air}}}{V_{\text{tot}}} \equiv 1 - \frac{(m_M + m_S)}{\rho lA} \quad [-]. \quad (A.2) \]

N.B. in the following analysis it will be assumed that the density of the solid material in both phases is the same, namely

\[ \rho_M = \rho_S = \rho \quad [\text{kg m}^{-3}]. \quad (A.3) \]

The volumetric moisture content is defined as

\[ \theta = \frac{V_{\text{liq}}}{V_{\text{tot}}} = \frac{\text{(total mass of compartment - mass of dry material)}}{\rho_{\text{water}}} \quad [-]. \quad (A.4) \]

The suspended solid load in the liquid is defined in terms of the mass of mobile solid material \( m_M \) kg and the volumes of the liquid and the mobile solid, which together make up the total saturated volume \( (V_p = V_{\text{liq}} + V_M) \) in the compartment:

\[ \alpha = \frac{m_M}{V_p} = \frac{m_M}{V_{\text{liq}} + V_M} \quad [\text{kg m}^{-3}]. \quad (A.5) \]

Experimentally, \( V_{\text{tot}} \) is defined by the initial sample of the compartment material. The total mass of solids in the sample \((m_S + m_M)\) and the porosity and volumetric moisture content \((\varepsilon \text{ and } \theta)\) are determined by drying the sample at 100 °C\(^1\). Similarly, drying the porewater in the sample gives \( \alpha \).

### A.2 Derivation of compartment concentrations from first principles

One of the problems in understanding modelling assumptions at the process level is that there is quite often a lack of clarity in defining and using the representation of terms in the mathematical model. The following derivation is given in considerable detail to make the steps as clear as possible.

The volume of the mobile (i.e. suspended) solid in the compartment is given by

\[ V_M = \text{volume of mobile solids} = V_p \frac{\alpha}{\rho} \quad [\text{m}^3], \quad (A.6) \]

and the volume of the immobile solids in the total compartment is defined as

\[ V_S = \text{volume of immobile solids} = (1-\varepsilon)l.A - V_M \quad [\text{m}^3]. \quad (A.7) \]

The volume of water in the pores containing the mobile solid is \( V_{p} \). This excludes the volume occupied by the mobile solids and is given by

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\(^1\) This neglects the mass of dissolved solids, but the contribution to the total mass is only likely to be significant in the case of non-agricultural soils with very high salt contents.
Substituting this into Equation (A.6) gives an expression for $V_M$ and hence for $V_S$:

$$V_M = \frac{\alpha \theta}{\rho - \alpha} \cdot l \cdot A \quad [m^3], \quad (A.9)$$

$$V_S = \left(1 - \varepsilon - \frac{\alpha \theta}{\rho - \alpha}\right) l \cdot A \quad [m^3]. \quad (A.10)$$

The partitioning between the contaminants in the solution and on the solid phase is modelled by the $k_d$ approach. In general, the $k_d$s of the mobile and immobile solids may not be the same. From the definition of $k_d$, the two quantities can be written as

$$k_S = \frac{C_S}{C_{liq}} \quad [m^3 \text{ kg}^{-1}], \text{ for the fixed material and}$$

$$k_M = \frac{C_M}{C_{liq}} \quad [m^3 \text{ kg}^{-1}], \text{ for the mobile.}$$

The total activity of a given radionuclide in the compartment is $N$ [Bq] and this is partitioned between the three components in the compartment:

$$N = N_{liq} + N_S + N_M = C_{liq} V_{liq} + C_S M_S + C_M M_M$$

$$= \left(V_{liq} + \rho k_S V_S + \rho k_M V_M\right) C_{liq} \quad [Bq],$$

which can further be written in terms of the volumes calculated in Equations (A.4), (A.9) and (A.10) above. The resulting expression can then be rearranged to give $C_{liq}$, the activity concentration of radionuclides in the solute in the compartment:

$$C_{liq} = \frac{1}{\theta \left(1 - \varepsilon - \frac{\alpha \theta}{\rho - \alpha}\right) \rho k_S + \frac{\alpha \theta}{\rho - \alpha} \rho k_M} \cdot \frac{N}{l \cdot A} \quad [Bq \ m^3]. \quad (A.11)$$

Note that this defines the compartmental retention coefficient\textsuperscript{2}, $R$:

$$R = 1 + \left(1 - \varepsilon - \frac{\alpha \theta}{\rho - \alpha}\right) \frac{\rho k_S}{\theta} + \frac{\alpha}{\rho - \alpha} \cdot \rho k_M \ [-].$$

The activity concentration of a radionuclide in the porewater, $C_p$ [Bq m$^{-3}$], is defined as the number of Bq in solution plus the number on the mobile solids, divided by the total volume of the porewater (solute plus suspended solid). Therefore, using Equations (A.8) and (A.9),

$$C_p = \frac{C_{liq} \theta A \left(1 + \frac{\alpha}{\rho - \alpha} \rho k_M\right)}{\theta A \left(1 + \frac{\alpha}{\rho - \alpha}\right)} = \left(1 + \alpha k_M - \frac{\alpha}{\rho}\right) C_{liq} \quad [Bq \ m^3],$$

\textsuperscript{2} This is the retardation coefficient relative to the total content of the compartment. More familiar in geosphere modelling and elsewhere is the retardation of the contents of the porewater: $R_{porewater} = R/\theta$. 

$$R_{porewater} = R/\theta.$$
which, including Equation (A.11), gives

\[ C_p = \frac{1 + \alpha k_M - \frac{\alpha}{\rho}}{\theta + \left(1 - e^{-\frac{\alpha}{\rho - \alpha}}\right) \rho k_s + \frac{\alpha}{\rho - \alpha} \rho k_M} \cdot \frac{N}{lA} \text{ Bq m}^{-3}. \] (A.12)

If the porewater is filtered to remove the suspended solid material, the concentration given by Equation (A.12) will be reduced. Assuming that a fraction \( \phi \) of the particulates remains in the porewater, the concentration is then given by \( C_p^\phi \) [Bq m\(^{-3}\)]

\[ C_p^\phi = \frac{N_{\text{liq}} + \phi N_M}{V_{\text{liq}} + \phi V_M} \text{ [Bq m}^{-3}\],

or

\[ C_p^\phi = \left[ \frac{1 + \phi \alpha k_M - \frac{\alpha}{\rho}}{1 + \phi \frac{\alpha}{\rho} - \frac{\alpha}{\rho}} \right] C_{\text{liq}} \] [Bq m\(^{-3}\)]. (A.13)

N.B.

\[ C_p = \frac{1 + \phi \alpha k_M - \frac{\alpha}{\rho}}{1 + \alpha k_M - \frac{\alpha}{\rho}} C_p^\phi \text{ [Bq m}^{-3}\].

There are two important limits to the expression for the filtered porewater concentrations: full filtration, for which \( \phi = 0 \), and no filtration, for which \( \phi = 1 \). Using these expressions gives

\[ C_p^\phi = \left(1 + \alpha k_M - \frac{\alpha}{\rho}\right) C_{\text{liq}} \text{ and } C_p^0 = C_{\text{liq}} \]

as required.

\[ ^3 \text{ N.B. the factor } \phi \text{ corresponds to the quantity } (1 - f_{\text{filter}}) \text{ used in equations (E.3), (E.4) and (E.5)}. \]
A.3 Practical considerations - the TAME representation

In practice, the suspended sediment loads are always small compared with the solid material density and the quotient $\alpha/\rho$ can be ignored, since $\rho = 2650\text{kg m}^{-3}$ and typically, in the terrestrial compartments, $\alpha \leq 10^{-3}\text{kg m}^{-3}$ (i.e. $\alpha/\rho \leq 4\times10^{-7}$). The concentrations calculated in Equations (A.11), (A.12) and (A.13) above can then be written more simply as

$$C_{\text{aq}} = \frac{1}{1 + \alpha(k_M - k_S)} \frac{\theta}{N} \frac{N}{lA} \text{[Bq m}^{-3}]\text{]}, \quad (A.14)$$

$$C_p = \frac{1 + \alpha k_M}{1 + \alpha(k_M - k_S)} \theta + (1 - \epsilon) \rho k_S \frac{N}{lA} \text{[Bq m}^{-3}]\text{],} \quad (A.15)$$

$$C_p^\phi = \frac{1 + \phi \alpha k_M}{1 + \alpha(k_M - k_S)} \theta + (1 - \epsilon) \rho k_S \frac{N}{lA} \text{[Bq m}^{-3}]\text{].} \quad (A.16)$$

In the TAME parameterisation, the $k_M$s on the mobile and immobile phases are not differentiated so that $k_S = k_M \equiv k_d$. The following simplifications to Equations (A.11), (A.12) and (A.13) are then possible:

$$C_{\text{aq}} = \frac{1}{\theta + (1 - \epsilon) \rho k_d} \frac{N}{lA} \text{[Bq m}^{-3}]\text{]}, \quad (A.17)$$

$$C_p = \frac{1 + \alpha k_d}{\theta + (1 - \epsilon) \rho k_d} \frac{N}{lA} \text{[Bq m}^{-3}]\text{],} \quad (A.18)$$

$$C_p^\phi = \frac{1 + \phi \alpha k_d}{\theta + (1 - \epsilon) \rho k_d} \frac{N}{lA} \text{[Bq m}^{-3}]\text{].} \quad (A.19)$$

The representations given here do not distinguish between the terrestrial and aquatic compartments and so are adequate for both. However, because the terrestrial compartments contain both mobile and immobile solids, whereas the surface water bodies do not (in other words, the mass of the immobile solid is zero), Equations (A.11) - (A.19) are more explicitly relevant to the terrestrial compartments. In surface water bodies, the following results are valid.

The aquatic compartments (rivers and lakes, etc.) are, by definition, saturated in the context of this compartment representation. The volumetric moisture content can then be written

$$\theta = \frac{lA - V_M}{lA} = 1 - \frac{\alpha \theta}{\rho - \alpha} \equiv 1 - \frac{\alpha}{\rho} + \cdots \quad [-],$$

so that Equations (A.11), (A.12) and (A.13) become, for the concentrations in surface water bodies,

$$C_{\text{aq}} = \frac{1}{1 + \alpha k_d} \frac{N}{lA} \text{[Bq m}^{-3}]\text{]}, \quad (A.20)$$
\[ C_p = \frac{N}{lA} \quad [\text{Bq m}^{-3}], \quad (A.21) \]

and
\[ C_p^* = \frac{1 + \phi \alpha k_d}{1 + \alpha k_d} \frac{N}{lA} \quad [\text{Bq m}^{-3}]. \quad (A.22) \]

A.4 The significance of different \( k_d \)s on mobile and immobile solids

The assumption in \textit{TAME} is that \( K_s = K_m = K_d \), i.e. it is assumed that the fixed and the mobile radionuclides in \textit{TAME} have the same sorbing properties on both fixed and mobile solids. This may not in fact be the case. The methodology of TITS \textit{et al.} (1996) bases the choice of appropriate \( k_d \) value for the \textit{TAME} compartments primarily on the particle size. In \textit{TAME} it is implicit that the mobile solids are derived from the same material as the immobile solids. This may not be the case - were colloids to form a component of the system, with particle sizes much smaller than the sand, clay and organic materials covered in the report, then differences between \( k_d \)s on the fixed and the mobile solids could arise.

It is not clear that colloids would play a rôle in the biosphere, although they have excited much interest in geosphere modelling (SMITH, 1993). This section discusses what the effects might be, based on the compartment properties identified in the example calculations in Section 3. The parameter values used for the investigation are given in Table A-1 and these correspond to soils in the example calculations.

For values of the mobile solid \( k_d \) in the range \( 10^{-3} \leq K_m \leq 10^6 [(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}] \), Figure A-2 plots the quantities \( \frac{C_{p,eq}(K_M \neq K_s)}{C_{p,eq}(K_M = K_s)} \) and \( \frac{C_p(K_M \neq K_S)}{C_p(K_M = K_S)} \) for the solute and total porewater concentrations respectively against \( K_M \).

The solute concentration, \( C_{p,eq} [\text{Bq m}^{-3}] \), is relatively unaffected for values of \( K_s \) up to \( 10 [(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}] \) and the influence is first felt in the \( K_m = 0 \) case, for which the solute concentration falls to 10% of the basic \textit{TAME} value for \( K_s = 10^4 [(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}] \). For the cases with \( K_m > 0 [(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}] \), the effect on the solute concentration is much less. For \( K_m = 0.1 \), the solute concentration is reduced to 50% only for \( K_s \approx 10^6 \) and for the highest value here, \( K_m = 0.1, K_s = 10^6 \) reduces the solute concentration by only around 10%. Clearly, as far as the solute concentrations are concerned, the effect of different \( k_d \)s on the two solids components in soils is limited, although for very weakly sorbing radionuclides, if the mobile solid \( k_d \) was greater than \( 10^3 [(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}] \) (a very high value) then some effect might be seen.

The solute concentration is relevant for the root uptake process. The inclusion of higher mobile phase \( k_d \)s might then have some effect on doses arising via this pathway; however there is considerable uncertainty about the experimental conditions used in the
transfer factors used in the current database. To be clear about this effect, it would have to be guaranteed that the soil-to-plant transfer factor was measured as the concentration in the plant relative to the concentration in the porewater of the soil, since root uptake operates via the solution in the soil. As the effect is so small, it is not necessary to include it in TAME at present. Alternative values for the other soil properties do not have a significant effect.

Of greater relevance to the transport calculations is the total porewater concentration. Here the effects are more significant for values of $K_M \gtrsim 10 \text{[(Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}]$, although still relatively minor for realistic enhancements to the fixed solid $k_d$. For $K_s = 0.1$ to $1.0 \text{[(Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}]$, the amount of activity transported in the porewater is increased by a factor of two for $K_M = 10^3 \text{[(Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}]$ and by a factor of ten for $K_M = 10^4 \text{[(Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}]$. Thus, a significant effect could be anticipated in the TAME transport calculations only if the mobile solid $k_d$ was around $10^3$ to $10^5$ times higher than the fixed solid $k_d$. Note that there is no effect on the porewater concentration for non-sorbing species since no activity is retained on the fixed solid and all the activity moves with the water flux, whether in solution or on the suspended solid load.

Overall this effect is not likely to be significant in the TAME calculations, although the methodology outlined here can be used should the database indicate that it is necessary - i.e. for enhancements of $10^3$ - $10^5$ times relative to the fixed solid $k_d$.

Table A-1: Parameter values for an investigation of the effects of different $k_d$s on mobile and immobile solids in TAME.

<table>
<thead>
<tr>
<th>parameter</th>
<th>value</th>
<th>units</th>
</tr>
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<tbody>
<tr>
<td>porosity, $\varepsilon$</td>
<td>0.4</td>
<td>[-]</td>
</tr>
<tr>
<td>volumetric moisture content, $\theta$</td>
<td>0.2</td>
<td>[-]</td>
</tr>
<tr>
<td>suspended solid load, $\alpha$</td>
<td>0.001</td>
<td>[kg m$^{-3}$]</td>
</tr>
<tr>
<td>solid material density, $\rho$</td>
<td>2650</td>
<td>[kg m$^{-3}$]</td>
</tr>
</tbody>
</table>

$K_s \text{[(Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}]$ | Corresponding example radionuclide |
<table>
<thead>
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</tr>
</thead>
<tbody>
<tr>
<td>0.0</td>
<td>$^{36}$Cl</td>
</tr>
<tr>
<td>0.1</td>
<td>$^{135}$Cs</td>
</tr>
<tr>
<td>1.0</td>
<td>$^{237}$Np, $^{233}$U, $^{229}$Th</td>
</tr>
</tbody>
</table>
Figure A-2: The effect of different $k_d$s on fixed and immobile solids on solute concentration (Equation (A.11)) and total porewater concentration (Equation (A.12)). Calculated values based on the soil compartments in the example calculations.
B. Parameterisation of the TAME transfer coefficients for water and solid material fluxes

B.1 General form of the transfer coefficient

The aim is to express the transfer coefficients used in Equation (1) of Section 2.3.1 in terms of observable characteristics of the system, and to illustrate how the same representations can be applied to the terrestrial components of the biosphere system as well as the aquatic. In the compartment model representation, contaminants are transferred from compartment $i$ to compartment $j$ at a fractional transfer rate defined by the transfer coefficient, $\lambda_{ij} \text{ y}^{-1}$, defined as

$$\lambda_{ij} = \sum_{\text{FEPs, }k} \lambda_{ij}^{(k)} = \sum_{\text{FEPs, }k} \frac{1}{N_i} \left( \frac{dN_{ij}}{dt} \right)^{(k)} \text{ [y}^{-1}]$$

(B.1)

where the summation takes into account all FEPs transferring activity from $i$ to $j$. The derivation of the $\lambda_{ij}$ requires a value for the amount of contaminant $N$ in compartment $i$ ($N_i$), as well as an estimation of the fluxes of $N$ to $j$ via all FEPs.

This Appendix deals with the parameterisation of the transfer coefficient for contaminants in solution in the compartmental water which then move with the water flux, and the fraction of contaminants sorbed onto the solid material in the compartment, which then move with the solid material flux. Equation (B.1) becomes, for these two FEPs,

$$\sum_{k=\text{water flux, solid flux}} \left( \frac{dN_{ij}}{dt} \right)^{(k)} = \frac{dN_{ij}^{(\text{liq})}}{dt} + \frac{dN_{ij}^{(\text{sol})}}{dt} \equiv C_{ij}^{(\text{liq})} F_{ij} + C_{ij}^{(\text{sol})} M_{ij},$$

(B.2)

where the suffix (liq) denotes the material in the liquid phase and (sol) denotes the solid phase. The concentrations of contaminant in each of these are, respectively, $C_{ij}^{(\text{liq})}$ [Bq m$^{-3}$] in solution in the water and $C_{ij}^{(\text{sol})}$ [Bq kg$^{-1}$] sorbed onto solid material. The water flux from compartment $i$ to $j$ is $F_{ij}$ [m$^3$ y$^{-1}$] and the solid material flux is $M_{ij}$ [kg y$^{-1}$].

For an element, the solid - liquid distribution coefficient ($k_d$) in compartment $i$ is defined as

$$k_i = \frac{N_{ij}^{(\text{sol})}}{N_{ij}^{(\text{liq})}} = \frac{C_{ij}^{(\text{sol})}}{C_{ij}^{(\text{liq})}} \left[ \frac{(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}}{C_i} \right],$$

(B.3)
(cf. Equation (A.1) in Appendix A), where the specific $k_d$ for the compartment under consideration is denoted by $k_i$. Here the total amount of contaminant $N$ in $i$ is given by the activity in solution plus the activity sorbed on the solid material:

$$N_i = N_i^{(\text{iq})} + N_i^{(\text{tot})} \quad \text{[Bq]}, \quad (B.4)$$

and the total mass of solid in the compartment is $m_i^{(\text{sol})}$ kg and the total volume of liquid is $V_i^{(\text{iq})}$ m$^3$. Therefore the content of the compartment can be written

$$N_i = \left(V_i^{(\text{iq})} + k_i m_i^{(\text{sol})}\right)c_i^{(\text{iq})}. \quad (B.5)$$

Use has been made of equation (B.3). This gives the general form of the transfer coefficient for transport by water and solid material fluxes, combining equations (B.2), (B.3) and (B.5),

$$\lambda = \frac{F_i^{(\text{iq})} + k_i M_i^{(\text{iq})}}{V_i^{(\text{iq})} + k_i m_i^{(\text{sol})}} \quad \text{[y$^{-1}$]}. \quad (B.6)$$

### B.2 Transfer coefficient in terms of compartment physical properties.

The detailed expressions for the water and solid material fluxes are given in terms of a mass balance scheme which is site- and scenario-specific. The TAME input parameters and data files define the general methodology by which $F_{ij}$ and $M_{ij}$ are defined. However, Equation (B.6) can be further expanded in terms of the physical characteristics of the biosphere compartments. Figure B-1 illustrates the composition of a typical compartment.

The total physical volume of a TAME compartment is $V_i^{(\text{tot})}$ and this can be defined in terms of the dimensions of the compartment dimensions:

$$V_i^{(\text{tot})} = A_i l_i = w_i d_i l_i \quad \text{[Bq m$^3$]}, \quad (B.7)$$

where

- $l_i$ [m] is the thickness of the compartment,
- $A_i$ [m$^2$] is the surface area of the compartment, itself the product of:
  - $w_i$ [m] width,
  - $d_i$ [m] length.

This volume is made up of solid material, water and air, so that

$$V_i^{(\text{tot})} = V_i^{(\text{sol})} + V_i^{(\text{iq})} + V_i^{(\text{air})} \equiv \left(1 - \varepsilon_i\right)V_i^{(\text{sol})} + \theta_i V_i^{(\text{iq})} + \left(\varepsilon_i - \theta_i\right)V_i^{(\text{air})}, \quad (B.8)$$
(a) General composition of a compartment in the terrestrial biosphere. Total mass of solid material in the compartment $= \rho_i (1 - \varepsilon_i) V_i^{(tot)}$, total volume of air-filled pore space $= (\varepsilon_i - \theta_i) V_i^{(tot)}$, where the porosity is $\varepsilon_i$ and the volumetric moisture content is $\theta_i$.

(b) General composition of a compartment in the aquatic biosphere. Porosity $\varepsilon_i = 1$, volumetric moisture content $\theta_i = 1$. Mass of suspended solid material $= \alpha_i V_i^{(tot)}$ where the suspended sediment load is $\alpha_i$. Volume of liquid $= V_i^{(tot)}$.

Figure B-1: Illustration of the physical parameters defining the terrestrial and aquatic compartments in TAME.
where here the porosity of the medium is \( \epsilon_i \). The medium may not be saturated with water, the volumetric moisture content is \( \theta_i \) and the difference between these two is the air-filled space. Substituting these definitions into Equation (B.6) gives the general form of the transfer coefficient to be

\[
\lambda_{ij} = \frac{1}{l_i A_i} \frac{F_{ij} + k_i M_{ij}}{\theta_i + (1 - \epsilon_i) \rho_i k_i} \text{[y}^{-1}\text{]},
\]

(B.9)

where the mass of solid material in the compartment has been expressed as the volume it occupies by use of the solid material density in the compartment, \( \rho_i \) [kg m\(^{-3}\)]. In this equation, all the parameters are observables in the biosphere system.

These relations hold for both the terrestrial compartments (soils, aquifer) and the aquatic compartments (rivers, lakes, aquatic sediments). In the case of the terrestrial compartments the interpretation is simple, as shown in Figure B-1 (a). In the case of the aquatic compartments, the parameters are somewhat different (Figure B-1 (b)). In this latter case, the porosity is almost identical to unity, and the compartment can be thought of as being completely saturated. The solid material in the compartment is in the form of a suspended sediment load, \( \alpha_i \) [kg m\(^{-3}\)]. The aquatic sediments are saturated, with a high porosity relative to soils, and represent an intermediate between soils and surface water. In general the local aquifer is less porous and more saturated than soils. In all cases the transfer coefficients take the same form.
C. Diffusive transport in compartment models

C.1 The rôle of diffusion in compartment models

Diffusion is a mechanism acting to equilibrate the concentrations of contaminants in compartments. In the absence of advective fluxes it is the dominant form of intercompartment transport for radionuclides in solution. While the process is not always the most important for solute driven contaminant transport, the formulation used here has indicated that diffusion could play a significant rôle in making waterborne radionuclides available in soils, particularly at early times following the start of release of radionuclides to the biosphere (KLOS et al., 1990; NEA, 1993). In the version of TAME used in the example calculations described in this report, a representation of diffusion is taken from the models of the UK's National Radiological Protection Board (KLOS, et al., 1988; MARTIN, et al., 1991) and although it is not the only one possible, this form is widely used in other similar biosphere models (BIOMOVS II, 1996b). As discussed in the main text, however (Section 3.5), the results from this implementation are not satisfactory, although the consequences for the model results are not significant.

The advection-diffusion equation is a second order partial differential equation. Strictly speaking, therefore, diffusion cannot be treated satisfactorily within the compartment model framework. For practical purposes, however, it is desirable to take diffusion into account in an approximate way and so it is necessary to find representations of the diffusive transfer coefficients which are compatible with the assumptions for mass transport in compartment representations. Conservative assumptions which might overestimate the effect or which might lead to earlier consequences are permissible, provided that they can be shown to give results which are not widely divergent from the correct solutions.

The original derivation of the transfer coefficients for the diffusive transfers as used by MiniBIOS is not available, but from the results in Section 4, the basis for the diffusive transfer coefficients (Equations (6) and (7) in Section 2.3.2) is clearly not correct in the version of TAME described in this report. The first part of this Appendix therefore seeks to arrive at the NRPB result from the parameterisation of the biosphere given in Appendices A and B and as used throughout this report. The factors affecting diffusion in soil porewater are considered and, finally, alternative representations suitable for use in compartment models are discussed in Section C.4.
C.2 Diffusive flux across a boundary

Figure C-1 illustrates the one-dimensional diffusion problem. This corresponds to diffusion across a compartment face in compartment models. A non-zero concentration gradient of particles exists such that there is a higher concentration at the bottom \( (x = 0 \text{ m}) \) than at the top \( (x = l_a + l_b \text{ [m]}) \). The concentrations at two points \( a \) and \( b \), separated by \( \delta x \text{ [m]} \), are known to be \( C_a \) and \( C_b \text{ [Bq m}^{-3}\text{]} \) at time \( t \). The time evolution of the concentrations in the system can be illustrated by calculating the movement of particles across the surface \( S \text{ (area } A_{ab} \text{ [m}^2\text{])} \), assuming a one-dimensional diffusion process.

![Figure C-1: One dimensional diffusion in a column. Transport of particles from a region of high concentration to one of low concentration is indicated by the bi-directional nature of the transfer coefficients. In this illustration the compartments can be thought of as containing solution only.](image)

Fick's law of diffusion states that the rate of transfer of the diffusant is proportional to the concentration gradient; so a short time \( (\delta t \text{ [y]}) \), later the amount of diffusant which has entered the shaded area through surface \( S \) is thus

\[
q_{a \rightarrow b} = D(C_a - C_b)A_{ab} \frac{\delta t}{\delta x} \text{ [Bq]},
\]

(C.1)

where \( D \text{ [m}^2\text{s}^{-1}\text{]} \) is the effective diffusion coefficient associated with the medium containing point \( a \) (allowing for retention, tortuosity, etc. in the two compartments). Thus the net rate of transfer across \( S \) is given, as \( \delta t \rightarrow 0 \), by

\[
\frac{dN_{a \rightarrow b}}{dt} = \lim_{\delta t \rightarrow 0} \frac{q_{a \rightarrow b}}{\delta t} = \left( \frac{D}{\delta x} \frac{C_a}{C_b} - \frac{D}{\delta x} \frac{C_b}{C_a} \right) A_{ab} \text{ [Bq y}^{-1}].
\]

(C.2)

1 N.B. this is a very generic formulation, intended to be the starting-point for the derivation of equation (C.9), which is the form of the transfer coefficient used in MiniBIOS (with additional refinements as a result of the TAME parameterisation). A more physically correct formulation is developed in Section C.5.
Note that the net rate of transfer of particles across the surface can be thought of as the sum of two terms, one acting in the positive direction and one in the negative direction. The transfer rate is proportional to the concentration of the particles in the region of origin. This model corresponds to a region of dynamic exchange in which particles from the region of lower concentration are free to diffuse to the region of higher concentration, but there are more particles at the higher concentration which are available for transport. Until a state of dynamic equilibrium is reached, there will therefore be a net flux from the high concentration region to the low concentration region. From the point of view of an exact solution of the diffusion problem, this statement seems trivial, but the key to representing the transfers in compartment models is to realise that diffusion must be included as a two-way process for all pairs of compartments.

Equation (C.2) is very close to the form of the transfer coefficient used in TAME (see Equation (C.7) below). In fact, because of the representation of dynamic equilibrium inherent in Equation (C.2), it can be seen that it represents two transfer coefficients acting in opposite directions and representing the net transfer rate:

$$\frac{dN_{a\rightarrow b}}{dt} = \frac{dN_+}{dt} + \frac{dN_-}{dt} \quad \text{[Bq y}^{-1}]\text{].} \quad (C.3)$$

The upwards (positive) transfer coefficient in the figure occurs at a rate given by

$$\lambda_+ = \frac{1}{N_a} \frac{dN_+}{dt} = D_\alpha \frac{A_{ab}}{\delta x_a A_{ab}'_a} = \frac{D_\alpha}{\delta x_a l_a} \quad \text{[y}^{-1}]\text{].} \quad (C.4)$$

Because the contents of the compartments are assumed to be well mixed, the instantaneous concentration in the lower region is

$$C_a(t) = \frac{N_a(t)}{A_{ab} l_a} \quad \text{[Bq m}^{-3}]\text{],} \quad (C.5)$$

where both the diffusion coefficient in the vicinity of point $a$, $D_\alpha$ [m$^2$ y$^{-1}$] and the appropriate diffusion length, $\delta x_a$ [m] depend on the properties of the medium around point $a$.

A similar expression holds for the reverse transfer coefficient:

$$\lambda_- = \frac{1}{N_b} \frac{dN_-}{dt} = D_\beta \frac{A_{ab}}{\delta x_b A_{ab}'_b} = \frac{D_\beta}{\delta x_b l_b} \quad \text{[y}^{-1}]\text{].} \quad (C.6)$$
C.3 Usage in compartment models: the NRPB formulation

The equivalent form of the TAME transfer coefficients for diffusion is seen in Equations (6) and (7) of Section 2.3.2 (which includes the full parameterisation of the effective diffusion coefficient, see Section C.4):

\[
\lambda_i = \frac{D_i}{l_i \min(l_i, l_j)} \\
\lambda_j = \frac{D_j}{l_j \min(l_j, l_i)}
\]

(C.7)

Thus the NRPB assumption is that the diffusion length terms - \( \delta_x \) - are adequately represented by the smallest of the two compartment thicknesses:

\[
\delta_x = \min(l_i, l_j), \quad \delta_x = \min(l_i, l_j) \text{ m}
\]

(C.8)

The \( \delta_x \) term corresponds to the dimension of the zone of dynamic exchange and so these quantities correspond to the distance the diffusant moves into the receptor box (thickness \( l_j \) [m]). It is, therefore, not unreasonable to take the minimum of the two thicknesses, since if a receptor compartment is much smaller than the donor, the diffusant cannot migrate by a distance greater than the thickness of the receptor.

C.4 Factors affecting the diffusion coefficient

In this way, the form of the NRPB expressions has been derived. However, as is clear from Section 4.3, there is an important factor missing from these expressions because the results for the non-sorbing \(^{36}\text{Cl}\) concentrations in the aquifer and aquatic bed sediments are not in agreement. A suitable representation is given in Section C.5, but first the parameterisation of the effective diffusion coefficient in TAME is given.

The molecular diffusion coefficient, \( D_0 \) [m\(^2\) y\(^{-1}\)], represents the properties of ions in solution. Diffusive transport takes place in the liquid phase of compartments so the retention factor can be used to distinguish between the sorbed and non-sorbed fractions in the compartment. In addition to this feature, the physical connectedness of the compartment's solid material also plays a part. The tortuosity (\( T \) [-]) is a measure of the increased diffusive path lengths caused by distribution of soil and rock particles and is used in TAME as a structure factor in representing the internal constitution of the solid material in the compartment. Thus the effective diffusion coefficient used in the derivation of the diffusive transfer coefficients above is
allowing for retardation in the compartment (see Equation (A.11) for the retardation factor).

The value of $D_0$ used for TAME is based on measured values for the colloidal environment (CREMERS, 1968):

$$D_0 = 1.2 \times 10^5 \text{ cm}^2 \text{ s}^{-1} = 3.8 \times 10^{-2} \text{ m}^2 \text{ y}^{-1}. \quad (C.10)$$

This value is assumed for all the radionuclides considered in TAME, irrespective of ionic radius.

The value of the tortuosity of a medium is defined by Burger's equation (CREMERS et al., 1966):

$$T_i = 1 + \mu \frac{1 - \epsilon_i}{\epsilon_i} \quad (C.11)$$

where $\epsilon_i$ is the porosity of the medium and $\mu$ is a parameter which depends on the solid particles, and hence on the mineral content of the medium. Soils (and the other media) can be classified as illitic with a corresponding value of $\mu = 1.9$. Pure illite has a $\mu$ value of 4.3 and montmorillonitic soils have different values. These are not immediately of interest for Swiss conditions, unless the future evolution of the site radically affects the mineralogy of the region.

The tortuosities of the TAME compartments based on $\mu = 1.9$ are given in Table C-1.

<table>
<thead>
<tr>
<th>compartment</th>
<th>porosity</th>
<th>tortuosity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Local Aquifer</td>
<td>0.2</td>
<td>8.6</td>
</tr>
<tr>
<td>Deep Soil</td>
<td>0.4</td>
<td>3.9</td>
</tr>
<tr>
<td>Top Soil</td>
<td>0.4</td>
<td>3.9</td>
</tr>
<tr>
<td>Aquatic Sediment</td>
<td>0.5</td>
<td>2.9</td>
</tr>
</tbody>
</table>

Table C-1: Tortuosity for the TAME compartments. Values are derived from Burger's equation, assuming an illitic mineralogy with $\mu = 1.9$. 

\[
D_i = \frac{1}{T_i} \frac{D_0}{R_i} = \frac{1}{T_i} \frac{\theta_i D_0}{\theta_i + (1 - \epsilon_i) \rho_i k_i} \quad [\text{m}^2 \text{ y}^{-1}], \quad (C.9)
\]
C.5 Proposal for a modified form of the diffusive transfer coefficient

C.5.1 The local aquifer / aquatic bed sediment problem in the example calculations

As discussed in Sections 3.2 and 4.3, the two modelling assumptions used to set up the generic TAME representation for the example calculations effectively isolate the river bed sediment porewater from all other parts of the system except by diffusion. For $^{36}$Cl, modelled in the example as having zero $k_d$ in all compartments, the only route into the bed sediment is via diffusion, as all other bed sediment interactions are mediated by solid material transport.

While these modelling assumptions are not inappropriate in the context of the modelled river (a fast flowing alpine river), the results obtained for the concentration in the aquatic sediments are physically impossible since, in the example, the porewater in the bed sediment and in the aquifer should have the same concentration. That they do not, in the results presented in Section 4.3, indicates an error in the representation of the diffusion process in the generic TAME representation used here.

Because of the isolation of the aquatic bed sediment in the case of $^{36}$Cl in the example, this part of the generic TAME system can be used as a test-bed for alternative formulations of diffusion driven transport in compartment models. The example of the $^{36}$Cl, aquifer - bed sediment system can be used to derive a more consistent form of the diffusive transfer coefficients, which can then be tested by comparing the results with a numerical solution for the problem as defined by the modelling assumptions.

It should be remembered that the situation as modelled is not a representation of reality - it is a representation of the modelled system, based on a set of modelling assumptions which are shown to be appropriate for the calculation of radiological exposure of a critical group of individuals in the example biosphere (Section 4.6). These assumptions justify the (admittedly unrealistic) isolation of the aquatic bed sediment porewater other than by diffusion. When considering the reality of the aquifer/bed sediment/river water system, there would be interaction of the river bed with the river water and there would be significant transport of bed sediment material downstream (which would be replenished by sedimentation and deposition on an annual cycle). The unphysical build-up of $^{36}$Cl in the sediment porewater would not occur.

It can therefore be considered fortunate that the modelling assumptions in this case were such that they not only illustrated the problem with the representation of diffusive processes, but also provided the means of improving the generic formulation in TAME.

To develop a better representation of the diffusive transfer coefficients, the correct physical properties of the system must be employed. As has already been noted, the effective diffusion coefficients in the two compartments depicted in Figure C-1 might be different but, more importantly, diffusive transport from the first medium to the second depends on the properties of both compartments. In this case Equation (C.2) should be rewritten as
\[
\frac{dN_{a\rightarrow b}}{dt} = \left( D_{a\rightarrow b} \frac{C_a}{\delta x_{a\rightarrow b}} - D_{b\rightarrow a} \frac{C_b}{\delta x_{b\rightarrow a}} \right) A_{ab} \quad [\text{Bq y}^{-1}], \quad (C.12)
\]

where \( D_{i\rightarrow j} \) are the diffusion constants (corrected for the physical properties of the compartments), \( A_{ab} \) is the area of the interface between the compartments and \( \delta x_{i\rightarrow j} \) represent regions of dynamic exchange on either side of the interface. Diffusion acts on the concentration in the porewater in each of the compartments, which is given by the results of Appendix A:

\[
C_a = \frac{1}{\theta_a + (1 - \epsilon_a) \rho_a k_a} \frac{N_a}{l_a A_a} \quad [\text{Bq m}^{-3}], \quad (C.13)
\]

\[
C_b = \frac{1}{\theta_b + (1 - \epsilon_b) \rho_b k_b} \frac{N_b}{l_b A_b} \quad [\text{Bq m}^{-3}]. \quad (C.14)
\]

N.B. the region of the compartments actively involved in the diffusive transport between the compartments is \( \delta x \) m thick. However, because the compartments are assumed to be instantaneously well mixed, the porewater concentrations used in calculating the diffusive fluxes are those that are applicable in the compartment as a whole. This is a property of the compartment model approach.

The porewater concentrations depend on the following physical properties:

- \( \epsilon_i \) [-] compartmental porosities,
- \( \theta_i \) [-] compartmental volumetric moisture contents,
- \( l_i \) [m] thicknesses of the compartments,
- \( A_i \) [m\(^2\)] compartmental plan areas,
- \( \rho_i \) [kg m\(^{-3}\)] density of solid material in the compartments,
- \( k_i \) [(Bq kg\(^{-1}\))(Bq m\(^{-3}\))\(^{-1}\)] solid - liquid distribution coefficients

and \( N_i \) [Bq] are the compartment inventories.

As before, Equation (C.12) is the sum of two fluxes acting in opposite directions, and so the corresponding transfer coefficients for diffusion between the two compartments can be written

\[
\lambda_{ab} = \frac{1}{N_a} \frac{dN_{a\rightarrow b}}{dt} = \frac{D_{a\rightarrow b}}{\delta x_{a\rightarrow b}} \cdot \frac{1}{\theta_a + (1 - \epsilon_a) \rho_a k_a} \frac{A_{ab}}{l_a A_a} \quad [\text{y}^{-1}] \quad (C.15)
\]

---

\(^2\) This is a reasonable assumption for soils, where mixing in the soil layers considered in TAME occurs as a result of bioturbation by resident flora and fauna. This is not the same bioturbation which causes transport from the deep soil to the top soil (and vice versa). The intercompartment transfer is a result of the migration of fauna between the two soil layers - see Appendix D.2.2. Bioturbation here is assumed to be wholly within each of the compartments. The assumption is less valid for the aquifer compartment, but is mildly conservative as it will tend to increase upward migration of radionuclides to the soils.
and \( \lambda_{b,a} = \frac{1}{N_b} \frac{dN_{b\rightarrow a}}{dt} = \frac{D_{b\rightarrow a}}{\delta x_{b\rightarrow a}} \cdot \frac{1}{\theta_b + (1 - \epsilon_b) \rho_b k_b l_b A_{ab}} \cdot [y^{-1}] \) \hspace{1cm} (C.16)

These are the most general form of the intercompartmental transfer coefficients for the diffusive process. Before they can be used in TAME, consideration must be given to the quantities

- \( A_{ij} \) [m²] the area of the interface between the compartments
- \( \delta x_{i\rightarrow j} \) [m] the diffusion lengths
- \( D_{i\rightarrow j} \) [m² y⁻¹] the effective diffusion constant for transport between the compartments.

There are several possibilities for the representations of diffusion. The example below has been set up to represent the transport of \(^{36}\)Cl between the local aquifer and the bed sediment in the TAME example given in the main text. Figure C-2 illustrates the aquifer-sediment system in the example calculations for \(^{36}\)Cl. The transfer coefficients for non-sorbing \(^{36}\)Cl are simplified with respect to Equations (C.15) and (C.16), since the terms involving \( k_d \) drop out. Having determined the appropriate form of the other parameters, however, the final version of the general transfer coefficient can then be reconstituted.

In the example biosphere described in Section 4.3, the aquatic sediment \( S \) is assumed to lie on top of the local aquifer \( L \) and the physical dimensions of the system are represented by

- \( l_L \) 15 m thickness of the local aquifer
- \( A_f \) 10⁶ m² plan area of the local aquifer
- \( d_S \) 0.1 m thickness of the aquatic sediment
- \( w_W \) 10 m width of the surface water and
- \( l_w \) 10³ m the length of the river section.

All other quantities used in Equations (C.15) and (C.16) above are denoted by the appropriate subscript, \( L \) or \( S \). Figure C-3 shows the results of the four forms of the diffusive transfer coefficients investigated for this system and compares them to the original results using Equation (C.7). These four representations are discussed below.

**Area of the interface: \( A_{ij} \):**

In deriving Equation (C.7), no allowance for the different areas of the interface between the aquifer and the sediment was made. Clearly they are not the same. The interface area must be represented by

\[ A_{ij} = \min(A_f, l_w w_W) \] \hspace{1cm} [m²]. \hspace{1cm} (C.17)

In situations such as between the soil compartments, this problem does not arise because the aquifer and the two soil layers have the same area, namely the area of the farmland \( A_f \) [m²]. This was the case in the original NRPB usage and is clearly incorrect for the situation modelled in TAME, where the aquifer underlies the soils and the river.
Figure C-2: The system for describing diffusion between the TAME aquifer - sediment compartments for the case of a non-sorbing radionuclide. In this system the equations to be solved are for the inventories in the sediment and the local aquifer:

\[
\frac{dN_s}{dt} = \lambda_{LS} N_L - \lambda_{SL} N_s
\]

\[
\frac{dN_L}{dt} = \lambda_{SL} N_s - \lambda_{LS} N_L - 1.55N_L + S(t)
\]

where the loss term from the aquifer comes from the numerical results given in Table 4-1 (as transport half-times). The source term is the top-hat function used in the example calculations (10^6 Bq y^{-1} for 10^4 years). The half-life of $^{36}$Cl is sufficiently long that it can be neglected in these calculations.
Diffusion Length: \( \delta x_{i\to j} \):

It is assumed that the diffusion length is related to the dimensions of the two compartments. There are two possibilities:

\[
\delta x_{L\to S} = \delta x_{S\to L} = \frac{1}{2} (l_L + d_S) \quad [\text{m}],
\]

(C.18a)

and

\[
\delta x_{L\to S} = \delta x_{S\to L} = \min\left(l_L, d_S\right) \quad [\text{m}].
\]

(C.18b)

The first is reasonable if the thicknesses of the compartments are similar - effectively the diffusion lengths in the compartments are the half-thicknesses of the boxes. The second alternative is more appropriate when the thicknesses are very different (as in this case). Diffusion then takes place over a dimension determined by the smaller of the two compartments. However, the assumption of rapid mixing in the receptor corresponds, in part, to diffusion within the receptor, with the other properties of the receptor applying.

Effective Diffusion Coefficient: \( D_{i\to j} \):

The effective diffusion constant can be calculated by recognising that, inside the individual compartments, the effective diffusion constant is given by

\[
D_i = \frac{\theta_i}{T_i} D_0 \quad [\text{m}^2 \text{y}^{-1}],
\]

as seen in Equation (C.7) (for the case of \( k_d = 0 \)). Again two possibilities exist for the representation of the intercompartment diffusion constant:

\[
D_L\to S = D_S\to L = \frac{1}{2} \left(D_L + D_S\right) = \frac{1}{2} \left(\frac{T_L \theta_L + T_S \theta_S}{T_L T_S}\right) D_0 \quad [\text{m}^2 \text{y}^{-1}]
\]

(C.20a)

or

\[
D_L\to S = D_S\to L = \frac{l_L D_L + d_S D_S}{l_L + d_S} = \left(\frac{T_L \theta_L + T_S \theta_S}{T_L T_S}\right) \frac{D_0}{l_L + d_S} \quad [\text{m}^2 \text{y}^{-1}].
\]

(C.20b)

These considerations lead to the four model representations for which numerical results have been calculated. The nomenclature used to distinguish them is given in Table C-2. The results of the models are shown in Figure C-3.

The biggest change between the original model and the new representation is that now the porewater concentrations in the compartments both tend to the same value. This results from the correct interpretation of the area of the interface between the two compartments (Equation C.17). N.B. the results for the concentration in the aquifer are unchanged because the source term and loss term to the elsewhere compartment dominate.

The effect of the various options for diffusion length and effective diffusion coefficient only affect the time taken to reach equilibrium. The longest is over \( 10^4 \) years (mean values of both parameters - model dif 1) and the shortest is around ten years (minimum diffusion length, volume-averaged diffusion coefficient- model dif 3). The use of the minimum value of the compartment thickness has the largest influence (compare dif 3 and dif 1).
Table C-2: Naming conventions for the model representations of the diffusive transfer coefficients in TAME.

<table>
<thead>
<tr>
<th>model name</th>
<th>Equations used</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>dif 0</td>
<td>C.7</td>
<td>Original TAME representation</td>
</tr>
<tr>
<td>dif 1</td>
<td>C.17, C.18a, C.20a</td>
<td>Mean diffusion length and diffusion constant</td>
</tr>
<tr>
<td>dif 2</td>
<td>C.17, C.18a, C.20b</td>
<td>Mean diffusion length, volume-averaged diffusion coefficient</td>
</tr>
<tr>
<td>dif 3</td>
<td>C.17, C.18b, C.20a</td>
<td>Minimum diffusion length, mean diffusion coefficient</td>
</tr>
<tr>
<td>dif 4</td>
<td>C.17, C.18b, C.20b</td>
<td>Minimum diffusion length, volume-averaged diffusion coefficient</td>
</tr>
</tbody>
</table>

Figure C-3: Results for the different parameterisations of the diffusion process.
C.5.2 Comparison with the solution of the diffusion equation

The question for the compartment model representation of the diffusion process is now which of these representations best fits reality as expressed by the diffusion equation

\[
\frac{\partial C}{\partial t} = D_{app} \frac{\partial^2 C}{\partial x^2}
\]  

(C.21)

To investigate this, a similar situation to the one discussed in Section C.5.1 has been modelled, using many of the same parameters describing the problem of the TAME aquifer and bed sediments. Figure C-4 illustrates the simplified problem, in which the only process involved in transport is diffusion.

Initially, there is a uniform concentration in the aquifer \((C_0(t) = 10^3 [\text{atom m}^{-3}])\) and zero in the bed sediment \((C_0(t) = 0 [\text{atom m}^{-3}])\). Radioactive decay is not taken into account, nor is there an external source term. The apparent diffusion coefficient in medium \(i\) is given by

\[
D_{app}^{(i)} = \frac{\theta_i D_0}{T_i} [\text{m}^2 \text{y}^{-1}],
\]  

(C.22)

with the volumetric moisture content \(\theta_i\) being equal to the compartmental porosity, \(\varepsilon_i\), because both media are saturated \((\varepsilon_L = 0.2 [-], \varepsilon_S = 0.5 [-])\). Otherwise, the problem has parameters as given in the earlier example.

Equation (C.21) has been solved numerically using the IMSL MOLCH routines, with a spatial discretisation of 0.01 m in both the aquifer and sediment compartments. Figure C-5 shows the behaviour for the exact solution compared with the four alternative compartment model representations, where the average concentration in each of the compartments has been evaluated from the MOLCH results. Results for both the concentration in the aquifer (practically constant at around \(10^3\) atom m\(^{-3}\)) and for the concentration in the bed sediment are given (bold curve). The results from the representations dif 1 to dif 4 are also shown (see key on figure).

The exact solution does not reach equilibrium until around \(10^3\) years, although 90% of the steady-state concentration in the bed sediment is reached around 100 years.

Of the four proposed representations, dif 2 is clearly inappropriate since its equilibrium is reached after 1500 years. Dif 1 reaches equilibrium at around the correct time, but at the earlier times it gives a lower concentration in the sediment (Figure C-5 (a)).

Both dif 3 and dif 4 reach equilibrium on relatively short timescales (Figure C-5 (b)): dif 3 before 10 years and dif 4 around 25 years.

Based on this analysis, the choice for the compartment model representation of diffusive transfers is between dif 1 and dif 4.
distribution at $t = 0$ \hspace{1cm} distribution at $t > 0$

\[
\frac{\partial C}{\partial x} \bigg|_{x=L_t,d_s} = 0, \forall t
\]

\[\begin{array}{c}
C_S(0) = 0 \\
C_L(0) = 10^3
\end{array}\]

\[\begin{array}{c}
x = l_L + d_s = 15.1 \text{ m} \\
x = l_L = 15 \text{ m}
\end{array}\]

\[
\frac{\partial C}{\partial x} \bigg|_{x=0} = 0, \forall t
\]

Figure C-4: Simple diffusion based on the TAME aquifer/sediment problem. Here the source term, radioactive decay and advective losses are switched off, leaving the diffusion equation as in Equation (C.21) to be solved. The initial conditions are that the initial concentration in the aquifer is uniform and in the bed sediment it is zero. (N.B. as there is no radioactive decay, the concentration can be thought of as atoms m$^{-3}$).
Figure C-5: Comparison of the exact solution of the diffusion equation with alternatives from the compartment model approach. The solutions over the long term show the approach to the equilibrium concentrations in each of the compartments. The short term behaviour is shown below.
C.5.3 Diffusive transfer coefficients for use in radioactive waste disposal performance assessments

Clearly none of the proposed compartmental modelling alternatives are completely satisfactory in their representation of the diffusion process. In choosing a representation for use in *TAME*, the assessment context can be used to screen suitable representations. Mild conservatism in the maximum consequence and an earlier time of occurrence are permissible, provided there are clear justifications.

Having included the area of the interface into the representations (Equation (C.17)), the consequences (i.e. concentration in the porewater in the two compartments) produce the correct solution. Only the timescales are different between the true solution and the compartment model results. Taking the dimensions of the compartments and the initial conditions into account, it can be seen that the situation investigated here is quite a severe test of the diffusion representation and the choice of the form of the diffusive transfer coefficients should be made with care. Rather than looking at the time to steady-state in the sediment compartment in the above example, an appropriate timescale for defining the most appropriate formulation might be the time taken for the true solution to reach 90% of the steady-state concentration in the bed sediment. This would favour dif 4, which gives earlier consequences.

In the model scenarios in *TAME*, contaminants enter the biosphere from below and consequences arise from the accumulation of concentrations physically higher in the system. In general terms, the results seen here for the aquifer/bed sediment combination mean that the use of the dif 4 formulation applied to diffusion between the aquifer and the deep soil or between the deep soil and the top soil would tend to overestimate contaminant concentrations in the soil compartments at earlier times (though not in the longer term). This is a minor conservatism in the model which is not inappropriate, especially as it is not anticipated that diffusive transport will be a major determining factor in the radiological consequences calculated by *TAME*. Another reason for accepting the dif 4 form is that, in the pure diffusion case modelled here, other factors which could lead to rapid mixing in the compartments are neglected. If there was additional mixing in the sediment compartment in the above examples, the time to reach equilibrium would be much less than appears in Figure C-5. Such factors are known to be present in the soil and aquatic systems modelled by *TAME* and they are major justifications for the use of compartment models in the first place. (For example there is bioturbation within each of the soil layers as well as between the deep and top soil and the porewater in the bed sediment would be well mixed by turbulence effects from the river water.) Only in the aquifer is the assumption of rapid mixing not a good approximation and the compartment model approach is clearly conservative in its effect on the time of occurrence.

The representation of diffusion given by the representation dif 4 will therefore be adopted in future *TAME* modelling. The diffusive transfer coefficient from compartment $i$ to compartment $j$ is
\[
\lambda_y = \frac{1}{l_i A_i} \frac{\min\left(\frac{v_i}{x_{i \rightarrow j}}, \frac{v_j}{x_{j \rightarrow i}}\right)}{\min(x_{i \rightarrow j}, x_{j \rightarrow i})} \left(\frac{T_i l_i \theta_i + T_j l_j \theta_j}{T_i T_j} \frac{D_0}{l_i + l_j} \theta_i + (1 - \varepsilon_i) \rho_i k_i\right) \quad [y^{-1}] \quad (C.23)
\]

and this must be complemented by an equivalent return transfer.

In this final form, the direction of diffusion is explicitly given as \( x_{i \rightarrow j} \) (and vice versa). This accounts for either vertical diffusion, in which case \( x_{i \rightarrow j} = l_i \), or horizontal, with \( x_{i \rightarrow j} = V_i / A_{ij} \). The interface area is similarly defined in terms of the volume and the corresponding diffusion length. This is not a circular definition of terms. In TAME the volume of the compartment is always defined either by one length and an area or by three lengths. The expressions to be used in (C.23) can therefore always be evaluated.
D. **TAME** input parameters and their relation to the parameterisation of the transfer coefficients

D.1 Parameters defining water fluxes in **TAME**

Section 2.3.2 of this report illustrates how the transfer coefficients in the **TAME** representation of the biosphere are derived from a matrix of water and solid fluxes between the compartments, \( F_{ij} \) [m\(^3\) y\(^{-1}\)] and \( M_{ij} \) [kg y\(^{-1}\)] respectively. The site characteristics are not usually defined in such terms; they include climatological parameters - precipitation, evapotranspiration; topological parameters - erosion rates; human behaviour parameters - irrigation rates, and so on.

**TAME** works internally with \( F_{ij} \) and \( M_{ij} \) to establish mass balance in the system. The site-specific input parameters in the model which define these largely determine how mass balance is maintained in the modelled biosphere. As the mass balance scheme is site-specific, general expressions for all the transfer coefficients valid for all applications, in the manner in which the exposure pathway factors are defined in Section 2.3.4, is not practical. However, in many situations similar relationships are employed in deriving \( F_{ij} \) and \( M_{ij} \) and these are shown in this Appendix.

D.1.1 Precipitation, irrigation and evapotranspiration

Precipitation and evapotranspiration combine to give an input at the soil surface from the atmosphere. In the mass balance scheme for a region this is represented by

\[
F_{AT} = (d_{AT} - ETP)A_f \quad [m^3 \text{ y}^{-1}]
\]  
(D.1)

where \( d_{AT} \) [m y\(^{-1}\)] is the mean annual precipitation, \( ETP \) [m y\(^{-1}\)] is the mean annual evapotranspiration and \( A_f \) [m\(^2\)] is the area of the farmland in the biosphere section.

Irrigation is potentially taken from the local aquifer (well abstraction) or from the surface water. In the model, the application to an area is considered (i.e. from the aquifer, the parameter is \( d_{LT} \) and from the surface water \( d_{WT} \) and both have units [m\(^3\) y\(^{-1}\) m\(^{-2}\)]):

\[
F_{LT} + F_{WT} \equiv (d_{LT} - d_{WT})A_f \quad [m^3 \text{ y}^{-1}]
\]  
(D.2)
D.1.2 Capillary rise

Following the implementation of this process by GROGAN et al. (1991), the water fluxes between the deep soil and the top soil and, similarly, between the local aquifer and the deep soil, are given by

\[ F_{DT} = d_{\text{capil}} A_f \text{[m}^3\text{y}^{-1}] \], and \( F_{LD} = d_{\text{capil}} A_f \text{[m}^3\text{y}^{-1}] \). \( \text{(D.3)} \)

where the mean annual capillary rise through the soil pores is \( d_{\text{capil}} \text{[m}^3\text{y}^{-1}] \).

The other water fluxes are defined in terms of the volumetric flows. These might include the inflow into the local aquifer (both contaminated and uncontaminated) and inflow from the river section. Other inputs to the system will depend on the particular site and application.

D.2 Parameters defining solid material fluxes in TAME

Transport of solid material can take place either as the bulk movement of solid material - e.g. erosion, bioturbation, etc. - or as a result of suspended solid material moving with water fluxes. The effects of these two mechanisms are best illustrated for each of the mass transport fluxes explicitly modelled in TAME.

D.2.1 Solid material in the water fluxes

The suspended solid load in the surface water compartment is \( \alpha_W \text{[kg m}^{-3}] \); consequently the throughflow of water in the surface water compartment corresponds to a mass transfer of

\[ M_{WE} = \alpha_W F_{WE} \text{[kg y}^{-1}] \]. \( \text{(D.4)} \)

Similarly, suspended solid material in the porewater gives rise to mass transport, and the following relationships hold

\[ M_{ij}^{(\text{suspended solids})} = \alpha_i F_{ij} \text{[kg y}^{-1}] \]. \( \text{(D.5)} \)

for the cases \( i = L, D, T \) and \( j = D, T, W, E \).

---

1 This is not exact but it is a very close approximation since \( V_{\text{sol}} \ll V_{\text{liq}} \) - see Appendices A and B.
D.2.2 Erosion and bioturbation

These processes constitute the bulk movement of solid material in the biosphere system. Erosion is determined by the regional erosion rate: \( m_e \) [kg m\(^{-2}\) y\(^{-1}\)]. This applies primarily to the top soil and, in accordance with the simplifications set out in Section 2.2, the mass transfer rate from the top soil to the surface water is therefore

\[
M^{(erosion)}_{TW} = m_e A_f \quad \text{[kg y\(^{-1}\)].} \tag{D.6}
\]

However the compartments in the model are assumed to have constant dimensions, so that the consequence of removing material from the top of the top soil is that a corresponding mass of deep soil is transferred to the top soil compartment, requiring the transfer from the local aquifer to the deep soil:

\[
M^{(erosion)}_{DT} = M^{(erosion)}_{LD} = m_e A_f \quad \text{[kg y\(^{-1}\)].} \tag{D.7}
\]

There must also be a corresponding transfer from the geosphere material below the local aquifer:

\[
M^{(erosion)}_{GL} = m_e A_f \quad \text{[kg y\(^{-1}\)].} \tag{D.8}
\]

The latter corresponds, in general, to a diluting input to the system since the geosphere is external to the biosphere system and it is usually assumed that only groundwater flows from the geosphere are responsible for the input of radionuclides to the biosphere.

Bioturbation is the term given to the mixing brought about by plants and animals in the soil system. In TAME the transfer process by animals is considered only in soils and, in particular, only the transport between the deep soil and top soil is explicitly modelled. Bioturbation taking place wholly within the top soil or deep soil is one of the mixing processes which justify the use of compartment models. N.B. Other modellers use bioturbation to model the mixing of different layers of sediments in the representation of the aquatic environment in their models; however, as there is only one sediment compartment in TAME it is not necessary to include it.

MÜLLER-LEMANS & VAN DORP (1995), have reviewed the literature available for characterising the bioturbation process and found that the main species involved in the transport from deep soil to top soil in Switzerland are earthworms. The mass transfer rate is modelled as

\[
M^{(bioturbation)}_{DT} = m_p W_D A_f \quad \text{[kg y\(^{-1}\)].} \tag{D.9}
\]

where \( m_D \) [kg m\(^{-2}\)] is the active biomass involved in the transport and \( W_D \) is the number of times the body mass carried by a single worm per year. Default values for these parameters are \( m_D = 0.1 \) kg m\(^{-2}\) and \( W_D = 20 \) y\(^{-1}\).

Generally, in TAME only the upward transfer is modelled in this way. The return from top soil to deep soil is derived from mass balance considerations (see the following section) and, unlike erosion, does not involve the movement of materials between
compartments because of the constant volume assumption. Other models (e.g. MARTIN et al., (1991); MARTIN et al., (1992); NEA, (1993)) model bioturbation as a diffusion-like process. The advantage of the TAME approach is that the parameterisation is based on quantities which can be obtained by observations and measurements.

D.2.3 Aquatic sediment processes

One of the conservative restrictions discussed in Section 2.2 concerns the downstream boundary of the surface water body. By default, only radionuclides in solution or in suspension in water flows are assumed to leave the system. Not all the material eroded from the top soil is transported downstream in suspension, some deposition being assumed on the bed sediments. Mass balance for the sediment compartment is governed by two processes: resuspension and bed sediment to land transfer. In the present default model description these are not well characterised, but their implementation is consistent with the conservative basis of the model. A better characterisation would require detailed information about the site of interest. However, as discussed in Section 4.6, the impact of these assumptions on the results from TAME are small in the context of the overall assessment end-points.

It is assumed, as a balance for the erosion from the top soil, that an equal amount of solid material is transferred to the top soil from the sediment as is eroded from the surface into the water compartment (sedimentation then being derived from mass balance). Furthermore it is assumed that an average turnover rate for bed sediments is once-per-year, representing the interaction of the surface layer of the sediment with the flowing water. These processes are parameterised as follows:

\[ M_{ST}^{\text{sediment-land}} = m_{sl}A_f \quad \text{[kg y}^{-1}] \quad \text{(D.10)} \]

where the sediment to land transfer parameter is \( m_{sl} \) [kg m\(^{-2}\) y\(^{-1}\)]. N.B. the area here corresponds to that of the farmland not the aquatic sediment compartment. As explained above, the default assumption is that \( m_{sl} = m_e \), the regional erosion rate. The interaction of the bed sediment with the water column is modelled by

\[ M_{SW} = \kappa_{SW}w_wl_wd_s(1-\varepsilon_s)\rho_S \quad \text{[kg y}^{-1}] \quad \text{(D.11)} \]

where

- \( w_wl_wd_s \) [m\(^3\)] is the volume of the sediment compartment
- \( \varepsilon_s \) [-] is the sediment porosity
- \( \rho_S \) [kg m\(^{-3}\)] is the density of the bed sediment solid
- \( \kappa_{SW} \) [y\(^{-1}\)] is the parameter representing the interaction with the water column. The default is 1 y\(^{-1}\).
D.2.4 Mass balance in TAME

Figure D-1 shows a typical example of the water balance scheme for the five TAME compartments. The inputs to the system are shown in Table D-1.

The corresponding scheme for solid material balance (for the same site) is shown in Figure D-2, with the input parameters shown in Table D-2.

It should be remembered that these mass balance schemes are not the only ones possible. They have been derived in accordance with the model assumptions given in Section 2.2 and also rely on other simplifications arising from imprecise knowledge about the fluxes at the site.
Table D-1: Input parameters defining the water flux balance in a typical TAME representation. The shaded entries are those for which the volumetric flows are entered without further parameterisation. All other water fluxes in the system are defined by mass balance - see Figure D-1.

<table>
<thead>
<tr>
<th>input parameter</th>
<th>units</th>
<th>water flux [m$^3$ y$^{-1}$]</th>
<th>comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>$d_{AT}$</td>
<td>m y$^{-1}$</td>
<td>$F_{AT}$</td>
<td>precipitation: $F_{AT} = d_{AT} A_f$</td>
</tr>
<tr>
<td>$ETP$</td>
<td>m y$^{-1}$</td>
<td>$F_{TA}$</td>
<td>evapotranspiration: $F_{TA} = ETP \cdot A_f$</td>
</tr>
<tr>
<td>$d_{capi}$</td>
<td>m y$^{-1}$</td>
<td>$F_{LD}, F_{DT}$</td>
<td>capillary rise: $F_{ij} = d_{capi} \cdot A_f$</td>
</tr>
<tr>
<td>$d_{LT}$</td>
<td>m y$^{-1}$</td>
<td>$F_{LT}$</td>
<td>irrigation with well water: $F_{LT} = d_{LT} A_f$</td>
</tr>
<tr>
<td>$d_{WT}$</td>
<td>m y$^{-1}$</td>
<td>$F_{WT}$</td>
<td>irrigation with surface water $F_{WT} = d_{WT} A_f$</td>
</tr>
<tr>
<td>$F_{PW}$</td>
<td>m$^3$ y$^{-1}$</td>
<td>$F_{PW}$</td>
<td>Input from upstream surface water</td>
</tr>
<tr>
<td>$F_{UL}$</td>
<td>m$^3$ y$^{-1}$</td>
<td>$F_{UL}$</td>
<td>Uncontaminated flow into the aquifer</td>
</tr>
<tr>
<td>$F_{CL}$</td>
<td>m$^3$ y$^{-1}$</td>
<td>$F_{CL}$</td>
<td>Contaminated flow into the aquifer (geosphere source term)</td>
</tr>
</tbody>
</table>

Figure D-1: A typical TAME mass balance scheme for water fluxes. The parameters used to describe this case are given in Table D-1 and the corresponding solid balance scheme is shown in Figure D-2.
Table D-2: Input parameters defining the solid material flux balance in a typical TAME representation. All other solid material fluxes in the system are defined by mass balance - see Figure D-2.

<table>
<thead>
<tr>
<th>input parameter</th>
<th>units</th>
<th>solid material flux [m$^3$ y$^{-1}$]</th>
<th>comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\alpha_w$</td>
<td>kg$^{-1}$ m$^{-3}$</td>
<td>$M_{PW}$</td>
<td>suspended solid flux: $M_{PW} = \alpha_w F_{PW}$</td>
</tr>
<tr>
<td>$\alpha_w$</td>
<td>kg$^{-1}$ m$^{-3}$</td>
<td>$M_{WT}$</td>
<td>suspended solid flux: $M_{WT} = \alpha_w F_{WT}$</td>
</tr>
<tr>
<td>$\alpha_L$</td>
<td>kg$^{-1}$ m$^{-3}$</td>
<td>$M_{LT}$</td>
<td>suspended solid flux: $M_{LT} = \alpha_L F_{LT}$</td>
</tr>
<tr>
<td>$m_e$</td>
<td>kg$^{-1}$ m$^{-2}$ y$^{-1}$</td>
<td>$M_{TW}$</td>
<td>erosion: $M_{TW} = m_e A_f$</td>
</tr>
<tr>
<td>$\alpha_D$</td>
<td>kg$^{-1}$ m$^{-3}$</td>
<td>$M_{DT}$</td>
<td>suspended solid flux, erosion and bioturbation: $M_{DT} = \alpha_D F_{DT} + (m_e + m_D) A_f$</td>
</tr>
<tr>
<td>$m_e$</td>
<td>kg$^{-1}$ m$^{-2}$ y$^{-1}$</td>
<td>$M_{LD}$</td>
<td>suspended solid flux and erosion: $M_{LD} = \alpha_L F_{LD} + m_e A_f$</td>
</tr>
<tr>
<td>$m_e$</td>
<td>kg$^{-1}$ m$^{-2}$ y$^{-1}$</td>
<td>$M_{GL}$</td>
<td>erosion: $M_{GL} = m_e A_f$</td>
</tr>
<tr>
<td>$\alpha_L$</td>
<td>kg$^{-1}$ m$^{-3}$</td>
<td>$M_{LW}$</td>
<td>suspended solid flux: $M_{LW} = \alpha_L F_{LW}$</td>
</tr>
<tr>
<td>$\kappa_{SW}$</td>
<td>y$^{-1}$</td>
<td>$M_{SW}$</td>
<td>bed sediment interaction with the water column: $M_{SW} = \kappa_{SW} w_{lu} d_{sl} (1 - \varepsilon_s) \rho_s$</td>
</tr>
<tr>
<td>$m_{st}$</td>
<td>kg$^{-1}$ m$^{-2}$ y$^{-1}$</td>
<td>$M_{ST}$</td>
<td>Mass of sediment transferred to area of farmland: $M_{ST} = m_{st} A_f$</td>
</tr>
</tbody>
</table>

Figure D-2: A typical TAME mass balance scheme for solid material fluxes. The parameters used to describe this case are given in Table D-2 and the corresponding water balance scheme is shown in Figure D-1.
E. Definition of the TAME exposure pathway sub-models

E.1 Exposure pathway sub-models for waste disposal assessments

Equation (13) in Section 2.3.3 gives the general form for the derivation of the doses received by individuals exposed to radionuclides in environmental media as:

\[ D_p^{(N)}(t) = \sum_{i,\text{exp}} E_p H_{\text{exp}}^{(N)} P_{p,i} N_i(t) \text{ [Sv y}^{-1}] \]  \hspace{1cm} (E.1)

In order to be used in practical assessments, the various parameters described in Section 2.3.3 need to be identified with features of the modelled biosphere in the context of the assessment.

The assessment context plays a key role in determining the form of the exposure pathway sub-model. In particular the definition of the critical group for the assessment identifies the potential exposure pathways as well as the range of exposure rates to be considered. Interpretation of the Swiss federal regulations (HSK/KSA, 1993) leads to identification of a small self-sufficient agricultural community as the critical group for whom it should be demonstrated that exposures do not breach the regulatory guidelines. This means that the exposure pathway sub-model in TAME consists of two distinct parts:

- expressions for the calculation of radiation doses following exposure to contaminated foodstuffs or environmental media,
- derivation of the exposure rates for each of the foodstuffs.

In TAME, the first of these is taken from the MiniBIOS model developed by the UK National Radiological Protection Board (MARTIN et al., 1991) and the second is based on an extension of the formulation of the critical group consumption rates used in the PSACOIN Level Ib exercise (KLOS et al., 1990; NEA, 1993), with data optimised for the Swiss biosphere database (JISKRA, 1985).

The expressions used in the model are given below in detail. This serves three purposes. Firstly, it is a reference point for the exposure pathway sub-model used in TAME. Secondly, it documents improvements to consistency implemented when the MiniBIOS model was adopted for the framework of the TAME transport sub-model and thirdly, the situations where the existing Swiss database - e.g. Projekt Gewähr (NAGRA, 1985) - has influenced the parameterisation of the present implementation can be seen.
E.2 Expressions for calculating radiological doses by exposure pathway

E.2.1 Concentrations in the TAME compartments

In the following definitions of the exposure pathways, it is convenient to express part of the processing factor in terms of the concentrations of radionuclides in each of the TAME compartments. The concentrations as used here are simply defined by the inventory divided by the physical compartment volume (see Table 2.4), so that for the local aquifer, the top soil and the surface water the physical concentrations are, respectively

\[ C_L \equiv \frac{N_L}{L_A f}, \quad C_T \equiv \frac{N_T}{T_A f}, \quad C_W \equiv \frac{N_W}{W_W d_W} \quad [\text{Bq m}^{-3}]. \quad (E.2) \]

The concentrations in solution in the compartments can then be derived from these as part of the processing factor. For example, the radionuclide concentration in the well water is finally given by

\[ C_{\text{drinking water}} = \frac{1 + (1-f_f i l t e r) \alpha_L k_L}{\theta_L + (1-\varepsilon_L)\rho_L k_L} C_L \quad [\text{Bq m}^{-3}]. \quad (E.3) \]

where the bracket in the numerator contains a factor quantifying whether the water is filtered to remove the suspended sediment or not (0 ≤ f_f i l t e r ≤ 1 gives the fraction of solid material removed). Radionuclides are sorbed onto the solid material both in the rock matrix and in the suspended solid particles according to the solid - liquid distribution coefficient in the compartment \( k_L \; ([\text{Bq kg}^{-1}](\text{Bq m}^{-3})) \). The denominator takes into account the total volume of porewater in the local aquifer, as well as sorption onto the rock matrix (see Appendices A and B).

E.2.2 Drinking-water consumption

Inhabitants of the region may take drinking-water either from the local aquifer (via a well) or from the surface water body. TAME models both possibilities in the following way.

The dose from consumption of water from the surface water body in the TAME representation of the biosphere is

\[ D_{\text{wat}} = H_{\text{ing}} I_{\text{wat}} (1-f_{\text{well}}) \frac{1 + (1-f_{\text{filter}}) \alpha_w k_w}{1 + \alpha_w k_w} C_w \quad \text{[Sv y}^{-1}]. \quad (E.4) \]
and the dose from the consumption of water from the local aquifer is

\[
D_{\text{well}} = H_{\text{ing}} I_{\text{wat}} f_{\text{well}} \frac{1 + (1 - f_{\text{filter}}) \alpha_{L} k_{L}}{\theta_{L} + (1 - \varepsilon_{L}) \theta_{L}} C_{L} \quad [\text{Sv y}^{-1}].
\]

(E.5)

In these expressions, the subscripts \(W\) and \(L\) refer to surface water and local aquifer. The other parameters are:

- \(\alpha_{W} [\text{kg m}^{-3}]\) is the suspended solid load in the water
- \(\alpha_{L} [\text{kg m}^{-3}]\) is the suspended solid load in the aquifer porewater
- \(k_{W} [(\text{Bq kg}^{-1}) (\text{Bq m}^{-3})^{-1}]\) is the nuclide solid - liquid distribution coefficient in the water compartment
- \(k_{L} [(\text{Bq kg}^{-1}) (\text{Bq m}^{-3})^{-1}]\) is the nuclide solid - liquid distribution coefficient in the local aquifer
- \(\theta_{L}\) and \(\varepsilon_{L} [-]\) are the volumetric moisture content and the porosity of the local aquifer
- \(f_{\text{filter}} [-]\) is a fraction between 0 and 1 which determines the amount of suspended solid material filtered out of the water before consumption,
- \(f_{\text{well}} [-]\) is the fraction of drinking-water obtained from a well in the local aquifer
- \(I_{\text{wat}} [\text{m}^{3} \text{y}^{-1}]\) is the exposure factor, the total annual intake of drinking-water by each of the two pathways
- \(H_{\text{ing}} [\text{Sv Bq}^{-1}]\) is the dose per unit intake on ingestion

### E.2.3 Fish consumption

In the MiniBIOS model, the equilibrium concentration in fish tissue is dependent on the filtered water concentration in the surface water compartment. The annual individual dose is

\[
D_{ff} = H_{\text{ing}} I_{ff} \frac{K_{ff} C_{W}}{(1 + \alpha_{W} k_{W})} \quad [\text{Sv y}^{-1}],
\]

(E.6)

where the processing factor includes the concentration in the surface water body used to calculate the drinking-water dose (allowing for filtering out of the radionuclides on the suspended solids in the water) and also the fish concentration factor, \(K_{ff} [(\text{Bq kg}^{-1}) (\text{Bq m}^{-3})^{-1}]\). The exposure factor \((I_{ff} [\text{kg y}^{-1}])\) for the intake of fish and the dose per unit intake are straightforward.
E.2.4 Grain consumption

Grain consumed by humans may be contaminated in three ways, namely internally by root uptake and by irrigation and externally by small amounts of soil adhering to the crop. The expression for the dose via this pathway is therefore

\[
D_{\text{gr}} = H_{\text{ing}} I_{\text{gr}} \left\{ C_{\text{root, uptake}}^{\text{gr}} + C_{\text{water-irrigation}}^{\text{gr}} + C_{\text{surface contamination}}^{\text{gr}} \right\} \text{[Sv y}^{-1}].
\]  
(E.7)

\( I_{\text{gr}} \text{[kg y}^{-1}] \) is the annual consumption of grain by the exposed individual and the three mechanisms giving rise to the grain contamination are:

Root uptake:

\[
C_{\text{root, uptake}}^{\text{gr}} = K_{\text{gr}} \frac{C_r}{\rho_r (1 - e_r)} \text{[Bq kg}^{-1}].
\]  
(E.8)

The concentration in the grain is derived from the concentration of the radionuclide in the dry soil using \( K_{\text{gr}} \text{[(Bq kg}^{-1}, \text{grain, fresh weight})(\text{Bq kg}^{-1}, \text{soil, dry weight})] \), the soil-grain transfer factor.

Interception of contaminated irrigation water:

\[
C_{\text{water-irrigation}}^{\text{gr}} = f_{\text{gr}} \left\{ 1 - \exp \left( -\mu_{\text{gr}} Y_{\text{gr}} \right) \right\} \left( \frac{F_{LT} \rho_r (1 - e_r) \mu_{\text{gr}}}{Y_{\text{gr}} (W_{\text{gr}} + H_{\text{gr}})} \frac{C_L + F_{WT} C_W}{A_f} \right) \text{[Bq kg}^{-1}].
\]  
(E.9)

\( f_{\text{gr}} \) is the food processing factor for grain, allowing for removal during preparation and cooking etc. The first bracket represents the interception of contaminated water by the crop leaves:

\begin{align*}
Y_{\text{gr}} \text{[kg m}^{-2}] & \quad \text{is the yield of grain} \\
\mu_{\text{gr}} \text{[m}^2 \text{kg}^{-1}] & \quad \text{is the irrigation mass-interception factor for grain} \\
W_{\text{gr}} \text{[y}^{-1}] & \quad \text{accounts for the removal of radionuclides from the external surfaces of the crop by weathering (the loss term is implicitly to the soil)} \end{align*}

and

\begin{align*}
H_{\text{gr}} \text{[y}^{-1}] & \quad \text{accounts for the removal of activity by harvesting, again the removed radionuclide is implicitly transferred to the soil in the transfer model. This is a consequence of assuming a closed agricultural system.} \\
F_{LT} \text{[m}^3 \text{y}^{-1}] & \quad \text{accounts for the removal of activity by harvesting, again the removed radionuclide is implicitly transferred to the soil in the transfer model. This is a consequence of assuming a closed agricultural system.} \\
F_{WT} \text{[m}^3 \text{y}^{-1}] & \quad \text{accounts for the removal of activity by harvesting, again the removed radionuclide is implicitly transferred to the soil in the transfer model. This is a consequence of assuming a closed agricultural system.} \end{align*}

The second bracket in Equation (E.9) gives the overall concentration of the radionuclide in the water used for irrigation. In the TAME transport model, this water comes potentially from the surface water (flux \( F_{WT} \equiv d_{WT} A_f \text{[m}^3 \text{y}^{-1}] \)) or a well in the local aquifer (flux \( F_{LT} \equiv d_{LT} A_f \text{[m}^3 \text{y}^{-1}] \)).
N.B. the translocation of contaminants on the external surfaces of the leaves to the edible portions of the crop is implicitly included in the interception factor. This is the recommendation of SIMMONDS & CRICK (1982) and clearly this representation of the process requires that the input data correspond to this assumption.

Surface contamination:

\[
C_{\text{surface contamination}}^{sv} = f_{sv} S_{sv} \frac{C_T}{\rho_T (1 - \varepsilon_T) + \varepsilon_T \rho_W} \quad \text{[Bq kg}^{-1}] \text{]. (E.10)}
\]

A fraction of the weight of the consumed crop is made up of wet soil on the external surfaces of the grain. The fraction is denoted by \(S_{sv}\). Again, only a fraction of the total contamination remains after food processing.

It is important to note here that, in some datasets, the soil - plant transfer factor takes this process into account. This representation might therefore introduce conservatism into the calculations since the intake via this mechanism is being counted twice.

E.2.5 Green vegetables consumption

The doses from the consumption of green vegetables are calculated assuming the same mechanisms as for the doses from grain consumption:

\[
D_{sv} = H_{\text{ing}} I_{sv} \left\{ C_{\text{root uptake}}^{sv} + C_{\text{water-irrigation}}^{sv} + C_{\text{surface contamination}}^{sv} \right\} \quad \text{[Sv y}^{-1}] \text{]. (E.11)}
\]

Root uptake:

\[
C_{\text{root uptake}}^{sv} = K_{sv} \frac{C_T}{\rho_T (1 - \varepsilon_T)} \quad \text{[Bq kg}^{-1}] \text{]. (E.12)}
\]

Interception of contaminated irrigation water:

\[
C_{\text{water-irrigation}}^{sv} = f_{sv} \left( \frac{1 - \exp(-\mu_{sv} Y_{sv})}{Y_{sv} (W_{sv} + H_{sv})} \right) \left( F_{LT} \delta_{sv} A_{sv} \frac{C_L + F_{WT} C_W}{A_f} \right) \quad \text{[Bq kg}^{-1}] \text{]. (E.13)}
\]

Surface contamination:

\[
C_{\text{surface contamination}}^{sv} = f_{sv} S_{sv} \frac{C_T}{\rho_T (1 - \varepsilon_T) + \varepsilon_T \rho_W} \quad \text{[Bq kg}^{-1}] \text{]. (E.14)}
\]
All the symbols have equivalent meanings to the case of grain and similar comments apply to the use of these formulæ.

E.2.6 Root vegetables consumption

Doses from this pathway are similarly given by root uptake, irrigation interception and surface contamination

\[ D_{rv} = H_{eq} I_{rv} \left\{ C_{\text{root uptake}}^{rv} + C_{\text{irrigation}}^{rv} + C_{\text{surface contamination}}^{rv} \right\} \quad [\text{Sv y}^{-1}]. \quad (E.15) \]

Expressions for root uptake and surface contamination are as used above.

Root uptake:

\[ C_{\text{root uptake}}^{rv} = K_{sv} \frac{C_T}{\rho_T (1 - \varepsilon_T)} \quad [\text{Bq kg}^{-1}]. \quad (E.16) \]

Surface contamination:

\[ C_{\text{surface contamination}}^{rv} = f_{rv} S_{rv} \frac{C_T}{\rho_T (1 - \varepsilon_T) + \varepsilon_T \rho_w} \quad [\text{Bq kg}^{-1}]. \quad (E.17) \]

Interception of contaminated irrigation water:

In the case of irrigation, the radionuclide concentration in the edible portion of the crop explicitly includes an additional process - translocation (SIMMONDS & CRICK, 1982). Activity entering the crop via the leaves is transported to the edible portion so that the concentration in the root via this process is given by

\[ C_{\text{water - irrigation}}^{rv} = \left( f_{rv} + \frac{T_{rv}}{H_{rv}} \frac{1 - \exp(-\mu_{rv} Y_{rv})}{Y_{rv} (W_{rv} + H_{rv} + T_{rv})} \right) \left( \frac{F_{LT} \frac{1}{\theta_{L} + [1 - \varepsilon_{L}] \rho_{L} k_{L}} C_{L} + F_{WT} C_{W}}{A_{f}} \right) \quad [\text{Bq kg}^{-1}]. \quad (E.18) \]

where \( T_{rv} \ [\text{y}^{-1}] \) is the translocation rate, giving the transfer of intercepted radionuclides to the internal, edible portions of the crop.
E.2.7 Meat consumption

Four mechanisms make up this pathway:

- water - livestock - man
- water - pasture irrigation - livestock - man
- soil - pasture - livestock - man
- soil - livestock - man

so that the dose from meat consumption is given by

$$D_{meat} = H_{ing} I_{meat} K_{meat} \left( I_{\text{livestock - water}} + I_{\text{livestock - irrigation - pasture}} + I_{\text{livestock - soil}} + I_{\text{meat}} \right) [\text{Sv y}^{-1}],$$

(E.19)

where again the dose per unit intake is clearly identified and the annual intake of meat $I_{meat}$ [kg y$^{-1}$] gives the exposure factor. The processing factor in the brackets is complex because of the four parallel mechanisms by which meat can become contaminated. The factor $K_{meat}$ [day kg$^{-1}$] is the concentration factor for the radionuclide in animal tissue. It models the transfer of the daily intake of the radionuclide to the tissue. The daily intake rates are given by the following expressions:

**Livestock intake in drinking-water:**

$$I_{\text{water}} = I_{wc} \left[ f_A \frac{1}{\theta_L + (1-\epsilon_L)\rho_L k_L} C_L + (1-f_A)C_W \right] [\text{Bq day}^{-1}],$$

(E.20)

where $I_{wc}$ [m$^3$ day$^{-1}$] is the daily water consumption by the animal

and $f_A$ [-] is the fraction of that water obtained from the well.

**Livestock consumption of pasture contaminated by irrigation:**

$$I_{\text{livestock - irrigation - pasture}} = Z I_{pc} \left( 1 - \exp\left( -\frac{\mu_p}{Y_p} \right) \right) \left( \frac{F_{LT} \theta_2 + (1-\epsilon_L)\rho_L k_L C_L + F_{WT} C_W}{\lambda_f} \right) [\text{Bq day}^{-1}],$$

(E.21)

where $Z$ [-] is the ratio by weight of fresh pasture to hay

and $I_{pc}$ [kg day$^{-1}$] is daily consumption of dry fodder by the animal.

The interception mechanism is modelled in the same way as for the other crop types. The removal by *harvesting* in this case is not so simple since the rate $H_{pc}$ [y$^{-1}$], which accounts for the removal of activity in growing plants as a result of cropping by livestock, is a continuous process depending on the stocking density of the livestock:

$$H_{pc} = n_d Z I_{pc} \frac{N_L}{Y_p} [y^{-1}],$$

(E.22)
where the daily intake of pasture is converted to an annual value using the number of days per year, $n_d$.

Translocation is not considered for this crop (SIMMONDS & CRICK, 1982).

Livestock consumption of pasture contaminated by root uptake:

$$I_{\text{soil-pasture}}^\text{livestock} = I_{\text{pc}} Z K_p C_T \left(\frac{1}{1 - \varepsilon_T} \right) \rho_T \left(1 - \varepsilon_T\right) \rho_T$$

[Bq day$^{-1}$].

(E.23)

The concentration in the pasture is derived from the concentration in the top soil using $K_p$ [(Bq kg$^{-1}$ crop, fresh weight) (Bq kg$^{-1}$ soil, dry weight)$^{-1}$], the soil-pasture transfer factor.

Livestock ingestion of soil during grazing:

$$I_{\text{soil}}^\text{livestock} = S_{pc} Z I_{pc} C_T \left(\frac{1}{1 - \varepsilon_T} \right) \rho_T + \varepsilon_T \rho_W$$

[Bq day$^{-1}$].

(E.24)

In this form, $S_{pc}$ is the fraction of the weight of pasture made up of wet soil.

E.2.8 Milk consumption

The form of the expression for the dose via milk consumption is almost identical to that of the meat consumption pathway, since the intake by livestock is by the same mechanisms as those described in Equations (E.19) to (E.24). The annual individual dose for this pathway is therefore given by

$$D_{\text{milk}} = H_{\text{mg}} I_{\text{milk}} \rho_W K_{\text{milk}} \left\{ I_{\text{livestock-water}} + I_{\text{livestock-water-irrigation-pasture}} + I_{\text{livestock-pasture}} + I_{\text{livestock-soil}} \right\} \left[\text{Sv y}^{-1}\right].$$

(E.25)

Here, $K_{\text{milk}}$ [(Bq kg$^{-1}$)(Bq day$^{-1}$)$^{-1}$] is the distribution factor for milk and $I_{\text{milk}}$ [m$^3$ y$^{-1}$] is the annual consumption of milk. The density of milk is taken to be equivalent to that of water.
E.2.9 Egg consumption

The dose from eggs is calculated in a similar way to the meat and milk doses, and the annual individual dose is given by the four following mechanisms by which activity can enter poultry

\[ D_{\text{eggs}} = H_{\text{egg}} I_{\text{egg}} K_{\text{egg}} \left[ I_{\text{poultry water}} + I_{\text{poultry water--grain}} + I_{\text{poultry soil}} \right] \text{[Sv y}^{-1}] \]  \hspace{1cm} (E.26)

\( K_{\text{egg}} \) [(Bq egg\(^{-1}\))(Bq day\(^{-1}\))\(^{-1}\)] is the distribution factor for the radionuclide in eggs and \( I_{\text{egg}} \) is the annual consumption rate of eggs. The equations below show the additional parameters required to model this pathway. Use is made of the expressions for the accumulation in grain described in Section E.2.4.

**Poultry intake in drinking-water:**

\[ I_{\text{water}} = I_{\text{wp}} \left[ f_{\text{poultry}} \frac{1}{\theta_L + (1 - \epsilon_L) \rho_L k_L} C_L + (1 - f_{\text{poultry}}) C_W \right] \text{Bq day}^{-1}. \]  \hspace{1cm} (E.27)

As with cattle and humans, the drinking-water for poultry can be obtained from both well and surface water. The fraction from the well is \( f_{\text{poultry}} \) and \( I_{\text{wp}} \) [m\(^3\) day\(^{-1}\)] is the daily water consumption of poultry.

**Poultry consumption of grain contaminated by irrigation:**

Using Equation (E.9), the poultry intake via this mechanism is given by

\[ I_{\text{poultry water--grain}} = I_{\text{gp}} \left( 1 - \exp \left( - \mu_L Y_{gr} \right) \right) \left[ \frac{F_L \tau (1 - \epsilon_L) \rho_L k_L C_L + F_W C_W}{A_f} \right] \text{Bq day}^{-1}. \]  \hspace{1cm} (E.28)

where the daily consumption rate of grain by the poultry is \( I_{\text{gp}} \) [kg day\(^{-1}\)].

**Poultry consumption of grain contaminated by root uptake:**

This is given using Equation (E.8) and the daily poultry grain consumption:

\[ I_{\text{poultry soil--grain}} = I_{\text{gp}} K_{gr} \frac{C_r}{\rho_T (1 - \epsilon_T)} \text{Bq kg}^{-1}. \]  \hspace{1cm} (E.29)

Similarly the expression for intake by direct ingestion of soil uses Equation (E.10).
Poultry ingestion of soil during feeding:

\[ I_{\text{poultry}} = I_p S_g \frac{C_T}{\rho_T (1 - \varepsilon_T) + \varepsilon_T \rho_w} \text{[Bq kg}^{-1}] \]  \hspace{1cm} (E.30)

E.2.10 External $\gamma$-irradiation

A semi-infinite plane of uniformly contaminated top soil is assumed and the annual dose rate is given by

\[ D_{\text{ext}} = GC_T \text{[Sv y}^{-1}] \]  \hspace{1cm} (E.31)

where the exposure factor in this case is the groundshine factor, \( G \) \([\text{Sv y}^{-1}\text{Bq m}^{-3}]\), which takes into account the $\gamma$-ray energy, the self-absorption factors for the soils and human tissue (as a function of emission energy) as well as the annual individual occupancy rate of the contaminated area. This latter factor assumes that an individual spends one year per year in the contaminated region so as to maximise the exposure by this pathway.

E.2.11 Inhalation Doses

The inhalation of radionuclides is potentially very important because of the high values for \( H_{\text{inh}} \text{[Sv Bq}^{-1}] \) the dose per unit intake on inhalation, particularly for the actinides. Radionuclides can be inhaled either in their gaseous form following degassing from the soil or on suspended soil particles which have concentrations determined by the sorption in the top soil. The dose from inhalation is therefore given by

\[ D_{\text{dust}} = H_{\text{inh}} I_{\text{air}} \left( O_a a_r + O_f a_f \right) \frac{C_T}{(1 - \varepsilon_T) \rho_T} + \lambda_{\text{at}} \frac{I_T I_W}{v_A l_A} C_T \text{[Sv y}^{-1}] \]  \hspace{1cm} (E.32)

where the volume of air inhaled by the exposed individual annually is \( I_{\text{air}} \text{[m}^3\text{y}^{-1}] \).

The first term in the square brackets gives the airborne radionuclide concentration due to the airborne dust loading. Two situations are identified - background dust concentrations \( (a_r \text{[kg m}^{-3}]) \) relevant to normal occupations and occupational dust concentrations \( (a_f \text{[kg m}^{-3}]) \), i.e. those higher than normal. These arise during ploughing and other occupation-related activities. The two occupancy rates, \( O_a \text{ and } O_f \text{[years per year]} \) are assumed to be exclusive and the total occupancy of the contaminated region is conservatively assumed to be one year per year.
The second term in the brackets gives the airborne concentration of radionuclides in the air passing over the modelled region, with the atmosphere represented by a compartment with thickness \( l_A \) [m] and an annual windspeed of \( v_A \) [m \( y \)^{-1}]. \( \lambda_{dg} \) [y^{-1}] is the degassing coefficient from the top soil. The direction of the wind is conservatively assumed to be in the direction of the downstream surface water section and the remaining compartment dimensions in Equation (E.32) reflect this (see Table 2.4). Table E.1 summarises the parameters used in TAME to model the doses arising from the individual pathways.

### E.3 The TAME exposure factors for food consumption

#### E.3.1 Parameterisation of the consumption rates

All the exposure pathways outlined above are combined, according to Equation (E.1), to give the total dose arising from the concentration of the radionuclide in the biosphere. This requires that the exposure factors be defined to give a consistent representation of the habits and behaviour of an individual member of the modelled community. The \( E_p \) factors here are the annual food consumption rates. The JISKRA (1985) database was based on the current consumption rates for Switzerland and correspond to an annual intake of 3000 [kcal day^{-1}]. These data could be used directly but there is no flexibility in imposing a fixed distribution of food consumption.

As discussed above, the exposure factors in TAME are chosen to represent a closed, self-sufficient agricultural community, in accordance with the TAME interpretation of the Swiss federal guidelines (HSK/KSA, 1993). All foodstuffs are therefore produced locally in the modelled region so that the dose calculated in the model is not reduced by the consumption of uncontaminated foodstuffs obtained from outside the modelled region. These features need to be included in the TAME expressions.

The approach taken is to start from a fixed total annual energy intake from food consumption and to distribute the consumption rates among the foodstuffs in the model (cf. NEA, 1993) so as to give the required total annual energy intake \( E_0 \) [kJ \( y \)^{-1}]. A more complete analysis of nutritional requirements could be carried out, but the system used in TAME has the advantage of simplicity and ease of use, particularly when alternative diets - corresponding to alternative lifestyles in, say, different climate states - are to be considered. The application of, and results from, a similar approach to that used in TAME are discussed by KLOS et al. (1990). Similarly a range of behaviour in the critical group could also be assessed by varying the amounts of different foodstuffs consumed. This is a possible approach for a stochastic assessment (where the total food consumption would also be varied). One disadvantage is that the relationships between the pathways are strictly determined and this necessarily favours a reduced number of exposure pathways. However the approach is internally consistent and also suitable for
use with the regulatory guidelines. In stochastic analyses (NEA, 1993), variable consumption rates are not seen to have a significant effect on the uncertainty in the dose.

Each of the ingestion rates used in the above expressions is then defined in terms of the fractional contributions to this total:

\[ E_0 = \sum_{\text{food pathways}} \eta_f I_f \text{ [kJ y}^{-1}] \]  

(E.33)

where \( \eta_f \) [kJ kg\(^{-1}\), [kJ m\(^3\)] or [kJ per item] is the energy content of the foodstuff and \( I_f \) [kg y\(^{-1}\)], [m\(^3\) y\(^{-1}\)] or [item y\(^{-1}\)] is the consumption rate of the foodstuff.

The fractional consumption rates used in TAME are then given by

\[ f_f = \frac{\eta_f I_f}{E_0} [-] \]  

(E.34)

The relationship between these can be quite complex. The model is set up to allow any combination of foodstuff consumption rates according to a set of simple rules. These are used to identify the main component of foodstuffs in the model biosphere.

Starting with the consumption of eggs:

\[ I_{\text{eggs}} = E_0 \frac{f_{\text{eggs}}}{\eta_{\text{eggs}}} \text{ [eggs y}^{-1}] \]  

(E.35)

where the energy content per egg is \( \eta_{\text{eggs}} \) [kJ egg\(^{-1}\)], the consumption rates of freshwater fish and milk are similarly determined by

\[ I_{\text{ff}} = E_0 \frac{f_{\text{ff}}}{\eta_{\text{ff}}} \text{ [kg y}^{-1}] \]  

(E.36)

and

\[ I_{\text{milk}} = E_0 \frac{f_{\text{milk}}}{\eta_{\text{milk}}} \text{ [m}^3\text{ y}^{-1}] \]  

(E.37)

where \( \eta_{\text{ff}} \) [kJ kg\(^{-1}\)] and \( \eta_{\text{milk}} \) [kJ m\(^3\)] are the energy contents of freshwater fish and milk respectively. These equations define the water intake and the total energy intake by vegetable consumption. Respectively, these are

\[ I_{\text{wat}} = I_{\text{fluid}} - I_{\text{milk}} \text{ [m}^3\text{ y}^{-1}] \]  

(E.38)

and

\[ \eta_{\text{gr}} I_{\text{gr}} + \eta_{\text{rv}} I_{\text{rv}} + \eta_{\text{gr}} I_{\text{gr}} = p_{\text{veg}} E_0 (1 - f_{\text{eggs}} - f_{\text{ff}} - f_{\text{milk}}) \text{ [kJ y}^{-1}] \]  

(E.39)
Table E.1: Summary of parameters used in the TAME representation of the individual exposure pathways. The parameters and symbols identified here are specifically those in the exposure pathway model which were not required for the definition of the transport model.

<table>
<thead>
<tr>
<th>category</th>
<th>parameter</th>
<th>symbol</th>
<th>units</th>
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<td>radiological</td>
<td>dose per unit intake on ingestion</td>
<td>$H_{ing}$</td>
<td>Sv Bq$^{-1}$</td>
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<tr>
<td></td>
<td>dose per unit intake on inhalation</td>
<td>$H_{inh}$</td>
<td>Sv Bq$^{-1}$</td>
</tr>
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<td>water</td>
<td>$I_{wat}$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
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<td></td>
<td>fish</td>
<td>$I_f$</td>
<td>kg y$^{-1}$</td>
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<tr>
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<td>grain</td>
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<td>kg y$^{-1}$</td>
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<td>kg y$^{-1}$</td>
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<tr>
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<td>$I_{milk}$</td>
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<td></td>
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<td>$I_{eggs}$</td>
<td>eggs y$^{-1}$</td>
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<td>poultry daily grain consumption</td>
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<td>local aquifer</td>
<td>$C_L$</td>
<td>Bq m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>deep soil</td>
<td>$C_D$</td>
<td>Bq m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>top soil</td>
<td>$C_T$</td>
<td>Bq m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>surface water</td>
<td>$C_{sw}$</td>
<td>Bq m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>aquatic sediment</td>
<td>$C_S$</td>
<td>Bq m$^{-3}$</td>
</tr>
<tr>
<td>drinking-water (well and surface)</td>
<td>drinking-water fraction taken from well</td>
<td>$f_{well}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>fraction of water consumption filtered prior to consumption</td>
<td>$f_{filter}$</td>
<td>-</td>
</tr>
<tr>
<td>fish</td>
<td>fish concentration factor</td>
<td>$K_f$</td>
<td>(Bq kg$^{-1}$) (Bq m$^{-3}$ fresh weight)$^{-1}$</td>
</tr>
<tr>
<td>grain</td>
<td>yield</td>
<td>$Y_{gr}$</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td></td>
<td>harvesting rate</td>
<td>$H_{gr}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>weathering loss rate</td>
<td>$W_{gr}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>spray irrigation interception factor</td>
<td>$\mu_{gr}$</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>food processing factor</td>
<td>$f_r$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>surface contamination factor</td>
<td>$S_{gr}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>soil - grain transfer factor</td>
<td>$K_{gr}$</td>
<td>(Bq kg$^{-1}$ fresh weight) (Bq m$^{-3}$ dry soil)$^{-1}$</td>
</tr>
</tbody>
</table>
Table E.1: Summary of parameters used in the TAME representation of the individual exposure pathways. The parameters and symbols identified here are specifically those in the exposure pathway model which were not required for the definition of the transport model (continued).

<table>
<thead>
<tr>
<th>category</th>
<th>parameter</th>
<th>symbol</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>green vegetables</td>
<td>yield</td>
<td>$Y_{gv}$</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td></td>
<td>harvesting rate</td>
<td>$H_{gv}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>weathering loss rate</td>
<td>$W_{gv}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>spray irrigation interception factor</td>
<td>$\mu_{gv}$</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>food processing factor</td>
<td>$f_{gv}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>surface contamination factor</td>
<td>$S_{gv}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>soil - plant transfer factor</td>
<td>$K_{pv}$</td>
<td>(Bq kg$^{-1}$ fresh weight)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(Bq m$^{-3}$ dry soil)$^{-1}$</td>
</tr>
<tr>
<td>root vegetables</td>
<td>yield</td>
<td>$Y_{rv}$</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td></td>
<td>harvesting rate</td>
<td>$H_{rv}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>weathering loss rate</td>
<td>$W_{rv}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>spray irrigation interception factor</td>
<td>$\mu_{rv}$</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>food processing factor</td>
<td>$f_{rv}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>surface contamination factor</td>
<td>$S_{rv}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>translocation factor (external $\rightarrow$ internal)</td>
<td>$T_{rv}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>soil - plant transfer factor</td>
<td>$K_{rv}$</td>
<td>(Bq kg$^{-1}$ fresh weight)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(Bq m$^{-3}$ dry soil)$^{-1}$</td>
</tr>
<tr>
<td>pasture</td>
<td>yield</td>
<td>$Y_{p}$</td>
<td>kg m$^{-2}$</td>
</tr>
<tr>
<td></td>
<td>harvesting rate</td>
<td>$H_{pc}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>number of days per year</td>
<td>$n_d$</td>
<td>days y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>weathering loss rate</td>
<td>$W_{p}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>spray irrigation interception factor</td>
<td>$\mu_{p}$</td>
<td>m$^2$ kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>surface contamination factor</td>
<td>$S_{pc}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>ratio of fresh weight to dry weight</td>
<td>$Z$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>soil - plant transfer factor</td>
<td>$K_{p}$</td>
<td>(Bq kg$^{-1}$ fresh weight)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(Bq m$^{-3}$ dry soil)$^{-1}$</td>
</tr>
<tr>
<td>meat, milk and poultry</td>
<td>fraction cattle drinking-water from well</td>
<td>$f_A$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>fraction poultry drinking-water from well</td>
<td>$f_{poultry}$</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>meat (cattle) concentration factor</td>
<td>$K_{meat}$</td>
<td>(Bq kg$^{-1}$)(Bq day$^{-1}$)$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>milk concentration factor</td>
<td>$K_{milk}$</td>
<td>(Bq m$^{-3}$)(Bq day$^{-1}$)$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>egg concentration factor</td>
<td>$K_{eggs}$</td>
<td>(Bq egg$^{-1}$)(Bq day$^{-1}$)$^{-1}$</td>
</tr>
<tr>
<td>external $\gamma$-irradiation</td>
<td>semi-infinite plane groundshine factor corrected for self-absorption in tissue.</td>
<td>$G$</td>
<td>(Sv y$^{-1}$)(Bq m$^{-3}$)$^{-1}$</td>
</tr>
<tr>
<td>dust inhalation</td>
<td>airborne dust load (residential)</td>
<td>$a_r$</td>
<td>kg m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>airborne dust load (working)</td>
<td>$a_f$</td>
<td>kg m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>residence factor (residential)</td>
<td>$O_r$</td>
<td>years per year</td>
</tr>
<tr>
<td></td>
<td>residence factor (working)</td>
<td>$O_f$</td>
<td>years per year</td>
</tr>
<tr>
<td></td>
<td>volatile element degassing parameter</td>
<td>$\lambda_{dg}$</td>
<td>y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>mean annual wind speed</td>
<td>$v_A$</td>
<td>m y$^{-1}$</td>
</tr>
<tr>
<td></td>
<td>thickness of atmosphere affected by degassing and associated with inhalation</td>
<td>$l_k$</td>
<td>m</td>
</tr>
</tbody>
</table>
Table E.2: Parameters used in the definition of the consumption rates appropriate for adult humans in the TAME representation of the biosphere. These parameters are used to define a consistent set of exposure and consumption rates for the exposure pathways given in Section 2.3.4.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Symbol</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total annual food energy intake. Adult member of the critical group</td>
<td>$E_0$</td>
<td>kJ y$^{-1}$</td>
</tr>
<tr>
<td>Total annual liquid intake. Adult member of the critical group</td>
<td>$I_{\text{fluid}}$</td>
<td>m$^3$ y$^{-1}$</td>
</tr>
<tr>
<td>Food energy content of eggs</td>
<td>$\eta_{\text{eggs}}$</td>
<td>kJ egg$^{-1}$</td>
</tr>
<tr>
<td>Food energy content of milk</td>
<td>$\eta_{\text{milk}}$</td>
<td>kJ m$^{-3}$</td>
</tr>
<tr>
<td>Food energy content of meat</td>
<td>$\eta_{\text{meat}}$</td>
<td>kJ kg$^{-1}$</td>
</tr>
<tr>
<td>Food energy content of fish</td>
<td>$\eta_{f}$</td>
<td>kJ kg$^{-1}$</td>
</tr>
<tr>
<td>Food energy content of grain</td>
<td>$\eta_{gr}$</td>
<td>kJ kg$^{-1}$</td>
</tr>
<tr>
<td>Food energy content of root vegetables</td>
<td>$\eta_{rv}$</td>
<td>kJ kg$^{-1}$</td>
</tr>
<tr>
<td>Food energy content of green vegetables</td>
<td>$\eta_{gv}$</td>
<td>kJ kg$^{-1}$</td>
</tr>
<tr>
<td>Fraction of $E_0$ derived from egg consumption</td>
<td>$f_{\text{eggs}}$</td>
<td>-</td>
</tr>
<tr>
<td>Fraction of $E_0$ derived from meat consumption</td>
<td>$f_{\text{meat}}$</td>
<td>-</td>
</tr>
<tr>
<td>Fraction of $E_0$ derived from milk consumption</td>
<td>$f_{\text{milk}}$</td>
<td>-</td>
</tr>
<tr>
<td>Fraction of $E_0$ derived from freshwater fish consumption</td>
<td>$f_{f}$</td>
<td>-</td>
</tr>
<tr>
<td>Fraction of $E_0$ derived from consumption of grain, root vegetables and green vegetables (after milk, fish and egg consumption is taken into account)</td>
<td>$p_{\text{veg}}$</td>
<td>-</td>
</tr>
<tr>
<td>Proportion of energy from vegetable consumption taken as grain</td>
<td>$p_{gr}$</td>
<td>-</td>
</tr>
<tr>
<td>Proportion of energy from vegetable consumption taken as root vegetables</td>
<td>$p_{rv}$</td>
<td>-</td>
</tr>
<tr>
<td>Proportion of energy from vegetable consumption taken as green vegetables</td>
<td>$p_{gv}$</td>
<td>-</td>
</tr>
</tbody>
</table>
Table E.3: The TAME annual consumption rates for a daily calorific intake of 3000 kcal ($4.6 \times 10^6$ kJ y$^{-1}$). The calorific intake by foodstuff are illustrated in Figure E-1.

<table>
<thead>
<tr>
<th>foodstuff</th>
<th>consumption rate</th>
<th>rate unit</th>
<th>rate value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eggs $I_{egg}$</td>
<td></td>
<td>eggs y$^{-1}$</td>
<td>200</td>
</tr>
<tr>
<td>Milk $I_{milk}$</td>
<td></td>
<td>m$^3$ y$^{-1}$</td>
<td>0.332</td>
</tr>
<tr>
<td>Water $I_W$</td>
<td></td>
<td>m$^3$ y$^{-1}$</td>
<td>0.73</td>
</tr>
<tr>
<td>Fish $I_f$</td>
<td></td>
<td>kg y$^{-1}$</td>
<td>2</td>
</tr>
<tr>
<td>Grain $I_{gr}$</td>
<td></td>
<td>kg y$^{-1}$</td>
<td>145</td>
</tr>
<tr>
<td>Root vegetables</td>
<td></td>
<td>kg y$^{-1}$</td>
<td>231</td>
</tr>
<tr>
<td>Green vegetables'</td>
<td></td>
<td>kg y$^{-1}$</td>
<td>61.0</td>
</tr>
<tr>
<td>Meat $I_{meat}$</td>
<td></td>
<td>kg y$^{-1}$</td>
<td>93.1</td>
</tr>
</tbody>
</table>

Figure E-1: The TAME diet: intake rates by contribution to calorific intake. These values have been derived from the average Swiss diet for the present-day, see JISKRA, (1985).
where $\eta_{gv}$, $\eta_{rv}$ and $\eta_{gr}$ are the energy contents of green vegetables, root vegetables and grain respectively and the fraction of the total energy from each of the vegetable consumption pathways is given by

$$
 p_i = \frac{\eta_i I_i}{\sum_{\text{pathways}} \eta_i I_i} \equiv \frac{\eta_i I_i}{p_{\text{veg}} E_0 \left( 1 - f_{\text{eggs}} - f_{\text{milk}} - f_{\text{FF}} \right)}
$$

(E.40)

The remaining annual food energy intake is assumed to be from meat consumption

$$
 I_{\text{meat}} = \frac{E_0}{\eta_{\text{meat}}} \left( 1 - p_{\text{veg}} \right) \left( 1 - f_{\text{eggs}} - f_{\text{FF}} - f_{\text{milk}} \right) \quad \text{[kg y}^{-1}].
$$

(E.41)

The parameters used to characterise the consumption pathways are summarised in Table E.2.

**E.3.2 Consumption rates in the TAME exposure pathways**

The data given in Table 3-2 for the TAME human behaviour parameters define the representative individual as an adult living in a small self-sustaining community. The fractional consumption rates have been derived on the basis of the diet employed in Projekt Gewährr (JISKRA, 1985). The corresponding consumption rates derived by TAME are given in Table E.3 and the make-up of the diet is shown schematically in Figure E-1. These consumption rates are therefore the same as used in earlier studies in Switzerland.
F. The TAME - BIOSPH intercomparison

F.1 Introduction to the test cases

To verify the correct implementation of the TAME mathematical model (code version 1b) in the TAME codes, a series of intercomparisons were carried out using the BIOSPH code (BÖHRINGER et al., 1986) as a benchmark. The advantages of using BIOSPH, compared to alternatives (say the PSACOIN Level 1b test scenario (NEA, 1993) or simple hand calculations) are that BIOSPH has a relatively detailed internal structure of compartments together with a multiple pathway exposure model which uses the same database for which TAME was set up. It is also a relatively simple code in that it provides only the steady-state distribution of radionuclides in the biosphere (and the corresponding doses). This is achieved by inverting the transfer matrix which represents the radionuclide fluxes between the biosphere compartments. Subsequently, doses are calculated on the basis of these distributions. The differences between TAME and BIOSPH are summarised in Table F.1.

Two pairs of calculations were carried out. Test case 1 (TEST_1) and test case 2 (TEST_2) were used to compare the behaviour of the transport sub-models and test cases 3 and 4 (TEST_3 and TEST_4) were used to perform an intercomparison of the exposure pathway sub-models.

In TEST_1 the TAME dataset was modified so that only those FEPs that are also considered in BIOSPH were included. TEST_2 had a dataset with all the additional FEPs switched on. Similarly the TEST_3 dataset used the TEST_1 values for the transport of radionuclides in the biosphere with a reduced TAME dataset for the exposure pathways. TEST_4 used the TEST_2 TAME dataset to illustrate the additional information in the complete TAME representation of the modelled biosphere. In TEST_1 and TEST_3, the aim was for TAME to reproduce the BIOSPH results as closely as possible. In TEST_2 and TEST_4 the differences between the two codes were to be highlighted. This would indicate the ways in which the new features of TAME might affect the results of future assessments.

The radionuclides chosen for the test cases are given in Table F.2. They are chosen for their variety of behaviour in the biosphere, their relevance to the Swiss waste disposal programme and their range of half-lives ($\ln 2/\lambda_0$). The source terms used in the intercomparison were continuous, constant releases of $10^6$ Bq y$^{-1}$ of the parents of the modelled decay chains. This constant release (with no cut-off) was used so that the quantities evaluated in the TAME calculations would eventually reach steady-state. In almost all cases this occurred before the end of the time-period for the calculations at $10^6$ years. This allowed the results from TAME to be compared directly with those from BIOSPH which are for the steady-state distribution of radionuclides in the biosphere only.

As in the example calculations given in Section 4, the daughters grow in as a result of the decay of the parent radionuclides. The contribution to the total dose for the chain is
negligible (in TEST_3 and TEST_4) but the ingrowth from very low inventories provides a useful comparison of the handling of the chain daughters in transport calculations (TEST_1 and TEST_2).

As far as the transport of radionuclides in the biosphere is concerned, the main difficulties in comparing TAME with BIOSPH are that the structures of the two models are rather different. The nature of the output from BIOSPH determined that the steady-state biosphere inventories and the steady-state doses should be compared. The lack of a fifth compartment in BIOSPH meant that the TAME aquatic sediment compartment was left disconnected from the rest of the TAME structure and, in BIOSPH, the compartments were chosen to represent the local aquifer, the deep soil, the top soil and the river water.

N.B., for TEST_1 and TEST_2, because of internal coding constraints BIOSPH was set up such that no sorption was assumed in the compartments representing the local aquifer and the river water. The TAME datasets for TEST_1 and TEST_2 correspondingly used $k_d = 0$ for the aquifer and the river water, to provide a basis for the intercomparison. In the later calculations (TEST_3 and TEST_4) where sorption in the aquifer was necessary for the comparison of the doses calculated in the exposure pathways, BIOSPH was run with non-zero $k_d$ in the aquifer, but with the deep soil not explicitly modelled.

Aside from the fact that TAME has more exposure pathways than BIOSPH, the main difference in the exposure pathway sub-models is that TAME represents the same exposure pathways in a more detailed way. This is a result of adopting the NRPB (National Radiological Protection Board) exposure pathway model from the model MiniBIOS. Since the dataset given by MARTIN et al. (1991) in the MiniBIOS description is tailored to conditions in the United Kingdom, it was decided to use the BIOSPH database with TAME. TEST_3 was therefore used to ensure that the same results for dose were obtainable from TAME following the reparameterisation of the BIOSPH inputs. TEST_4 employed the same transport dataset as TEST_2 so that the final results in TEST_4 (as presented in Sections E-3 and E-4) represent operation of the full set of FEPs in TAME.

The results of the intercomparison, discussed below, focus mainly on the transport model, since, as is shown below, the TAME results from the dose calculations in TEST_3 were identical to those from BIOSPH. This was because the TAME input values were derived from the BIOSPH input. Table F.3 summarises the four test cases.

Of interest here is (i) the confirmation that with the same input data TAME and BIOSPH give the same results and (ii) an indication of what the differences are when the restrictions placed on TAME when it is configured to behave like BIOSPH are removed. The exact input data files are not fully relevant here - only the differences are. The data used in the intercomparison are broadly the same as those given in Section 3 of this report. The differences are given below, together with some discussion on the relevant points of difference between the TAME and BIOSPH representations of the same biosphere site.
Table F.1: The main differences between the *TAME* and *BIOSPH* models.

<table>
<thead>
<tr>
<th></th>
<th><strong>TAME</strong></th>
<th><strong>BIOSPH</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Dynamic, time-dependent distribution of radionuclides in the biosphere</td>
<td>Steady-state distribution of radionuclides in the biosphere</td>
<td></td>
</tr>
<tr>
<td>Transfer coefficients based on conservation of water and solid material fluxes and porewater diffusion</td>
<td>Transfer coefficients based on the conservation of water fluxes</td>
<td></td>
</tr>
<tr>
<td>Five compartments plus a sink</td>
<td>Four compartments plus a sink</td>
<td></td>
</tr>
<tr>
<td>Partially saturated soils and aquifer</td>
<td>Fully saturated soils and aquifer</td>
<td></td>
</tr>
<tr>
<td>Non-zero $k_d$ in all compartments</td>
<td>Non-zero $k_d$ only possible in some compartments</td>
<td></td>
</tr>
<tr>
<td>Exposure by inhalation and external $\gamma$-irradiation considered</td>
<td>Exposure by inhalation and external $\gamma$-irradiation not considered</td>
<td></td>
</tr>
<tr>
<td>All <em>TAME</em> exposure pathways included (cf. <em>MiniBIOS</em>)</td>
<td>Diverse FEPs present in the <em>MiniBIOS</em> exposure pathway sub-model <strong>not</strong> included in <em>BIOSPH</em>. See Section F-3.</td>
<td></td>
</tr>
</tbody>
</table>
Table F.2: The radionuclides used in the TAME - BIOSPH intercomparison. Shaded entries indicate radionuclides not in the BIOSPH database, and so not compared. They are included here to indicate that they are an integral part of the current TAME database. The \( k_d \)-values used here are for test purposes only and have since been superseded by those given by TITS et al. (1996).

<table>
<thead>
<tr>
<th>nuclide</th>
<th>daughters</th>
<th>deep soil ( k_d ) [m(^3)/kg]</th>
<th>top soil ( k_d ) [m(^3)/kg]</th>
<th>Included in the TEST_n calculations</th>
<th>decay constant, ( \lambda_0 ) [y(^{-1})]</th>
</tr>
</thead>
<tbody>
<tr>
<td>( ^{14}C )</td>
<td>-</td>
<td>5x10(^{-3} )</td>
<td>5x10(^{-3} )</td>
<td>1,2</td>
<td>1.21x10(^{4} )</td>
</tr>
<tr>
<td>( ^{36}Cl )</td>
<td>-</td>
<td>0</td>
<td>0</td>
<td>1,2,3,4</td>
<td>2.31x10(^{6} )</td>
</tr>
<tr>
<td>( ^{99}Tc )</td>
<td>-</td>
<td>1x10(^{-3} )</td>
<td>1x10(^{-3} )</td>
<td>1,2</td>
<td>3.30x10(^{6} )</td>
</tr>
<tr>
<td>( ^{126}Sn )</td>
<td>-</td>
<td>0.1</td>
<td>0.1</td>
<td>1,2</td>
<td>6.93x10(^{6} )</td>
</tr>
<tr>
<td>( ^{129}I )</td>
<td>-</td>
<td>0</td>
<td>0.01</td>
<td>1,2</td>
<td>4.42x10(^{8} )</td>
</tr>
<tr>
<td>( ^{135}Cs )</td>
<td>-</td>
<td>1.0</td>
<td>0.5</td>
<td>1,2,3,4</td>
<td>3.47x10(^{7} )</td>
</tr>
<tr>
<td>( ^{137}Cs )</td>
<td>-</td>
<td>1.0</td>
<td>0.5</td>
<td>1,2</td>
<td>2.30x10(^{2} )</td>
</tr>
<tr>
<td>( ^{237}Np )</td>
<td></td>
<td>5.0</td>
<td>0.1</td>
<td>1,2,3,4</td>
<td>3.24x10(^{7} )</td>
</tr>
<tr>
<td>( ^{233}U )</td>
<td></td>
<td>5.0</td>
<td>0.1</td>
<td>1,2,3,4</td>
<td>4.36x10(^{6} )</td>
</tr>
<tr>
<td>( ^{229}Th )</td>
<td></td>
<td>5.0</td>
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Table F.3: Summary of the test cases used in the TAME - BIOSPH intercomparison.

<table>
<thead>
<tr>
<th>test case dataset</th>
<th>input description</th>
<th>comments on results</th>
</tr>
</thead>
<tbody>
<tr>
<td>TEST_1 biosphere transport</td>
<td>Common dataset. TAME set up to reproduce exactly the compartmental inventories calculated by BIOSPH. TAME sediment compartment isolated, zero $k_{dS}$ in aquifer and river water. Solid material fluxes all set to zero. Diffusion switched off.</td>
<td>Tables F-5 - F-8 compare the results of TAME with BIOSPH. Error in BIOSPH discovered: evapotranspiration was acting to transfer radionuclides out of the top soil. When corrected, perfect agreement obtained.</td>
</tr>
<tr>
<td>TEST_2 exposure pathways</td>
<td>TAME transport model set up to include all of the new FEPs allowed by the TAME conceptual model. Differences between this and the TEST_1 dataset are highlighted.</td>
<td>A comparison of the intercompartment transfer rates in Table F.9 illustrates the rôle played by transport on solid material, particularly between the soils. Table F.10 illustrates the effects on the compartment inventories and the times to steady-state.</td>
</tr>
<tr>
<td>TEST_3</td>
<td>Test to ensure compatibility of datasets for the exposure pathways in TAME and BIOSPH. Comparison based on equal inventories in aquifer, river and top soil. Only the exposure pathways common to both models were considered.</td>
<td>The results of this exercise ensured the results from TAME were equivalent to those from BIOSPH. This is achieved by setting generic TAME input parameters to appropriate values to give identical results.</td>
</tr>
<tr>
<td>TEST_4</td>
<td>Full TAME dataset (including the database from TEST_3) plus additional pathways - inhalation and external $\gamma$-irradiation. Realistic compartment $k_d$ values.</td>
<td>Results reported in Section 4 of the main text in this report.</td>
</tr>
</tbody>
</table>
F.2 Intercomparison of the transport models: Datasets TEST_1 and TEST_2

F.2.1 Results for TEST_1

In setting up the comparison of TAME and BIOSPH, the majority of the simplifications are in the SOLIDFLUX section of the TAME biosphere description file (see Table 2-4). As described in Appendix D, many of the solid fluxes are mediated by the water fluxes (which were the same in the two test cases) via the suspended solid load. These parameters are the ALPHA_n (n = L, W, D and T) in the table; they are zero in TEST_1 but are assigned realistic values in TEST_2 (cf. Section 3 of this report). Other changes include the biomass of worms in the deep soil (MD) and the parameters for the transfer of river bed sediment to land (SL) and the interaction rate for the bed sediment with the river water (KAPPA_SW). Note that the diffusion coefficient (DO) is switched off in TEST_1.

Also of interest is the treatment of the saturation in the soils. In TAME, the porosity of the compartments is defined by the parameter EPS_n. The degree of saturation is defined by the volumetric moisture content of the total compartment, THETA_n, so that

\[
\text{degree of saturation} = \frac{\text{THETA}_n}{\text{EPS}_n}.
\]

BIOSPH does not distinguish these two quantities, so that THETA_n = EPS_n for all n and implicit saturation of all media is hardwired into the code. A similar point of difference concerns the treatment of the aquifer in the BIOSPH for this intercomparison. In this case BIOSPH treated the aquifer as a compartment of pure water - the \( k_d \) was equal to zero which effectively meant that there was no solid material in the aquifer at all. In the TAME aquifer there is solid material and the other properties of the compartment are taken into account, see Table 3.1. The volumetric flow to the various other connected compartments is the same in the datasets of both models, but the physical dimensions of the BIOSPH aquifer are only those required to allow the volumetric water fluxes. This plays no rôle in determining the radionuclide transfer coefficients and hence the distribution of radionuclides in the biosphere when steady-state has been reached. With \( k_d = 0 \) in the local aquifer, the contents of the local aquifer compartment correspond exactly to the quantity of radionuclides in the TAME local aquifer porewater (see Appendix A) and this is exactly the total amount in the BIOSPH aquifer compartment.

For each of the four compartments compared, the results of the intercomparison of TAME and BIOSPH are given in Tables E-5 to E-8. When undertaking model intercomparisons such as these, it is frequently the case that the results are in good agreement with each other. In this case, the care with which the datasets had been prepared to be completely compatible for the two models and the transparency of the implementation of the FEPs, particularly in TAME, lead to exact agreement for the steady-state distributions of the radionuclides considered. On the basis of these tables, for a broad
range of radionuclides it can be stated that the TAME transport model used in TEST_1 is correctly implemented.

F.2.2 The effect of the additional FEPs - TEST_2

TAME produces a log-file at runtime which contains the precise input data values used in the calculations. As an illustration of the differences, the log-files generated for $^{135}$Cs in both TEST_1 and TEST_2 have been compared (using the DOS file compare utility) and the results of the comparison are shown in Table F.4.

As can be anticipated from Table F.4, the radionuclides showing the greatest difference in results from the TEST_1 dataset will be those with the highest $k_d$s. The highly sorbing elements bind to the solid material in the system, giving rise to additional transport routes. Table F.9 shows the TAME transfer coefficients for $^{135}$Cs, with the white boxes showing where there are differences between TEST_1 and TEST_2.

The way in which the boundary conditions for solid material transport are applied means that the only solid material loss from the modelled system is via the suspended solid load in the river water. Aquatic bed sediment is confined to the system by fixing a recirculation between the top soil, the river water and the bed sediment. However, in these test cases, the $k_d$ in the river water is zero, so that even the transfer coefficient for this term is zero. None of the entries for the transfer to elsewhere column changes between the two datasets.

The greatest change can be seen in the transfer coefficient between the deep soil and the top soil, where the factor is 22:1. This is purely as a result of the bioturbation caused by the earthworms in the system (see Appendix D). However the overall effect of erosion in the model cannot be seen here, because the $k_d$ value in the local aquifer is zero in both cases. Clearly, the increased mobility of the sorbing radionuclides is an important consequence of the new TAME parameterisation.

Also present in the transfer coefficients in TEST_2 is the transfer from the local aquifer to the aquatic sediment as a result of diffusion in the compartmental porewater. However, as discussed in Appendix C, the representation of this process needs to be amended in future versions of TAME (a life history of the TAME codes can be found in Appendix G).

In this comparison it can be seen that TAME and BIOSPH behave the same when the same data are input, activating the same FEPs. As far as modelling using either dynamic or steady-state solution methods is concerned, the only overhead is that the dynamic system requires a slightly increased run-time. On modern computers the difference between the instantaneous derivation of the steady-state radionuclide distribution and the almost instantaneous dynamic solution is trivial in comparison with all the work required to correctly parameterise the FEPs in the models (and prior screening of the FEPs for inclusion).
Table F.4: Differences in the TAME log-files for $^{135}$Cs in the TEST_1 and TEST_2 datasets. These listings show the parameter values switched out in making TAME behave like BIOSPH. This is a direct comparison of the TAME datafiles, showing the line numbers and nearest similar data lines. The differences are indicated by bold characters. For an explanation of parameter names, see Appendix F. Further details are given in the text.

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Table F.5: Comparison of activity content of the river water compartment at steady-state, showing the perfect agreement between TAME 1b and BIOSPH. Also given is the time taken to reach steady-state. These can largely be explained by the different $k_d$ values in the soil compartments - see Table F.2.

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<tr>
<td></td>
<td>$^{231}$Pa</td>
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</table>
Table F.6: Comparison of activity content of the top soil compartment at steady-state, showing the perfect agreement between TAME_1b and BIOSPH. Also given is the time taken to reach steady-state. These can largely be explained by the different $k_d$ values in the soil compartments - Table F.2.

<table>
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<th>time to steady-state [y]</th>
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</table>
Table F.7: Comparison of content of the deep soil compartment at steady-state, showing the perfect agreement between TAME_1b and BIOSPH. Also given is the time taken to reach steady-state. These can largely be explained by the different $k_d$ values in the soil compartments - Table F.2.

<table>
<thead>
<tr>
<th>nuclide</th>
<th>compartment inventory [Bq]</th>
<th>time to steady-state [y]</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>parent daughter</td>
<td>TAME 1b</td>
</tr>
<tr>
<td>$^{14}$C</td>
<td>-</td>
<td>2.55×10^5</td>
</tr>
<tr>
<td>$^{36}$Cl</td>
<td>-</td>
<td>1.22×10^4</td>
</tr>
<tr>
<td>$^{99}$Tc</td>
<td>-</td>
<td>6.09×10^4</td>
</tr>
<tr>
<td>$^{126}$Sn</td>
<td>-</td>
<td>4.87×10^6</td>
</tr>
<tr>
<td>$^{129}$I</td>
<td>-</td>
<td>1.22×10^4</td>
</tr>
<tr>
<td>$^{135}$Cs</td>
<td>-</td>
<td>4.87×10^7</td>
</tr>
<tr>
<td>$^{137}$Cs</td>
<td>-</td>
<td>1.19×10^6</td>
</tr>
<tr>
<td>$^{237}$Np</td>
<td>$^{233}$U</td>
<td>2.43×10^8</td>
</tr>
<tr>
<td></td>
<td>$^{229}$Th</td>
<td>4.70×10^6</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1.39×10^6</td>
</tr>
<tr>
<td>$^{238}$U</td>
<td>$^{234}$U</td>
<td>2.43×10^8</td>
</tr>
<tr>
<td></td>
<td>$^{230}$Th</td>
<td>3.09×10^6</td>
</tr>
<tr>
<td></td>
<td>$^{226}$Ra</td>
<td>1.20×10^5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>4.67×10^3</td>
</tr>
<tr>
<td>$^{239}$Pu</td>
<td>$^{235}$U</td>
<td>4.72×10^7</td>
</tr>
<tr>
<td></td>
<td>$^{231}$Pa</td>
<td>2.57×10^2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>4.94×10^1</td>
</tr>
</tbody>
</table>
Table F.8: Comparison of activity content of the local aquifer compartment at steady-state, showing the perfect agreement between TAME_1b and BIOSPH. Also given is the time taken to reach steady-state. These can largely be explained by the different $k_d$ values in the soil compartments - Table F.2.

<table>
<thead>
<tr>
<th>nuclide</th>
<th>compartment inventory [Bq]</th>
<th>time to steady-state [y]</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>TAME_1b</td>
<td>BIOSPH</td>
</tr>
<tr>
<td>parent</td>
<td>daughter</td>
<td></td>
</tr>
<tr>
<td>$^{14}$C</td>
<td>-</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td>$^{36}$Cl</td>
<td>-</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td>$^{99}$Tc</td>
<td>-</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td>$^{126}$Sn</td>
<td>-</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td>$^{129}$I</td>
<td>-</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td>$^{135}$Cs</td>
<td>-</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td>$^{137}$Cs</td>
<td>-</td>
<td>$6.39 \times 10^5$</td>
</tr>
<tr>
<td>$^{237}$Np</td>
<td>$^{233}$U</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td></td>
<td>$^{229}$Th</td>
<td>$7.17 \times 10^2$</td>
</tr>
<tr>
<td>$^{238}$U</td>
<td>$^{234}$U</td>
<td>$6.85 \times 10^5$</td>
</tr>
<tr>
<td></td>
<td>$^{230}$Th</td>
<td>$4.71 \times 10^2$</td>
</tr>
<tr>
<td></td>
<td>$^{226}$Ra</td>
<td>$1.83 \times 10^1$</td>
</tr>
<tr>
<td>$^{239}$Pu</td>
<td>$^{235}$U</td>
<td>$3.54 \times 10^1$</td>
</tr>
<tr>
<td></td>
<td>$^{231}$Pa</td>
<td>$6.84 \times 10^5$</td>
</tr>
<tr>
<td></td>
<td>$^{235}$U</td>
<td>$3.95 \times 10^2$</td>
</tr>
<tr>
<td></td>
<td>$^{231}$Pa</td>
<td>$3.75 \times 10^3$</td>
</tr>
</tbody>
</table>
Table F.9: Comparison of the *TAME* transfer coefficients ($\lambda_{ij} \ [y^{-1}]$) generated for the moderately highly sorbing $^{135}$Cs in TEST_1 and TEST_2. The shaded regions indicate no difference between the two cases.

<table>
<thead>
<tr>
<th></th>
<th>from \ to</th>
<th>Local</th>
<th>Deep soil</th>
<th>Top soil</th>
<th>Water</th>
<th>Sediment</th>
<th>Elsewhere</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>TEST_1.LOG</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local</td>
<td></td>
<td>3.33x10^{-2}</td>
<td>5.00x10^{-2}</td>
<td>2.13x10^{-1}</td>
<td>0</td>
<td>1.25</td>
<td></td>
</tr>
<tr>
<td>Deep soil</td>
<td></td>
<td>1.17x10^{3}</td>
<td></td>
<td>8.38x10^{-5}</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Top soil</td>
<td></td>
<td>0</td>
<td></td>
<td>7.04x10^{-3}</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Water</td>
<td></td>
<td>0</td>
<td></td>
<td>3.00x10^{1}</td>
<td>0</td>
<td>6.01x10^{4}</td>
<td>0</td>
</tr>
<tr>
<td>Sediment</td>
<td></td>
<td>0</td>
<td></td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>from \ to</th>
<th>Local</th>
<th>Deep soil</th>
<th>Top soil</th>
<th>Water</th>
<th>Sediment</th>
<th>Elsewhere</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>TEST_2.LOG</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local</td>
<td></td>
<td>3.37x10^{-2}</td>
<td>5.00x10^{-2}</td>
<td>2.13x10^{-1}</td>
<td>2.95x10^{-3}</td>
<td>1.25</td>
<td></td>
</tr>
<tr>
<td>Deep soil</td>
<td></td>
<td>1.27x10^{3}</td>
<td></td>
<td>1.85x10^{-3}</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Top soil</td>
<td></td>
<td>0</td>
<td></td>
<td>1.24x10^{2}</td>
<td>2.51x10^{4}</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Water</td>
<td></td>
<td>0</td>
<td></td>
<td>3.00x10^{1}</td>
<td>0</td>
<td>6.01x10^{4}</td>
<td>0</td>
</tr>
<tr>
<td>Sediment</td>
<td></td>
<td>1.31</td>
<td></td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>0</td>
</tr>
</tbody>
</table>
Table F.10: Comparison of the steady-state results for $^{135}$Cs in the two test cases, TEST_1 and TEST_2, illustrating the effect of the additional FEPs introduced into TAME. Note that the top soil inventory is increased by a factor of 1.65 and that its time to steady-state is increased by three thousand years. N.B. these datasets should only be used for intercomparisons - the simplifications imposed by BIOSPH mean these are not good indicators of the results in a real assessment.

<table>
<thead>
<tr>
<th>compartment</th>
<th>inventory</th>
<th>time to reach steady-state</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>TEST_1</td>
<td>TEST_2</td>
</tr>
<tr>
<td>local aquifer</td>
<td>$6.85 \times 10^5$</td>
<td>$6.83 \times 10^5$</td>
</tr>
<tr>
<td>deep soil</td>
<td>$4.87 \times 10^7$</td>
<td>$4.32 \times 10^7$</td>
</tr>
<tr>
<td>top soil</td>
<td>$5.45 \times 10^6$</td>
<td>$9.02 \times 10^6$</td>
</tr>
<tr>
<td>river water</td>
<td>2.43</td>
<td>2.46</td>
</tr>
<tr>
<td>bed sediment</td>
<td>-</td>
<td>1.54 $\times 10^3$</td>
</tr>
</tbody>
</table>
The time-dependence of the dynamic solution also brings benefits when carrying out safety assessments, particularly for releases which are of short duration in comparison with the time to steady-state. Such a situation can occur for L/ILW repositories which are not placed at great depth. Using a steady-state solution method will always overestimate the consequences in such cases, no matter how sophisticated the FEPs the model contains.

F.3 TEST_3: Harmonisation of the TAME and BIOSPH database for the exposure pathway sub-model

F.3.1 Scope of TEST_3

Unlike TEST_1, TEST_3 was not a direct comparison of TAME_1b and BIOSPH. The main purpose was to ensure that the modified MiniBIOS structure used as the basis for the TAME exposure pathway model could reproduce the restricted set of pathways implemented in BIOSPH. This was achieved by converting the BIOSPH database into a form suitable for input to TAME. The TEST_4 dataset was then a complete best-estimate representation of the Central Swiss valley biosphere, as reported in Sections 3 and 4 of this report).

Use was made of spreadsheet representations of the BIOSPH and TAME models before the final coding of the operational versions of TAME was completed. The use of spreadsheets enabled the details of the FEPs to be compared on a parameter-by-parameter basis. The end result was that the TAME and BIOSPH codes gave exactly the same results for those exposure pathways that they have in common.

Rather than showing tables such as those in Section F.2.1, which show perfect agreement between the two transport models, this section focuses on the work that was required to harmonise the results of the exposure pathway sub-models. This proceeded by a combination of setting certain parameters in the TAME input to zero (to switch-off FEPs not in BIOSPH) and evaluating the suitable values for the other TAME parameters so that the results for both codes would be the same. This approach was chosen for two purposes:

a). Intercomparison purposes - the correct implementation of TAME could be verified when the results were the same both from the codes and from the spreadsheets, and

---

1 In contrast to the TEST_1 database, TEST_3 used a non-zero \(k_d\) in the aquifer. This was achieved by ignoring the deep soil compartments in the BIOSPH calculations. This did not affect the intercomparison since results for the exposure pathways can be compared by assuming equal inventories in each of the top soil, aquifer and river compartments.
b). The BIOSPH database is customised for Swiss biosphere conditions. Rather than adopt the default MiniBIOS database (for United Kingdom conditions) the new TAME database would also be relevant to generic Swiss biospheres.

The following sections gives selected examples of how suitable values for the TAME input database were constructed.

**F.3.2 Irrigation mass interception factors**

In TAME, the generic form of the concentration of a radionuclide in crop $c$ as a result of irrigation from contaminated water from the aquifer or the river is given by

$$C_{\text{water-irrigation}}^c = \left( f_c + \frac{T_c}{H_c} \right) \frac{1 - \exp(-\mu_c Y_c)}{Y_c (W_c + H_c + T_c)} \left( \frac{F_{LT} \theta_c (1 - \epsilon_c) \rho_{k_L} C_L + F_{WT} C_W}{A_f} \right). \quad (F.1)$$

BIOSPH assumes only irrigation from the aquifer, so that the term in $F_{WT}$ disappears, leaving only the applied irrigation

$$d_{\text{irr}} = \frac{F_{LT}}{A_f}, \; [\text{m y}^{-1}], \quad (F.2)$$

The translocation factor, $T_c$, [$\text{y}^{-1}$], is not included in the BIOSPH representation and similarly, BIOSPH does not distinguish between the removal rate of radionuclides from the crop by harvesting ($H_c$, [$\text{y}^{-1}$]) and by weathering ($W_c$, [$\text{y}^{-1}$]) - only a single generic residence time is used.$^2$ This leaves what BIOSPH calls $N_c$, the fraction of the nuclide which remains on the crop, i.e. the fraction intercepted:

$$N_c = f_c \left(1 - \exp(-\mu_c Y_c)\right), \quad (F.3)$$

where the yield of the crop is $Y_c$ [kg m$^{-2}$]. TAME includes two additional factors in determining this quantity - $f_c$ is the fraction of the external contamination remaining after food processing (concept and data are taken from IAEA, 1992). The irrigation interception factor is $\mu_c$ [m$^2$ kg$^{-1}$]. As discussed in Section 2, this is a measure of the interception area per unit weight of crop. Equation (F.3) can therefore be used to derive suitable values for the crop dependent $\mu_c$. This procedure was carried out for the TAME database and the values derived are quoted in Table 3-3.

The TAME implementation of the MiniBIOS equations for this FEP provides for greater flexibility in the model response. It is known that irrigation is a very important mecha-
nism for some radionuclides and the exponential factor is, in turn, sensitive to both $\mu_c$ and $Y_c$.

Other factors leading to the accumulation of radionuclides in crops for both human and livestock consumption (Appendix E) are common to both models.

**F.3.3 Soil consumed by grazing cattle**

In *TAME*, the intake rate of radionuclides into cattle by direct consumption of contaminated soil during grazing is given in Appendix E.2.7 by

$$I_{soil}^{\text{livestock}} = S_{pc}ZI_{pc} \frac{C_T}{(1 - \varepsilon_T)\rho_T + \varepsilon_w \rho_w}, \text{[Bq day}^{-1}]. \quad (F.4)$$

Here the dimensionless factor $S_{cp}$ gives the amount of wet soil consumed as a fraction of the total mass of fresh pasture ($ZI_{pc}$ - see Section E.2.7 for details of the parameterisation and Appendix F for a glossary of terms).

The *BIOSPH* expression for this FEP is

$$I_{soil}^{\text{livestock}} = I_{soil} \frac{C_T}{(1 - \varepsilon_T)\rho_T}, \text{[kg day}^{-1}], \quad (F.5)$$

where dry soil is assumed and a fixed amount of soil is consumed - $I_{soil}$ [kg day$^{-1}$]. Equations (F.4) and (F.5) can be combined to give an appropriate value for $S_{cp}$ in the *TAME* database, commensurate with the original *BIOSPH* value. A similar calculation has been carried out for the grain consumption pathway for poultry.

The difference between the wet weight or dry weight consumption is of no consequence for the resulting doses. Both models give the same results because of the manipulation of the data. The question of the correct interpretation of the FEP is a matter for debate, although it has hardly been a dominant theme in the *BIOMOVS II Complementary Studies* or *Reference Biospheres* working groups! This case is a prime example of where unnecessary detail in the biosphere models might be removed, without jeopardising the calculated consequences.
F.4 Completion of the TAME database for the example calculations: Dataset TEST_4

As a final point concerning the TAME datasets in this document, it is worth mentioning how the TEST_4 dataset used in the example calculations in Section 4 of this report differed from the first three test cases. TEST_4 was not simply TEST_2 for biosphere transport with best-estimate enhancements to the TEST_3 database for the exposure pathways.

In TEST 4, the main difference was that the nuclide $k_d$ database from TITS et al., (1995) was used to provide realistic information for the compartments in the example biosphere. In addition to the soils, $k_d$ values were given for the local aquifer, the river water and the river bed sediment. Another difference was that in TEST_4 irrigation and drinking-water were abstracted from both the river and from the local aquifer. In BIOSPH only the aquifer could be a source of irrigation water.

In the calculation of the total annual individual dose over all pathways, data allowing the calculation of inhalation doses and the dose from external $\gamma$-irradiation were the main enhancement. Data for these were taken from the MiniBIOS database, as were many of the other parameters cited in Section 3, for which no corresponding values were available in the existing Swiss database. The potential importance of the inhalation pathways can be seen in Table 4-7 for the chain daughters.
G. Glossary of terms used in the TAME mathematical and computer models

As can be seen in Section 2 of this report, a large number of input parameters required to fully characterise the TAME model. Each of the quantities in the mathematical model has a corresponding FORTRAN parameter in the computer implementation. This Appendix provides a glossary of all terms in the TAME model and equates the names of the mathematical symbols with the FORTRAN names. The units used in the model and brief descriptions of the quantities are given.

The lists below are split into four sections, one for each of the TAME input files (see Appendix H, Figure H-1).

Table G.1 lists the nuclide-specific parameters used in the transport model and Table G.2 gives the radionuclide-specific parameters which are additionally required in the exposure pathways model. Table G.3 lists the parameters used to characterise the biosphere and Table G.4 lists the parameters used to characterise human behaviour which affects calculated doses.

N.B. the nomenclature used here is generally applicable to the mathematical formulation used throughout this report. In certain parts of the Appendices indices are not explicitly written, but this should not cause confusion when using the following tables as a reference. The information is correct up to TAME version 3a, October 1995. See Appendix G for a review of the life history of the TAME codes.
Table G.1: Parameters describing radionuclide characteristics used in the transport calculations (the time dependent part of the TAME system).

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Mathematical Symbol</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>NI</td>
<td>-</td>
<td>-</td>
<td>Number of radionuclides in the decay chain</td>
</tr>
<tr>
<td>NUC_NAME</td>
<td>-</td>
<td>-</td>
<td>Name of radionuclide in the TAME format (5 characters)³</td>
</tr>
<tr>
<td>DECAY</td>
<td>$\lambda_N$</td>
<td>$y^{-1}$</td>
<td>Radionuclide decay constant.</td>
</tr>
<tr>
<td>K_D</td>
<td>$k_D$</td>
<td>(Bq kg$^{-1}$)(Bq m$^{-3}$)$^{-1}$</td>
<td>Solid - liquid distribution coefficient in the deep soil.</td>
</tr>
<tr>
<td>K_L</td>
<td>$k_L$</td>
<td>(Bq kg$^{-1}$)(Bq m$^{-3}$)$^{-1}$</td>
<td>Solid - liquid distribution coefficient in the local aquifer.</td>
</tr>
<tr>
<td>K_W</td>
<td>$k_W$</td>
<td>(Bq kg$^{-1}$)(Bq m$^{-3}$)$^{-1}$</td>
<td>Solid - liquid distribution coefficient in the surface water.</td>
</tr>
<tr>
<td>K_S</td>
<td>$k_S$</td>
<td>(Bq kg$^{-1}$)(Bq m$^{-3}$)$^{-1}$</td>
<td>Solid - liquid distribution coefficient in the sediment.</td>
</tr>
<tr>
<td>K_T</td>
<td>$k_T$</td>
<td>(Bq kg$^{-1}$)(Bq m$^{-3}$)$^{-1}$</td>
<td>Solid - liquid distribution coefficient in the top soil.</td>
</tr>
<tr>
<td>L_DG</td>
<td>$\lambda_{dg}$</td>
<td>$y^{-1}$</td>
<td>Degassing coefficient for volatile radionuclides in the top soil. (This parameter was not used in the WLB calculations).</td>
</tr>
</tbody>
</table>

³ Only five characters are currently allowed for nuclide names in TAME - two characters for the element symbol and three for the nucleon number. This causes problems for radionuclides such as $^{108m}$Ag and $^{93m}$Nb. Such meta-stable nuclides are identified in TAME by adding 100 to the nucleon number and deleting the m suffix, so that $^{108m}$Ag becomes Ag208 and $^{93m}$Nb NB193.
Table G.2: Parameters describing radionuclide- and element-specific characteristics used in the calculation of annual individual dose.

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Mathematical Symbol</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>K_FF</td>
<td>$K_{ff}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq m}^{-3})^{-1}$</td>
<td>Fish concentration factor - fresh weight of fish</td>
</tr>
<tr>
<td>K_G</td>
<td>$K_{gr}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq kg}^{-1})^{-1}$</td>
<td>Soil to grain transfer factor: fresh weight crop / volume of dry soil</td>
</tr>
<tr>
<td>K_GV</td>
<td>$K_{gv}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq kg}^{-1})^{-1}$</td>
<td>Soil to green vegetables transfer factor: fresh weight crop / volume of dry soil</td>
</tr>
<tr>
<td>K_MEAT</td>
<td>$K_{meat}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq day}^{-1})^{-1}$</td>
<td>Meat concentration factor</td>
</tr>
<tr>
<td>K_MILK</td>
<td>$K_{milk}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq day}^{-1})^{-1}$</td>
<td>Milk concentration factor</td>
</tr>
<tr>
<td>K_P</td>
<td>$K_p$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq kg}^{-1})^{-1}$</td>
<td>Soil to pasture transfer factor: fresh weight crop / volume of dry soil</td>
</tr>
<tr>
<td>K_RV</td>
<td>$K_{rv}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq kg}^{-1})^{-1}$</td>
<td>Soil to root vegetables transfer factor: fresh weight crop / volume of dry soil</td>
</tr>
<tr>
<td>K_EGG</td>
<td>$K_{eggs}$</td>
<td>$(\text{Bq kg}^{-1})(\text{Bq day}^{-1})^{-1}$</td>
<td>Eggs concentration factor</td>
</tr>
<tr>
<td>W_G</td>
<td>$W_{gr}$</td>
<td>$y^{-1}$</td>
<td>Weathering rate for grain - loss of external contamination</td>
</tr>
<tr>
<td>W_P</td>
<td>$W_{p}$</td>
<td>$y^{-1}$</td>
<td>Weathering rate for pasture - loss of external contamination</td>
</tr>
<tr>
<td>W_GV</td>
<td>$W_{gv}$</td>
<td>$y^{-1}$</td>
<td>Weathering rate for green vegetables - loss of external contamination</td>
</tr>
<tr>
<td>W_RV</td>
<td>$W_{rv}$</td>
<td>$y^{-1}$</td>
<td>Weathering rate for root vegetables - loss of external contamination</td>
</tr>
<tr>
<td>T_RV</td>
<td>$T_{rv}$</td>
<td>$y^{-1}$</td>
<td>Translocation rate for root vegetables - external contamination transferred to plant interior</td>
</tr>
<tr>
<td>D_ING</td>
<td>$D_{ing}$</td>
<td>$\text{Sv Bq}^{-1}$</td>
<td>Dose per unit intake on ingestion (includes contributions from short-lived daughter)</td>
</tr>
<tr>
<td>D_INH</td>
<td>$D_{inh}$</td>
<td>$\text{Sv Bq}^{-1}$</td>
<td>Dose per unit intake on inhalation (includes contributions from short-lived daughter)</td>
</tr>
<tr>
<td>G</td>
<td>$G$</td>
<td>$(\text{Sv} y^{-1})(\text{Bq m}^{-3})^{-1}$</td>
<td>Semi-infinite plane groundshine factor corrected for self-absorption (includes contributions from short-lived daughter)</td>
</tr>
</tbody>
</table>
Table G.3: Parameters in the biosphere characteristics BIO-file.

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Mathematical Symbol</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>NTER</td>
<td>-</td>
<td>-</td>
<td>Number of sets of the five TAME compartments in the model.</td>
</tr>
<tr>
<td>SCENARIO</td>
<td>-</td>
<td>-</td>
<td>Name of the data scenario (8 characters)</td>
</tr>
<tr>
<td>AF</td>
<td>$A_f$</td>
<td>m$^2$</td>
<td>Surface area of the biosphere region</td>
</tr>
<tr>
<td>A_LE</td>
<td>$A_{LE}$</td>
<td>m</td>
<td>Area of interface between local aquifer and Elsewhere compartment</td>
</tr>
<tr>
<td>DO</td>
<td>$D_0$</td>
<td>m$^2$ y$^{-1}$</td>
<td>Ionic diffusion constant in pure water</td>
</tr>
<tr>
<td>ETP</td>
<td>$d_{TA}$</td>
<td>m$^2$ y$^{-1}$</td>
<td>Evapotranspiration rate - water loss from top soil to atmosphere</td>
</tr>
<tr>
<td>RAINFALL</td>
<td>$d_{AT}$</td>
<td>m$^2$ y$^{-1}$</td>
<td>Annual rainfall</td>
</tr>
<tr>
<td>ME</td>
<td>$m_e$</td>
<td>kg m$^{-2}$ y$^{-1}$</td>
<td>Erosion rate in the biosphere</td>
</tr>
<tr>
<td>V_A</td>
<td>$v_A$</td>
<td>m$^2$ y$^{-1}$</td>
<td>Annual average windspeed in the biosphere region</td>
</tr>
<tr>
<td>L_A</td>
<td>$l_A$</td>
<td>m</td>
<td>Mixing depth for the atmosphere - inhalation of volatile nuclides</td>
</tr>
<tr>
<td>ALPHA_D</td>
<td>$\alpha_D$</td>
<td>kg m$^{-3}$</td>
<td>Suspended solid load in the deep soil porewater</td>
</tr>
<tr>
<td>EPS_D</td>
<td>$\varepsilon_D$</td>
<td>-</td>
<td>Porosity of the deep soil</td>
</tr>
<tr>
<td>L_D</td>
<td>$l_D$</td>
<td>m</td>
<td>Thickness of the deep soil</td>
</tr>
<tr>
<td>M_D</td>
<td>$m_D$</td>
<td>kg m$^{-2}$</td>
<td>Biomass in the deep soil per square metre of surface</td>
</tr>
<tr>
<td>RHO_D</td>
<td>$\rho_D$</td>
<td>kg m$^{-3}$</td>
<td>Dry density of deep soil material</td>
</tr>
<tr>
<td>THETA_D</td>
<td>$\theta_D$</td>
<td>-</td>
<td>Volumetric moisture content of the deep soil</td>
</tr>
<tr>
<td>T_D</td>
<td>$T_D$</td>
<td>-</td>
<td>Tortuosity of the deep soil material</td>
</tr>
<tr>
<td>W_D</td>
<td>$w_D$</td>
<td>y$^{-1}$</td>
<td>Activity of earthworms: number of round trips deep to top soil per year</td>
</tr>
<tr>
<td>ALPHA_L</td>
<td>$\alpha_L$</td>
<td>kg m$^{-3}$</td>
<td>Suspended solid load in the local aquifer porewater</td>
</tr>
<tr>
<td>EPS_L</td>
<td>$\varepsilon_L$</td>
<td>-</td>
<td>Porosity of the local aquifer</td>
</tr>
<tr>
<td>L_L</td>
<td>$l_L$</td>
<td>m</td>
<td>Thickness of the local aquifer</td>
</tr>
<tr>
<td>RHO_L</td>
<td>$\rho_L$</td>
<td>kg m$^{-3}$</td>
<td>Dry density of local aquifer material</td>
</tr>
<tr>
<td>THETA_L</td>
<td>$\theta_L$</td>
<td>-</td>
<td>Volumetric moisture content of the local aquifer</td>
</tr>
<tr>
<td>T_L</td>
<td>$T_L$</td>
<td>-</td>
<td>Tortuosity of the local aquifer material</td>
</tr>
<tr>
<td>ALPHA_W</td>
<td>$\alpha_W$</td>
<td>kg m$^{-3}$</td>
<td>Suspended solid load in the surface water</td>
</tr>
<tr>
<td>D_W</td>
<td>$d_W$</td>
<td>m</td>
<td>Depth of the surface water</td>
</tr>
<tr>
<td>L_W</td>
<td>$l_W$</td>
<td>m</td>
<td>Length of surface water compartment (length of TAME section)</td>
</tr>
<tr>
<td>RHO_W</td>
<td>$\rho_W$</td>
<td>kg m$^{-3}$</td>
<td>Density of water (also used for milk)</td>
</tr>
<tr>
<td>W_W</td>
<td>$w_W$</td>
<td>m</td>
<td>Width of surface water compartment</td>
</tr>
</tbody>
</table>
Table G-3: Parameters in the biosphere characteristics BIO-file (continued).

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Mathematical Symbol</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS_S</td>
<td>$e_S$</td>
<td>-</td>
<td>Porosity of sediment</td>
</tr>
<tr>
<td>KAPPA_Sw</td>
<td>$\kappa_S$</td>
<td>y$^{-1}$</td>
<td>Turnover rate: bed sediment to suspended sediment in water column</td>
</tr>
<tr>
<td>D_S</td>
<td>$d_S$</td>
<td>m</td>
<td>Thickness of the bed sediment layer</td>
</tr>
<tr>
<td>SL</td>
<td>$m_{sl}$</td>
<td>kg m$^{-2}$ y$^{-1}$</td>
<td>Application rate: mass of bed sediment to area of land in region</td>
</tr>
<tr>
<td>RHO_S</td>
<td>$\rho_S$</td>
<td>kg m$^{-3}$</td>
<td>Dry density of bed sediment material</td>
</tr>
<tr>
<td>THETA_S</td>
<td>$\theta_S$</td>
<td>-</td>
<td>Volumetric moisture content of the bed sediment</td>
</tr>
<tr>
<td>T_S</td>
<td>$T_S$</td>
<td>-</td>
<td>Tortuosity for bed sediment particles</td>
</tr>
<tr>
<td>ALPHA_T</td>
<td>$\alpha_T$</td>
<td>kg m$^{-3}$</td>
<td>Suspended solid load in the top soil porewater</td>
</tr>
<tr>
<td>EPS_T</td>
<td>$e_T$</td>
<td>-</td>
<td>Porosity of the top soil</td>
</tr>
<tr>
<td>T_L</td>
<td>$l_T$</td>
<td>m</td>
<td>Thickness of the top soil</td>
</tr>
<tr>
<td>RHO_T</td>
<td>$\rho_T$</td>
<td>kg m$^{-3}$</td>
<td>Dry density of top soil material</td>
</tr>
<tr>
<td>THETA_T</td>
<td>$\theta_T$</td>
<td>-</td>
<td>Volumetric moisture content of the top soil</td>
</tr>
<tr>
<td>T_T</td>
<td>$T_T$</td>
<td>-</td>
<td>Tortuosity of the top soil material</td>
</tr>
<tr>
<td>F_ij</td>
<td>$F_{ij}$</td>
<td>m$^3$ y$^{-1}$</td>
<td>Water fluxes: compartments $i - j; i,j = L, D, T, W, S, E$</td>
</tr>
<tr>
<td>M_ij</td>
<td>$M_{ij}$</td>
<td>kg y$^{-1}$</td>
<td>Solid material fluxes: compartments $i - j; i,j = L, D, T, W, S, E$</td>
</tr>
</tbody>
</table>
Table G.4: Parameters in the DSE-file which stores data on habits and practices for the model region.

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Mathematical Symbol</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>DAYS PER YEAR</td>
<td>( n_d )</td>
<td>day y(^{-1} )</td>
<td>Number of days per year</td>
</tr>
<tr>
<td>ENERGY</td>
<td>( E_0 )</td>
<td>kJ y(^{-1} )</td>
<td>Calorific value of food consumed annually</td>
</tr>
<tr>
<td>F_MILK</td>
<td>( f_{milk} )</td>
<td></td>
<td>Fraction of annual food energy intake obtained from milk</td>
</tr>
<tr>
<td>I_MILK</td>
<td>( I_{milk} )</td>
<td>m(^3) y(^{-1} )</td>
<td>Annual consumption rate of milk</td>
</tr>
<tr>
<td>I_FLUID</td>
<td>( I_{fluid} )</td>
<td>m(^3) y(^{-1} )</td>
<td>Annual intake of liquids from all sources</td>
</tr>
<tr>
<td>FILTER</td>
<td>( f_{filter} )</td>
<td></td>
<td>Filtration factor for water consumption</td>
</tr>
<tr>
<td>H_DRINK</td>
<td>( f_{well} )</td>
<td></td>
<td>Fraction of water taken from the well in the local aquifer</td>
</tr>
<tr>
<td>I_W</td>
<td>( I_{wat} )</td>
<td>m(^3) y(^{-1} )</td>
<td>Annual consumption rate of water (calculated)</td>
</tr>
<tr>
<td>F_EGG</td>
<td>( f_{eggs} )</td>
<td></td>
<td>Fraction of annual food energy intake obtained from eggs</td>
</tr>
<tr>
<td>I_EGG</td>
<td>( I_{eggs} )</td>
<td>egg y(^{-1} )</td>
<td>Annual consumption rate of eggs</td>
</tr>
<tr>
<td>I_FF</td>
<td>( I_{ff} )</td>
<td>kg y(^{-1} )</td>
<td>Annual consumption rate of fish (calculated)</td>
</tr>
<tr>
<td>F_FF</td>
<td>( f_{ff} )</td>
<td></td>
<td>Fraction of annual food energy intake obtained from fish</td>
</tr>
<tr>
<td>I_G</td>
<td>( I_{gr} )</td>
<td>kg y(^{-1} )</td>
<td>Annual consumption rate of grain (calculated)</td>
</tr>
<tr>
<td>I_RV</td>
<td>( I_{rv} )</td>
<td>kg y(^{-1} )</td>
<td>Annual consumption rate of root vegetables (calculated)</td>
</tr>
<tr>
<td>I_GV</td>
<td>( I_{gv} )</td>
<td>kg y(^{-1} )</td>
<td>Annual consumption rate of green vegetables (calculated)</td>
</tr>
<tr>
<td>P_VEG</td>
<td>( p_{veg} )</td>
<td></td>
<td>Fraction of annual food energy intake obtained from vegetables (after milk, eggs and fish have been taken into account)</td>
</tr>
<tr>
<td>P_G</td>
<td>( p_{gr} )</td>
<td></td>
<td>Fraction of energy intake from vegetable consumption coming from grain</td>
</tr>
<tr>
<td>P_GV</td>
<td>( p_{gv} )</td>
<td></td>
<td>Fraction of energy intake from vegetable consumption coming from green vegetables</td>
</tr>
<tr>
<td>P_RV</td>
<td>( p_{rv} )</td>
<td></td>
<td>Fraction of energy intake from vegetable consumption coming from root vegetables</td>
</tr>
<tr>
<td>I_MEAT</td>
<td>( I_{meat} )</td>
<td>kg y(^{-1} )</td>
<td>Annual consumption rate of meat (calculated)</td>
</tr>
<tr>
<td>I_AIR</td>
<td>( I_{air} )</td>
<td>m(^3) y(^{-1} )</td>
<td>Annual inhalation rate of air (breathing rate)</td>
</tr>
<tr>
<td>O_R</td>
<td>( O_r )</td>
<td></td>
<td>Fraction of year spent at normal airborne dust concentrations (calculated)</td>
</tr>
<tr>
<td>O_F</td>
<td>( O_f )</td>
<td></td>
<td>Fraction of year spent at high dust concentrations</td>
</tr>
<tr>
<td>ETA_MILK</td>
<td>( \eta_{milk} )</td>
<td>kJ m(^{-3} )</td>
<td>Food energy content: milk</td>
</tr>
<tr>
<td>ETA_EGG</td>
<td>( \eta_{eggs} )</td>
<td>kJ egg(^{-3} )</td>
<td>Food energy content: eggs</td>
</tr>
<tr>
<td>ETA_FF</td>
<td>( \eta_{ff} )</td>
<td>kJ kg(^{-1} )</td>
<td>Food energy content: fish</td>
</tr>
<tr>
<td>ETA_GV</td>
<td>( \eta_{gv} )</td>
<td>kJ kg(^{-1} )</td>
<td>Food energy content: green vegetables</td>
</tr>
<tr>
<td>ETA_RV</td>
<td>( \eta_{rv} )</td>
<td>kJ kg(^{-1} )</td>
<td>Food energy content: root vegetables</td>
</tr>
<tr>
<td>ETA_G</td>
<td>( \eta_{gr} )</td>
<td>kJ kg(^{-1} )</td>
<td>Food energy content: grain</td>
</tr>
<tr>
<td>ETA_MEAT</td>
<td>( \eta_{meat} )</td>
<td>kJ kg(^{-1} )</td>
<td>Food energy content: meat</td>
</tr>
</tbody>
</table>
Table G-4: Parameters in the DSE-file which stores data on habits and practices for the model region (continued)

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Mathematical Symbol</th>
<th>Units</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$S_{G}$</td>
<td>$S_{gr}$</td>
<td>-</td>
<td>Surface contamination factor for grain</td>
</tr>
<tr>
<td>$M_{U,G}$</td>
<td>$\mu_{gr}$</td>
<td>m$^2$ kg$^{-1}$</td>
<td>Spray irrigation interception factor for grain</td>
</tr>
<tr>
<td>$Y_{G}$</td>
<td>$Y_{gr}$</td>
<td>kg m$^{-2}$</td>
<td>Yield of grain</td>
</tr>
<tr>
<td>$H_{G}$</td>
<td>$H_{gr}$</td>
<td>y$^{-1}$</td>
<td>Harvesting rate for grain</td>
</tr>
<tr>
<td>$S_{G,V}$</td>
<td>$S_{gv}$</td>
<td>-</td>
<td>Surface contamination factor for green vegetables</td>
</tr>
<tr>
<td>$M_{U,G}$</td>
<td>$\mu_{gv}$</td>
<td>m$^2$ kg$^{-1}$</td>
<td>Spray irrigation interception factor for green vegetables</td>
</tr>
<tr>
<td>$Y_{G,V}$</td>
<td>$Y_{gv}$</td>
<td>kg m$^{-2}$</td>
<td>Yield of green vegetables</td>
</tr>
<tr>
<td>$H_{G,V}$</td>
<td>$H_{gv}$</td>
<td>y$^{-1}$</td>
<td>Harvesting rate for green vegetables</td>
</tr>
<tr>
<td>$S_{R,V}$</td>
<td>$S_{rv}$</td>
<td>-</td>
<td>Surface contamination factor for root vegetables</td>
</tr>
<tr>
<td>$M_{U,R}$</td>
<td>$\mu_{rv}$</td>
<td>m$^2$ kg$^{-1}$</td>
<td>Spray irrigation interception factor for root vegetables</td>
</tr>
<tr>
<td>$Y_{R,V}$</td>
<td>$Y_{rv}$</td>
<td>kg m$^{-2}$</td>
<td>Yield of root vegetables</td>
</tr>
<tr>
<td>$H_{R,V}$</td>
<td>$H_{rv}$</td>
<td>y$^{-1}$</td>
<td>Harvesting rate for root vegetables</td>
</tr>
<tr>
<td>CATTLEFRAC</td>
<td></td>
<td>-</td>
<td>Fraction of farmed area devoted to cattle (not used)</td>
</tr>
<tr>
<td>$N_{C}$</td>
<td>$N_c$</td>
<td>animals m$^{-2}$</td>
<td>Stocking density of cattle</td>
</tr>
<tr>
<td>I_WC</td>
<td>$W_{wc}$</td>
<td>m$^3$ day$^{-1}$</td>
<td>Water consumption rate for cattle</td>
</tr>
<tr>
<td>I_PC</td>
<td>$I_{pc}$</td>
<td>kg day$^{-1}$</td>
<td>Pasture consumption rate for cattle</td>
</tr>
<tr>
<td>A_DRINK</td>
<td>$f_A$</td>
<td>-</td>
<td>Fraction of cattle water consumption taken from the well in the local aquifer</td>
</tr>
<tr>
<td>Z</td>
<td>$Z$</td>
<td>-</td>
<td>Ratio of weight of fresh pasture to hay</td>
</tr>
<tr>
<td>$S_{CP}$</td>
<td>$S_p$</td>
<td>-</td>
<td>Surface contamination factor for pasture</td>
</tr>
<tr>
<td>$M_{U,P}$</td>
<td>$\mu_p$</td>
<td>m$^2$ kg$^{-1}$</td>
<td>Spray irrigation interception factor for pasture</td>
</tr>
<tr>
<td>$Y_{P}$</td>
<td>$Y_p$</td>
<td>kg m$^{-2}$</td>
<td>Yield of pasture</td>
</tr>
<tr>
<td>$H_{PC}$</td>
<td>$H_p$</td>
<td>y$^{-1}$</td>
<td>Harvesting rate for pasture</td>
</tr>
<tr>
<td>I_WH</td>
<td>$I_{wp}$</td>
<td>m$^3$ day$^{-1}$</td>
<td>Water consumption rate for poultry</td>
</tr>
<tr>
<td>I_GH</td>
<td>$I_{gp}$</td>
<td>kg day$^{-1}$</td>
<td>Grain consumption rate for poultry</td>
</tr>
<tr>
<td>P_DRINK</td>
<td>$f_{poultry}$</td>
<td>-</td>
<td>Fraction of poultry water consumption taken from the well in the local aquifer</td>
</tr>
<tr>
<td>A_R</td>
<td>$a_r$</td>
<td>kg m$^3$</td>
<td>Airborne dust concentration: value for normal background conditions</td>
</tr>
<tr>
<td>A_F</td>
<td>$a_f$</td>
<td>kg m$^3$</td>
<td>Airborne dust concentration: value for high and/or occupational conditions</td>
</tr>
</tbody>
</table>
H.  **TAME implementation and life history**

H.1  **Computer implementation**

The mathematical description of TAME in Section 2 shows that there is a clear distinction between the biosphere transport and exposure pathway sub-models. The computer implementation of TAME uses this division to calculate:

1. The transfer coefficients for the biosphere transport sub-model
2. The compartment inventories as a function of time
3. Environmental concentrations
4. Foodstuff concentrations
5. The annual individual dose arising from the exposure pathways as well as the sum over pathways and (where appropriate) the sum of members in the decay chain.

Figure H-1 shows the relationship between the various modules used in TAME calculations. Execution of the model portions is controlled by machine-specific job-control procedures. To date TAME has been implemented under MS-DOS, VMS, UNIX, OS2 and Windows 95, and has given consistent results. The TAME codes themselves are written in enhanced FORTRAN 77.

Since its earliest implementations in 1992, TAME has been upgraded and modified a number of times.

H.2  **Life history of the TAME codes**

Early versions of TAME (version 1a) were first run in 1991. Since then there has been an almost constant tide of development which has considerably enhanced the capabilities of the code. The main sequence of development has been TAME 1 - TAME 2 - TAME 3, but along the way there have been numerous sub-variants.

Figure H-2 lists the different versions of TAME. Each new assessment project generates a slightly different version of the codes. The major changes are indicated by a change in the identification number, less significant changes are designated by a change to identification letter. The model results presented in Section 4 were produced using TAME 1c. At each stage of development, the new versions is not accepted into service before the results of the earlier version have been duplicated. This is achieved using a similar process to that described in Appendix F. Currently TAME 3b is being developed for the next phase of the Wellenberg assessments. The versions of TAME are summarised below.
Figure H-1: Organisation of the TAME computer codes. The three main sections of the TAME codes are shown on the left of the figure. The control routines TAMEexec and DOSEexec have been created for TAME to handle input, output and processing and interfacing with the numerical integration routines† used to solve Equation (1). The right-hand side illustrates the input and output files used by TAME. User-defined input files are denoted by italics, output files containing model results are underlined. The log-files generated by TAMEexec and DOSEexec contain lists of all input parameters as well as the values of derived parameters.

† Two numerical integration methods are available: ACTIVI from the BIOPATH code package (BERGSTRÖM et al., 1985) and SIEM (Semi-Implicit Extrapolation Method) (KLOS et al., in preparation). This latter has been developed recently as an alternative to ACTIVI and is more suited to working with TAME. A detailed discussion is outside the scope of this report.
Since 1991, the abbreviations used are: K3 - Kristianfjell-2 assessment, Swiss - Swiss Institute for Ecological Research, W/E - Wellcome Environmental Research Group, SEM - Semi-empirical extrapolation method, and SSS - Stochastic Sampling System.
• **TAME\_Ia**

The prototype version of TAME. This was used to demonstrate the feasibility of using input files which were readable by both humans and computers. Job control and runtime log-files also set up. Radionuclide transport only defined on the basis of the biosphere transport model used in the PSACOIN Level 1b intercomparison (NEA, 1993). Mass balance not included in the data definition. Solution to the compartment model equations obtained by interfacing the TAME codes to the ACTIVI routines from the BIOPATH codes (BERGSTROM et al., 1985).

• **TAME\_Ib**

Radionuclide transport based on mass balance for water and solid material fluxes included. Pre-calculated mass balance data input to the code in the data-file describing the biosphere. Implementation of the MiniBIOS exposure pathways sub-model. Verified against BIOSPH (see Appendix F). Operational Version used in the Kristallin-1 and Wellenberg safety assessments (NAGRA, 1994a, NAGRA, 1994b).

• **TAME\_Ic**

The case specification of the BIOMOVS II working group on Complementary Studies (KLOS, 1993) called for the comparison of compartment inventories, doses and foodstuff concentrations. This latter was added to the output options. Otherwise the version is the same as TAME 1b. Used in the calculations of the Complementary Studies deterministic cases. Good agreement achieved with comparable models.

• **TAME\_Id**

In the Kristallin-1 assessment a special version was required to calculate potential doses arising from a tundra climate biosphere (NAGRA, 1994b). Specially modified, non-standard exposure pathway sub-model for lichen and reindeer. Modified database. Demonstration of the feasibility of performing such calculations with TAME.

• **TAME\_Ie**

Export version supplied to the Rez Nuclear Institute, plc, Czech Republic. Minor modifications relative to TAME\_Ic - exposure pathways sub-model of principle interest. Used in the IAEA NSARS study (see IAEA, 1995).

• **TAME\_2a**

Additional routines to interface TAME with the Swiss Stochastic Sampling System (SSSS) codes were added to TAME\_2b to verify the feasibility of performing multiple-run probabilistic calculations using TAME.

• **TAME\_2b**

Before full stochastic runs could be carried out using TAME, the codes had to be able to handle potentially uncertain biosphere data and to derive for themselves the mass balance schemes for use in calculating the radionuclide transfer coefficients. This was achieved in TAME\_2b and full implementation of the stochastic capability was made possible. Advantage was taken of the upgrade to install a few additional options into the codes. Inhalation doses for volatile radionuclides emanating from the top soil (with corresponding consequences for the transport of radionuclides in the soil/aquifer system) were added. Additional routines were inserted into the transport...
and dose sub-models to allow for special coding for special radionuclides. This has not so far been developed, but the potential to treat some radionuclides differently, e.g. $^3$H or $^{14}$C, differently is now possible. Several investigative cases have been run with TAME$_{2b}$ to test the capabilities of SSSS and TAME together.

- **TAME$_{3a}$**
  Experience with TAME in the Kristallin-1 and Wellenberg assessments showed that the ACTIVI integration routines could be unreliable and uninformative under certain, unpredictable circumstances. Also, as seen in Figures 4-1, 4-8 and 4-13, there were numerical problems near to sharp edges in the source term, producing rounded-off results starting well before the edges themselves. Consequently, a new set of integration routines - SIEM the Semi-Implicit Extrapolation Method (KLOS et al., in preparation) - was commissioned to replace the old version. This necessitated a revision of the interface routines between the setting up of the TAME transfer coefficients and the integration routines. This greatly streamlined the interface since SIEM was designed to be more user-friendly than ACTIVI. Otherwise the codes were identical to TAME$_{2b}$. The new integration routines are also around a factor of ten faster than their old ACTIVI counterparts. The calculations for the stochastic phase of the BIOMOVS II Complementary Studies exercise have been carried out using TAME$_{3a}$.

- **TAME$_{3b}$**
  Currently under development and testing is TAME$_{3b}$. This will use SIEM and will also include a number of refinements, many of which result from a better understanding of the biosphere in the Engelberger-Tal. TAME$_{3b}$ will become the standard for future safety assessments. For some time the fate of radionuclides leaving the release region has been of concern. TAME$_{3b}$ will be capable of performing calculations for chains of biosphere regions, so that the effects of the release to the biosphere downstream from the source can be evaluated. Similarly, temporal biosphere chains will also be possible - i.e. calculations up to a given time can be performed with one set of transfer coefficients, before changing to a set of values representing different biosphere conditions are read in.