9. BIOSPHERE MODELLING

9.1. Aim

This section considers biosphere modelling. In particular, it:

- provides a definition of the biosphere and considers its importance, and associated key issues, in safety assessments (Section 9.2);
- identifies the key radionuclide transport and exposure mechanisms that are typically identified when developing a conceptual model of the biosphere (Section 9.3);
- considers their representation in mathematical models (Section 9.4);
- identifies some example computer codes that are used for implementing the mathematical models (Section 9.5); and
- discusses typical data requirements for model formulation and implementation (Section 9.6).

9.2. Introduction

9.2.1. Definition

The biosphere can be defined the physical media (atmosphere, soil, sediments and surface waters) and the living organisms (including humans) that interact with them (Figure 9.1).

As part of the IAEA's BIOMASS programme, consideration has been given to the identification and description of the biosphere for the purpose of the assessment of solid radioactive waste disposal [IAEA, 2002a, b, c]. IAEA [2002b] identifies the following components of the biosphere that are described further in Section 4:

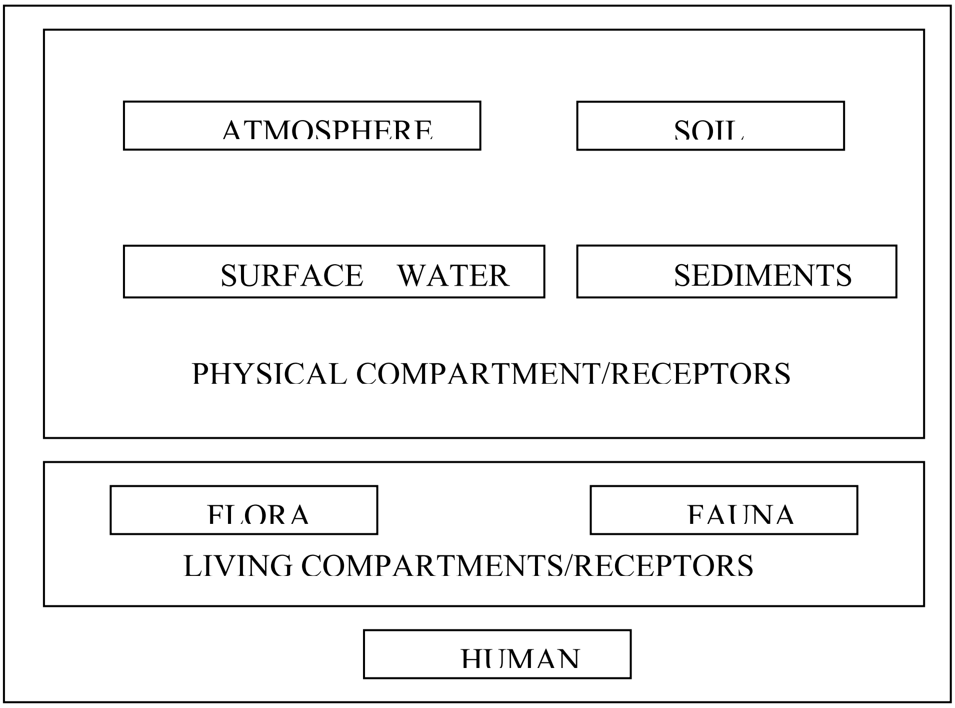
- climate and atmosphere;
- water bodies;
- human activity;
- biota;
- near-surface lithostratigraphy;
- topography;
- geographical extent; and
- location.

9.2.2. Importance and Key Issues

Most of the end-points that are considered in safety assessments require the modelling of the migration and accumulation of radionuclides in the biosphere (see Section 3). Furthermore, those relating to the impact of radionuclides on humans (i.e. dose and risk) also require consideration of the habits and activities of humans. Thus it is necessary to analyse and quantify the transport and radiological impact of radionuclides released from the disposal facility into the biosphere in order to demonstrate the acceptability of radioactive waste disposals.

The biosphere (and associated chemical, physical, and biological processes) forms the link between the near-field and geosphere (and associated processes which affect radionuclide release and transport from the disposal facility) and the ultimate radiological impacts to be assessed. The boundary between the near-field and the biosphere and the geosphere and biosphere will be context and system specific and must be consistently defined within the assessment. As described in Sections 7 and 8, radionuclides can be released into the biosphere from the near-field and geosphere in liquid, gaseous and solid form. When analysing and quantifying such releases, the assessor is faced with a number of challenges that need to be addressed:

Figure 9.1: A Simplified Graphical Representation of the Biosphere Components [Torres and Simón, 1997]



- in common with the other components of near surface disposal systems, the biosphere can be subject to change, even over relatively short timescales (for example decades), due to natural mechanisms and human actions [IAEA, 2002b];

- biosphere modelling potentially requires consideration of the future activities and behaviour of humans over timescales of many thousands of years; and

- a number of different interfaces can exist between the biosphere and the near-field and the geosphere depending on the nature of the release.

  - For liquid releases, the release might be from the geosphere via groundwater discharge to a well, soil and/or surface water body and associated sediment (lake, river, estuary, sea). In addition, the release might be direct from the near-field to the biosphere if bathtubbing occurs (see Section 7).

  - For gaseous releases, the release might be from the near-field and/or geosphere into the atmosphere and/or a building.

  - For solid releases, the release might be from the near-field and/or geosphere. Releases from the near-field might result from disruptive events such as human intrusion, biotic intrusion and erosion (see Section 7). Whilst releases from the geosphere might result from colloidal transport and/or the erosion of geosphere previously contaminated by the sorption of radionuclides in groundwater.

The complexity of this interface issue can be further compounded by the fact that its nature can change with time. For example consider the discharge of radionuclides to a surface water body through bed sediments. Radionuclide concentrations in and transport through sediments could be important because of the implications for the rate of release into surface waters, but also because sediments may later be converted by natural or human influenced processes (e.g. siltation or land reclamation) into a substrate on which crops and other plants can grow.

9.3. *Conceptual Models*

Conceptual models for the migration of radionuclides in the biosphere and the resulting exposures are discussed below. In common with transport from the near-field (Section 7) and through the geosphere (Section 8), radionuclide decay and in-growth has to be considered when modelling transport through the biosphere and subsequent exposure. However, the timescales of interest can be significantly shorter than those considered in the near-field and geosphere and often any daughter with a half-life greater than 25 days is explicitly modelled (see for example IAEA [2002c]). The selection of a half life of greater than 25 days is justified because biosphere transport processes that might disturb secular equilibrium of the decay chains can operate over such timescales. Daughter radionuclides with a half-life less than 25 days are assumed to be in secular equilibrium with their parents in all parts of the biosphere (for example Bi-210 with Pb-210). Many radionuclides which have a half-life of less than 25 days have generally lower dose per unit intake than their longer lived parents and cannot accumulate in any part of the biosphere to a level significantly different from that of their parents because of their short half-life. One radionuclide of possible interest with a half-life of less than 25 days is Rn-222. It has a half-life of 3.8 days and is very mobile and in some circumstances cannot be considered to be in secular equilibrium with its Ra-226 parent. Consideration of Rn-222 requires explicitly modelling of the gaseous phase.

9.3.1. Transport in Surface Waters

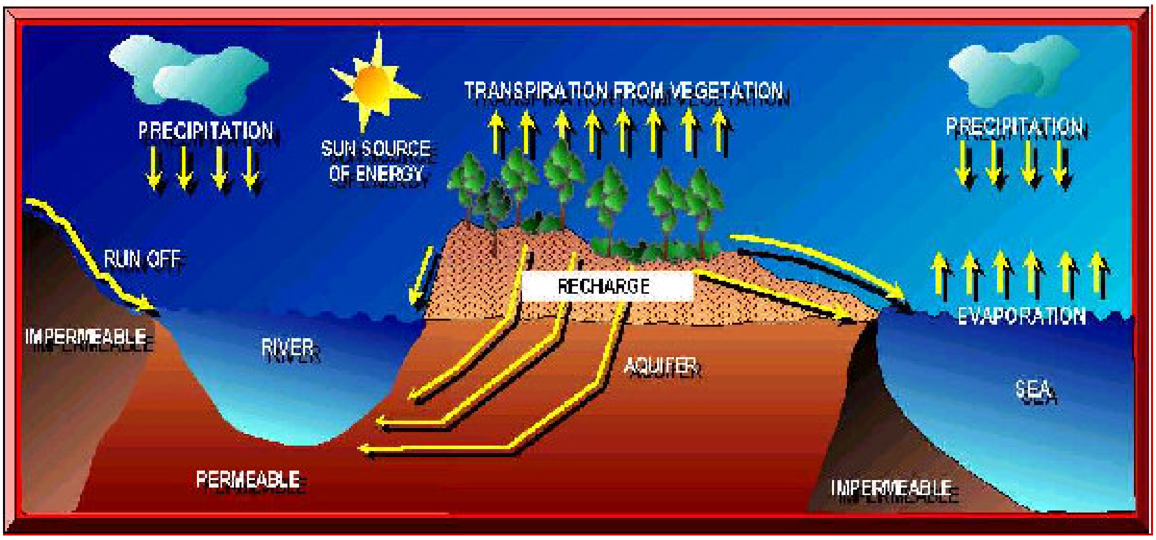
Figure 9.2 summarises the hydrological cycle which is the driving force for the key processes that affect the transport of radionuclides into and between surface water bodies. Radionuclides released to surface waters are subject to a series of physical and chemical processes which affect their transport from the source point. These processes include:

- flow processes
  - down-current transport (advection)
  - mixing processes (turbulent dispersion)

- sediment processes
  - sorption/sorption by suspended, shore/beach and bottom sediment
  - down-current transport, deposition, and resuspension of sediment, which absorbs radionuclides

- other processes
  - mechanisms that will reduce concentrations in water such as radionuclide volatilisation

Figure 9.2: The Hydrological Cycle [Torres and Simón, 1997]



These processes are, in general, three-dimensional and transient in nature. However, for the purposes of safety assessments, these processes are often assumed to be at a steady state, with mixing considered to be either complete or to change in one or two dimensions.

Four different types of surface water body can be distinguished (Figure 9.3) [Till and Meyer, 1983].

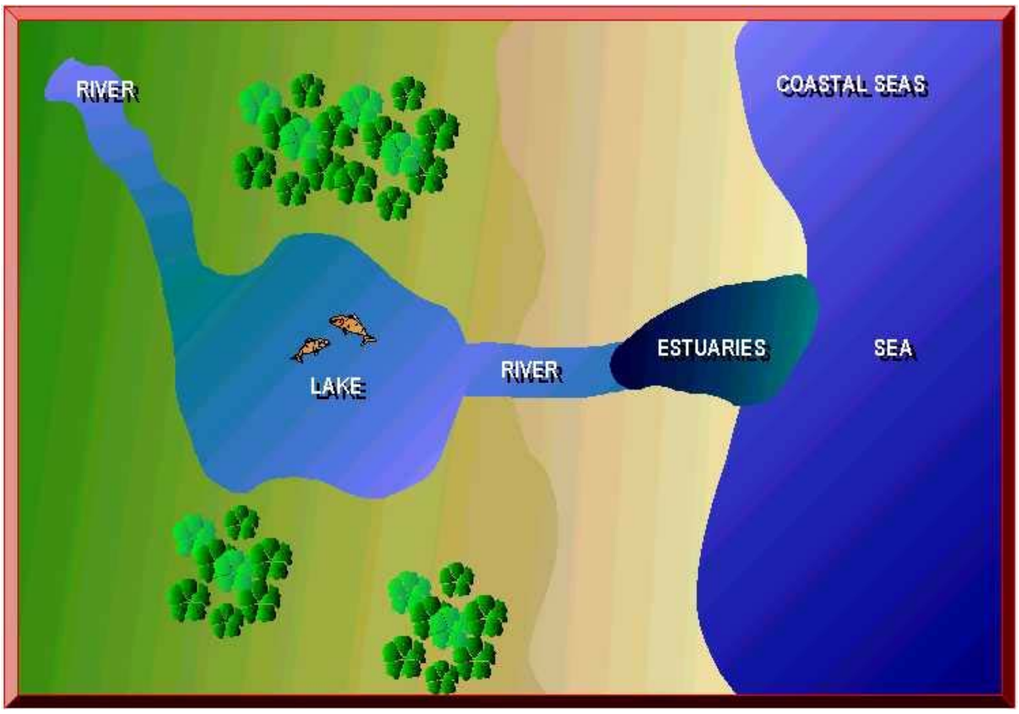
- **Rivers** - The movement of radionuclides in a river is mainly affected by advective flow and radionuclide sorption (with suspended sediments and bed load sediments). Rivers are dynamic systems, and so it is often necessary to simplify their representation in terms of their geometry and dominant processes, for modelling purposes.

- **Lakes** - The radionuclide transport processes in lakes are, in principle, similar to the ones in rivers, although some differential characteristics must be considered. In particular, the residence time in the lake which can sometimes be long, and the stratification in the lake, due for example to seasonal temperature differences. In such cases the lake cannot be considered as one unique water body; it is then necessary to model the lake as two or more separated compartments considering their relative volumes and transfer rates (see for example BIOMOVS II, 1996a].

- **Estuaries** - Being intermediate between rivers and seas, estuaries have a number of peculiar characteristics. They are zones within which there is a substantial change in salinity, typically from 0.5 parts per thousand dissolved solids (or less) in a river to about 35 parts per thousand dissolved solids for the open sea. This change in salinity has a profound influence on concentrations of radionuclides in both sediments and biota. The salinity is clearly subject to variation within an estuary, and values are generally somewhat lower for an estuary than they are in the sea because of the dilution effect of fresh water from the river. However, except for the upper reaches of the estuary, salinity values will be closer to those for sea water than those for fresh water. Estuaries are energetic systems, especially where there is a strong tidal influence. This leads to good mixing, of both water and sediment, and often results in high suspended-sediment concentrations, thus promoting conditions favourable to depletion from the water of those radionuclides which have affinity for sediment. In any case, estuaries tend to be sediment-rich areas, with material brought down by the rivers and perhaps also derived from the sea.

- **Seas** – The consideration of coastal seas necessarily covers a range of different situations, from those which are part of ocean basins to those which, by being nearly land-locked, have only restricted interchange of sea water. The salinity of the latter is generally subject to wide variation, whereas in the open ocean seas it is more nearly uniform. Just as salinity is variable, so too are suspended loads and other sediment conditions. In general, suspended sediment loads are lower than in estuaries. However, in seas with a strong tidal influence and, for example, in shallow water near to coasts, conditions do exist which act to promote strong interaction between bed sediment and sea water, leading to considerable depletion of activity from the water and accumulation of radionuclides in sediment. At the coast, this sediment may be transferred to the terrestrial environment due to flooding and wind erosion.

Figure 9.3: Surface Water Bodies [Torres and Simón, 1997]



9.3.2. Transport in Soils

Radionuclides in an unsaturated soil are subject to a number of transport mechanisms [Torres and Simón, 1997]:

- the mass flow of dissolved radionuclides with the movement of soil water; vapour diffusion of gaseous radionuclides through both the vapour and liquid phases of the soil;
- molecular diffusion along a concentration gradient;
- diffusion along potential gradients induced by temperature differences in the soil;
- movement in association with fine particles, microbes or colloids; and
- mechanical mixing through processes such as ploughing and bioturbation.

Since water flow controls radionuclide transport, radionuclides can move upward or downward through the soil profile. For radionuclides deposited on the soil surface from the air or with irrigation water, root-zone concentrations are determined primarily by downward leaching and sorption. For radionuclides reaching the bottom of the soil profile with groundwater, capillary rise also plays a role. These radionuclides move upward through the profile during dry periods, and some absorb to the soil. A portion is leached back down following the next rainfall, but the remainder continues to rise to the surface in subsequent dry periods. In this way, radionuclides can reach the root zone from the watertable even in areas where the net flow of water through the soil is downward.

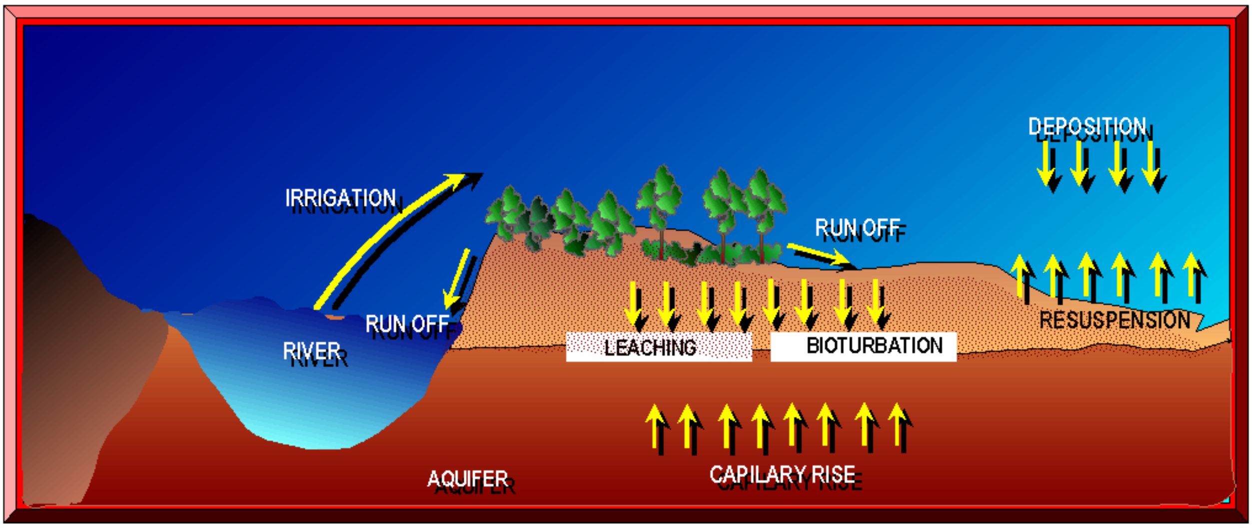
Hence, transport of radionuclides depends on the interactions of the deposited materials with various soil compounds; the association of trace substances in soils can be summarised broadly as:

- precipitation with soil components to form a new solid phase;
- occlusion during formation of new solid phase;
- incorporation into organic matter and micro-organisms;
- sorption onto charged surfaces of clays, precipitates and organic matter;
- inclusion in soil materials (e.g. during mineralization or later via solid state diffusion).

These associations will affect the mobility and transport of elements within and from a soil and hence the supply to plants via root uptake.

A number of processes act to deplete radionuclide concentrations in soil (Figure 9.4). Radionuclides may flow out of the bottom of the profile with drainage water and be lost to the regional groundwater system. During wet weather, some may escape the soil with surface runoff. Radionuclides may be lost to the atmosphere through suspension of contaminated particulate matter, or though gaseous evasion. Finally, radionuclides may be taken up by plants through their roots, although a portion of these may return to the soil when the plant dies and decays.

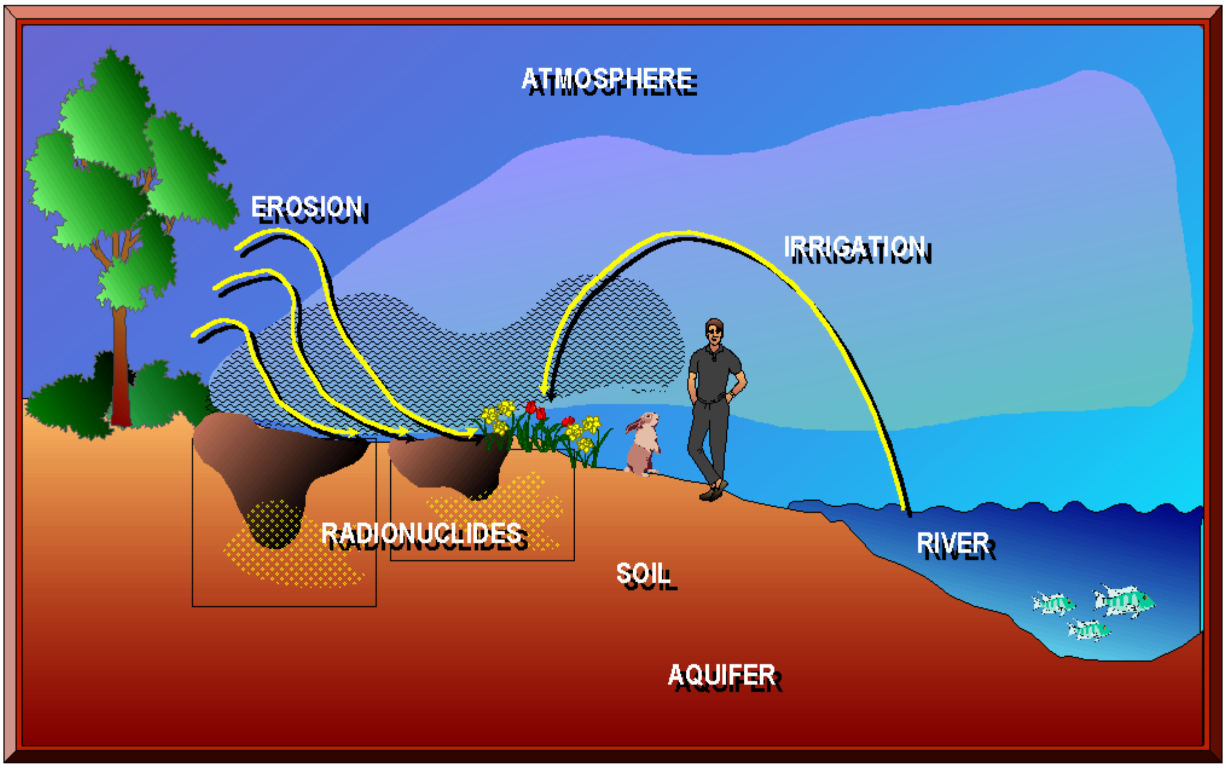
Figure 9.4: Example Soil Transport Processes [Torres and Simón, 1997]



9.3.3. Transport in the Atmosphere

Radionuclides can be transported into and through the atmosphere in the solid (e.g. attached to suspended material), liquid (e.g. dissolved in spray irrigation water or sea-spray) and gaseous (e.g. radioactive gases) phases. Radionuclides in the air in particulate and gaseous form will be transported by advection and dispersion. Dry deposition and wet deposition due to precipitation can deplete the concentrations of radionuclides in the air, and deposit them on the surface soils, the surface water bodies, and the surfaces of vegetation the resultant exposure pathways for dose (see Figure 9.5).

Figure 9.5:Example Transport Processes in the Atmosphere [Torres and Simón, 1997]



9.3.4. Transport to/within Flora and Fauna

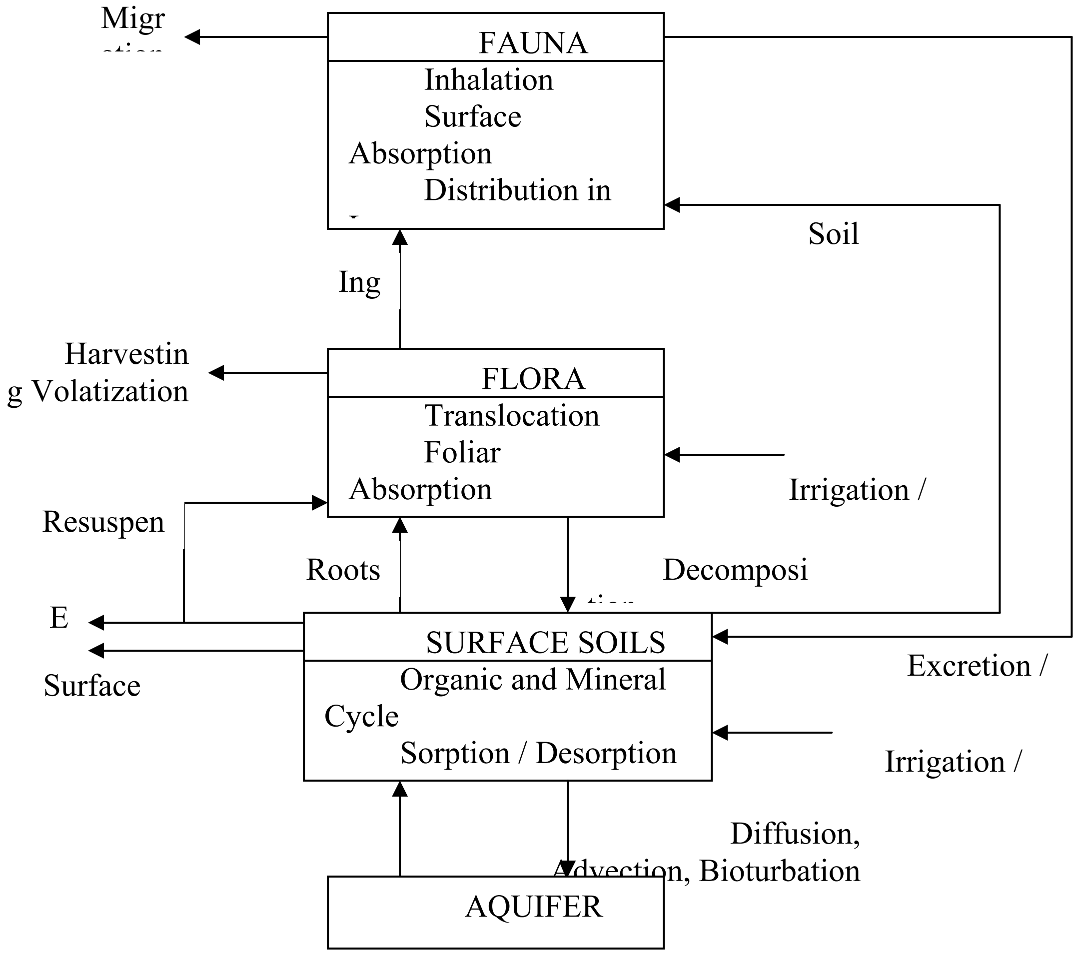
Radionuclides in the physical compartments of the biosphere (surface water, sediment, soil and the atmosphere) may be taken up by living organisms, both plant and animal (Figure 9.1). These radionuclides may affect the organisms and may move along the foodchain and eventually be ingested by humans. It is helpful to distinguish between the flora and fauna associated with terrestrial ecosystem and the aquatic ecosystem.

When considering terrestrial flora and fauna, it is necessary to consider the following processes either explicitly or implicitly (Figure 9.6) [Torres and Simón, 1997]:

- deposition by dry or wet processes;
- interception and retention by vegetation surfaces;
- translocation from sites of deposition to the edible tissues of vegetation;
- post-deposition retention by vegetation and soil surfaces;
- uptake by roots;
- direct ingestion of surface soil by grazing animals;

- transfer of contamination from soil, air, water and vegetation into the milk and meat of grazing animals;
- transfer of contamination from surface water to the terrestrial system via spray;
- transfer of contamination from surface water to sediment and to aquatic biota; and
- transfer of contamination from groundwater to the terrestrial system.

Figure 9.6: Example Transport Processes in a Terrestrial Ecosystem [Torres and Simón, 1997]

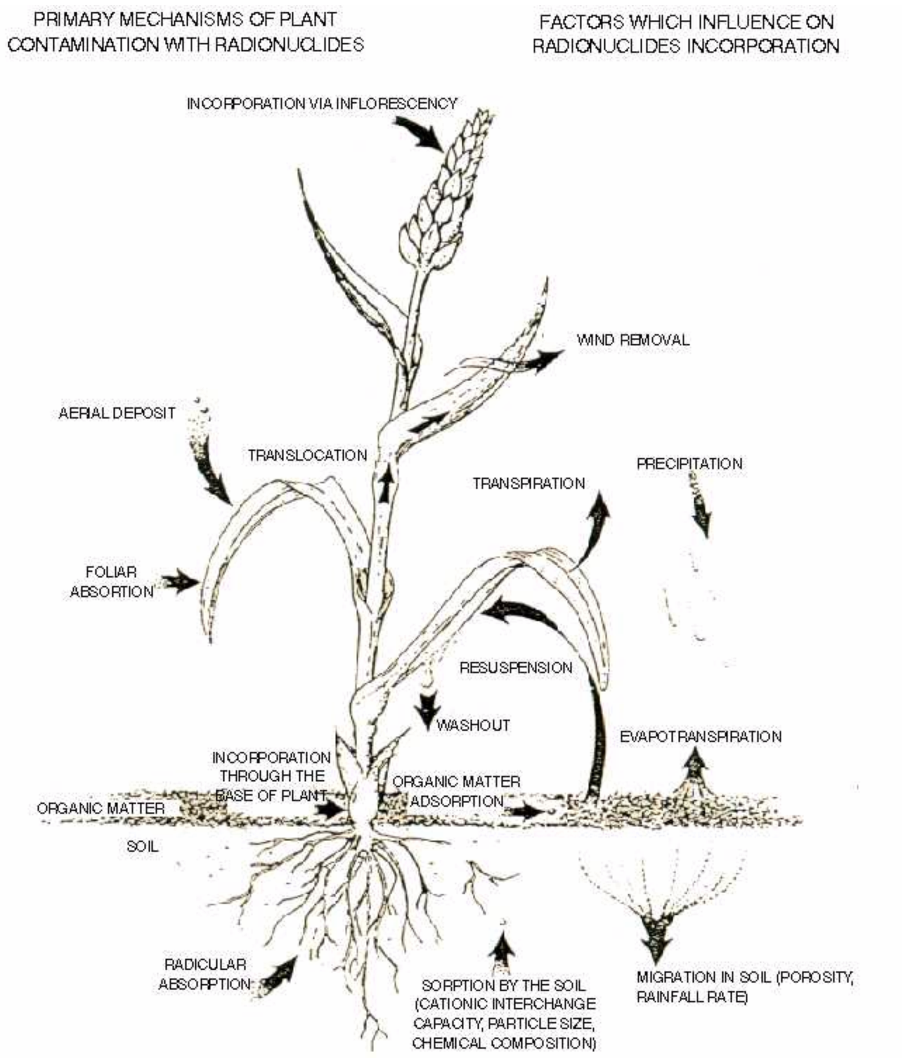


Contamination of vegetation (Figure 9.7) may result from the interception of radionuclides from either the atmosphere or *hydrosphere*[1]. Radionuclides retained on vegetation may come from fallout, washout, rainout, from irrigation with contaminated water, and from resuspended matter. External deposits can be taken up by foliar absorption into plants. Another source of plant contamination is the uptake of radionuclides from soil via roots and internal redistribution of the various parts of the plants. Processes in addition to radioactive decay can lead to the reduction of radionuclide concentrations in vegetation. These processes include growth dilution, wash-off of externally deposited radionuclides, leaching

[1] Definitions for all terms appearing in italics are given in the glossary to these notes. Only terms that have not been used in previous sections are italicised in this section.

during rainfall and soil fixations. Further removal of radioactive material from vegetation can occur due to grazing and harvesting.

Figure 9.7: Crop Processes [Till and Meyer, 1983]

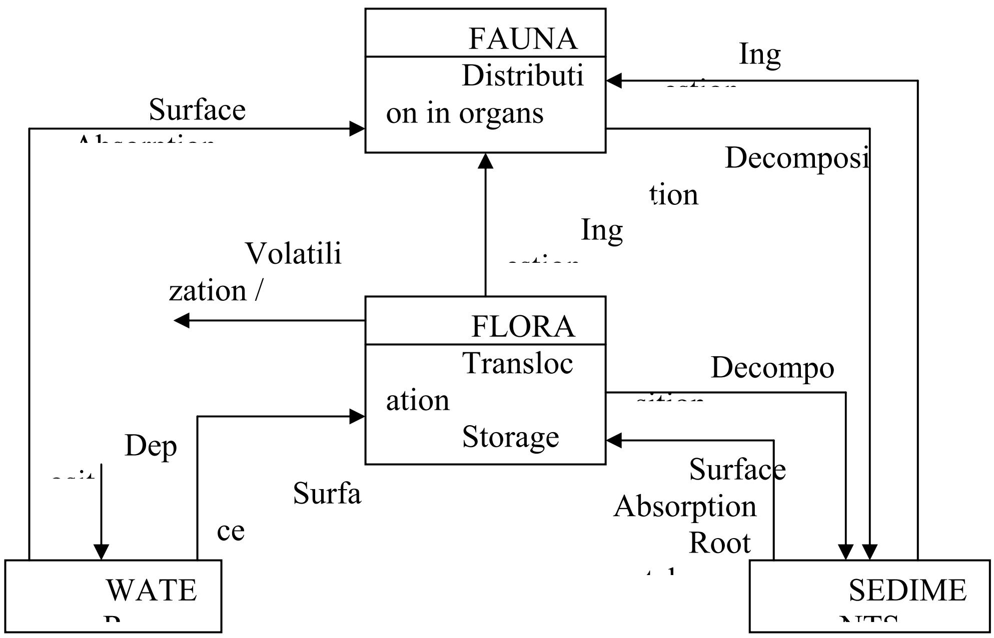


It is often difficult to separate those losses which are attributable to leaching during rainfall from those due to abscission, mortality and resuspension. In this regard, it is emphasised that all processes of uptake and loss from plants can be expected to be related not only to the matturity of the plant but also to the temperature, surface wetting, humidity, chemical form of the substance concerned and general nutritional status.

Radionuclides are taken in by animals via ingestion of contaminated feed, soil or water. Other, usually less important, exposure pathways for animals are inhalation of contaminated material and external irradiation from contaminated material. The relationship between concentration in animal products, e.g. meat, milk and eggs, and the daily intake of contaminated material can be obtained experimentally or via use of compartment models representing the distribution of radioactivity in the body and animal products after ingestion/inhalation. As is the case for all biological systems, natural variation is inevitable. The relationship between activity in animal products and feed is uncertain because uptake and retention in the body is dependent among others on age and condition of the animal, morphological characteristics of the plant, chemical form of elements in plants, the difficulty in representing short term fluctuations in activity content of food and uncertainties in the value of gastro-intestinal transfer. The transfer of a radionuclide from animal feed to a food product depends upon the metabolism of the animal.

The aquatic ecosystem comprises the freshwater (aquatic), saltwater (marine) and brackish water (estuarine) environments. The estuarine ecosystem usually acts as a bridge between the other two environments. This interface happens most frequently when rivers and streams flow into bays and other arms of the sea. Although different species may occupy the same niche in different systems, the components of the systems are similar and can be modelled and considered together. Key processes are summarised in Figure 9.8.

Figure 9.8: Example Transport Processes in a Aquatic Ecosystem [Torres and Simón, 1997]



The behaviour of radioactive materials in the various waters is not necessarily the same. The physicochemical form of the radionuclide is generally more important in determining bioaccumulation in these ecosystems than in the terrestrial ecosystem and may influence the behaviour and transport of radionuclides by changing their physical or chemical characteristics. Microbes can change the oxidation state of some elements through metabolic activity, and thereby increase or decrease their mobility. They can create toxic mobile organic compounds through methylation in sediments and soils. Various microbially generated chelating agents can target otherwise immobile elements and form organic complexes with increased mobility and bioavailability. Microbes may change the absorption of some elements during root uptake by plants and digestion by herbivores. The exact extent by which microbes affect radionuclide transport in the biosphere is variable. For example, root uptake of elements by many plants depends on mycorrhizal associations between the host plant and soil-living fungi.

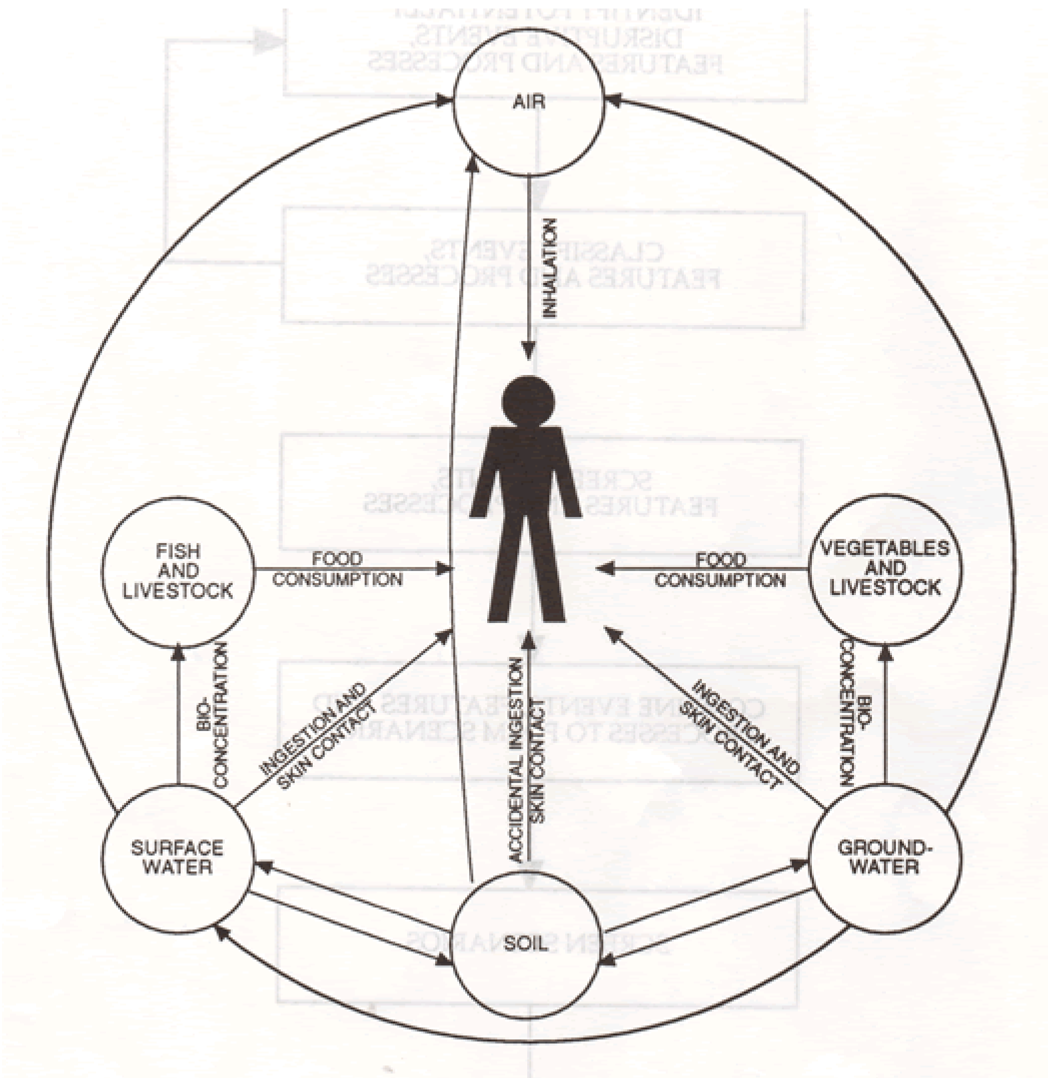
9.3.5. Human Exposure

Radiation doses to humans can occur via any of four exposure modes:

- ingestion;
- inhalation;
- external irradiation; and
- by transfer of radionuclides through the skin, by puncture or absorption.

Although, all four modes can be considered, the fourth one is not usually considered on the basis of its low significance [BIOMOVS II, 1996b]. Humans living in a contaminated environment can receive a radiological dose via a multitude of exposure pathways (see for example Figure 9.9) depending on the characteristics of the release, the biosphere media, and the human habits. As noted in Section 9.2.2, assumptions also have to be made concerning the particular behaviour of the individuals or populations for whom a calculation of health impact is required. Of special interest are the assumptions for the behaviour of groups whose exposure is representative of the highest that may be expected, commonly referred to as the critical group (see Section 3).

Figure 9.9: Example Exposure Pathways for Humans [Kountzman and Tucker, 1989]



In addition to the calculation of exposure of humans, consideration in some assessments is being given to the impact of exposure on non-human biota (see Section 3). For example there is the FASSET programme of the European Commission that has the aim of providing a reference set of models, dosimetric factors, etc, for generic organisms and ecosystems [Strand and Larsson, 2001].

9.4. Mathematical Models

Mathematical models that have been developed to represent biosphere transport and exposure processes range in complexity from simple expressions to highly complex mathematical algorithms [Torres and Simón, 1997]. The models can be split according to their degree of complexity in two categories.

Mechanistic Models. They describe processes in a physically realistic manner and are normally specific to a given process (e.g. erosion of soil). For example, in the case of erosion, they would calculate the flux of radionuclide from the source to the atmosphere, and then let this material be dispersed according to typical fluid motions. In most cases, mechanistic models are quite complicated and reflect the state of the art knowledge about the process. This often makes them specific, and not always applicable to a large range of processes. For example, UK Nirex have developed the SHETRAN code [Thorne, 1995] to model the migration of radionuclides within a hydrological catchment area within three dimensions. Such codes tend to be resource intensive and their contribution to an assessment of a facility is restricted to increasing the understanding of certain processes which can then be incorporated into the more simplified code used for the assessment.

Transfer Coefficient Models. Simple transfer coefficient models do not describe the physical processes in a detailed mechanistic way, but instead are based on measurements made of contaminants in two different media. The transfer coefficient is then inferred from these relationships. Although this approach is often not scientifically rigorous it considers many parts of a process implicitly, it is simple, and it is usually based on real data. Thus, it can be appropriate for assessment purposes especially given the considerable uncertainties associated with the future evolution of the biosphere.

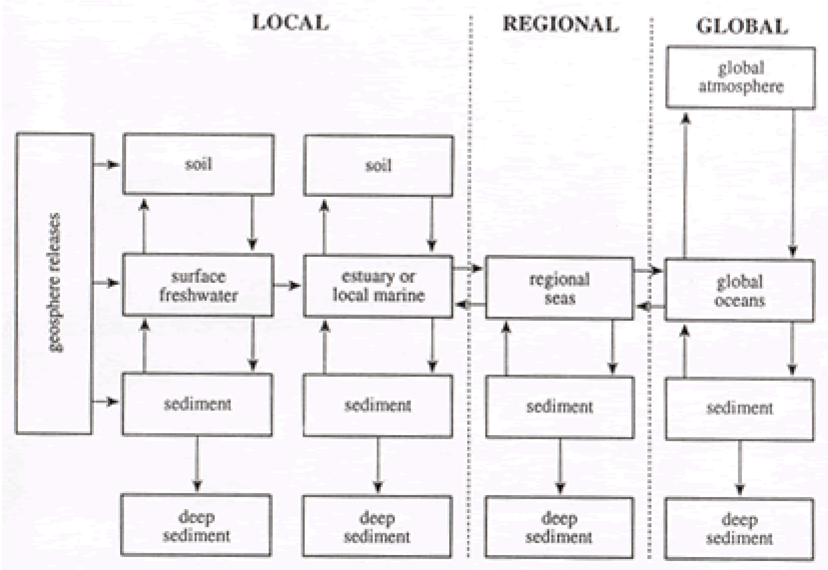
The focus of the following sections is on the transfer coefficient models.

9.4.1. Transfer between Physical Media

Most of the biosphere assessment models are based on linear donor-controlled compartment models (see for example NEA [1993] and BIOMOVS II [1996c]). Such models assume that a system may be represented by breaking it down into compartments, each of which may represent a medium which is distinct from other associated media. It is assumed that, as soon as material (in this case radionuclides) enters a compartment, instantaneous mixing occurs so that there is a uniform concentration over the whole compartment. Each compartment must be chosen to represent a region of the environment for which this assumption is reasonable. Figures 9.10 and 9.11 show some example compartments. The regional and global compartments shown in Figure 9.10 are generally only necessary if it is intended to calculate collective doses over large populations. Special models have been developed for this purpose specifically for very long lived and mobile radionuclides such as I-129 [Smith and White, 1983].

Radionuclides in one compartment may be transferred to another by various processes. The transfer is described by transfer coefficients that represent the fraction of the activity in a particular compartment transferred from that compartment to another one in unit time. Radionuclides can also be lost from the system altogether (by radioactive decay).

Figure 9.10: Compartments from a Generic Biosphere Model used for the Calculation of Collective Doses [Lawson and Smith, 1985]



The mathematical representation of the intercompartmental transfer processes (shown graphically in Figures 9.10 and 9.11) takes the form of a matrix of transfer coefficients that allow the compartmental amounts to be represented as a set of first order linear differential equations. For the i^{th} compartment, the rate at which the compartment inventory changes with time is given by:

$$\frac{dN_i}{dt} = \left(\sum_{j \neq i} \lambda_{ji} N_j + \lambda_N M_i + S_i(t) \right) - \left(\sum_{j \neq i} \lambda_{ij} N_i + \lambda_N N_i \right)$$

where:

N_i is the activity of radionuclide N in biosphere compartment i, Bq,

M_i is the amount of radionuclide M in biosphere compartment i (M is the precursor radionuclide of N in a decay chain), Bq,

$S_i(t)$ is an external source term of radionuclide N to compartment i, Bq y^{-1},

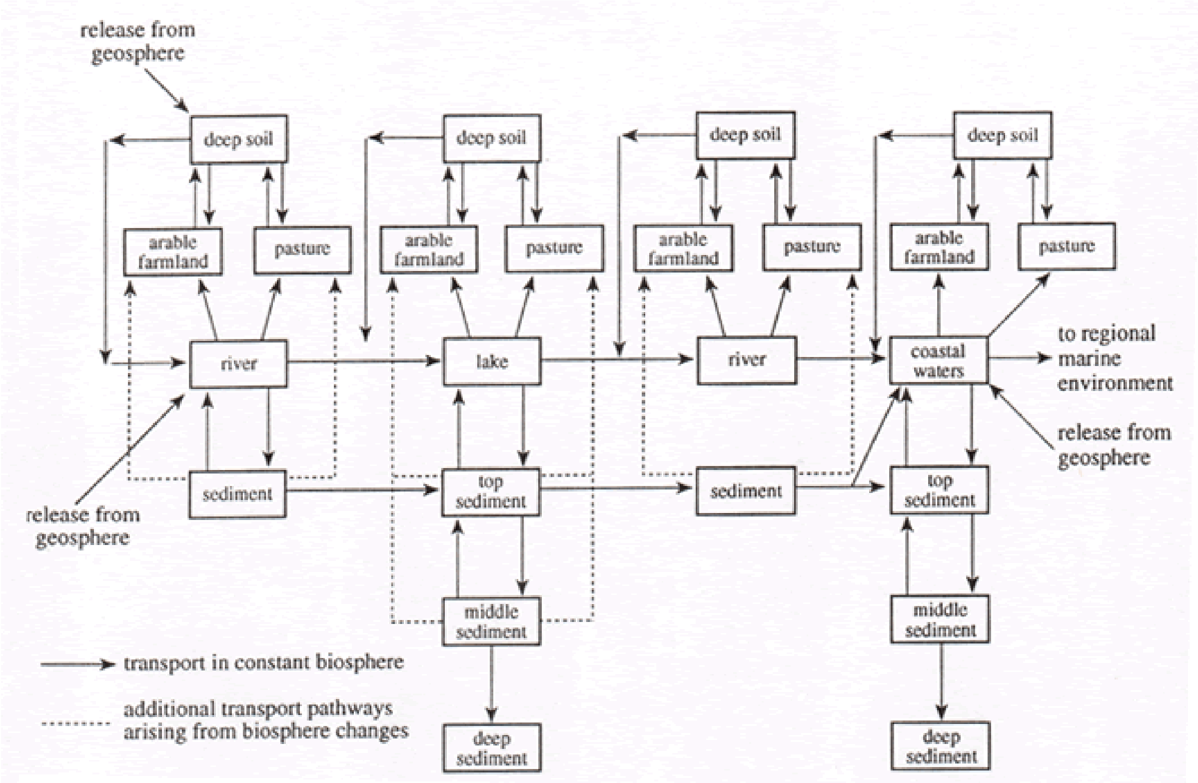
λ_N is the decay constant for radionuclide N, y^{-1},

λ_{ji} is the transfer coefficient to compartment i from compartment j or to sinks, y^{-1}

The solution of the matrix of equations given above provides the time-dependent inventory of each compartment. Assumptions for compartment sizes then result in estimates of concentrations in the corresponding media.

The transfer coefficients can represent the single or multi-phase movement of radionuclides. Some examples are given below based on information given in IAEA [2002c]. Further examples are provided in publications such as IAEA [2002c], and BIOMOVS II [1996c].

Figure 9.11: Example Local Biosphere Model used for the Calculation of Individual Doses [Lawson and Smith, 1985]



9.4.1.1. Example of Liquid Phase Transfer

The transfer of radionuclides to the sub-soil compartment from the surface soil compartment due to infiltration (and other downward processes), λ_D [y^{-1}], is given by:

$$\lambda_D = \frac{D}{R\theta\, V}$$

where D is the volume of recharge from the surface soil compartment [$m^3\ y^{-1}$]
R is the retardation coefficient for the surface soil compartment [-]
θ is the water filled porosity of the surface soil compartment [-]
V is the volume of the surface soil compartment [m^3]

The R term is calculated using the following equation:

$$R = 1 + \frac{(1-\theta_t)\rho}{\theta} K_d$$

where θ_t is the total porosity of the surface soil compartment [-]
ρ is the grain density of the surface compartment [$kg\ m^{-3}$]
K_d is the sorption coefficient of the soil in the surface soil compartment [$m^3\ kg^{-1}$]

9.4.1.2. Example of Solid Phase Transfer

The transfer rate of radionuclides from the surface water compartment of the lake to the sediment compartment of the lake due to sedimentation, λ_{hL} [y^{-1}], is given by:

$$\lambda_{hL} = \frac{K_{dLb} h_L Area_L}{(1 + K_{dLb} \alpha_{Ls}) V_{Ls}}$$

where: K_{dLb} is the sorption coefficient of the soil in the lake bed compartment [m^3 kg^{-1}]

h_L is the gross sedimentation rate from the water compartment to the associated sediment compartment [kg m^{-2} y^{-1}]

$Area_L$ is the area of the lake [m^2]

α_{Ls} is the suspended sediment load in the surface water compartment of the lake [kg m^{-3}]

V_{Ls} is the volume of the surface water compartment of the lake [m^3]

9.4.1.3. Example of Multi-phase Transfer

The transfer rate of radionuclides from the river compartment to the lake, λ_{SRL} [y^{-1}], is given by:

$$\lambda_{SRL} = \frac{S_{RL}}{V_R}$$

where:

S_{RL} is the discharge rate from the river to the lake [m^3 y^{-1}]

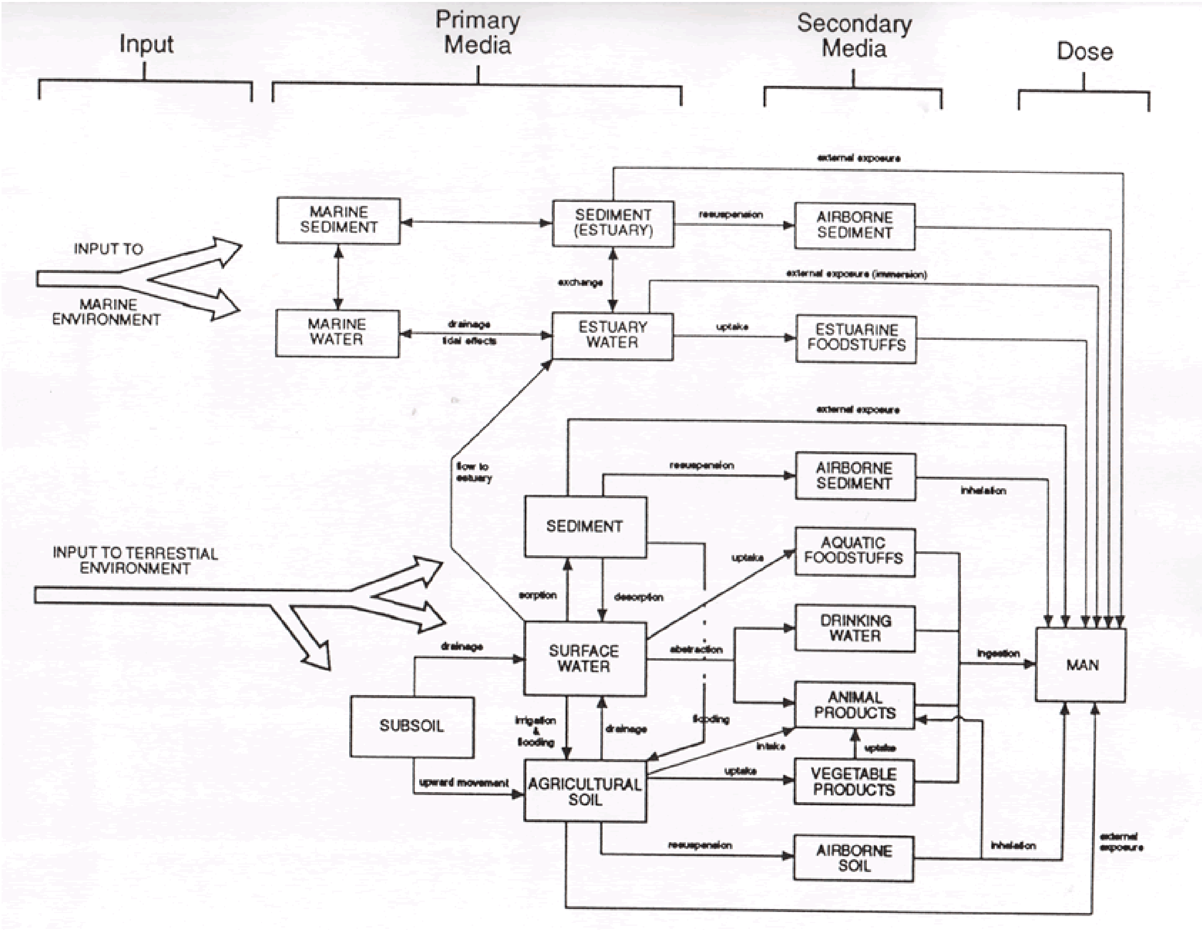
V_R is the volume of the river compartment [m^3]

9.4.2. Transfer to/within Flora and Fauna

For the purposes of long-term assessments of radioactive waste disposal, as opposed to the short-term routine/accidental discharge assessments [IAEA, 2001], concentrations of radionuclides in certain biosphere media (for example crops and animals) can often be assumed to be in equilibrium with their donor media and their concentrations are assumed to be a linear function of the concentration in the donor media. Therefore, they and their associated FEPs do not need to be modelled dynamically using the first order differential equation given in Section 9.4.1, instead this equilibrium assumption is sufficient. For example, the concentration in a crop grown in the soil can be assumed to be in equilibrium with the concentration in the soil and any irrigation water applied. This approach is valid because the processes affecting the concentrations in such media are rapid compared with those affecting concentrations in the donor media, particularly because of the long-term nature of the release.

Thus it is helpful to distinguish, at both the conceptual and mathematical model level, between media for which the temporal variation of radionuclide concentration needs to be calculated using the first order differential equation given in Section 9.4.1 (i.e. dynamic media – the "primary media" in Figure 9.12), and those for which the radionuclide concentration is a linear function of the concentration in the dynamically modelled media (i.e. equilibrium media – the "secondary media" in Figure 9.12). Data for the derivation of this linear function are often derived from models and monitoring of current day releases into the environment.

Figure 9.12: Example Model of Radionuclide Transport Pathways in the Biosphere and Associated Exposure Mechanisms [Little et al, 1993]



This approach has been used in a large number of biosphere modelling studies, for example IAEA [2002a] and BIOMOVS II [1996c]. Two examples are given below, based on IAEA [2002c], for the calculation of radionuclide concentrations in flora (crops) and fauna (fish).

9.4.2.1. Example of Concentration in Flora

The radionuclide concentration in the edible part of the crop, C_{crop} [Bq kg^{-1} (fresh weight of crop)], is calculated using the following equation:

$$C_{crop} = \frac{(F_{p2} CF_{crop} + F_{p1} Soil_{plant}) C_s}{(1-\theta_t)\rho}$$

where F_{p2} is the fraction of the internal contamination associated with the edible part of the plant at harvest that is retained after food processing has occurred [-]

CF_{crop} is the concentration factor from root uptake to the edible portion of the plant [Bq kg^{-1} (fresh weight crop)/Bq kg^{-1} (dry weight soil)]

F_{p1} is the fraction of external soil contamination on the edible part of the crop retained after food processing [-]

$Soil_{plant}$ is the soil contamination on the crop [kg (dry weight soil) kg^{-1} (fresh weight of crop)]

C_s is the radionuclide concentration in the soil compartment [Bq m^{-3}]

This particular model assumes that:

- the crop can be contaminated due to internal uptake of contaminants from the surface soil compartment into the crop via the roots (represented by the $\frac{CF_{crop}C_s}{(1-\theta_t)\rho}$ term);
- the crop can be contaminated due to external contamination of the crop due to deposition of re-suspended sediment from the surface soil compartment (represented by the $\frac{Soil_{plant}C_s}{(1-\theta_t)\rho}$ term);
- contamination can be lost due to food preparation (represented by F_{p1} and F_{p2} terms).

Extra terms can be added to represent other sources and losses of radionuclides such as contaminated irrigation water (see for example IAEA [2002c]).

9.4.2.2. Example of Concentration in Fish

The radionuclide concentration of the aquatic food, C_{aqfood} [Bq kg^{-1}], is calculated using the following equation:

$$C_{aqfood} = FF_{Ls}C_{Ls}CF_{aqfood}$$

where FF_{Ls} is the fraction of activity in the filtered lake water [-]
C_{Ls} is the radionuclide concentration in the surface water of the lake [Bq m^{-3}]
CF_{aqfood} is the concentration factor for the aquatic foodstuff [Bq kg^{-1} (fresh weight of edible fraction)/Bq m^{-3} (filtered water)]

The FF_{Ls} term is calculated using the following equation:

$$FF_{Ls} = \frac{1}{1+K_{dLb}\alpha_{Ls}}$$

where K_{dLb} is the sorption coefficient for the lakebed sediment [m^3 kg^{-1}]
α_{Ls} is the suspended sediment load in the surface water compartment of the lake [kg m^{-3}]

9.4.3. Dose Calculations

Having determined radionuclide concentrations in the various media, it is possible to calculate the associated dose using equations of the form:

$$D = C \quad U \quad DCF$$

where C is the radionuclide concentration in the environmental media that acts as the source of contamination
U is a use factor that describes the utilisation rate of the media by human

DCF is a dose conversion factor for the radionuclide and any shorted lived daughters[2].

Examples for ingestion, inhalation and external irradiation are given below based on IAEA [2002c]. Other examples are given in IAEA [2002c] and BIOMOVS II [1996c].

9.4.3.1. *Example of Ingestion*

The annual individual dose from the consumption of a crop is given by:

$$D_{crop} = ING_{crop} DC_{ing} C_{crop}$$

where D_{crop} is the individual dose from consumption of the crop [Sv y^{-1}]
ING_{crop} is the individual ingestion rate of the crop [kg y^{-1}]
DC_{ing} is the dose coefficient for ingestion [Sv Bq^{-1}]
C_{crop} is the radionuclide concentration in the edible part of the crop [Bq kg^{-1} (fresh weight of crop)]

The annual individual dose from the consumption of fish is given by:

$$D_{aqfood} = ING_{aqfood} DC_{ing} C_{aqfood}$$

where D_{aqfood} is the individual dose from consumption of the aquatic foodstuff [Sv y^{-1}]
ING_{aqfood} is the individual consumption rate of the aquatic foodstuff [kg y^{-1}]
C_{aqfood} is the radionuclide concentration of the aquatic food [Bq kg^{-1}]

9.4.3.2. *Example of Inhalation*

The annual individual dose from the inhalation of dust, during occupancy of the soil compartment, is calculated for both normal and dusty conditions using:

$$D_{dust} = DC_{inh} BR\, O_s C_{airs}$$

where D_{dust} is the individual dose from the inhalation of dust [Sv y^{-1}]
DC_{inh} is the dose coefficient for inhalation [Sv Bq^{-1}]
BR is the breathing rate of the human in the soil compartment [m^3 h^{-1}]
O_s is the individual occupancy in the soil compartment [h y^{-1}]
C_{airs} is the radionuclide concentration in the air above the soil compartment [Bq m^{-3}]

9.4.3.3. *Example of External Irradiation*

The annual individual dose from external irradiation from soil/sediment, during occupancy of the soil compartment, is given by:

[2] As noted in Section 9.3, daughter radionuclides with a half-life less than 25 days are often assumed to be in secular equilibrium with their parents. Their radiological effects, e.g., the dose per unit activity ingestion, are taken into account by adding them to those of their parents.

$$D_{exsoil} = O_s DC_{exts} C_s$$

where D_{exsoil} is the individual dose from external irradiation from the soil [Sv y^{-1}]
DC_{exts} is the dose factor for external irradiation from soil [Sv h^{-1}/Bq m^{-3}]
C_s is the radionuclide concentration in the soil compartment [Bq m^{-3}]

9.5. *Computer Codes*

Computer codes for the solution of the above mathematical models have been in existence since the late 1970's and early 1980's (see for example Bergstrom et al [1982], Lawson and Smith [1985] and Grogan [1985]). Some of the codes were originally relatively simple extensions of codes developed for other purposes, such as estimating the consequence of routine discharges from nuclear power plants. In addition, further codes have been developed and applied for routine and accidental discharges from nuclear facilities (see for example IAEA [1993]), and relevant experience has been transferred across to the codes used for waste disposal assessment. Projects such as BIOMOVS [BIOMOVS, 1993; BIOMOVS II, 1996d], BIOMASS [IAEA, 2002a, b, c] and the Level 1B exercise of the PSACOIN (Probabilistic Safety Assessment Code Intercomparison) [NEA, 1993], have contributed to a greater understanding of relevant long-term processes and how to model them. Therefore an increasing number of codes have been developed specifically for the biosphere assessment of radioactive waste disposals. See for example Table 9.1 taken from IAEA [2002d]; further examples are given in BIOMOVS II [1996c].

Table 9.1: Some Computer Codes Used for Biosphere Modelling (based on information in IAEA [2002d])

Code	Description	Reference
AMBER	AMBER is a flexible windows-based application, which allow users to build their own dynamic compartment models to represent the migration and fate of contaminants in a system, using an intuitive Graphical User Interface. Radioactive and non-radioactive contaminants in solid, liquid and gaseous phases can be considered. AMBER can be used to assess routine, accidental and long-term contaminant releases. It can be used to represent the near-field and geosphere, as well as the biosphere.	Enviros QuantiSci and Quintessa [2002]
GENII	GENII calculates radiation doses for acute and chronic releases to air and water, from ground level and stacks, and includes doses due to initial contamination of soil and surfaces. It evaluates exposure pathways via air immersion, water immersion, ground surface (surface and buried sources), inhalation, and ingestion.	Napier et al [1988]
RESRAD	RESRAD uses a pathway analysis method in which the relation between radionuclide concentrations in soil and the dose to a member of a critical population group is expressed as a pathway sum, which is the sum of products of "pathway factors". Pathway factors correspond to pathway segments connecting compartments in the environment between which radionuclides can be transported or radiation emitted. Radiation doses, health risks, soil guidelines and media concentrations are calculated over user-specified time intervals. The source is adjusted over time to account for radioactive decay and ingrowth, leaching, erosion, and mixing. RESRAD uses a one-dimensional groundwater model that accounts for differential transport of parent and daughter radionuclides with different distribution coefficients.	Yu et al [1993]

TAME	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.	Klos et al [1996]

9.6. Data

The broad range of data required to allow the modelling of the biosphere is described in Section 4.5. The specific parameters for which data need to be obtained will depend upon the precise models that are used but typically include:

- compartment characteristics (dimensions, porosities, bulk densities)
- water and sediment flow rates;
- human and animal habits including ingestion rates of water and foodstuffs;
- radionuclide distribution coefficients;
- radionuclide transfer factors;
- radionuclide half-life;
- radionuclide dose coefficients.

IAEA [2002d] provides example compilations of such data. Further example data are provided in IAEA [2002c and e].

9.7. Summary

The biosphere includes the physical media (atmosphere, soil, sediments, surface waters) and the living organisms (including humans) that interact with them. Most of the end-points that are considered in safety assessments require the modelling of the migration and accumulation of radionuclides in the biosphere. Thus it is necessary to analyse and quantify the transport and radiological impact of radionuclides released from the disposal facility into the biosphere in order to demonstrate the acceptability of radioactive waste disposals.

When analysing and quantifying such releases, the assessor is faced with a number of challenges that need to be addressed:

- the biosphere can be subject to change, even over relatively short timescales;
- biosphere modelling potentially requires consideration of the future activities and behaviour of humans over timescales of many thousands of years
- there are many potential interfaces between the biosphere and the other components of the disposal system (i.e. the near-field and the geosphere). It is particularly important to ensure that there are consistent boundary conditions at the interface.

When considering biosphere transport and exposure processes, it is helpful to distinguish between:

- transport in surface waters;
- transport in soils;
- transport in the atmosphere;
- transport with and between flora and fauna; and
- resulting exposure of humans.

The mathematical models that have been developed to represent biosphere transport and exposure processes range in complexity from simple expressions to highly complex mathematical algorithms. However, most of the biosphere assessment models are based on linear donor-controlled compartment models that use transfer coefficients. Such models assume that a system may be represented by breaking it down into compartments, each of which may represent a medium which is distinct from other associated media. The models estimate the transfer between compartments and then derive concentrations in the various environmental media to which humans are exposed. These concentrations are then used to calculate resulting doses.

9.8. References

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