FASSET

Framework for Assessment of Environmental Impact

Deliverable 1: Appendix 1

Ecological characteristics of European terrestrial ecosystems

Overview of radiation exposure pathways relevant for the identification of candidate reference organisms

November 2001

A project within the EC 5th Framework Programme

FASSET will bring to radiation protection a framework for the assessment of environmental impact of ionising radiation. The framework will link together current knowledge about sources, exposure, dosimetry and environmental effects/consequences for reference organisms and ecosystems. Relevant components of the framework will be identified on an ecosystem basis through systematic consideration of the available data. The application of the framework in assessment situations will be described in an overall report from the project. The project started in November 2000 and is to end by October 2003.

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Swedish Radiation Protection Authority	SSI
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German National Centre for Environment and Health	GSF
Spanish Research Centre in Energy, Environment and Technology	CIEMAT
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1. Background

Traditionally, radiological protection systems have focused on the protection of man. This approach and constraint is being increasingly questioned and the requirement for an internationally agreed rationale to the protection of the environment to ionizing radiation has been recognized. The overall aim of the FASSET project is to develop a framework within which assessment models can be applied and results analyzed for European ecosystems.

One of the objectives to be met in achieving this aim is: to provide a set of *reference organisms* relevant to different exposure situations.

Environmental compartments where radionuclides can be expected to accumulate and organisms for which enhanced exposure (both external and internal) is likely to occur have been identified. To aid this a range of different European ecosystems have been considered, namely, forests, semi-natural pastures and heathlands, agricultural ecosystems, wetlands, freshwaters and marine and brackish waters. Compilation of relevant data on the distribution of radionuclides within these ecosystems has been undertaken.

The work on ecosystem characterization has been divided into two appendices to FASSET Deliverable 1, concentrating on the terrestrial and the aquatic ecosystems respectively. These two appendices are the foundation on which the selection of the candidate reference organisms in Deliverable 1, ‘Identification of candidate reference organisms from a radiation exposure pathways perspective’ is based (available at www.fasset.org). This appendix (Appendix 1 to Deliverable 1) focuses on terrestrial ecosystems.

The resulting candidate reference organisms presented in Deliverable 1 are suggested, primarily on radioecological criteria (i.e. those organisms which are likely to be the most exposed). To reflect the behaviour of different radionuclides, and conditions of chronic or acute exposure, candidate reference organisms for the soil, canopy and herbaceous layer of the terrestrial ecosystems have been suggested. For aquatic ecosystems candidate reference organisms have been suggested for both benthic (associated with bed sediments) and pelagic foodchains (associated with the water column). In conditions of chronic exposure organisms most likely to be the most exposed are those in closest contact with soil or sediments.

The approach taken towards the selection of the candidate reference organisms should ensure that suitable reference organisms are available for a range of scenarios (chronic and acute exposure) and different European ecosystems. In total 31 candidate reference organisms have been suggested representing marine, freshwater and a variety of terrestrial ecosystems. These candidate reference organisms will be used for development of dosimetric models and will be assessed against radiosensitivity and ecological criteria to select a final set of reference organisms for use within the FASSET project.

Full documentation on the FASSET project is available at the project website, www.fasset.org

2. Forest ecosystems

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2.1 Ecosystem description

This section describes forest ecosystems, which are the most common terrestrial ecosystems [Perry, 1994]. A forest ecosystem is a community dominated by trees which interact with other organisms, and that develops at the soil-atmosphere interface. It should be noted that, unlike an organism, an ecosystem does not have clearly defined, but rather diffuse boundaries. Ecosystems are defined by connections that extend through space and time, integrating every local ecosystem within a network of successively larger structures that compose landscapes, regions, and eventually, the entire Earth (Figure 2-1). Local forest ecosystems also exhibit biological diversity at many scales, from the individual organism species, genotype and individual, through the fine-scale structure of soils and canopies. As we will discuss later in this chapter, any given forest both influences and is influenced by the 'global ecosystem'.

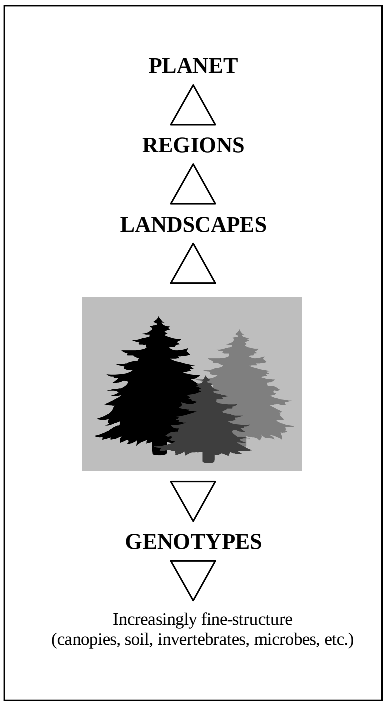

Figure 2-1 Position of a local forest ecosystem in a simplified view of the hierarchy of nature. Adapted from Perry [1994].

Areas covered by forests

The FAO (Food and Agriculture Organization of the United Nations) defines forest as land with tree crown cover (or equivalent stocking level) of more than 10 % and area of more than 0.5 hectares (ha). The trees should be able to reach a minimum *in situ* height of 5 meters (m) at maturity. They may consist either of closed forest formations where various tree storeys and undergrowth cover a high proportion of the ground; or open forest formations with a continuous vegetation cover in which tree crown cover exceeds 10 %. Young natural stands and all plantations established for forestry purposes which have yet to reach a crown density of 10 % or tree height of 5 m are included under forest, as are areas normally forming part of the forest area which are temporarily unstocked as a result of human intervention or natural causes, but which are expected to revert to forest.

The Global Forest Resources Assessment 2000 [FRA, 2000] collected and synthesized a vast amount of information on the status and trends in forest cover worldwide. The FRA 2000 concluded that the total forest area, according to the FAO definition of forest given above, for year 2000 was 3.9 billion ha, of which 95 % was natural forest and 5 % was forest plantation. A summary of forest cover in Europe is presented in Table 2-1.

Table 2-1 Forest cover in European countries [FRA, 2000].

Country	Area 1 000 ha	Percentage of land area
Russia	851 392	65.9
Sweden	27 134	50
Finland	21 935	72
France	15 341	27.9
Spain	14 370	28.8
Germany	10 740	30.7
Italy	10 003	34
Ukraine	9 584	16.5
Belarus	9 402	45.3
Poland	9 047	29.7
United Kingdom	2 794	11.6
Other	56 509	
Total	**1 039 251**	

Types of forests

Forests are often classified into three major types according to the latitude in which they occur: (a) tropical, (b) temperate and (c) boreal (also called taiga). Various subdivisions of the major forest types, however, exist as influenced by precipitation patterns, types of trees, etc. By the end of 2000 about 47 % of forests worldwide were tropical, 9 % subtropical, 11 % temperate and 33 % boreal [FRA, 2000]. In the report FRA 2000 twelve European terrestrial ecological zones were defined. In most of these zones forest ecosystem are an important part of the landscape (Table 2-2). According to the Interpretation Manual of European Union Habitats, 6 main classes of forests with 58 subtypes can be determined [EC, 1999]:

1. Forests of Boreal Europe (with 8 subtypes),

2. Forests of Template Europe (with 19 subtypes),

3. Mediterranean Deciduous Forests (with 13 subtypes),

4. Mediterranean Sclerophyllous Forests (with 7 subtypes),

5. Temperate Mountainous Coniferous Forests (with 3 subtypes),

6. Mediterranean and Macronesian Mountainous Coniferous Forests (with 8 subtypes).

2.1.1 Typical species

Forest ecosystems are highly diverse and there are striking differences in the biological diversity of the major forest types. There is also large variation in the species composition among different types of forests. The typical tree species found in forests growing in different European ecological zones are listed in Table 2-2. A large variability in tree species composition between ecological zones is noticeable from this table. Trees, together with the forest soil provide the energy and basic habitat structure for all organisms living in the forest. Hence, a high variability in species composition in general should also be expected between different ecological zones. Thus, lists of typical species for different hierarchical levels in the forest would be required for each particular ecological zone and type of forest. Such information is not yet readily available in a systematic form. In this report we have focused on describing the interactions between species, and their role for forest ecosystem functioning, in a more generic sense (see Sections 2.1.2, 2.1.3 and 2.1.4).

2.1.2 Ecological niches and habitats

In its simplest terms, a habitat is a place (or type of place) that provides the basic necessities for an organism, including energy, nutrients, water and shelter. Habitats may be either extensive or localized. For example broad habitats such as elevation ranges are composed of mosaics of smaller, more suitable habitats for some species. In this sense, it is difficult to provide a complete list of habitats in a forest ecosystem. For the purpose of systematisation a common approach is to consider habitats in different layers of the forest (Table 2-3): the tree layer, the understorey layer and the soil-litter layer. For example, plants growing in a shaded forest understorey occupy a different habitat (at least in terms of light) than upper canopy trees, although the soil is a source of nutrients for both.

Forest soils have a marked vertical structure termed the soil profile. This profile is composed of horizons. Current designations for soil horizons are described by Guthrie & Witty [1982]:

- organic Horizon (O) – litter in various stages of decay,
- first mineral horizon (A) – characterized by accumulation of organic matter,
- second mineral horizon (E) – characterized by loss of clay,
- third mineral horizon (B) – characterized by various factors, usually involving accumulation of clay,
- fourth mineral horizon (C) – characterized by chemical weathering, but not modified by biological activity.

Table 2-2 Typical species of trees found in European ecological zones covered with forests [FRA, 2000].

Ecological zone	Typical tree species found in the forests
Boreal coniferous forest	Dominated by a few conifer tree species, primarily spruce (*Picea abies*) on moister ground and pine (*Pinus sylvestris*) on drier ground. East of the White Sea, mainly closer to the Ural Mountains, Siberian conifer species like *Pinus sibirica*, *Abies sibirica* and *Larix sibirica* may also occur.
Boreal mountain systems	More or less open *Betula pubescens* subsp. *czerepanovii* forests, partly with pine forests (*Pinus sylvestris*) in the eastern parts.
Boreal tundra woodland	Predominantly composed of the trees *Betula pubescens* subsp. *czerepanovii* and *Picea obovata*.
Subtropical dry forest	Only one species dominates the canopy, often one of the evergreen oak species. *Quercus ilex* compete most successfully at humid-subhumid sites; it is represented on the Iberian peninsula outside of Cantabrica and Catalonia by *Q. ilex* subsp. *rotundifolia*, in the rest of the Mediterranean area by *Q. ilex* subsp. *ilex*.
Subtropical dry forest	Information not available.
Subtropical mountain systems	Mainly deciduous oak species (*Quercus pyrenaica, Q. faginea, Q. petraea, Q. frainetto, Q. pubescens*).
Temperate continental forests	Spruce forests (*Picea abies*) cover most of the northern area. In the east *Picea obovata* may occur. In these hemiboreal coniferous forests broad-leaved trees like *Quercus robur, Tilia cordata, Ulmus glabra* and *Acer platanoides* play an important role in the canopy. Further south the trees forming stands include *Quercus robur*, *Q. petraea*, *Carpinus betulus* and *Tilia cordata*. Associating species like *Fraxinus excelsior* and *Acer campestre* are also important.
Temperate mountain system	Beech forests and particularly mixed beech forests with *Abies alba*, *Picea abies*, *Acer pseudoplatanus*, *Fraxinus excelsior* and *Ulmus glabra* characterise the vegetation of the lower belt in this region. At higher altitudes fir and spruce forests (*Abies alba, A. borisii-regis, A. nordmanniana, Picea abies, P. orientalis, P. omorika*) replace the beech forests. *Abies* and *Picea* dominate with alternating portions. *Pinus sylvestris*, *Fagus sylvatica*, partly *Quercus robur*, and pioneer species like *Sorbus aucuparia*, *Populus tremula*, *Betula pendula* play a minor role. In western parts of the Iberian peninsula oak forests (*Quercus robur, Q. pyrenaica, Q. petraea*) with *Betula pubescens* subsp. *celtiberica*, ericoides and other acidophilous species cover the top of the mountains. In the Ural the altitudinal zonation starts with lime-oak forests (*Quercus robur, Tilia cordata*), followed by herb-rich fir-spruce forests (*Abies sibirica, Picea obovata*) with broad-leaved trees like *Ulmus glabra* and *Tilia cordata* as well as pine forests (*Pinus sylvestris*) with *Larix sibirica*.
Temperate oceanic forest	Dominated by various types of beech forests and mixed beech forests (*Fagus sylvatica*). Outside the distribution area of beech, oak-ash forests (*Quercus robur*, *Fraxinus excelsior*) with *Corylus avellana* and a relatively rich herb layer occupy base-rich, often calcareous soils. Oak-hornbeam forests (*Carpinus betulus*, *Quercus petraea*) dominate periodically moist soils. Azonal vegetation types include flood plain and alluvial forests with *Quercus robur, Ulmus laevis, U. minor, Fraxinus excelsior*, in combination with willow and poplar alluvial forests (*Salix alba, S. fragilis, Populus nigra, P. alba*). Mires and, concentrated in oceanic parts, blanket bogs may occur as well as swamp and fen forests (*Alnus glutinosa, Betula pubescens*).

Table 2-3 Groups of organisms that habit different forest layers. The presented division of the forest into layers is conventional. The purpose is to illustrate the vertical stratification observed in forests. In reality organisms may habit more than one layer, for example trees habit all three layers.

Forest layer	Groups of organisms
Tree layer	Trees, birds, reptiles, insects, micro-organisms, epiflora
Understorey layer	Shrubs, herbs and cryptogams, fungi, birds, mammals, reptiles, amphibians, micro-organisms
Soil-litter layer	Fungi, soil microbes (bacteria and actinomycetes), soil invertebrates, burrowing mammals

Surface organic layers (O), often called forest floor or litter layer, consist of litter in various stages of decay [Pritchett, 1979]: litter that is relatively fresh and still clearly recognizable in its original form (L), litter that is partially decomposed (i.e. fragmented) but still recognizable as to origin (F) and litter in advanced stage of decomposition, amorphous and not recognizable as to origin (H).

The habitat is an aspect of the niche – a broader concept introduced into the ecological literature by Grinnell [1914] and Elton [1927]. For them, a niche was an opening in community structure or function that was filled by a species. Hutchinson [1957] took the opposite view and considered the niche as set of attributes identified directly with a species rather than an ‘opening’ of some kind that the species adapted to. Perry [1994] means by niche, both a) an ‘opening’ or opportunity for a species to find suitable living (and reproducing) conditions within a given system, and b) the set of attributes that a given species bring to that opening. The niche can be seen as a characteristic with multiple dimensions, one of which is the habitat. In forest ecosystems, through its influence on nutrient and water availability, soil is a major component of the habitat niche. Requirement for light is also a part of the habitat niche; shade-tolerant species capable of growing in forest understories occupy a different habitat than those shade-intolerant species that require relatively high light levels. Another example is the life-form niche that refers to differences in inherent size (e.g. shrubs versus trees) or morphological characteristics (e.g. rooting depth, branching pattern).

2.1.3 Food webs

Food webs are a conceptual representation of the flow of substance and energy within an ecosystem. Food webs are divided into trophic levels, which represent each stage of energy transfer. A highly generalised forest food web is presented in Figure 2-2.

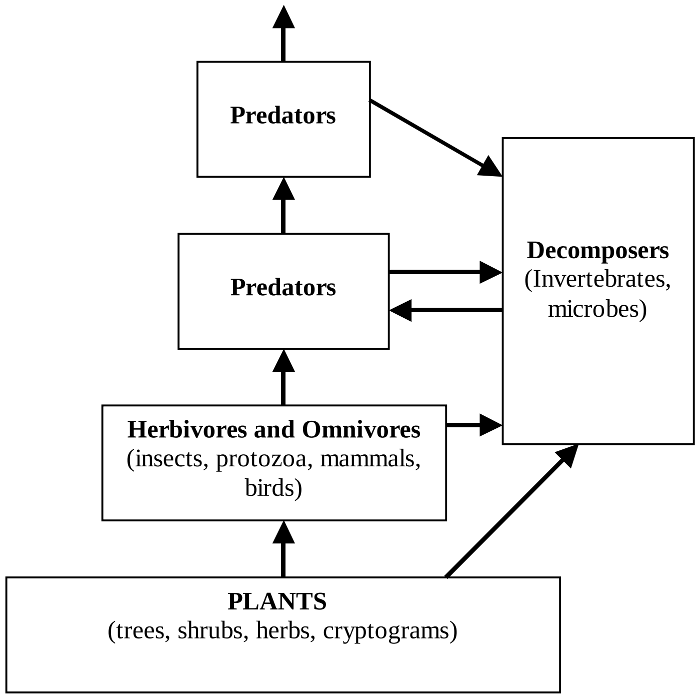

Figure 2-2 Schematic representation of a highly generalised forest food web. Symbiotic micro-organisms and parasites (not represented) are found in all trophic levels. Adapted from Perry [1994].

The first (or basal) trophic level is occupied by green plants, i.e. by trees and understorey plants. The second trophic level comprises three distinct functional groups of organisms:

1. Herbivores and pathogens: animals, microbes, and viruses that feed on living plants, which yield no direct benefit to the plant.
2. Plant mutualists: animals and microbes that feed or otherwise obtain energy from living plants and that directly benefit from plants in some fashion.
3. Decomposers: invertebrate animals, fungi and other organisms that feed on dead tissue such as fallen leaves (detritivores and saprophytes).

Except during relatively infrequent periods when populations of herbivores or pathogens reach epidemic levels, in forests, decomposers and mutualists dominate the second trophic level. Predators, parasites, and mutualists of second-level organisms occupy the third trophic level; the fourth level is occupied by predators, parasites and mutualists of third-level organisms, and so on.

By convention, energy is said to flow upward from lower to higher trophic levels. As is typical of all forests, by far most energy goes directly from plants to fungi and bacteria (decomposers and mutualists). A basal species is one that eats no other species (e.g. most plants). A top species is one that is not eaten by any species. Omnivores, which include many invertebrates and vertebrates (e.g. bears, humans, etc.), are species that occupy at least two trophic levels. Decomposers are the ultimate omnivores, feeding on the faecal matter of insect grazers, the body of a dead predator, fallen leaves, or essentially on all organic matter from which energy and other resources can be extracted. The unique role of decomposers produces loops within forest food webs. Note that the transfer of energy from one organism to another does not always involve death and consumption in the sense that we think of eating. Parasites live on or in the body of their host, as do viruses. Mutualists such as mycorrhizal fungi, bacteria, and plant pollinators, obtain energy from their hosts in payment for providing valuable services. Below, these relationships are discussed in some more detail.

Schoenly *et al.* [1991] have estimated that the number of connections in a forest food web, as a proportion of the total number possible, ranges from less than 10 % to approximately 30 %. Connectance has been frequently used as a measure of relational diversity in ecosystems. However, the focus has been on direct links between trophic levels, i.e. only on a subset (albeit an important one) of the rich tapestry of interactions that characterise the community (see below). Food webs are compartmentalised, which is to say that there are certain channels of energy flow that some species participate in and that other do not. For example the below-ground food web consists of at least three distinct compartments: a) species that eat only dead organic matter (decomposers), b) species that live from living roots and mycorrhizal hyphae, and c) pathogens.

Diverse types of ecosystems (including all terrestrial as well as aquatic systems) are remarkably similar in the basic structure of their food webs [Briand & Cohen, 1987; Cohen *et al*., 1986; Lawton, 1989; Pimm, 1982]. Some general patterns that have been observed are [Perry, 1994]:

- Although up to 10 trophic levels have been reported, most ecosystems have no more than three to five.
- Intermediate trophic levels in forests are dominated by insects and microbes (especially decomposers and plant mutualists).
- With rare exceptions, each species above the basal trophic level feeds on more than one species below it, and each species below the top is fed on by more than one species above it, creating food webs instead of food pipelines.
- When one compares the number of species in each trophic level, food webs tend to be spindle-shaped. The number of species increases as one moves from basal to the intermediate levels, then declines again from intermediate levels to the top vertebrate predators. Parasites, however, are an exception; these small-bodied creatures are quite diverse, even when feeding at the top of the food chain [Schoenly *et al*., 1991]. Decomposers are another exception.
- The biomass of organisms within a given trophic level drops exponentially from the base to the top.
- As a general rule the biomass of animals (at least those that live aboveground) correlates positively with aboveground primary productivity [McNaughton *et al*., 1989].

It should be noted that the above patterns are based mainly on studies heavily biased towards the larger, more easily measured plant and animal species and toward aboveground as opposed to belowground food webs. Invertebrates and microbes (which account for most of the energy flow within ecosystems) are frequently lumped together with no attempt to distinguish individual species or even major species groups.

2.1.4 Interactions between species

Interactions between two species include predation, parasitism, competition, amensalism, mutualism, commensalisms and neutralism. A good description of interactions in forest ecosystems can be found in Perry [1994]. Among them mutualism is one of the most common in forest ecosystems and probably in nature. Janzen [1985] distinguishes the following general types of mutualisms: harvest, propagule mutualisms, protective, pollination and human agricultural and animal husbandry. All first four types of mutualisms, and in some cases even the fifth one, can be found in forest ecosystems.

Mycorrhizae

Most plant species exploit the soil with the help of beneficial micro-organisms called mycorrhizal fungi. The fine threads that make up the fungus branch between soil particles grow into decomposing organic matter and even explore the shells of dead insects, where they find phosphorus and other vital nutrients. The nutrients are then passed back to the roots of the plant.

Mycorrhiza is a harvest mutualism that plays an important role for the forest nutrition and influences the behaviour of radionuclides and other contaminants in the forest soils and their transfer to plants. The word *mycorrhiza* comes from the Greek words *myco*, for fungus and *rhiza* for root. It refers to any of several types of associations between plant roots and soil fungi.

In a mycorrhiza, root and fungal tissue combine in such a way that both plant and fungus benefit. The fungus receives carbohydrates and other essential organic substances from the plant in the form of sugars and the organic molecules, and performs numerous services in return, such as:

- Enhancing water and nutrient uptake, particularly nutrients such as phosphorous that do not readily move in soils. Some mycorrhizal fungi decompose organic matter and cycle the nutrients contained therein directly back to their host plants [Read, 1987; 1991a; 1991b].
- Protecting plants against root pathogens [Marx, 1972].
- Extending the lifetime of small roots.
- Binding soil particles together into large aggregates, thereby producing favourable structure for water retention and gas exchange.
- Linking individual plants of the same and different species together in a common hyphal network, through which carbon and nutrients pass [Read *et al*., 1985].
- Reducing competition between plants of different species [Perry & Amaranthus, 1992].

Roughly 90 % of the world's plant species form mycorrhizae with at least one (and frequently many) species of fungi [Molina *et al*., 1992]. Conifers are always mycorrhizal in the wild. Among species of angiosperms that have been studied, 70 % are consistently mycorrhizal and 12 % are apparently facultatively mycorrhizal, that is, sometimes forming mycorrhizae and sometimes not [Trappe, 1987]. The numbers and types of mycorrhizae supported by trees vary with soil fertility. Fewer mycorrhizae are formed in fertile than in unfertile soils. Few if any forest soils are sufficiently fertile that trees require no mycorrhizae at all.

Guilds

A guild is a group of species within a given community that share some common interest and constitute an important high order interaction between species in an ecosystem. A guild is any group whose members overlap in one or more niche dimensions. Any given species is likely to belong to several different guilds, resulting in a complex intertwining of relationships within communities. For example, plant species that form mycorrhizae with one or more species of fungi, along with the fungi themselves, form a guild. At the same time, some of them may be part of a network of plants that require the same pollinators and that form a guild along with the pollinators.

Guild structure plays a critical role in ecosystem processes, especially regarding system stability. The ability of more than one species to perform the same function (a property called redundancy) maintains functional integrity of the system even though a species may be lost from it. Guilds that are related to overlap in functional niches represent the redundancies in ecosystems. For example, the loss of a foliage gleaning bird species from a community does not mean that foliage-eating insects are free of predators, so long as other species in the foliage-gleaning guild remain [Perry, 1994].

Keystones

Where evolutionary and other pressures act to fragment guilds by reducing niche overlap (niche diversification) the end result can be keystones, or species that perform some unique function. Keystones are points of vulnerability in ecosystems. When they are lost, so is their function.

In a more general sense keystones are species, groups of species, habitats (e.g. large dead wood), or biotic factors (e.g. fire) that play a pivotal role in ecosystem (or landscape) processes and 'upon which a large part of the community depends' [Noss, 1991]. Some landscape features, such as riparian zones or migration corridors, may also be keystones. Loss of a keystone produces cascade effects (i.e. leads to the loss of other species or disruption of processes). Below we discuss candidates of species and groups of species for keystones in a forest ecosystem.

Plants

Clearly, trees compose a keystone group in forest ecosystems; they provide the energy and basic habitat structure for all living organisms within the system. In monospecific forests, which occur naturally as well as because of human simplification, the health of the entire system turns on the health of one single tree species. Green plants are also a keystone group, due to their role in photosynthesis. In forests with only one major nitrogen-fixing plant species, that species is certainly a keystone. An individual plant species may be part of one keystone group, with regard to one function and part of another group with regard to another. For example, two tree species may host the same mycorrhizal fungi, but different pollinators.

Predators

The argument on whether or not predators can be keystone species centres on the degree to which predators actually limit populations of their prey. The experience shows that predators contribute to the control of prey populations, but they seldom are the sole control agent. The relative importance of predators may vary from one ecosystem to another and over time in any single system. There are several clear examples in marine ecosystems [Krebs, 1988]. To mention one, Paine [1974] removed a top predator (i.e. starfish) from an intertidal community; the starfish prey, a mussel, grew in numbers and eventually pushed out most other species in the system.

When it comes to forest ecosystems, such well-documented and convincing examples have not however been found. At the same time, forest food chains are often structured in such a way that redundancy is relatively higher at the bottom than at the top (Figure 2-3). Hence, the system might be better buffered against the loss of a plant species than against loss of a top predator. This would not be true, however, if the lost plant species played some key role in the ecosystem.

The issue has a special significance for large predators. Being at the top of the food chain, they are few in number and require more territory to make their living than species of lower trophic levels. Therefore, they are the first to feel the impact of habitat loss. Moreover, top predators generally have no natural enemies and therefore no innate flight reaction that may save them from humans. Finally, predators may accumulate toxic substances that pass up the food chain.

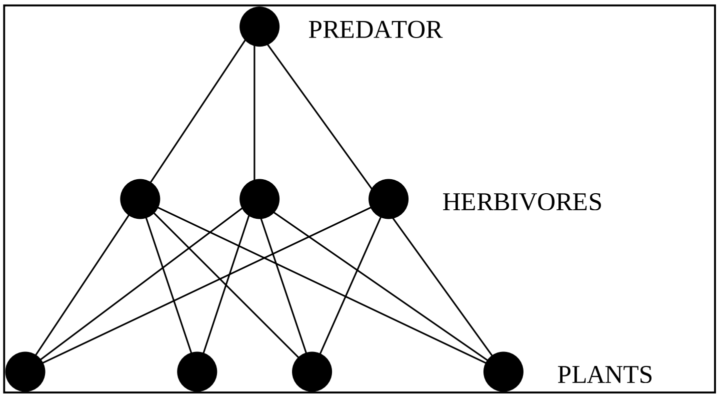

Figure 2-3 Conceptual representation of a forest food web with relatively high redundancy at the first trophic level and no redundancy at the highest trophic level.

Herbivores

Like predators, herbivores affect system structure and processes through what they eat. Herbivores promote diversity by preventing one or a few plant species from dominating. A given herbivore species plays a keystone role when it is so large (or so abundant) as to account for a major portion of the plant material that is consumed. Large mammals like elephants, moose, deer and wapiti (the North American elk) are the most obvious examples [Naiman, 1988], but invertebrates can also play a keystone role.

Builders

Builders, through activities that benefit themselves, provide benefits to others. Noss [1991] discussed one of the most striking examples, the gopher tortoise, which is native to longleaf pine forests in the southeastern United States: 'The gopher tortoise digs burrows up to 30 ft.

long and 15 ft. deep, in which some 32 species of vertebrates and invertebrates have been found. Some of these species are absolutely dependent upon the threatened tortoise'. Beavers are also keystones because of the unique habitat provided by their pounds.

From the discussion above it follows that there are undoubtedly keystone species within forest ecosystems, but it is far from straightforward in many cases to define what these are. Potential keystones can be identified on the basis of studies of the structure of interactions and interdependencies in the system, although in most cases more knowledge might be required than there is available. Verification, though, requires an experiment in which the hypothesized keystone is removed, and these critical experiments have seldom been performed in forest ecosystems [Perry, 1994]. There are also 'natural' experiments, such as the disappearance of top predators (e.g. wolves) from much of their former range. However, natural experiments are usually difficult to interpret unambiguously, due to the diversity and complexity of the system and the large number of factors involved that need to be taken into account.

2.1.5 Linkage to other ecosystems

The global ecosystem is tied together by cycles of water and nutrients and through the dynamics of the climate, all of which are significantly influenced by forests. Although occupying approximately one third of the Earth's land area, forests account for over two thirds of the leaf area of land plants and contain roughly 70 % of the carbon contained in living things [Perry, 1994]. Forest canopies reflect less solar energy than other vegetation types; hence the extent of forest cover affects the Earth's heat balance [Sagan *et al.*, 1979]. Because of their high leaf area, forests account for most of the Earth's photosynthesis and the bulk of water evapotranspiration from land to the atmosphere. By keeping water in circulation, forests effectively create rainfall. Forests also play a vital role in stabilizing soils and regulating the inputs of nutrients and sediments to surface waters.

2.2 Behaviour of radionuclides in forest ecosystems

In this section we present an overview of the behaviour of radionuclides of interest in forest ecosystems. The behaviour in forests of caesium (^{134}Cs and ^{137}Cs), and to a lesser extent strontium (^{90}Sr), isotopes have been broadly studied and have been summarised in several review papers [Fraiture, 1992; Myttenaere *et al.*, 1993; Thiry & Myttenaere, 1993; Tikhomirov & Shcheglov, 1994; Nimis, 1996], proceedings of conferences and other publications [Avila, 1998; Linkov & Schell, 1999]. For other radionuclides, knowledge and publications are much scarcer. Nevertheless, by combining the existing knowledge for radionuclides and stable elements it is possible to identify several patterns of the behaviour of radionuclides in the system. The focus of this section will be on the description of these patterns.

2.1.6 Dynamics of the radioactive contamination of forests

A conceptual view of the migration of radionuclides in a forest ecosystem is presented in Figure 2-4 in a matrix form. The diagonal elements of the matrix are components of the system, while the off-diagonal elements are interactions between components resulting in radionuclide transfer. The matrix in Figure 2-4 was proposed by the BIOMASS Forest Working Group [IAEA, 2001] on the basis of a similar matrix described in Avila & Moberg

[1999]. It represents the consensus reached by the group on which transfer processes are relevant and necessary to describe the migration of ^{137}Cs in a forest ecosystem. It is also a suitable conceptualisation for representing the interactions. Although this matrix was developed for ^{137}Cs it can also be used for other radionuclides. The intensity and importance of the interactions might, however, differ from radionuclide to radionuclide. The matrix also aims at covering a broad spectrum of forest types of the northern hemisphere in a generic way.

The significance of different transfer processes may also vary depending on the type of contamination; season of the year and time elapsed since the contamination event. In the case of an aerial contamination (fallout), three main phases of evolution of the radioactive contamination can be identified [Shcheglov, 1999]: a phase lasting 2–3 months after the fallout (initial phase), a phase lasting 2–3 years after the fallout (intermediate phase) and a phase characterised by the soil-plant system gradually approaching a steady-state condition, that can last several years to several decades depending of the radionuclide and forest type (long-term phase). Below we discuss the main processes and factors that dominate the initial and long-term phases. The intermediate phase is a transition between these two phases and is, thus, characterised by a gradual decrease of the significance of processes dominating in the initial phase and a gradual increase of the significance of the factors dominating in the long-term phase.

The initial phase of an aerial contamination

The processes dominating in the initial phase of an aerial contamination are shown underlined in Figure 2-4. Among them, the most important are interception of radionuclides by plant surfaces, translocation to internal parts of the plant and leaching of radionuclides from the plant surfaces to the forest floor by weathering processes. Under special local conditions, resuspension of radionuclides deposited on the forest floor might also have some significance, but in most cases radionuclides reaching the forest floor are either readily fixed by the soil moss and litter cover or they percolate down the soil profile.

In forests affected by the Kysthym and Chernobyl accidents the fraction intercepted by tree crowns varied between 40 to 90 % [Tikhomirov & Shcheglov, 1994; Nimis, 1996] depending on the radionuclide, type of forest and climatic conditions. Assessments made with the model FORESTLAND [Avila *et al.*, 1999] point to strong seasonal variations in the fraction of the deposited radionuclide (in this case ^{137}Cs) that is initially retained in tree crowns (Figure 2-5). The seasonal variations could be more or less accentuated depending on the crown closure and type of trees.

Trees, mainly through their leaves, branches, twigs and the living bark are the most exposed living organisms during this phase of the contamination. The degree of contamination of understorey plants will depend of the fraction intercepted by the trees and the rate of decontamination of the trees. The radionuclide composition in contaminated plants in this phase is very similar to the composition of the fallout. Herbivores grazing on trees and understorey plants may also be highly exposed at this phase. In the Nordic countries, for example, high levels of ^{137}Cs were observed in moose and roe deer directly after the Chernobyl deposition [Rantavaara *et al.*, 1987; Johanson & Bergström, 1994].

The radionuclides intercepted by trees and understorey plants are rapidly transferred to the forest floor by weathering, i.e. by the action of rain and wind. According to Tikhomirov &

Shcheglov [1994], in areas contaminated by the Chernobyl accident *circa* 90 % of the ^{137}Cs activity intercepted by trees was transferred to the forest floor within the first six months after the deposition. Other authors have also reported a very rapid transfer of most part of the intercepted radiocaesium to the forest floor [Nimis, 1996]. A small fraction of the radionuclides can, nevertheless, be translocated inside the plants and fixed by woody tissues.

Atmosphere	intercept. rainfall snowfall	intercept. rainfall snowfall			intercept. rainfall, snowfall			intercept. rainfall snowfall	intercept. rainfall snowfall	intercept. inhalation
transpir.	**Tree leaves**	weathering	translocation	translocation	leaf fall, weathering			weather. intercept.	weather. intercept.	ingestion
		Bark	translocation		weather. intercept.			weather. intercept.	weather. intercept.	ingestion
	translocation	translocation	**Living wood**	translocation		fertilisation	fertilisation	mycorrhizae transfer		ingestion
			translocation	**Dead wood**						
resuspension		rain splash	root uptake		**Litter (Ol)**	decomposition, leaching soil biota		uptake	rain splash, root uptake	ingestion
			root uptake			**Organic soil (Of, Oh)**	percolat. diffusion advect., soil biota	uptake	root uptake	
			root uptake			diffusion, capillary rise, soil biota	**Mineral soil**	uptake	root uptake	
			root upt. (mycorrhizae)		fertilisation	fertilisation	fertilisation	**Fungi**	root upt. (mycorrhizae)	ingestion
transpir.					leaf fall, weather. intercept.	fertilisation	fertilisation	mycorrhizae transfer	**Understorey plants**	ingestion
					fertilisation					**Herbivores, predators**

Figure 2-4 Conceptual representation with an interaction matrix of the migration of radionuclides in a forest ecosystem. The diagonal elements are components of the system and the off-diagonal elements are interactions between them (transfer processes between compartments). In order to identify the transfer processes the matrix should be read clockwise. The underlined interactions correspond to those dominating in the initial phase of an aerial contamination. Adapted from IAEA [2001].

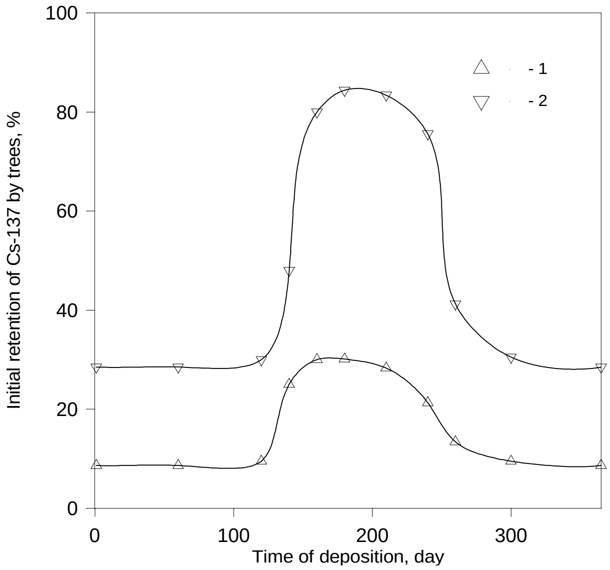

Figure 2-5 Initial retention of ^{137}Cs in a deciduous forest as a function of the season of the year and the crown closure (1 – the crown closure is 0.3; 2 – the crown closure is 1.0). Adapted from Avila et al. [1999].

The long-term behaviour of radionuclides in forests

The radionuclides that either enter the forest soil-litter layer, or following a superficial contamination are absorbed by trees or understorey plants, can be readily incorporated in the cycling of nutrients that is continuously occurring in the forest. During the annual cycles, the nutrients, and with them the radionuclides are partially removed from the soil in the period of plant growth and are subsequently returned to it during senescence with falling leaves and by weathering processes. The radionuclides also enter the animal food chains, from where they are partially or totally returned to soil. The compound result of annual cycles, occurring in sequence over years, determines the long-term behaviour of radionuclides in the system. All interactions shown in Figure 2-4 can, in principle, take place during an annual cycle, but root uptake, litter and organic matter decomposition, vertical migration and sorption-desorption in soil, leaves fall and translocation in trees dominate the behaviour of radionuclides in this phase [Riesen *et al.*, 1999].

2.1.7 Cycling characteristics of nutrients in forests

Radionuclides that are isotopes of plant nutrients (K, C, H, Cl, Ni), as well as those with a nutrient analogue (Cs – analogue of K, Sr and Ra – of Ca, Tc – of N in NO_3^- form) behave in a way that resembles, the behaviour of the corresponding nutrient[1]. This can be explained by the fact that, as a rule, radioactive and non-radioactive isotopes do not differ in their physico-

[1] This is probably more valid for analogous of macronutrients (C, H, O, N, S, P, K, Ca, Mg).

chemical properties[2]. Some of the main cycling characteristics of the nutrients and of their content in different forest components are summarized below [summarized and adapted from Perry, 1994; Mengel & Kirby, 1978; Bowen, 1979; Clarkson & Hanson, 1980]:

- Plants primarily take up C and O from the atmosphere as CO_2 and O_2, and H is obtained when the water molecule is split during photosynthesis. Because they are released from organic forms as gases and therefore escape to the atmosphere, these elements are not cycled to any large degree within local ecosystems. Some local cycling does occur, for example, plants can take up organic molecules released during decomposition of dead tissues. The acidity generated by H^+ ions significantly influences a number of soil processes, including the availability of other nutrients. Other macronutrients (N, S, P, K, Mg, Ca) are cycled tightly within forest ecosystems. Among them N, S, P are associated in soils very closely to organic matter, while K, Mg and Ca are usually not integral constituents of the soil organic matter (SOM). The cycling of the last three, however, depends on cation exchange sites provided by SOM and clays. They are also quite soluble, thereby enhancing both their availability to plants and their susceptibility to leaching from the system.
- The micronutrients (Fe, Mn, Cu, Zn, Mo, B, Cl, Ni), except for Fe, occur in very small quantities within parent material; however; plants require only small amounts of them. The metals in this group are insoluble to one degree or another at pH values normally found in forest soils; hence; their abundance is not a good measure of their availability to plants. Solubility of the metals occurs through complexation (chelation) with humic and fulvic acids and a wide variety of compounds released into the soil by roots, mycorrhizae and microbes.
- On a dry-weight basis, over 90 % of the foliar mass consists of C, O and H. The remaining nutrients compose only a small fraction of biomass, because their function in living organisms is as regulators and mediators of processes rather than as components of structure. Nutrients other than C, H and O occur in greatest concentrations in living tissues (i.e., foliage, cambium, growing root tips) and in lowest concentrations in stem wood. As a rough guide, element concentrations in small branches and small roots are 50–70 % of those in foliage, in large branches and roots 20–30 % of those in foliage, and in stems 10–20 % of those in foliage. In evergreens, foliar concentration of most nutrients decline with leaf age; Ca is an exception, tending to increase with leaf age.
- There is a wide variation of foliar nutrient concentrations among tree species within any given forest type. Despite of this, it is possible to conclude that temperate deciduous trees average the highest foliar concentrations, particularly when compared with conifers. The relatively low nutrient concentration of evergreen trees is often seen in evergreen shrubs also. Ericaceous plants, for example, which commonly associate from conifers in many temperate and boreal forests, also tend to have low foliar nutrients. It is also apparent that trees growing in infertile soils are usually genetically adapted to conserve and use nutrients with high efficiency.
- N, K and Ca are accumulated in highest amounts by tree species, followed by P, S and Mg. According to Mengel & Kirby [1978], in all species of higher plants, ‘the N and K content

[2] Elements with low atomic number, such as H, C and S are exceptions from this rule.

of green plant material is about 10 times higher than that of P and Mg which is in turn about 10–1 000 times higher than the content of the micronutrients'.

- Animals, richer in protein than plants, contain two to three times higher concentrations of N and S than the tree foliage. Bacteria are also relatively high in N and S, while values for fungi fall between those for plants and bacteria. The ash content of bacteria, invertebrates and mammals is also higher than of the tree foliage. In general, plant tissues have lower concentrations of nutrients than other life forms, except for O. It should be noted that animals, in addition to the plant nutrients mentioned above, require Na as a macronutrient and F, Si, Cr, Co, As, Se, Sn and I as micronutrients.

2.1.8 Distribution of radionuclides between forest compartments

Even in case of high radioactive contamination, the chemical concentrations of radionuclides are much lower than the concentrations of their stable analogues. Hence, the radioactive contamination does not lead to changes in fluxes of chemicals prevailing in the system. The radionuclides rather tend to distribute in the system in the same way as their stable analogue. But since the cycles of elements in a forest are very slow, it usually takes long time for the radionuclides to achieve a homogeneous distribution with the stable analogue. This peculiarity of the radionuclides, in combination with the high heterogeneity and biodiversity that characterise forests, results in a very high variability of the concentrations of radionuclides in different forest components. Hence, in order to identify general patterns of the distribution of radionuclides between forest component extensive long-term experimental studies are needed. Such comprehensive studies has been performed only for ^{137}Cs, and to a lesser extent for ^{90}Sr, in forested areas of Russia contaminated by the Kysthym accident and in forested areas all-over the world contaminated by the radioactive fallout from test of nuclear weapons in the atmosphere and the Chernobyl accident. The presentation that follows below concerns, therefore, mainly the patterns of ^{137}Cs distribution between forest components. A discussion on the applicability of these patterns to other radionuclides of interest is included at the end of the chapter.

The forest litter layer

As mentioned earlier, nearly 90 % of the ^{137}Cs intercepted by aboveground vegetation is transferred to the soil-litter layer within a few months after the deposition event. Long-term observations carried out in areas contaminated by the Kysthym and Chernobyl accident [Tikhomirov & Shcheglov, 1994; Shcheglov, 1999; Fesenko *et al.*, 2001a; 2001b] have shown that decades after an aerial deposition more than 90 % of the total deposition could still be found in the upper horizons of the soil-litter layer.

An important fraction of ^{137}Cs inventory in the soil-litter layer might be associated with the soil biomass. Estimations made by Olsen *et al*. [1990] indicate that soil mycelia could retain, on average, 32 % of the radiocaesium in the soil-litter layer. Brückmann & Wolters [1994] concluded from a study by the fumigation extraction method, that the microflora contained between 1 and 56 % and on average 13 % of the total amount of ^{137}Cs found in the organic layer of the soil. According to Shcheglov [1999] up to 24 % of ^{137}Cs inventory in forests near Chernobyl is fixed by the soil mycobiota.

Vertical migration of ^{137}Cs in forest soils

Forest soils are characterised by a vertical structure that is termed the soil profile (see Section 2.1.2). The soil-litter serves as a barrier preventing the migration of nutrients and radionuclides down the soil profile, especially in nutrient poor forests [Lieser & Stinkopf, 1989; Melin & Wallberg, 1991; Bunzl *et al.*, 1992; Tikhomirov & Shcheglov, 1994; Fawaris & Johanson, 1994; Bürmann *et al.*, 1994; Fesenko *et al.*, 2001a]. However, the thickness, structure and composition of the forest litter vary widely depending on the type of trees and the climatic conditions. This results in a large variation of the vertical distribution of radionuclides in the soil.

In a study performed in five different sites of the Bryansk region, Fesenko *et al*. [2001a] could observe a decrease of the fraction of ^{137}Cs leached from the litter layer with the thickness of the layer (Figure 2-6). A similar observation was made by Shcheglov [1999] in contaminated forest near the Chernobyl NPP.

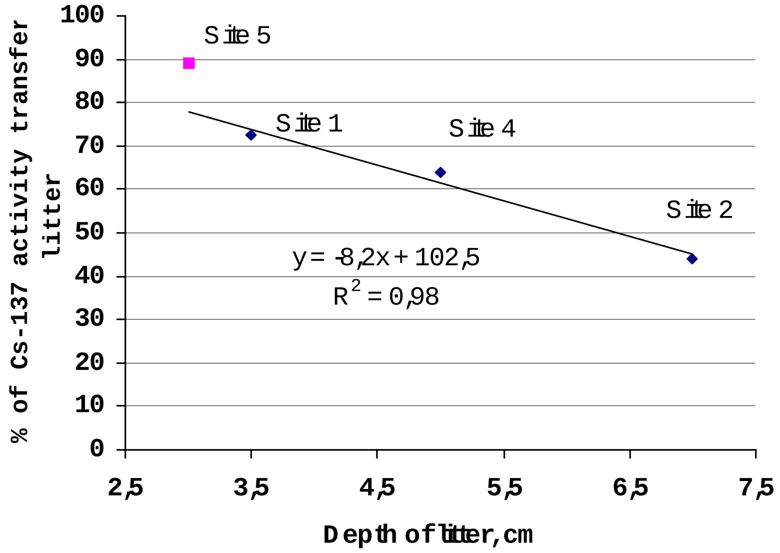

Figure 2-6 Dependence between the quantities of ^{137}Cs leached from the forest litter layer (percentage of the total activity in the soil-litter layer) and the thickness of this layer. Adapted from Fesenko et al. [2001a].

The effect of the litter layer thickness alone cannot fully explain the observed variability in the vertical distribution of radiocaesium in soils. In the study carried out by Fesenko *et al.* [2001a], for example, two sites with the same thickness of the litter layer, showed significant difference in the leaching rate from this layer. This was explained by the fact that one site was covered by coniferous, while the other was covered by deciduous trees. The litter formed from deciduous forest is known to have a higher rate of transfer of potassium, and thus radiocaesium, with soluble organic compounds than the litter formed from coniferous forests. Another factor that has to be taken into account is the fixation of radionuclides by the soil biota. The distribution of fine roots of trees in soil varies substantially with depth (Figure 2-7) and has been shown to correlate well with the ^{137}Cs vertical distribution in soils [Fesenko *et al.*, 2001b]. The fixation of ^{137}Cs by fungi mycelia was already mentioned above. To this, it should be added that upward transport in soils of ^{137}Cs by microflora has been reported [Brückmann & Wolters, 1994]. According to Llauradó *et al*. [1994] a high faunal mixing of ^{137}Cs among the forests floor layers and between them and the upper mineral layers is a distinctive characteristic of Mediterranean forests. Shcheglov [1999] also observed an extremely fast migration of ^{137}Cs in steppe forests on chernozems, which he attributed to the activity of soil invertebrates, which are especially abundant in these soils.

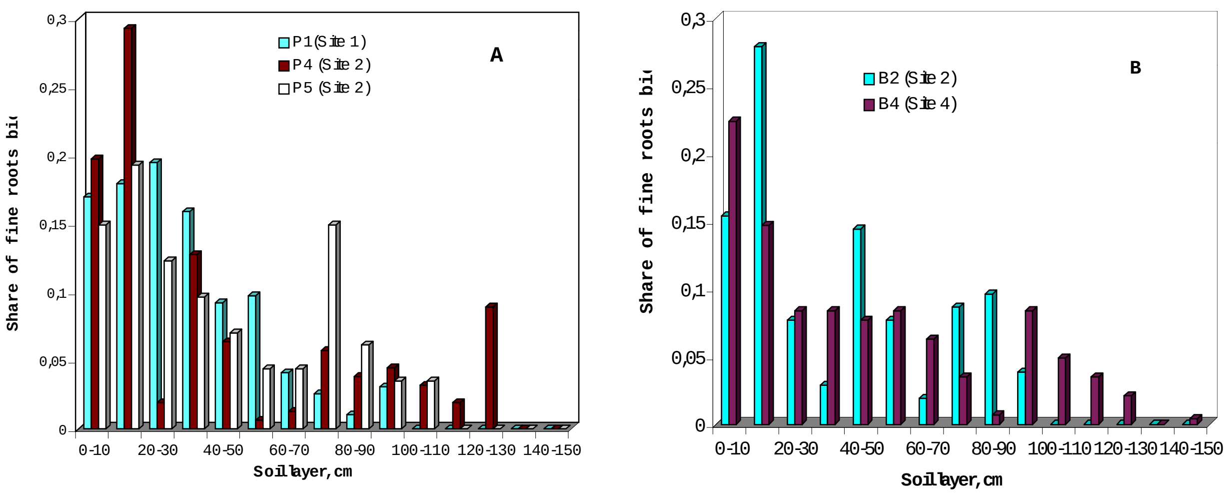

Figure 2-7 Distribution of fine roots biomass of pine (A) and birch (B) trees in different soil horizons. Adapted from Fesenko et al. [2001b].

The tree layer

The transfer of ^{137}Cs from soil to trees as well as the radionuclide distribution within the tree involves multiple components and processes interrelated in a complex way (see Figure 2-4). For the same tree species, variations between sites of up to three orders of magnitude have been observed depending on the soil conditions and type of landscape [Shcheglov, 1999]. In order to identify patterns in the radionuclides distribution in trees, experimental data for radionuclide concentrations in different parts of the trees should be collected in the same site, at the same time after deposition and a large number of factors need to be controlled. To illustrate this point in we present predictions of the activity levels in wood of pine trees made with the model FORESTLAND [Avila *et al.*, 1999] for different types of soils and age of trees at the moment of the deposition (Figure 2-8). This figure reveals a very large variation not only in the activity levels, but also in the form of the predicted time dynamics. The regularities in ^{137}Cs distribution in trees presented below are based on a long-term study that has been conducted in five sites of the Bryansk region of Russia that were heavily contaminated by the Chernobyl deposition [Fesenko *et al.*, 2001b]. This study fulfils the above-mentioned conditions.

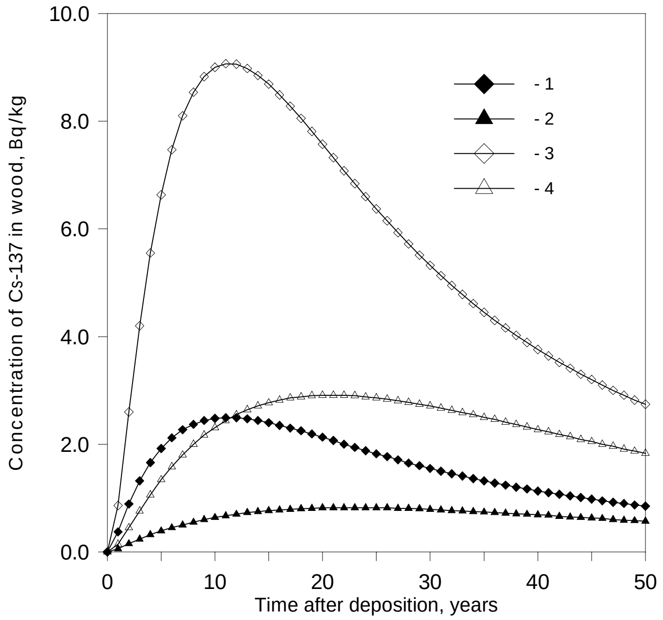

Figure 2-8 Activity levels in wood for different types of forest soil and ages of trees – deposition 1 kBq/m² of ¹³⁷Cs (1 and 2 – automorphic soil; 3, 4 – semi-hydro-morphic soil; 1 and 3 – initial age 20 years; 2 and 4 – initial age 80 years). Adapted from Avila et al. [1999].

^{137}Cs distribution within trees

The ratios of ^{137}Cs concentration in different parts of the pine and birch trees with respect to the concentrations in trunks at the height of 1.3 m are presented in Table 2-4. The highest ratios were observed for roots, current needles (pine trees), current branches (pine trees), leaves (birch) and bark. The lowest values were observed for wood, older pine needles and branches. These ratios are unstable throughout the year, for example ^{137}Cs activity concentrations in needles and leaves vary considerably across seasons. Nevertheless, they do reflect peculiarities in the physiological functioning of different tree parts. For instance newly created tissues behave like sink compartments during the growth period and act as sources of nutrients at the end of this period, while roots act as a final sink for potassium and thus radiocaesium. In general these results are in concordance with the general patterns of nutrients cycling in trees presented above.

The observed variation in concentrations in the trunk wood at different heights can be explained by the fact that the tree trunk serves as a natural filter for radionuclides transported from roots to needles or leaves with the xylem sap. The radionuclides moving along the xylem are absorbed in the newly formed annual ring and this leads to a reduction of their concentration in the xylem sap. Hence, a decrease in concentration with height is observed until about two thirds of the tree height. Starting from this height the diameter, and consequently the cross section, of the trunk gets smaller and smaller, which leads to a new increase in the concentrations. This effect is obviously more pronounced for tall trees.

Table 2-4 Ratios between ^{137}Cs concentrations in different parts of the tree and ^{137}Cs concentration in the tree trunk at the height of 1.3 m. Adapted from Fesenko *et al.* [2001b].

Parts of trees	Mean	95 % C.I.	Range
Pine			
Needles 1 year	25.7	19.3–34.2	14.5–44.3
Needles 2 year	5.5	3.2–9.3	1.6–11.7
Needles 3 year	3.9	2.3–6.5	1.3–9.1
Needles 4 year	4.6	3.0–7.0	2.0–7.7
Branches 1 year	22.0	14.3–33.9	9.2–40.3
Branches 2 year	8.2	5.8–11.5	3.9–13.7
Branches 3 year	6.4	5.5–7.4	4.5–7.6
Branches 4 year	5.3	4.5–6.1	4.3–6.5
Roots (wood)	6.9	3.0–15.8	1.5–18.5
Wood 1.3 m	1.00	–	–
Wood M	0.69	0.51–0.93	0.34–1.0
Wood T	0.91	0.67–1.25	0.60–1.5
Bark 1.3	4.6	3.4–6.1	2.6–7.3
Birch			
Leaves	26.0	11.8–57.4	9.7–58.9
Branches	4.4	2.8–7.0	2.4–7.1
Roots (wood)	39.4	17.6–88.3	18.4–87.2
Wood 1.3 m	1	–	–
Wood M	0.82	0.72–0.95	0.71–0.96
Wood T	1.0	0.8–1.3	0.76–1.4
Bark 1.3	6.5	5.1–8.4	4.9–8.3

A radial movement of ^{137}Cs in the tree trunk has also been observed [Fesenko *et al.*, 2001b]. This results in a pattern of ^{137}Cs concentration decreasing exponentially with the age of the ring. In the same study that provided the results shown in Table 2-4, the ^{137}Cs concentration in the outer ring of pine trees exceeded that in the second and third ring by a factor of 1.4–1.7 and 2.1–2.6 respectively. The corresponding factors for birch trees were 1.7–1.8 and 2.8–3.2 respectively. A similar pattern was observed for potassium [Fesenko *et al.*, 2001b].

The results presented above clearly indicate that ^{137}C is highly mobile inside the tree and that this radionuclide behaves in the tree in a very similar way to potassium.

The understorey layer

The characteristics and species composition of the understorey layer in a specific site will largely depend on the characteristics of the tree and soil-litter layer at the site. An infinite number of niches of different character exist in this layer, which is reflected in a very large biodiversity. It is possible to affirm that the very high variability of ^{137}Cs accumulation by different organisms is the main feature of the behaviour of this radionuclide in the understorey layer. Reported transfer factors of ^{137}Cs to understorey plants, game animals and especially for mushrooms vary within several orders of magnitude [see for example IAEA, 1994]. Below we discuss some regularities of ^{137}Cs transfer to different organisms, which explain, to some extent, the high variability observed in this layer.

Understorey plants

The rooting depth, which is a life-form niche (see Section 2.1.2), has been considered as a key factor in the differences in ^{137}Cs levels observed in understorey plants [Römmelt *et al.*, 1990; Guillitte *et al.*, 1994; Shcheglov, 1999; Fesenko *et al.*, 2001a]. This can been explained by a different content of available ^{137}Cs in different soil-litter layers. According to Nimis [1996] the microhabitat created by depressions in the soil can also influence on the radionuclide accumulation. In addition to root depth Guillitte *et al.* [1994] postulated that in some plants mycotrophism might also result in higher contamination values.

Mushrooms

The fact that mushrooms effectively and selectively concentrate radionuclides was already known before the Chernobyl accident [Roehleder, 1967; Seeger & Schweinshaut, 1981; Eckel *et al.*, 1986]. More evidence of mushrooms being accumulators of radionuclides has been obtained in studies carried out after the Chernobyl accident [Eckel *et al.*, 1986; Mazcanzoni, 1987; 1990; Byrne, 1988; Römmelt *et al.*, 1990; Guillitte *et al.*, 1990; 1994; Heinrich, 1992; Strandberg, 1994; Yoshida & Muramatsu, 1994].

High differences in the capacity of accumulation of ^{137}Cs between species of mushrooms and sites have been reported. The following main explanations of this phenomenon have been put forward:

- The fungi mycelia habit different soil layers in combination with a non-uniform distribution of ^{137}Cs in the soil profile [Eckel *et al.*, 1986; Mazcanzoni, 1990; Olsen *et al.*, 1990; Heinrich, 1992].
- Symbiotic fungi accumulate more ^{137}Cs than saprophytic fungi, due to, on one hand, a better capacity of the former to accumulate nutrients in general and on the other hand because mycelia of symbiotic fungi occupy more contaminated horizons inhabited by mycorrhizae [Guillitte, 1994; Strandberg, 1994; Yoshida & Muramatsu, 1994].

Mammals (herbivores and predators)

Taking into account the large variability of ^{137}Cs levels observed in trees, understorey plants and mushrooms, a large variation of ^{137}Cs levels in herbivores grazing on them should be expected. Studies of ^{137}Cs levels in moose and roe dear carried out during the last 15 years in forested areas contaminated by the Chernobyl accident has confirmed this. Below we summarize some patterns in the variability of ^{137}Cs levels in these herbivores that can be inferred from these studies:

- The levels of ^{137}Cs in roe deer and moose in areas contaminated by the Chernobyl accident show a rather slow long-term reduction and in some cases are decreasing at a rate close to the physical half-life of ^{137}Cs [Bergman *et al.*, 1991; Johanson & Bergström, 1994; Avila, 1998]. This has been explained by a long persistence of ^{137}Cs availability for uptake by plants and fungi eaten by game [Bergman & Johanson, 1989]. However, pronounced yearly variations may be obscuring the real character of the long-term trends [Nylén, 1996; Avila, 1998].
- Large seasonal and yearly variations in ^{137}Cs activity concentrations in roe deer have been observed. There is evidence that changes in the rates of fungi ingestion is a major reason

for these variations [Karlén *et al*., 1991; Strandberg & Knudsen, 1994; Kiefer *et al.*, 1996; Avila, 1998].

- An increase of the levels of ^{137}Cs in moose harvested during summer and autumn has been observed. Nylén [1996] discussed a possible connection between these changes and moose feeding during summer and autumn on plants and fungi with high ^{137}Cs activity concentrations. In a study carried out in Central Sweden, Avila [1998] observed correlation between ^{137}Cs levels in roe deer harvested in August and in moose harvested in October in the same area. This was attributed to both roe deer and moose feeding on mushrooms.
- ^{137}Cs activity concentrations in moose show large non-systematic variation from year to year [Nelin & Nylén, 1994; Nylén, 1996]. Palo & Wallin [1996] suggested that such variation is due to changes in the share of plants with different ^{137}Cs activity concentrations in the animal's diet. These changes in the diet might be related to changes in plant abundance and/or in the animal habitat. Currently, there is not satisfactory understanding of the prevailing relationships. Yearly variations in the ingestion of fungi could also be a reason for the observed variations [Avila, 1998].
- The individual variability of ^{137}Cs activity concentrations in roe deer and moose is high. There is often a correlation between the standard deviations and mean values, which are of the same order of magnitude [Nylén, 1996]. In the case of moose, part of the variability can be explained by different accumulation by animals of different age and sex [Nelin, 1994]. Calves usually show higher (up to 40 % higher) ^{137}Cs activity concentrations than moose. This was explained by differences in the ^{137}Cs biological half-life of animals of different age (size), a larger daily ingestion of food in relation to body weight for calves than for adult moose and differences in food selection. Females often accumulate ^{137}Cs more readily than males, although differences by sex are less pronounced than differences by age. Correlation between ^{137}Cs levels in moose and the characteristics of the particular place were the animals was shot has been found [Nylén, 1996].

2.1.9 Other radionuclides

From the regularities of ^{137}Cs behaviour in forest ecosystems presented above, it can be concluded that this radionuclide is highly mobile and biologically available in the system. Caesium radioisotopes entering a forest ecosystem are readily incorporated into the potassium cycle continuously going on in the system and can, in principle, be transferred to living organisms of all trophic levels of the forest food chains (see Section 2.1.3). It is, therefore, reasonable to assume that the transfer processes and pathways (see Figure 2-4) identified for radiocaesium constitute a rather complete list also for most of the radionuclides of interest[3]. The intensity and importance of the transfer processes and pathways might, however, vary from radionuclide to radionuclide. Below we discuss the relative importance (with respect to caesium) of different transfer processes for the radionuclides of interest. The discussion is based on one hand on the scarce knowledge available on the behaviour of these radionuclides in the environment [adapted and summarized from Coughtrey & Thorne, 1983] on and on the other hand on the knowledge of the cycling of their stable analogues in the system.

[3] The validity of this assumption is more questionable for isotopes of C and H due to possible isotopic effects and because these elements, along with O, are the main structure components of living organisms.

Strontium ($^{89, 90}$Sr)

Strontium is an analogue of Ca, which is an essential micronutrient for both plants and animals. Hence radioisotopes of strontium, as caesium, can be transferred to living organisms of all trophic levels of forest food chains. Strontium usually shows more mobility in soils and is more readily taken up by plants than caesium. Marked differences might be observed in the distribution of caesium and strontium isotopes in aboveground parts of the plant. For instance, in evergreens, foliar concentrations of caesium (as potassium) decline with leaf age, while the concentrations of strontium (as calcium) tend to increase with leaf age. Strontium and calcium tend to concentrate in bone tissues of animals, while caesium and potassium tend to be homogenously distributed in soft tissues. ^{90}Sr in forest soils persists in the organic horizons and shows a very low migration rate down the soil profile. The qualitatively different vertical profiles of ^{90}Sr and ^{137}Cs suggest an efficient transfer from soil to plants and a continuous supply of ^{90}Sr to the litter horizon via leaf-turnover [see Bruchertseifer *et al.*, 2001]. The accumulation of Sr in fungal fruit bodies is lower that the accumulation of Cs [Yoshida *et al.*, 1998]. It should be noted that a model of ^{90}Sr transfer in forest ecosystems has been developed [see Alexakhin *et al.*, 1994].

Actinides ($^{238, 239, 240, 241}$Pu, ^{241}Am, ^{237}Np, ^{242,243,244}Cm)

The behaviour of actinides in soils is characterized by the formation of chelates with fulvic and humic acids that remain relatively stable. They usually show low mobility in soil and limited root uptake by plants. The actinides have also low mobility in plants and if incorporated in the plant by root uptake will remain in the roots. Among them Np is the most mobile in soil, most available to plants and more easily translocated within the plants. Soil is the major repository of Pu in the forest. Only less than 0.25 % of existing inventory of Pu in forest ecosystems resides in forest biota. It should be noted that a model of Pu transfer in forest ecosystems has been developed [see Garten *et al.*, 1978].

Niobium (^{95}Nb)

Niobium behaves in soils as zirconium and is usually forming insoluble oxides. Niobium and zirconium in soil show low availability for plant uptake. Of the ^{95}Nb that is absorbed by plants, only 10 to 15 % is translocated to shoots.

Ruthenium (^{106}Ru)

After an aerial contamination, 10 % retention of ruthenium by plants can be expected. Considerable uptake of ruthenium by plant roots has been reported. Soluble ruthenium nuclides added to soil are translocated to plant shoots and accumulated by leaves in particular. Ruthenium complexes appear to be localised in the roots with little translocation to shoots.

Technetium (^{99}Tc)

Technetium uptake by plants is characterised by high transfer factors and depends strongly on the bioavailability of the radionuclide in soil. In aerobic soils, contaminated with technetium in the pertechnetate form, 90 % of added technetium can be assumed to remain in soil solution either as the free ion or weakly absorbed to ion-exchange sites. In very acid soils about 60 % can be expected to be bound rapidly to soil minerals. It is considered that technetium is very mobile in plants and can readily be transferred from roots to shoots and subsequently to developing fruit and seed tissues. The metabolism of technetium in animals closely resembles that of iodine. In mammals, technetium competes with iodine in thyroid uptake and the fractional uptake to this organ is, therefore, dependent on the level of dietary iodine. However, since technetium cannot be utilised in the production of thyroid hormone, it is lost rapidly from the thyroid. Technetium can also accumulate in the gastrointestinal tract and the liver.

Iodine ($^{129, 130, 131}$I)

Soil iodine availability to plants is influenced by soil type. Soils rich in iodine, such as clay and peat, hold iodine, whereas it is rapidly leached from sandy soils. Plants usually take up radioiodine more readily than caesium, but less readily than strontium. Very fast translocation of iodine within trees and even between adjacent trees has been reported. Among the vertebrates iodine accumulates in the thyroid gland. Iodine in hormonal combination has been reported as important to the metamorphosis of amphibians. Radioiodine transfer from the mother to embryo has been demonstrated in rabbits and other vertebrates. Generally the fetal thyroid concentration is higher than of the dam.

Chlorine (^{36}Cl)

Chlorine is a plant micronutrient and an essential macronutrient for animals. Hence, ^{36}Cl is probably mobile in forest food chains and its concentrations will be higher in animals than in plants.

Nickel ($^{59, 63}$Ni)

When Ni enters a soil system it can be assumed that 90 % is rapidly adsorbed or occluded by the mineral lattice and is hence relatively unavailable for plant uptake. From the scarce data available it appears that Ni translocation within the plant can readily occur.

3. Semi-natural pastures and heathlands

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Semi-natural pastures and heathlands incorporate a broad range of ecosystems including mountain (e.g. Alpine pastures) and upland grasslands (e.g. those characteristic of many upland areas of the UK), heath- and shrublands (e.g. Mediterranean garigue), saltmarshes and some Arctic ecosystems. Many of these ecosystems, most especially those in the Mediterranean, are species rich with considerable biodiversity [e.g. EEA, 1998]. These ecosystems are termed 'semi-natural' since, whilst they comprise natural species not introduced by man, they have been influenced by human use, for instance by the grazing of livestock. Indeed many natural semi-grasslands would revert to scrub and woodlands if it were not for their utilisation by man.

3.1 Ecosystem descriptions and typical species

The European Union *Natura 2000* natural habitats classification system [EC, 1999[4]] lists 79 habitat classes, with further sub-classes, which could be considered within the broad description 'semi-natural pastures and heath-/shrublands' as defined here; Arctic ecosystems are not considered within *Natura 2000*. A detailed description of all these habitats is outside the scope (and requirements) of this work. Within the following, brief descriptions of typical species and characteristics of these ecosystem types are noted. If more detailed species listings are required there are a number of European atlases for fauna [e.g. Hagemeijer & Blair, 1997; Gasc *et al*., 1997; Mitchell-Jones, 1999]. Flora characteristics of the relevant habitats have been identified by the *Natura 2000* natural habitats classification system [EC, 1999].

The characteristics of soils present within a given ecosystem are generally determinant factors in the bioavailability of radionuclides (see Section 3.2.1). The European Soil Bureau database [1999] contains information on various chemical and physical properties for georeferenced soil profiles within Europe.

3.1.1 Saltmarshes [see Dijkema, 1984; Adam, 1990]

Found from the Mediterranean to the Arctic, coastal saltmarshes can be defined as areas, vegetated by herbs, grasses or low shrubs, which are subjected to periodic flooding by adjacent saline water bodies. Coastal saltmarshes occupy the interface between sea and land and consequently their flora and fauna have both marine and terrestrial elements. They are highly dynamic environments subject to erosion, accretion and progradation. Because saltmarsh soils are often waterlogged, and anaerobic, they represent a very different physiochemical environment to free draining soils. Saltmarshes differ from most other terrestrial ecosystems in that, potentially, *in situ* production by plants is not the only carbon

[4] Available from http://www.mrw.wallonie.be/dgrne/sibw/N2000/home.html

and energy source; an alternative supply is provided by material washed into the marsh by the tide although production within the marsh usually dominates.

Saltmarshes do not generally have a great diversity of vegetation species, although usually species richness increases from the lower to upper marsh. Species typical of the *circa* 5 000 occurring within European saltmarshes include: *Armeria* spp., *Festuca*, *Glaux maritima*, *Halimione* spp., *Juncus* spp., *Salicornia* spp. and *Spartina* spp. Vegetation communities characteristic of saltmarshes of the different biogeographical regions of Europe are described by Dijkema [1984]; diversity generally decreases with increasing latitude.

The pioneer zone (sparsely vegetated with species such as *Spartina* spp. and *Salicornia* spp.) of saltmarshes may contain 300 species of meio- and micro-zoobenthos with *circa* 200–500 individuals cm^{-2} (example data for Wadden Sea, [Adam, 1990]). The macro-zoobenthos comprises about 100 species, of which species such as *Nereis diversicolor* (polychaete), *Macoma baltica* (bivalve mollusc), *Hydrobia ulvae* (gastropod) and *Corophium* spp. predominate, with, for example, 120 000 to 400 000 individuals m^{-2}. The majority of species are either detritivores or consumers of the microflora (algae, cyanobacteria and bacteria). The continuously vegetated marsh supports a richer fauna with insects, including phytophagous species. Spiders are the major invertebrate carnivores. Crabs may be present within the lower marsh.

Saltmarshes provide an important habitat for breeding birds such as: mallard (*Anas platyrhynchor*), shelduck (*Tadorna tadorna*), oystercatcher (*Haematopus ostralegus*), black-headed gull (*Larus ridibundus*) and redshank (*Tringa totanus*). Well-grazed saltmarshes are favoured by a range of wildfowl, including wigeon (*Anas penelope*), teal (*Anas crecca*), brent goose (*Branta bernicla*) and pink-footed goose (*Anser brachyrhynchus*), which feed on saltmarsh grassess (e.g. *Puccinella* spp.) and the seeds of saltmarsh plants. Raptors such as hen harrier (*Circus cyareus*), merlin (*Falco colvarbarius*) and short-eared owl (*Asio flammeus*) hunt extensively over saltmarshes.

The predominant large herbivores grazing saltmarshes are domestic livestock; red deer (*Cervas elephas*) are known to graze saltmarshes in Scotland. Where sand dunes are adjacent to saltmarshes grazing by rabbits (*Oryctolagus cuniculus*) can be intense, hares may graze lower marshes. In upper marshes voles (*Microtus* spp.) are common. Reptiles and amphibians are not normally characteristic of saltmarshes.

3.1.2 Semi-natural grasslands [see Coupland, 1979; 1993]

Occurring under a wide range of climatic conditions, semi-natural grasslands are found largely in deforested uplands or where physical factors are unfavourable for forest development (e.g. river valleys subject to flooding). They may be utilised as hay meadows and as pastures for domesticated animals; differences in management regime determining plant species. If mowing and grazing were to cease many of these areas would revert to woodlands by natural succession. In areas with mild winters, they are practically evergreen; even in areas with severe frosts they do not loose all their biomass.

The formation of semi-natural grasslands is dependent upon climatic conditions but in general they can be found where the mean annual precipitation is between 430–2 330 mm and the mean annual temperature between 0 ^{o}C and 10 ^{o}C. The soils are predominantly acid to neutral (pH range 5.5–6.5) and rich in organic matter (5–27 %) with a wide range in C:N ratios (8–

29). Neutral grasslands are the most widespread in Europe, mostly occurring on clays or loams.

The carbon and nitrogen cycles are highly specific in grassland ecosystems, differing from forest and agricultural ecosystems because of the abundant and biologically active underground biomass. The return of 60–80 % of plant biomass to the soil through the detritus (decomposer organisms) and grazing food chain maintains soil fertility and structure.

Flora

Semi-natural grasslands are dominated by perennial graminoids (grass and grass like plants) mostly native to a given region. The vegetation is multi-layered, the upper layer containing tall grasses and forbs, the lower layers being dominated by species of shorter stature that dominate early in the season before being overtopped by the taller grasses. Table 3-1 gives examples of the differing types of semi-natural grasslands found in Europe together with some of the more common species found within them. The presence of 80 species of vascular plants in a stand is not exceptional, although commonly two or three species provide 60 % or more of the total shoot biomass.

Table 3-1 Common vegetation species of semi-natural grassland ecological types in Europe [Coupland, 1993].

Grassland type	Habitat	Floristic description	Common species	Management
Plains and slopes on drier plateaux	Rich soils in sites of former oak forests typical of the sub-Mediterra-nean zone	Floristically diverse, numerous orchids	*Bromus erectus*, *Koeleria pyramidata*, *Brachypodium pinnatum*, *Festuca ovina*, *Carex humilis,*	
Plains and slopes on drier plateaux	Sub continental climate of Western Europe	Floristically diverse	*Xerbromiom* spp., *Iris* spp., *Scabiosa* spp.	
Lowlands and moister plains	Rich and mostly fertilised soils, water is supplied by rainfall or from underground sources, flooding does not occur	Tall productive grass meadows	*Arrhenatherum elatus, Alopecurus pratensis, Dactylis glomerata, Trisetum flavescens, Festuca rubra, Agrostis tenuis*	Mown 2–3 times per year for animal fodder
Lowlands and moister plains	Rich and mostly fertilised soils, water is supplied by rainfall or from underground sources, flooding does not occur	Short grass pastures	*Lolium perenne, Cynosurus cristus, Festuca rubra, Phleum pratense, Trifolium repens*	Pasture
Uplands, submontane and montane grasslands	Rich fertilised soils	Tall grass forb communities, species rich	*Trisetum flavescens, Festuca rubra, Alopecurus pratensis, Agrostis tenuis, Nardus stricta*	Mown meadows

Table 3-1 (continued) Common vegetation species of semi-natural grassland ecological types in Europe [Coupland, 1993].

Grassland type	Habitat	Floristic description	Common species	Management
Uplands, submontane and montane grasslands	Poor acid soils	Short grass forb communities	*Festuca rubra, Agrostis tenuis, Anthoxanthum odoratum, Briza media, Alchemilla vulgaris*	Mown meadows
Uplands, submontane and montane grasslands	Pastures on acid, minimally fertilised soils	Short grass communities	*Festuca rubra, Agrostis tenuis, Cynosurus cristatus, Festuca ovina, Nardus stricta*	Grazed
Uplands, submontane and montane grasslands	Pastures on acid, unfertilised soils growing on the site of former forests, extend above the tree line	Short grass communities	*Nardus stricta, Festuca ovina, Agrostis tenuis, Festuca rubra, Calluna vulgaris*	Grazed
River valleys and basins	Nutrient rich soils prolonged flooding,	Hygrophytic tall grass communiti-es; on terraces near to a river	*Phalaris arundinacea, Glyceria maxima, Carex gracilis, Poa palustis, Agrostis stolonifera*	
River valleys and basins	Nutrient rich soils, ground water retreat in summer	Mesohygrophytic tall grass forb communities; on terraces near to a river	*Alopecurus pratensis, Cirsium oleraceum, Festuca pratensis, Poa pratensis, Festuca rubra*	
River valleys and basins	Rich, calcareous unfertilised soils, with a sharp decline in the underground water in the summer, more common in continental Europe	Mesohygrophytic short grass-forb communities	*Molinia coerulea, Carex pinicea, Bromus erectus, Deschampsia caespitosa, Serratula tinctoria*	Once well distributed but increased fertilisation and mowing has changed them to more fertile lands
River valleys and basins	Acid soil mainly in Atlantic/ Sub-Atlantic part of Europe	Mesohygrophytic short grass-forb communities	*Molinia coerulea, Holcus lanatus, Festuca rubra, Juncus effusus, Succisa pratensis*	
River valleys and basins	Sub continental	Mesohygrophytic short grass-forb communities	*Festuca rupicola, Festuca rubra, Anthoxanthum odoratum, Galium boreale, Galium verum*	

Microflora and invertebrates

The grazing and detritus (decomposer organisms) food chains of grasslands terminate with the decomposing activity of bacteria, yeasts, actinomycetes, microscopic fungi and protozoa. The most abundant groups of microorganisms in grassland soils are bacteria, actinomycetes and fungi, algae are less numerous although more abundant than protozoa. The numbers present are greater than in forest or arable soils.

The soil fauna of semi-natural grasslands is richer both in quantity and diversity than that of arable soils; the biomass of earthworms in semi-natural soils is three times that of arable soils. The most important soil invertebrates are: earthworms (typically 200–800 m^{-2}), nematodes (10^6–10^{10} m^{-2}), collembola (10^5 m^{-2}), mites (10^5 m^{-2}) and enchytraeids (10^5 m^{-2}). Of the invertebrates present in the vegetative layer, grasshoppers and other members of the order Orthoptera are the most common herbivores, followed by bugs (hemiptera), aphids and leaf hoppers (homoptera); spiders being important predators.

Vertebrates

The diverse range of vertebrate species includes ungulates, rodents, predators of the cat and dog family, birds, lizards and snakes. Although a great variety of birds inhabit semi-natural grasslands only few nest within them. For example, Pelekàn [1985] found that as few as six species of birds nested or foraged in (Czech) meadows, whereas 31 species, which nested in other biotopes, fed in the meadows. Consequently, a significant amount of biomass is removed and distributed between ecosystems.

3.1.3 Heathlands [see Specht, 1993; Horrill, 1990]

Heathlands and related ecosystems are found in all parts of the world, from the tropics to Polar Regions, from lowland to sub alpine altitudes. They are generally thought of as areas dominated by Ericaceous species (e.g. *Calluna vulgaris*, *Erica tetralix*), open and generally without trees, but which can also include areas of gorse (*Ulex gallii*) and birch (*Betula* spp.) scrub and bracken (*Pteridium aquilinum*).

The climatic conditions in which many heathlands occur could support forests, to which heathlands would revert if they were not managed. Indeed, many were originally developed as forests but were destroyed for charcoal production, hence more open spaces occurred and the grazing of domestic animals began.

Most western European heathlands require moist temperate conditions with mild winters and long spring and autumn periods. The mean temperature for four months of the year is above 10 ^oC with an annual rainfall of 600–1 100 mm. Well developed heaths of this type are found in northwest Scotland, the extreme west of southern Norway and western Ireland. In northern Scotland, persistent cloud and mist, low summer temperatures and a short growing season confine heathlands to lower altitudes (360–850 m). Heathlands do not survive late lying snow although they are well adapted to cope with a protective snow cover in winter.

In general heathlands are not floristically rich, as the species present are basically woodland fauna less those species that need trees, those that get eaten by domestic livestock, and those that do not tolerate periodic burning. For instance a square metre of heather moorland may contain as few as five species of plants (including mosses and lichens) the average being 10 species per m^2; there are very few annual plants. This can be compared to about 20 species per m^2 found in woodlands.

In the south of Europe, where it is hotter and drier, dwarf shrub communities such as Mediterranean garigue occur. Garigue is widely distributed and is known by many different names in various countries. In general shrubs 50 cm high (e.g. *Juniperus oxycedrus*) can be found amidst patches of bare earth. Many of the species are not heavily grazed as a consequence of being chemically resistant (e.g. *Euphorbia* spp., *Cystus* spp.), aromatic (e.g. *Lavendula* spp.) or having spines (e.g. *Carlina* spp.). If ungrazed these areas would develop into oak or pine woodland. The soils are very different from those found in north and west Europe often being thin and rocky, and subject to erosion. The floristic composition is also very different from more western heathlands in that many of the plants are annuals, which die down completely in the summer. Amongst the many plant families that are particularly well adapted to survive the hot and dry conditions are members of the pea (Leguminosae), mint (Labiatae*),* daisy (Compositae) and the orchid (Orchidaceae) families.

Most temperate heathlands can be sub-divided into three broad categories: dry heathlands, wet heathlands and mountain heathlands.

Dry heathlands

Dry heathlands are frequently found on freely drained, nutrient poor, podzolic soils. Commonly, dense cover is provided by evergreen ericaceous species (generally *Calluna vulgaris*). Other low shrub species include *Erica cinerea* and *Vaccinium myrtillus*. Below this layer there may be some species of partially shade tolerant creeping dwarf shrubs such as *Vaccinium vitis idaea* and *Empetrum nigrum*, herbs (e.g. *Viola riviniana*, *Thymus* spp.) and graminoid plants (*Galium saxatile*, *Potentilla erecta*), with bryophytes (e.g. *Pleurozium schreberi*) and lichens dependant upon the habitat and age of the heath. About 75 % of the total number of green plants found in dry heathlands is bryophytes or lichens.

There are substantial differences between northern and southern heathlands and between western and eastern heathlands. For example *Juniperus communis* is lacking in most British heathlands, whilst *Vaccinium vitis idaea* is characteristic along the western fringes of southern Sweden, Denmark and the Netherlands. *Arctosaphylos uva-ursi* is frequent in southwest Norway and Scotland and where sub-continental conditions prevail in southwest Sweden and Denmark. In northern France and some parts of Britain *Ulex europeaus* is abundant and can dominate the heath. In transitional areas between north and south *Genista angelica* can be found. *Erica vagans* is significant in parts of France and Spain together with *Daboecia cantabrica.* In the more southerly extremes such as southwest France and in the coastal lowlands of northern and northwestern Spain and Portugal *Erica umbellata* and other *Erica* spp. dominate, *Cistus* spp. can be found, and there are fewer bryophytes.

Wet heathlands

Wet heathlands occur on soils with impeded drainage and are found on margins of peat bogs and wet hillsides in high rainfall areas. Tall shrubs (e.g. *Salix* spp.) are sometimes present, low shrubs are less dense than in dry heathlands. In habitats where there is a moderate moisture status (humid heath) *Calluna vulgaris* and *Erica tetralix* are found together with abundant species of sedge and some *Sphagnum* spp. mosses. In wetter habitats there is less

Calluna vulgaris and greater amounts of *Erica tetralix*, *Ledum palustre* and *Myrica gale*. These shrubs are distributed amongst a mixture of tall grasses such as *Molinia caerulea* or tufted sedges such as *Eriophorum* spp. and *Trichophorum caespitosum*. There is usually a dense ground cover of *Sphagnum* spp. and other bryophytes together with herbs but few, if any, lichens. Wet heathlands do not show the same geographical diversity as dry heathlands although *Empetrum nigrum* is common in northern areas whereas *Erica ciliaris* and *Erica scoparia* are common in the south

Mountain heathlands

Mountain heathland is widespread above the tree line. Dwarf shrubs (< 10 cm high) such as *Salix* spp. and Ericaceae form a low patchy mat. Graminoid species, bryophytes and lichens are also present. This type of heathland is common in the low alpine zones (1 200–1 500 m) of European mountains. Usually occurring on ranker soil, the floristic composition changes with altitude and climatic conditions. *Vaccinium* species are typical as is dwarf juniper at the transition between subalpine scrub and low alpine scrub together with dwarf *Calluna vulgaris* which becomes dominant above the tree line disappearing at higher altitudes. In Britain *Calluna vulgaris* and the moss *Rhacomitrium langinosum* are found together on the mountainous areas whilst *Calluna vulgaris* with *Vaccinium* spp. are more widespread in Scandinavia, Germany and France.

Soils

Heathlands generally occur on acid soils of low fertility (e.g. podzols, rankers, peats and humic gleys) especially those deficient in phosphorous and nitrogen. The pH is normally between 3.4 and 6.5, with a high C:N ratio and low levels of exchangeable cations. Highly organic peats and podzols generally occur in northern and western Europe. In more southerly temperate countries high quality brown earths are common with dry, stony, eroded soils in the Mediterranean regions.

Microflora and invertebrates

Heathland litter has a low soluble carbohydrate and persistent tannin content which is unpalatable and avoided by many organisms (e.g. earthworms and millipedes) that would otherwise cause decay. Consequently litter is not readily mixed with the mineral material within soil, and densities of soil micro and macro fauna are less than in woodlands. To compensate for the lack of soil micro and macro fauna, peptone and pectin decomposing yeasts are important in the decomposition of *Calluna vulgaris* litter.

Poor soil conditions and harsh climates ensure that plants grow slowly; hence there is a limited food supply and a limited number of characteristic species. Examples include the heather beetle (*Lochmea suturalis*) and its predator the ladybird; the upland reed beetle (*Plateumaris discolor*) usually associated with *Eriophorum* spp.; more than 60 species of spider; the emperor and eggar moths; mountain agus (butterfly); frog hoppers; crane flies. The variety and number of invertebrate species is greater in managed (periodically burnt) heaths than in unmanaged heaths. In general insect diversity decreases with increasing altitude.

Vertebrates

Vertebrate herbivores are not very numerous in European heathlands due to the relatively recent origin of the habitat, the low productivity of non-woody vegetation and grazing by domestic herbivores. The mountain hare (*Lepus timidus*) is characteristic of upland and northern heathlands; *Calluna vulgaris* providing almost 100 % of the diet of this species in winter. The brown hare (*Lepus europus*) and the rabbit (*Oryctolagus cuniculus*) are found on lowland heaths. Red deer (*Cervus elaphus*) are found on upland heathland in Britain where densities are high in comparison to other areas of western Europe.

In Scandinavia the willow grouse (*Lagopus lagopus*) can be found on heathland; open heathland in Britain is the natural habitat of the red grouse (*Lagopus lagopus scoticus*). Red grouse feed almost exclusively on *Calluna vulgaris* in winter, which also provides 50 % of their summer diet together with the fruit and leaves of *Vaccinium myrtillus*, *Empetrum nigrum*, *Oxycoccus* spp., *Erica* spp., and crane flies. In Britain large areas of *Calluna vulgaris* heathland are managed as 'grouse moors' rather than extensively grazed by domestic livestock. Mountain heathlands support the ptarmigan (*Lagopus mutus*) which also consumes *Calluna vulgaris* but to a lesser extent than the red grouse. The black grouse (*Lyrurus tetrix*) inhabits heathland bordering woodland and is found in the Netherlands, Belgium, Scandinavia, Britain and upland areas of southern Europe.

Heathland invertebrates attract a wide variety of insectivores, although few inhabit heathlands exclusively. Examples include the common shrew (*Sorex araneus*) and the viviparous lizard (*Lacerta vivipera*, below 760 metres), and in the more grassy areas wood mice (*Apodemus sylvaticus*) and the common vole (*Microtus agrestis*). Frogs (*Rana temporia*, below *circa* 900 metres) and newts (*Tritrus vulgaris*) are found in wetter habitats.

There are few predators of the larger herbivores. Foxes (*Vulpes vulpes*) and stoats (*Mustela erminea*) feed on rabbits, hares and other small mammals. The buzzard (*Buteo buteo*), hen harrier (*Circus cyaneus*), peregrine falcon (*Falcon peregrinus*), golden eagle (*Aquila chrysaetos*), kestral (*Falco tinnunculus*), merlin (*Falco columbarius*), and the short-eared owl (*Asio flammeus*) also prey on small mammals, birds and invertegrates. Two crow species, the hooded crow (*Corevus cornix*) in the north of Scotland and northwards from the Netherlands and Belgium, and the carrion crow (*Corvus corone*), eat the eggs of ground nesting birds and small animals as carrion. Adders (*Vipera berus*) can be plentiful in any heathland habitat below 600 metres and are particularly associated with *Molinia* spp. grassland. The wild cat (*Felis sylvestris*) is present in the more northern areas of Scotland and hunts small mammals and rabbits. In continental areas, species including the wolf (*Canis lupus*) and the brown bear (*Ursus arctos*), may hunt within these systems.

3.1.4 Arctic grasslands and scrublands [see AMAP, 1998; CAFF, 2000; Stonehouse, 1989]

The extreme conditions (cold temperatures, extensive snow and ice cover, seasonal variations in solar radiation and short growing seasons) found in Arctic ecosystems dramatically influence their productivity, species diversity and behaviour of wildlife.

Different regions in the Arctic can be recognised:

- High Arctic – The most northern regions of the Arctic have a growing season of only 1–2.5 months with mean July temperatures of 4–8 °C. Around 360 vascular plants and 8 terrestrial mammals have been recorded. Vascular plant cover is typically 0–20 % with lichens and mosses increasing this to 50–80 % in some areas. The High Arctic is often referred to as the Polar desert as it can appear to be composed primarily of bare ground or rock, lacking the necessary moisture and warmth to sustain vegetation growth. In Eurasia the polar deserts are restricted to islands in the Arctic Ocean.

- Low Arctic – Has a growing season of between three to four months with mean July temperatures of 4–11 °C. Around 600 vascular plants have been recorded. Plant cover is typically 80–100 %. Broadly, the Low Arctic occurs where summer temperatures are above freezing with sufficient moisture to support vegetation growth. In Eurasia, low willow and birch shrub tundra forms a wide transition zone from the forest-tundra areas and often extends to the shore of the Arctic Ocean.
- Subarctic – Represents the transition between boreal forest and treeless tundra with a growing season of 3.5–12 months and a plant cover of 100 %. Often referred to as the forest-tundra zone.

The short length of the growing season is the most significant factor influencing the biological productivity in the Arctic. Growing seasons in the Low Arctic can range between three to four months, but can be as little as one month in the High Arctic. Growing seasons are therefore intense and there is little time over which herbivores can consume quality forage. As a consequnce, biota exhibits a wide range of physiological and behavioural adaptations.

Arctic soils (termed cryosols as they have frozen layers at their base) are poorly developed and can be frozen for most of the year; only for brief periods during the summer can shallow upper soil layers be unfrozen. Soils are low in nutrients due to cold temperatures reducing the rate of decomposition; the cold also reduces the rate of nutrient uptake through plant roots. Carbon accumulates in Arctic soils and nutrients such as N and P remain bound in decaying organic matter and are unavailable for plant growth. Soils in the Polar Desert can be pure sands and gravels with traces of organic matter, in the tundra, soils are usually characterised by a thick layer of undecomposed organic matter.

The growth and survival of many plants is often limited by the availability of moisture or the protection from extremes in climate offered by snow cover. Arctic ecosystems receive relatively little precipitation, mostly in the form of snow. The majority of the annual runoff occurs during snowmelt over periods that can be as short as only two to three weeks. Significant areas of wetlands can develop during the summer due to low rates of evaporation and frozen soil layers.

Survival of many plants and animals in the Arctic, particularly during the winter, is dependent upon their ability to exploit conditions during the summer – organisms store energy and nutrients when food is available which is then used when food is scarce. With differing environmental conditions both within and between years (e.g. differences in temperature, moisture and food availability), many species adjust their feeding habits, growth rates and reproduction. Indeed, some species will be opportunistic feeders with a number of possible positions within food chains. Alternatively, some Arctic organisms migrate to over-wintering, feeding or breeding areas. Due to short food chains, environmental changes can lead to the rapid growth or decline of Arctic organisms.

Whilst the total number of organisms in the whole of the Arctic may appear to be large, in small areas the number of species is limited, forming simple food chains (e.g. lichen →

reindeer → wolf). Overall, in northern latitudes the number of species declines as latitude increases, a general response to the increasing severity of environmental conditions.

Flora

The plants found in the Arctic can vary greatly over short distances in response to local environmental characteristics.

Microfauna, including bacteria, algae and fungi, are responsible for a significant part of primary production and decomposition in Arctic ecosystems, due to their diverse nature and ability to adapt to a wide variety of conditions. Lichens are pioneer organisms colonising bare ground and rock. Parasitic and saprophytic fungi are essential to decay processes in the Arctic.

In the northern Polar desert vegetation is present in the form of a thin, single layer consisting of algae, lichens and mosses. Patchy ground cover is composed largely of lichens (e.g. *Neuropogon sulpureus*, *Collema*, *Ochrolechia*, *Pertusaria* and *Toninia* species) and also mosses. Blue-green algae are present amongst the lichens. Moss species found in Polar Desert include those of the genera *Bryum*, *Pohlia*, *Myurella*, *Rhacomitrium*, *Andreaea* and *Onchophorus*; in wet areas where mires exist, *Orthothecium chryseum*, or species of *Campylium* and *Byrum* can be found. Around 60 species of angiosperms can be found growing in isolated clusters and include cushion plants (e.g. *Dryas integrifolia*, *Saxifraga oppositifolia*, *Silene acaulis* and *Papaver* spp.), small tufts (e.g. grasses *Phippsia algidae* and *Poa abbreviata*) and prostrate shrubs of *Salix arctica* and rosette species of *Saxifraga*, *Draba*, and *Minuartia*. In the transition between Polar Desert and tundra some vascular cryptogams (ferns) occur.

Plants typical of the Low Arctic include low shrubs (e.g. *Alnus*, *Salix* and *Betula*), dwarf shrubs of heath species (e.g. *Ledum*, *Vaccinium*, *Cassiope* and *Empetrum*), sedges (e.g. *Carex* and *Eriophorum*), rushes (*Juncus* and *Luzula*), grasses (e.g. *Poa* and *Arctagrostis*), cushion plants (e.g. *Dryas*), chickweeds (Stellaria spp.), wintergreen (*Pyrola grandiflora*), willow-herb (*Epilobium latifolium*), mountain vetch (*Astragalus alpinus*), Labrador-tea (*Ledum decumbens*), the heather *Cassiope tetragonal*, ferns (e.g. *Woodsia* spp.), lupins (*Lupinus arcticus*), buttercups (*Ranunculus lapponicus*), windflowers (*Anenome parviflora*), louseworts (*Pedicularis* spp.), thin grasses, lichens (e.g. *Cladonia* spp.) and mosses.

The Subarctic contains those plants found in the Low Arctic, some boreal species (e.g. *Deschampsia flexuosa*, *Epilobium angustifolium*, *Empetrum hermaphrodicum*, *Vaccinium myrtillus* and *V. uliginosum*) and stands of trees (e.g. *Pinus sylvestris*, *P. pumila*).

Fauna

The Arctic is home to a small number of well-adapted land animals. During the winter some Arctic mammals will hibernate (e.g. marmots) whilst the majority of insect species are dormant.

The diversity of soil invertebrates in the Arctic is low and earthworms tend to be absent. In warmer areas of the tundra, beetles, moths, butterflies, ichneumon flies, bumblebees, cranefiles, blowflies and other diptera can be found. Warble flies parasitise reindeer and mosquitoes are common. Larger invertebrate herbivores, such as browsing and grazing insects, are not found in Arctic ecosystems.

Amphibia and viviparous reptiles (e.g. *Vipera berus*) are rarely found in the Arctic.

The Arctic has over 150 species of breeding birds, but only eight are resident in tundra areas for the whole year. The majority of resident bird species in the Arctic breed in the far north during the summer, but move to areas of the Arctic such as warmer tundra in the south or the coast, with more hospitable conditions during the winter. Resident bird species in the Arctic include rock ptarmigan (*Lagopus mutus*), willow grouse (*Lagopus lagopus*), hazel grouse (*Tetrastes bonasia*), capercaillie (*Tetrao urogallus*), raven (*Corvus corax*), snowy owl (*Nyctea scandiaca*), merlin (*Falco columbarius*), gyrfalcon (*Falco rusticolus*), rough legged buzzard (*Buteo lagopus*), white-tailed sea eagle (*Haliaeetus* albicilla) and golden eagle (*Aquila chrysaetos*).

In the summer over 120 bird species migrate to Arctic areas, taking advantage of lower population densities and plentiful forage, for breeding. Migratory birds include small, insect eating birds (e.g. white wagtail, *Motacilla alba*; sedge warbler, *Acrocephalus schoenobaenus*; arctic warbler, *Phylloscopus borealis*; reed bunting, *Emberiza schoeniclus*; brambling, *Fringilla montifringilla*; pine grosbeak, *Pinicola enucleator*; Siberian jay, *Perisoreus infaustus*), waders or shorebirds (e.g. Baird's sandpiper, *Erolius bairdii*; sandpipers, *Calidris* spp.; plover, *Charadruis hiaticula*; the golden plover, *Pluvialis apricaria*), songbirds (e.g. Lapland bunting, *Calcarius lapponicus*; snow bunting, *Plectrophenax nivalis*; common redpoll, *Carduelis flammea*; Arctic redpoll, *Carduelis hornemanni*), wigeon (*Anas penelope*), loons (e.g. red-throated loon, *Gavia stellata*), ducks (e.g. common eider, *Somateria mollissima*; teal, *Anas crecca*; the long-tailed duck, *Clangula hyemalis*), geese and swans (e.g. snow goose, *Chen caerulescens*; lesser white-fronted goose, *Anser erythropus*; bean goose, *Anser fabilis*; the whooper swan, *Cygnus cygnus*) and birds of prey (e.g. long-tailed jaeger, *Stercorarius longicaudus*; peregrine falcon, *Falco peregrinus*).

Around 50 species of land mammal can be found in the Arctic. Herbivores include lemmings (e.g. *Lemming sibiricus*, *Dicrostonyx groenlandicus*, *Myopus schisticolor*), voles (e.g. *Microtus oeconomus*, *M. gregalis*, *Clethrionomys rufocanus*), marmots (e.g. *Marmota camtschatica*), red squirrels (*Sciurus* vulgaris), arctic ground squirrels (*Spremophilus undulates*), northern pika (*Ochotona hyperborean*), mountain hare (*Lepus timidus*), moose (*Alces alces*), reindeer (*Rangifer tarandus*) and muskox (*Ovibos moschatus*). Carnivorous animals in the Arctic include the stoat (*Mustela erminea*), least weasel (*Mustela* nivalis), European mink (*Mustela* lutreola), pine martin (*Martes martes*), otter (*Lutra canadensis*), Eurasian beaver (*Castor fiber*), red fox (*Vulpes vulpes*) and the arctic fox (*Alopex lagopus*), lynx (*Felis* l*ynx*), wolverine (*Gulo gulo*), grey wolf (*Canis lupus*), and brown bear (*Ursus arctos*).

3.2 Radionuclide behaviour in semi-natural pastures and heathlands

To our knowledge there are no bespoke models for these ecosystem types as have been developed for ecosystems considered in other chapters of this report. However, for some of the ecosystems (most especially those in northern Europe) there are data originating from studies following weapons testing and the Chernobyl accident. This data is largely restricted to radiocaesium and with an emphasis on those ecosystems and species utilised by man.

This discussion will pertain to:

1. the comparative vulnerability of ecosystems coming within the category 'semi-natural pastures and heath-/shrublands' ,
2. observations of transfer through food chains,
3. highlight some species/food chains known to have high radionuclide transfers within the ecosystems considered,
4. suggested reference organisms.

3.2.1 Behaviour of radionuclides in soils – comparative ecosystem vulnerability

The semi-natural ecosystems considered here cover a diverse range of soil types and environmental conditions. The mobility for transfer through food webs of the radionuclides considered in the project will be determined by their chemical behaviour and various soil properties [e.g. Desmet *et al.*, 1991]. The uptake of elements by plant roots occurs mainly through the soil solution. Bruemmer *et al.* [1986] suggested three important parameters to be considered to characterise the availability of elements for plant uptake: (1) the total amount of potentially available elements (the quantity factor), (2) the concentration of the elements in soil solution (the intensity factor), and (3) the rate of element transfer from soil solids to soil solution and to the roots (reaction kinetics).

The radionuclides can be grouped on the basis of: (1) periodic classification of the elements, or (2) availability of literature data on soil-to-plant transfer for certain groups – such as the radionuclides in the U-Th decay series and the transuranics (Table 3-2). The important interactions of any chemical species in solution, which can influence its mobility in soils and eventual root uptake, include: (1) charge interactions, (2) complexation and precipitation reactions with other chemical species, such as organic and inorganic ligands, (3) oxidation-reduction (redox) transformations, and (4) specific interactions with soil components including soil biota. Several references on soil processes [Wolt, 1994; Greenland & Hayes, 1981], chemical speciation and geochemical modelling in the environment [e.g. Jenne, 1979; Broekaert *et al.*, 1990], plant nutrition [e.g. Kabata-Pendias & Pendias, 1984] and phytoremediation [Saxena *et al.*, 1999; Wenzel *et al.*, 1999] provide relevant information on most of the radionuclides considered here. A summary of the important chemical properties for these radionuclides, including key species in soil solution, is given in Table 3-2.

The review will focus on information gained from investigations on the soil-to-plant transfer of the radionuclides (elements), particularly on measurements of the concentration ratio, CR (the concentration of the element in the plant divided by the concentration of that element in the soil). Various workers use different ways of expressing the radionuclide activity concentrations in soil or plant (e.g., Bq kg^{-1} wet or dry material; the IAEA [1994] recommends expressing both measurements on a dry weight basis to reduce variability in the calculated CR). The processes affecting plant uptake (hence CR) for any radionuclide are controlled by two sets of factors: those associated with the physiological requirements of the plant, and the physico-chemical factors influencing the distribution of radionuclides between the soil solid phase and soil solution. This review will focus mainly only the soil factors.

A large database of soil-to-plant CR values for various radionuclides and soil types has been compiled by the International Union of Radioecologists [IUR, 1989]. The database contains

CR values for the majority of the radionuclides considered in this project. However, the data are largely for crops in agricultural soils and may be inappropriate to apply to native plants in natural or semi-natural environments [Sheppard & Evenden, 1997]. Nevertheless, an evaluation of the data has revealed important soil and plant factors that influence CR values [e.g. Bell *et al.*, 1988; van Bergeijk *et al.*, 1992; Noordijk *et al.*, 1992; Shaw & Bell, 1994]. Statistical analyses of the compiled CR data have also provided important information on the use of CR values for stochastic modelling [e.g. Sheppard & Sheppard, 1989; Konshin, 1992; Sheppard & Evenden, 1997]. However, Sheppard & Sheppard [1989] suggest that for more accurate assessments, it would be preferable to mechanistically relate plant uptake processes with the underlying chemical and physical processes, and describe these relationships as explicit equations rather than as correlation coefficients. Mechanistic information obtained from different investigations of the soil to plant transfer of the elements considered in this project (either as radionuclides or as naturally occurring forms) will be summarised here, to provide an insight into their expected behaviour in semi-natural environments.

Table 3-2 Grouping of selected radionuclides on basis of periodic classification and prevalent species in soil solution.

Element	Valence state	Prevalent solution species
H	+I	HTO, H^+, organically bound
C	+IV, -IV	H_2CO_3, HCO_3^-, CO_3^{-2}, DOC
Alkali		
K	+I	K^+
Cs	+I	Cs^+
Alkaline earths		
Sr	+II	Sr^{+2}
Ra	+II	Ra^{+2}
Halides		
Cl	-I	Cl^-
I	-I	I^-, $IO3^-$, (I_3^-, IO^-, IO_6^{-3})
Transition metals		
Ni	+II	Ni^{+2}, $NiSO_4$, organic
Ru	II, III, IV, VI, VII, VIII	
U-Th series		
U	+VI	UO_2^{+2}
Th	+IV	Th^{+4}, $Th(OH)_2^{+2}$
Po	+IV	Po^{+4}
Pb	II	Pb^{+2}, organic, $PbSO_4$
Ra	II	Ra^{+2}
Transuranics		
Am	+III	Am^{+3}
Cm	+III	Cm^{+3}
Np	+III to +VI	Np^{+3}, Np^{+4}, NpO_2^+, NpO_2^{+2}
Pu	+III to +VI	Pu^{+3}, Pu^{+4}, PuO_2^+, PuO_2^{+2}

Systematic variations in transfer factors for the specific radionuclides/elements in relation to soil properties

Frissel *et al.* [1990] used the IUR database to extract information on relevant parameters that influence the uptake of Cs, Sr, Pu, Am, Np and Co, and extrapolated these to 'extreme' soil conditions such as would be found in natural or semi-natural ecosystems. As mentioned previously, the CR database is based mainly on agricultural conditions; Frissel *et al.* [1990] considered the following as the most important differences between agricultural and natural systems: (1) pH considerably lower in natural systems compared to agroecosystems; (2) organic matter (OM) content: natural ecosystems often have either extremely high (peat soils) or extremely low (sand) OM contents; (3) soil moisture content in natural systems also tends to extreme values (waterlogged bogs versus deserts); (4) litter layer agroecosystems are ploughed and the litter layer is almost nonexistent, whereas litter layer is important in natural ecosystems; and (5) soil nutrient status fertilisation (agroecosystems) versus no fertilisation in natural ecosystems. Of these five parameters, insufficient data did not allow an evaluation of the effects of OM and soil moisture contents on Am, Np and Co uptake. Increasing soil pH increased the uptake of Am only; the reverse was found for the other radionuclides. Organic matter content did not have an impact on Pu uptake; Sr uptake decreased with increasing OM content, whereas Cs uptake increased with OM > 15 %, but no effect on Cs uptake where OM < 15 %. Increasing soil moisture content increased Sr uptake, but its effect on Cs uptake varied with the plant species.

Van Bergeijk *et al*. [1992] applied a nonparametric method to assess the effect of soil parameters on the transfer of Cs and Sr from soil to edible plant parts, using CR values from the IUR database. Increasing soil organic matter content was found to increase the transfer of Cs, whereas the reverse trend was observed for Sr. Soil pH did not affect Cs transfer, whilst Sr transfer decreased with increasing pH. Soil type (clay, loam or sandy soils) also appears to be an important factor for both radionuclides, but this was difficult to generalise because of differences in the nutritional characteristics of these soil types. Other studies suggest that the Ca and K contents of the soil would influence the transfer of Sr and Cs, respectively; but this was difficult to evaluate due to limited information of these elements in the IUR database. In a separate evaluation of the database [Noordijk *et al*., 1992], the effects of ageing on Cs transfer was observed, possibly due to decline in bioavailability as Cs is progressively fixed by clay minerals. Weather conditions affect Cs and Sr transfer, as reflected by annual and seasonal fluctuations in CR values.

A compilation of the available soil-to-plant CR values for U, Th and Pb from field and pot experiments in various settings and soil types was analysed to select the most appropriate CR values for environmental assessments in Canada [Sheppard & Evenden, 1988]. A substantial variability of the CR data was noted; CR values ranged between 1 000- to 30 000-fold, and significantly differed among soil and plant types. An overview of the mobility of U, Th and Pb in soils was provided, as this is a major determinant of plant uptake. The cationic species UO_2^{+2}, Th^{+4}, $Th(OH)_2^{+2}$ and Pb^{+2} are the dominant forms of these elements in soils, and are strongly sorbed on soil solids, including the organic matter component. However, organic complexes and colloids can increase their mobility in mineral soils. Within the plant itself, these elements adsorb strongly on cell wall materials, and are found to be highest on root surfaces. In general, root crops have higher CR values than cereal grain crops. Another review of the soil-to-plant relationships for naturally occurring radionuclides in the U and Th decay series is that of Mortvedt [1994]. This includes information not only on U and Th but also Pb, Po and Ra. In general, plant uptake of these radionuclides was affected by soil pH, which may be related to indirect effects of competition from Ca and Mg (particularly for Ra, which belongs to the alkaline earth series). There is evidence of increased uptake of the radionuclides on the clay fraction compared to the sand fraction of soils contaminated by U mine tailings. However, uptake by different grasses growing on such soils was not different from those on uncontaminated soils, except for U and Ra. Soil organic matter appears to form strong complexes with Th and increases its mobility in soil, similar to observations noted by Sheppard [1980]. There is evidence that Se, available K, nitrate, carbonate and phosphorus can increase the plant uptake of U [e.g. van Netten & Morley, 1983].

The transuranic radionuclides Am, Cm, Np and Pu have relatively low mobility due to their strong tendency to sorb onto soil particles. Am and Cm generally exist in the +III valence state, whereas both Np and Pu can be found in any of four oxidation states (+III, +IV, +V, +VI) depending on the redox conditions of the soil system. The corresponding ionic forms for these various valence states are shown in Table 3-2. Due to their low mobility, plant uptake of the transuranic elements is generally low, and follows the order: Np(V) > Am(III) ~ Cm(III) > Pu (IV) ~ Np (IV) [Bulman, 1983]. The observed CR values are low, in the range of 10^{-6} to 10^{-8} for Pu and 10^{-3} to 10^{-7} for Am [e.g. Pimpl & Schuttelkopf, 1981; Livens *et al.*, 1994]. Where reported CR values are higher (particularly for data from field measurements), contamination of the vegetation fraction with resuspended soil particles appears to be an important factor [e.g. McLendon *et al.*, 1976; Cawse, 1983]. Changes in soil pH and Eh (redox conditions) and the presence of complexing agents such as dissolved organic compounds and extracellular metabolites of microflora are known to increase the mobility of Pu in soils [e.g. Negri & Hinchman, 2000, and references therein]. Several investigator have shown that naturally occurring Th (IV) and the rare earth elements La, Ce and Nd (valence state III) can be considered as analogues of Pu (IV) and Am (III) and Cm (III) respectively [e.g. Linsalata *et al.*, 1989].

Ruthenium has a complex chemistry and can exist in several oxidation states in the environment (Table 3-2). It is known to change valence and form a variety of complexes (including nitrosyls, amines, oxides and halogenated complexes) relatively easily. Prosser [1994] reviewed the literature data on the availability of ^{106}Ru in soils for uptake by plants. Although the reviewed data indicated that plant uptake of ^{106}Ru was found to be dependent on several interrelated soil properties such as clay and mineral contents, soil texture, cation exchange capacity and pH and organic matter content, the trends in Ru plant uptake with any of these properties was not identified. The limited data suggest that enhanced uptake occurs on sandy soils compared to clay or loam soils. The review recommended a generic CR of 1 x 10^{-2} (expressed as fresh weight plant to dry weight soil) for use in radiological assessments; the uncertainties associated with reported soil to plant CR values are potentially large.

Nickel belongs to the Group VIII metals subgroup of the transition elements, and has similar geochemical properties as Fe and Co. Ni is a component of the enzyme urease, and has known beneficial effects on plant growth, but it is not clear if Ni is an essential plant nutrient [Kabata-Pendias & Pendias, 1984]. It occurs as the divalent cation Ni^{+2}, and also in the form of organic compounds and complexes. Available results suggest that both the speciation of Ni in soil solution and its rate of replenishment from the soil phase are important factors affecting Ni availability for root uptake.

Investigations on Tc have shown that the predominant form in aerobic environments is the pertechnetate ion, TcO_4^-. This species is highly mobile in soils and is taken up readily by plants [e.g. Hoffmann *et al.*, 1982; Harms *et al.*, 1999]. An excess of NO_3^- in the soil significantly decreased TcO_4^- transfer to the root, presumably due to the competition between these two ions for root uptake [Echevarria *et al.*, 1998]. The redox or aeration status of the soil, and organic matter content, are key factors affecting Tc mobility. Substantial sorption of reduced Tc (presumably as TcO_2) occurs in anaerobic environments, especially in the presence of organic matter [Sheppard *et al.*, 1990]. The observed decrease in plant uptake of ^{95m}Tc from soil with time after initial application was attributed to the reduction of soluble Tc(VII) to less soluble forms by the organic matter fraction of soils [Hoffmann *et al.*, 1982].

Sheppard *et al.* [1993] reviewed the soil-to-plant transfer pathways for the halides, including Cl and I, and performed carefully designed lysimeter experiments to investigate plant uptake of these nuclides. Iodine could potentially be present in volatile form, and soil-to-plant transfer would take place via two routes: directly by root uptake and indirectly through the leaves (soil-atmosphere-plant transfer). This is similar to the situation for ^{14}C. Due to the role of atmospheric transport, the design of soil-to-plant transfer experiments with I (and to some extent Cl) must take this pathway into account. In general, the lysimeter experiments showed that CR values for Cl were higher than those for I, in agreement with trends observed by other workers. Across the range of concentrations for Cl and I used in the experiments, the plant concentrations of I were found to be linearly related to the soil concentration, but this was not observed for Cl. In terms of radionuclide mobility in the lysimeter soils (via measurements of the distribution coefficient, K_d), the sorption of I was found to be higher than that for Cl; the role of microorganisms and/or interactions with organic matter in increasing I sorption with time has been postulated. The CR values derived for I from these experiments were higher compared to data reported by other workers in the IUR database. Sheppard *et al.* [1993] suggest that CR values for plant uptake of I may be underestimated if the atmospheric pathway is not taken into account.

Plants fix C as CO_2 from the atmosphere during photosynthesis and may also take up inorganic and organic forms of soil C directly through their roots. Most radiological models consider mainly ^{14}C from atmospheric sources, as little is known about direct C uptake through the roots. Sheppard *et al.* [1991] investigated the plant uptake of ^{14}C in lysimeters, using an inorganic and organic C source in two soils with different retention properties for C, to derive CR values for ^{14}C. The interpretation of results from these experiments was complicated by two factors: (1) the interaction of the C reservoir in the soil with the ^{14}C tracers, and (2) the role of the soil-atmosphere-plant pathway (similar to that described previously for I). The measured CR values were about 25-fold lower than values commonly used in assessment models. The major factor was the transfer of C through the atmospheric pathway, which was particularly important for the inorganic ^{14}C-CO_3. In addition, the retention of ^{14}C-CO_3 by the carbonate fraction of one of the soils studied suggested that this has important implications regarding the availability of ^{14}C for plant uptake in the long-term.

Spatial variability in radiocaesium uptake within semi-natural ecosystems

To determine the comparative mobility of radiocaesium within the ecosystems being considered, the *SAVE-IT* spatial modelling package as presented by Gillett *et al*. [2001] was implemented (note saltmarshes have not been included within this assessment). This model utilises a recently developed semi-mechanistic temporal model [Absalom *et al*., 1999; 2001] to predict radiocaesium activity concentrations in vegetation using soil characteristics (exchangeable K, pH, percent clay and percent organic matter content) known to determine radiocaesium mobility and which are available from spatial soil databases [European Soil Bureau, 1999]. To obtain an indication of the comparative radiocaesium mobility within the ecosystems considered, the model has been run for a single deposition event resulting in 1 Bq m^{-2} deposition to ground and radiocaesium activity concentrations predicted in 'pasture grass'. Obviously this vegetation type is not representative of the ecosystems being considered. However, it does enable a comparative assessment from which is it evident that the mobility of radiocaesium for transfer through food webs is likely to be considerably higher in northern than southern Europe.

3.2.2 Radionuclide behaviour in selected ecosystems

Saltmarshes

As a consequence of marine discharges from the Sellafield reprocessing plant there are considerably elevated concentrations of radionuclides (e.g. ^{137}Cs, ^{106}Ru, ^{237}Np, $^{239+240}$Pu, ^{241}Pu and ^{241}Am) on many saltmarshes of the Irish Sea [Howard *et al*. 1996; Sanchez *et al*. 1998]. Consequently there has been a number of radioecological studies conducted within these ecosystems, with an emphasis on the saltmarshes of the Ravenglass Estuary (West Cumbria, UK). The mechanism of transfer of many radionuclides from the marine discharges to saltmarsh vegetation is via the deposition of contaminated silt during tidal inundation. Vegetation type can influence the amount of silt trapped during tidal inundation and consequently the amount of radionuclides accumulated; higher deposition being associated with areas of *Halimione portulacoides* on an ungrazed saltmarsh [Horrill, 1983].

The activity concentrations determined within samples of vegetation are found to be associated with the amount of adherent silt; Howard [1985] found that the activity concentrations of 11 artificial radionuclides within vegetation samples were correlated with concentrations of titanium (a commonly used soil marker). Silt can therefore contribute to the majority of the radionuclide intake of grazing animals for some elements (e.g. Cs and Pu). Studies with sheep have shown that the absorption of radiocaesium from ingested silt is *circa* 10–20 % of that of radiocaesium incorporated within vegetation [see review by Beresford *et al*., 2000]. It is likely that this low bioavailability will result in comparatively low transfer of radiocaesium to other animals grazing saltmarshes contaminated by marine discharges. The percentage of Pu absorbed from saltmarsh vegetation fed to sheep and rabbits has been estimated to be 0.01 % [Beresford *et al*., 2000] and 0.004 % [Stather *et al*., 1978] of that ingested respectively.

Transfer coefficients for radiocaesium to the muscle of greylag geese (*Anser anser*) and wigeon (*Anas penelope*) grazing saltmarshes of the Ravenglass estuary of 0.59 d kg^{-1} and 0.57 d kg^{-1} respectively were estimated by Lowe & Horrill [1986]. In a later paper Lowe [1991] determined radionuclide levels in the tissues of a range of bird species from Ravenglass and also some invertebrate species know to be important sources of food for (some of the) birds. Although transfer parameters were not estimated a comparison of the results for the different species is of potential interest for the current assessment and is presented in Table 3-3.

Recently, unexpectedly high ^{3}H activity concentrations have been observed in wildfowl close to the Amersham Nycomed facility in south Wales. The mechanisms for these observations have not yet been identified.

The transfer of radionuclides through the *sediment/vegetation–invertebrate–small mammal* food chains of the Ravenglass saltmarshes has been studied by Copplestone [1996]. Activity concentrations of the radionuclides studied (^{137}Cs, ^{238}Pu, $^{239/240}Pu$ and ^{241}Am) were lowest in the predatory Carabidae generally being highest (by one to two orders of magnitude) in the detritivorous Isopoda. A summary of the estimated concentration factors for sampled invertebrate species is presented in Table 3-4. The author notes that, whilst the ^{137}Cs concentration factors are in agreement with previously derived values for other ecosystems, the values derived for the actinides are higher than expected. It is acknowledged that external contamination may contribute although sample preparation should have minimised this. Transfer of radiocaesium to small mammal species was noted as being lower than that measured in a nearby woodland site; this is probably a reflection of the low bioavailability of the radionuclides studied in the saltmarsh ecosystem. Concentration factors derived for *Sorex araneus* (common shrew) and *Microtus agrestis* (field vole) are summarised in Table 3-4.

Upland semi-natural ecosystems (pastures and heathlands)

Many northern European upland semi-natural ecosystems received comparably high levels of radiocaesium deposition as a consequence of nuclear weapons testing and following the Chernobyl accident. As a consequence of the relatively high mobility of radiocaesium within these ecosystems there have been a number of studies conducted within them [e.g. Desmet *et al*., 1990], although the emphasis has been on transfer to man. With respect to radiocaesium an important radioecological characteristic of these ecosystems is the long ecological half-life observed for many compartments; values approaching that of the radioactive decay of ^{137}Cs now being determined [e.g. Smith *et al*., 2000].

There is considerable variability in the uptake of radiocaesium between different higher plant species present in upland semi-natural ecosystems, as illustrated by data presented in Table 3-5. A number of studies have demonstrated a high transfer of radiocaesium to plants of the Ericaceae family [e.g. Bunzl & Krake 1984a; Colgan *et al*., 1990; Horrill *et al*., 1990] compared with other species of higher plants present within upland environments; *Calluna vulgaris* (ling heather) being reported to have the highest radiocaesium uptake of the Ericacae [Bunzl & Krake 1984a; Horrill *et al*., 1990].

Table 3-3 A comparison of radionuclide activity concentrations determined in invertebrate prey species and the tissues of different bird species at Ravenglass (1980–1984) adapted from Lowe [1991].

Species	Tissue	Bq kg^{-1} fresh weight*		
		^{137}Cs	$^{239/240}Pu$	^{238}Pu
Invertebrates				
Arenicola marina		321	72	16
Nereis diversicolor		358	49	11
Mytilus edulis		27	16	4
Macoma balthica		35	36	8
Corophium volutator		133	78	19
Birds				
Greylag goose (*A. anser*)	Pectoral muscle	58	0.1	0.03
	Liver	28	13	3
Shelduck (*Tadorna tadorna*)	Pectoral muscle	295±56	0.67±0.33	0.15±0.06
	Liver	215±62	12±2	2.7
Wigeon (*A. penelope*)	Pectoral muscle	158±94	0.56±0.37	0.16±0.12
	Liver	100±36	8±6	2±1
Mallard (*Anas brachyrhynchus*)	Pectoral muscle	168±5	n/a	n/a
	Liver	126±24	3	1
Black-headed gull (*Larius ridibundus*)	Pectoral muscle	14±14	0.32±0.77	< 0.03
	Liver	11±9	0.54±0.67	0.17±0.19
	Whole chicks	25	0.47	0.09
	Egg (contents)	11	0.05±0.04	0.02±0.002
Curlew (*Numenius arquata*)	Pectoral muscle	140±138	0.34±0.34	0.09±0.09
	Liver	103±120	2.4±2.5	0.14±0.12
Bar-tailed godwit (*Limosa lapponica*)	Pectoral muscle	478	0.03	< 0.02
	Liver	510	0.91	0.17
Oystercatcher (*Haematopus ostralegus*)	Pectoral muscle	613±34	0.53±0.30	0.2
	Liver	463±67	4.1±2.2	1.8
Carrion crow (*Corvus corone*)	Pectoral muscle	162±158	0.17±0.14	< 0.04
	Liver	131±80	0.7±0.6	< 0.08
	Egg (contents)	n/a	0.06	0.03

* Errors quoted where available are standard deviation (n = 2 to 9).
n/a – not available

Table 3-6 presents data for the transfer ratios of radiocaesium from diet to muscle for a range of vertebrate species characteristic of the ecosystems being considered here. For comparative purposes T_{ag} values and transfer coefficients (F_f) are also shown where available. There is a marked concentration of radiocaesium from prey (i.e. rabbit) to carnivorous species (i.e. fox). Although Lowe & Horrill [1991] report significant differences between concentration ratios on the basis of species, sex, breeding condition and age, values for all herbivores (including, bird, ruminant and mono-gastric species) are within the range 1.4–3.8. There is considerably greater variation in the available F_f values between species differences.

Table 3-4 A summary of concentration factors (dry weight activity concentration in tissue:dry weight activity concentration in diet) derived for invertebrate and small mammal foodchains at the Ravenglass estuary. Adapted from Copplestone [1996]*.

Species or Group	Concentration factor**			
	^{137}Cs	^{238}Pu	$^{239/240}Pu$	^{241}Am
INVERTEBRATES				
Detritivores				
Orchestia spp.	0.08–0.34	0.09–0.54	0.08–0.23	0.04–0.13
Isopoda	0.08–0.54	0.12–0.47	0.08–0.13	0.09–0.28
Herbivores				
Coleoptera	0.33–1.31	0.05–1.27	0.53–9.08	0.30–2.33
Predators				
Araneida	0.11–5.53	0.12–11.22	0.08–8.15	0.06–4.65
Carabidae	0.13–1.34	0.05–1.36	0.07–1.28	0.12–8.81
Coleopteran larvae	0.88–14.08	0.49–1.47	0.27–2.59	0.83–12.66
SMALL MAMMALS				
Herbivore				
M. agrestis	0.11–2.44	0.02–2.13	0.01–1.13	0.01–4.60
Insectivore				
S. araneus	0.10–1.45	0.01–0.53	0.01–0.24	0.01–0.19

* Refer to Copplestone [1996] for details of assumptions made in estimation of transfer.
** Range summarised over different sampling sites and times for both invertebrates and small mammals, and also potential dietary components in the case of small mammals.

Because of the high transfer of radiocaesium to *C. vulgaris*, heather ecosystems have been the subject of a number of studies. The red grouse (*Lagopus lagopus scoticus*) feeds more or less exclusively on (comparatively highly contaminated) *C. vulgaris* at certain times of year. Consequently high T_{ag} values have been derived for red grouse compared with other species of grouse (e.g. compare values for willow and red grouse in Table 3-6). The mountain (or blue) hare (*L. timidus*) also often feeds on *C. vulgaris* and high radiocaesium activity concentrations have been reported (up to 20 kBq kg^{-1} [Johanson, 1994]). Brown [1990] reports the transfer of radionuclides to the leaf eating heather beetle (*Lochmaea suturalis*); concentrations ratios of 0.3 (Cs), 0.5 (Ru) and 1.5 (Sr) were estimated. Concentration ratios for the phloem feeding leaf hopper (*Ulopa reticulate*) were also measured values of 0.48 and 0.38 being derived for Cs and Sr respectively [NERC, 1993]. The comparative transfers of Sr and Cs were in contrast to previous studies that had shown the transfer of Cs to insects to be 1–2 orders of magnitude higher than that of Sr [Crossley, 1963]. No concentration of radiocaesium through the *crane fly (Tilapia spp.)–beetle (Carabus problematicus)–pigmy shrew (Sorex minutes)* food chain of a heather ecosystem was observed [NERC, 1993].

Table 3-5 Concentration ratios (dry weight activity concentration in plant to dry weight activity concentration in 0–10 cm soil layer) estimated for ^{137}Cs at different European upland sites in summer 1989. Adapted from Livens *et al.* [1991].

Location	Ecosystem	Soil type	Species	^{137}Cs concentration ratio
Stolvizza, Italy	Alpine meadow	Calcareous brown earth	*Arrhenatherum elatius*	0.365
			Galium sp.	0.214
			Ranunculus sp.	0.250
			Stachys alpina	0.046
			Trifolium dubium	0.299
Norway	Open hillside	Ranker	*Juniperus* sp. (shoots)	0.359
			Vaccinium sp.	0.254
			Geranium sp.	0.061
Scotland, Glen Shirra	Calluna moor	Podzol	*Calluna vulgaris*	1.75
Scotland, Creag Meagaidh	Open hillside	Ranker	*Deschampsia caespitosa*	0.434
			Empetrum nigrum	0.868
			Nardus stricta	0.376
			Trichophorum caespitosum	0.595
			Vaccinium myrtillus	1.62
			Calluna vulgaris	3.27
			Juncus squarrosus	0.676
			Molinia caerulea	0.677

The transfer of radiocaesium to invertebrates and small mammals inhabiting a semi-natural (unmanaged lowland) grassland was studied by Rudge *et al*. [1993a; b] during 1986–87. Radiocaesium concentration ratios for different trophic levels of invertebrate species were in the order: detritivores > herbivore ≈ predators; estimated concentration ratios for the three groups were: detritivores (0.53–1.77[5]), herbivores (0.24–0.46), predators (0.17–0.58) [Rudge *et al*., 1993a]. The study period included that coinciding with deposition from the Chernobyl accident. Radiocaesium levels were initially higher in insectivorous shrews (*Sorex* spp.) and subsequently declined more rapidly compared with herbivorous/omnivorous species sampled [Rudge *et al*., 1993b]. Estimated dry weight concentration ratios were approximately: 2–6 (Sorex spp.); 0.6 (*M. agrestis,* field vole); 2 (*Apodemus sylvaticus*, wood mouse); 2 (*Clethrionomys glareolus*, bank vole). Increases in the radiocaesium activity concentrations of all four small mammal species were observed in autumn (associated with composition and radiocaesium content changes in their diets).

[5] Dry weight concentration ratios, range for different sampling periods.

Table 3-6 Transfer parameters for radiocaesium to different wild animals. Concentration ratios (CR) for herbivores are the ratio of the fresh weight activity concentrations in muscle and the gut [Lowe & Horrill, 1991]. For fox the concentration ratio is derived from the fresh weight activity concentration in rabbit meat. T_{ag} and transfer coefficient (F_f) values are shown for comparison where available; details of T_{ag} and F_f values can be found in the footnotes.

Species	Sex	CR	Tag (m^2 kg^{-1})	F_f [e] (d kg^{-1})
Red deer (*Cervus elaphus*)	Milk hind	2.29		0.49[f]
Red deer (*Cervus elaphus*)	Calf	2.59		
Red deer (*Cervus elaphus*)	Yeld hind	1.34		
Red deer (*Cervus elaphus*)	Stag	3.80		
Red deer (*Cervus elaphus*)			0.25[a]	
Red grouse (*Lagopus lagopus scoticus*)	Male	1.70		
Red grouse (*Lagopus lagopus scoticus*)	Female	1.71		
Red grouse (*Lagopus lagopus scoticus*)			0.27–0.76[b]	10[g]
Willow grouse (*Lagopus lagopus*)			0.006[c]	
Arctic hare (*Lepus arcticus*)			0.006–0.079[d]	
Brown hare (*Lepus capensis*)	Female	3.31	0.0005–0.0013[d]	
Blue/Mountain hare (*Lepus timidus*)	Male	2.07		
Blue/Mountain hare (*Lepus timidus*)	Female	2.50		
Rabbit (*Oryctolagus cuniculus*)	Male	1.86		
Rabbit (*Oryctolagus cuniculus*)	Female	1.49		
Rabbit (*Oryctolagus cuniculus*)				0.29[h]
Black grouse (*Tetrao tetrix*)	Male	1.39		
Fox (*Vulpes vulpes*)	Male	11.73		
Fox (*Vulpes vulpes*)	Female	7.19		

[a] Estimated from Horrill *et al*. [1995] for UK in 1986/7.
[b] From Howard & Howard [1997] values for UK 1990; lower value advised for peaty podzol and upper value peat soils.
[c] Data for Norway (1987/90) quoted by Johanson [1994].
[d] From Rantavvaara [1990] data for Finland 1988 (data for other years available in original reference).
[e] Where transfer coefficient (F_f) is defined as the ratio of the equilibrium fresh weight radiocaesium activity concentration in muscle to daily dietary intake of radiocaesium.
[f] Mayes *et al*. [1994].
[g] Moss & Horrill [1996].
[h] Jandl & Sladovnik [1993].

Bunzl & Krake [1984a] report ^{210}Po, ^{210}Pb and ^{137}Cs activity concentrations in plants and soil of a German heath. Concentration ratios have been estimated from this data and are presented for different heath species in Table 3-7. However, the comparatively high values for ^{210}Po and ^{210}Pb are probably the result of the interception of fallout on vegetation surfaces.

Table A3-7 Concentration ratios (dry matter activity concentration in plant:dry weight activity concentration in 0–15 cm soil layer) for ^{137}Cs, ^{210}Po and ^{210}Po estimated for plant species of a German heath from the data of Bunzl & Krake [1984].

Radionuclide	Grass leaves	Heather leaves	Juniper shoots	Broom shoots	Bilberry leaves
^{137}Cs	4.67	28.67	1.20	1.60	15.33
^{210}Pb	3.07	5.50	2.00	2.00	0.93
^{210}Po	1.81	4.00	1.88	1.31	0.51

The transfer of ^{226}Ra to grouse has been determined with reported concentration ratios in excess of unity for ^{226}Ra to bone compared with its major food source (aspen in this study); concentration ratios for other species such as beaver, muskrat and snowshoe hare of up to 9 have been estimated (in Canada) [Clulow *et al*. 1991; 1992; 1996]. Concentration ratios for ^{226}Ra to soft tissues of these animals were generally below 1. In an extensive survey in an area of high natural radioactivity Maslov *et al*. [1967] divided animals into three groups on the basis of the extent of their "contact" with the contamination. Thus animals with close contact – especially moles (*Talpa europaea*) and other burrowing animals – had higher whole body ^{226}Ra activity concentrations than those with moderate contact (with prolonged habitation, but not permanent in contaminated area) that were in turn more contaminated than those within the weak contact group (who do not have much contact with area). The authors proposed that various indicators of ecological health such as fatness and number of parasites were correlated with the groupings, but cause and effect is difficult to test.

Comparatively high Pu activity concentrations have been noted in the livers of chamois (*Rupicapra rupicapra*) sampled from the German Alps [Bunzl & Krake, 1984b].

Arctic ecosystems

The Arctic food chain *lichen–reindeer* has received considerable attention following deposition from both weapons fallout and the Chernobyl accident [see AMAP, 1998]. The large surface area of lichens means that they intercept atmospheric radionuclides more efficiently than other vegetation; accumulation is by the slow turnover of lichen. Thus airborne radionuclides, particularly ^{137}Cs, ^{210}Pb and ^{210}Po are efficiently trapped and retained on lichens, which form the main food source for reindeer in winter. Radiocaesium activity concentrations in the range of 10^4 Bq kg^{-1} have been commonly recorded for both lichen species [AMAP database] and reindeer [Gaare & Staaland, 1994]. Transfer coefficients have been derived for reindeer feeding on lichen of 0.65 d kg^{-1} [Jones, 1989]; the comparative value for summer feeding on pasture was 0.3 d kg^{-1}. Aggregated transfer coefficients for radiocaesium and ^{90}Sr have recently been reviewed by Howard and Wright [*in preparation*]. For radiocaesium a T_{ag} value of 1.4 m^2 kg^{-1} was derived from > 900 measurements of reindeer in Finnmark. A T_{ag} value for ^{90}Sr of 0.0014 m^2 kg^{-1} based upon available data for the transfer of ^{90}Sr to lichen and from lichen to reindeer. Hanson [1967] reports ^{90}Sr activity concentrations in the bone of reindeer to be 6–15 times higher than those in lichen.

In addition to the well-known accumulation of ^{137}Cs, high levels of ^{210}Po and ^{210}Pb have been reported in reindeer. Both ^{210}Pb and ^{210}Po, levels were highest in bone; of the soft tissues concentrations were highest in liver and kidney. Concentration ratios have been reported for reindeer muscle compared to lichen of 0.01–0.16 for U, 0.06–0.25 for ^{226}Ra, 0.01–0.02 for ^{210}Pb, 0.06–0.26 for ^{210}Po; these values can be compared with that of 2.60–3.70 for ^{137}Cs [Thomas & Gates, 1999]. Concentration ratios greater than unity were also determined for ^{226}Ra in bone, in part due to its long biological half-life in this tissue.

The transfer of radiocaesium to Eurasian woodcock (*Scolopax rusticola*) from their main food source earthworms was determined in a Norwegian sub-alpine ecosystem (1986–90) [Kålås *et al.*, 1994]. The ratio of radiocaesium concentrations in woodcock to those in earthworms decreased from 6.1 in 1986 to < 1 in 1988–90. Radiocaesium activity concentrations in woodcock were 5–10 times higher than in willow grouse and rock ptarmigan (*Lagopus mutus*) collected from the same area.

Radiocaesium activity concentrations in the flesh of wolves was observed to be *circa* 2-fold higher than that in reindeer flesh during studies of weapons fallout radionuclides in Alaskan Arctic ecosytems [Hanson, 1967]. In contrast, ^{90}Sr concentrations in wolf flesh were approximately 0.3 times those in reindeer meat [Hanson, 1967] although ^{90}Sr activity concentration in wolf bone were 260 times higher than those in reindeer meat (^{90}Sr activity concentration in wolf bone were *circa* 60 % of those in reindeer bone) [Hanson *et al.*, 1967]. As for ^{90}Sr, the (limited) data presented by Hanson *et al.* [1967] for ^{110m}Ag (which like the actinides accumulates in the liver) and ^{228}Th (which accumulates in bone) suggests that radionuclides that are localised in a given tissue will not be accumulated within the muscle of carnivores.

3.3 Discussion

There is a limited amount of data available to demonstrate wild plant and animal species that may accumulate high levels of some radionuclides, especially radiocaesium. A number of animals inhabiting the semi-natural ecosystems considered here have been shown to have especially radionuclide (largely radiocaesium) levels as a consequence of their dietary habits – e.g., radiocaesium concentrations in the range of 10^3–10^4 Bq kg^{-1} have been relatively commonly recorded in reindeer, red grouse and mountain hare. However these species are not ubiquitous being restricted to rather specific habitats.

Some authors have reported transfer parameters of radionuclides to biota. Whilst the above discussion is not exhaustive with respect to presenting the results of these studies, the available transfer parameters values are limited, and also expressed in a number of disparate ways (indeed it is sometimes difficult to be ascertain if dry or fresh weight transfer parameters are being reported). Although not discussed here, there are some biological and ecological half-life values available for wild species.

There is clear evidence of a concentration of radiocaesium from the flesh of prey to carnivorous species. Whilst data for foxes and wolves have already been discussed above there are a number of other examples of this concentration process in terrestrial and aquatic ecosystems. Maximum radiocaesium activity concentrations of 87 000 Bq kg^{-1} were observed in Norwegian lynx in 1989; this was considerably higher than in their prey species [Gaare & Staaland, 1994]. Previously, *circa* 3-fold higher radiocaesium activity concentrations had been observed in the flesh of cougars compared to the flesh of mule deer [Pendleton *et al*., 1964]. A similar approximately 3-fold increase in radiocaesium activity concentrations was also observed in *small mammal–snake* food chains [Brisbin *et al*., 1974]. However, there is (perhaps) less evidence of a concentration of radiocaesium from invertebrate prey species to the mammals and birds consuming them; only the data of Rudge *et al*., 1993b indicating a concentration process of the four studies discussed above [i.e. NERC, 1993; Rudge *et al*., 1993a; Kålås *et al*., 1994; Copplestone, 1996]. This may, in some instances, be the result of the ingestion of soil together with prey species.

There is less data on the transfer of other radionuclides through terrestrial *prey–carnivore* food chains. Limited data for wolves discussed above suggests that radionuclides that accumulate in organs such as bone or liver, will not show elevated concentrations from the muscle of prey to that of carnivores (as the muscle of the prey species, which contributes most to the dietary intake of the carnivore, has comparatively low levels of such radionuclides). In an extensive survey of biota across eight "background" sites in the former Soviet Union Pokarzhevskii & Krivolutzkii [1997] reported that CR values for ^{226}Ra for *soil–plant*, *plant–animal* and *prey–carnivore* were usually close to or less than unity.

A number of studies of the movement of radionuclides through invertebrate food chains have demonstrated that detritivorous species have higher concentrations of radionuclides (Cs, Pu, Am) than herbivore and predatory species [Crossley, 1963; Rudge *et al*., 1993a; Copplestone, 1996; Copplestone *et al*., 1999].

Candidate reference organisms

Based upon the above discussion we can suggest some candidate reference organisms for semi-natural ecosystems from purely radioecological considerations.

When considering chronic exposure the organisms most exposed to external irradiation will be those living totally or partially within the soil. These will include: microorganisms that on account of their small size receive some dose from external alpha-emitters; soil invertebrates; burrowing mammals. Roots of plants growing and hyphae of fungi will be more exposed than above ground plant parts. Birds and mammals consuming soil dwelling invertebrates may be prone to ingestion of comparatively high rates of radionuclides.

Lichens will accumulate many aerially deposited radionuclides, whereas Ericaceous shrub species characteristic of many of the ecosystems considered here are likely to be amongst the plants with highest internal contamination (with radiocaesium). Of the above, ground invertebrate detritivorous species should be considered as candidate reference organisms. For larger animals, in the case of radiocaesium there is clear evidence of a concentration through food chains. Therefore predatory species should be candidate reference organisms. For organ seeking radionuclides (e.g. Pu, Am, Ru and Sr) higher concentrations may be found in herbivores compared with carnivores.

Candidate reference organisms are further discussed within Chapter 3 of FASSET Deliverable 1, 'Identification of candidate reference organisms from a radiation exposure pathways perspective'.

Data requirements and solutions

There may be a requirement to estimate the transfer of radionuclides to vegetation species for which there is currently little, or no, data. Broadley *et al*. [1999] recently analysed the available data on the transfer of radiocaesium to plant tissues on the basis of taxa, Caryophyllidae being identified as having the highest uptake. Similar analyses are currently being performed for ^{131}I, ^{106}Ru, ^{36}Cl and ^{90}Sr [Willey, 2001]. Such methodologies could be used to classify species of wild plants on their likely radionuclide transfers.

Many radiological protection models [e.g. Müller & Pröhl, 1993] utilise equilibrium transfer coefficients to describe the transfer of dietary radionuclides to animal tissue. Whilst there are few of these values available for wildlife species, allometric relationships related to live-weight have been suggested to extrapolate available data to different animal species [Sheppard, 2001]. For instance, MacDonald [1996] presented a relationship for the radiocaesium transfer coefficient (F_f):

$$F_f = 8.89(\text{Live weight})^{-0.73}$$

However, it is questionable as to if utilising daily food intake rates adds an unnecessary level of uncertainty. The similarity of radiocaesium CR values determined for a range of ruminant, bird and mono-gastric herbivores (see the data of Lowe & Horrill [1991] summarised in Table 3-6) might be worth further consideration. Perhaps CR is a more robust parameter than F_f for our purposes? Galeriu *et al*. [*in press*] have recently demonstrated that CR values are a more robust parameter than F_f to estimate the transfer of ^{3}H to farm animals; the generic models derived by these authors could be applied to derive CR values for wildlife species if the appropriate input parameters were available.

Where specific animal/plant data is lacking the compilation of relative concentrations of a number of radioisotopes in the organs of different groups of biota presented by Yankovich & Beaton [2000] should be utilised.

4. Agricultural ecosystems

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4.1 Ecosystem descriptions

4.1.1 Mediterranean ecosystem description

The Mediterranean region lies in the transitional area between the temperate, wet climate of Central and Northern Europe and the extremely arid area of North Africa.

Mediterranean climate belongs to middle latitude climates (named type C_{sa} and C_{sb} by Köppen) [Trewartha, 1968]. All of these are characterised by a very definite seasonal rhythm in temperature conditions, the amplitude of the ranges reaching a maximum for the earth in the north intermediate zone.

Mediterranean climate depends on rainfall distribution. Rainfall is highest between autumn and spring, and minimal in summer. This seasonal pattern is intensified by intense solar radiation, which causes high evapotranspiration and aridity during summer. Another important feature of this climate is the shortness of both autumn and spring, which only serves to separate the longer wet winters and the hot summers.

The Mediterranean Sea, with an area of some 2.5E+06 km^2, lies within latitudes dominated by anticyclones in summer and a succession of storms in winter. Although there is great variability in the climatic conditions of the region due to differences in altitude, proximity to the sea and natural barriers, which create a wide range of micro-climates, the combined topographic features of the mainland surrounding the Mediterranean Sea give rise to certain common characteristics. In general, it is possible to say that, as one moves from North to South and from West to East, the summers get hotter; there are fewer storms and less annual rainfall. Variability is also reflected in other climatic components such as atmospheric humidity, solar radiation, evapotranspiration and winds, all of which have conditioned the wide ecological diversity of the region.

Type location

The Mediterranean climate is located on the tropical margins of the middle latitudes (30–45 º), along the western sides of the continents. Currently occurs in the borderlands of the Mediterranean Sea, the Macaronesia (Azores, Madeira and Canary Island), central and coastal southern California, central Chile, the southern tip of South Africa, and parts of southern Australia. Mediterranean climate is characteristic of only 1.7 per cent of the earth's land area, but it is well known, since it is extensively developed within the Mediterranean Basin.

Climate parameters

Temperature

Winter months have average temperatures, between 4.4 ºC and 10 ºC, and the summer months between 21 ºC and 26.6 ºC [Pearson, 1992].

The oscillation between absolute maximum and minimum is also much more marked inland. The coldest winter month throughout the region is January, although February is equally cold along the central coast of the Mediterranean.

Precipitation

Rainfall is generally less than moderate, 380 to 635 mm being a typical average. More characteristic than the *amount* of rain is the *distribution* of rain over the year; there is a pronounced maximum during the cooler months, summer being nearly if not absolutely dry. This feature of dry summers and wet winters causes the Mediterranean climate to be unique among the climates of the world.

Every year, between September and October, and sometimes in November, intensive rainfall occurs and can cause floods and torrents. After the long and dry summer, the Mediterranean Sea, and especially the surface zones near the coast, reaches very high temperatures of approximately 30 ºC at the beginning of September. During this time, storms with very intensive rainfall, hail and lightning can occur. These squalls can deposit more than one hundred litres of water per square meter in less than one hour over a limited zone.

Relative humidity

The relative humidity is low in most of the Mediterranean region, with the exception of the coastline where high humidity, due to the maritime influence, is trapped by a mountain barrier. There are some areas where the sea mist or heavy dew-fall also gives rise to high humidity. In these areas crop production potential is high, although these same conditions are very favourable for the spread of pests and diseases.

Sunshine

Cloud cover in the European Mediterranean is under 20 % whereas in North Africa and the eastern Mediterranean it rarely reaches 10 %. The number of sunny days per year is about 250, with approximately 3 000 hours of annual sunshine.

Evapotranspiration

High potential evapotranspiration rates due to high solar radiation, dry wind and low relative humidity, together with scarce and ineffective rainfall, cause great water stress throughout the area. During the high summer temperatures this water stress becomes a serious constraint on growth and plant production, and hence irrigation becomes an important factor in summer cropping.

Soil development under Mediterranean conditions

There appears to be no single dominant mature soil type characteristic of the Mediterranean realm. This probably results from the unusual variety of surface configuration, geological formations, and vegetation conditions, which prevail there. Soils are likely to be thin, coarse in texture and somewhat droughty in nature.

Based on data from pleistocene paleosoils [ITGE, 1993], the most characteristic soils are brown and red fersiallitique, sometimes with a calcareous crust included.

A *Terra rossa* soil profile is usually found in the chalky regions, consisting of a litter layer, a dark humus horizon, and beneath this, a 1 to 2 m deep clay containing, plastic, bright red horizon. This type of soil is currently being generated under Mediterranean conditions.

Hydrological characteristics

A general condition of this climate is the alternation of a dry season (summer) with a wet season (winter or spring), which give the watercourses a seasonal regime. The mechanical erosion is globally much higher than the chemical erosion, at least seasonally.

Even where streams are not permanent, irrigation water often can be obtained from subterranean supplies contained in the unconsolidated and porous alluvial materials.

Natural vegetation

Mediterranean woodland is predominantly a mixed forest of low, or even stunted, trees and woody shrubs. Between the trees, the ground is completely or partially covered by pale, dusty, bush vegetation. This bush thicket is known as *chaparral* in California and *maqui* in lands bordering the Mediterranean Basin.

The vegetation consists mainly of sclerophyllous woody plants typical of winter-rain regions with only sporadic frost; plants cannot tolerate cold periods of greater duration. The most favoured time for growth is spring, when the soil is moist and the temperatures are rising, or autumn, after the first rain. The winter temperatures of 10 ºC and lower are too cool for growth.

The original zonal vegetation was evergreen sclerophyllus forest with *Quercus rotundifolia*. Small remnants of this association forest have provided the following data concerning the original forests [Walter, 1985]:

- *Tree layer* – 15–18 m tall, closed canopy composed exclusively of *Quercus rotundifolia*,
- *Shrub layer* – 3–5 m tall, *Buxus sempervirens*, *Viburnum tinus*, *Phillyrea media*, *Ph. angustifolia*, *Pistacia lentiscus*, *P. terebinthus*, etc.,
- *Herb layer* – approximately 50 cm tall, sparse, *Ruscus aculeatus*, *Rubia peregrina*, *Asparagus acutifolius*, *Asplenium adiatum-nigrum*, *Carex distachya*, etc.,
- *Moss layer* – very sparse.

The pine tree is another representative species in the arboreal stratum: however the species vary in accordance with the environmental conditions of each zone. Thus, *Pinus nigra* and *Pinus pinea* trees are typical of the Andalusian coast, and *Pinus nigra* are abundant in the Valencian region where *Pinus halepensis* is also found though punctually. The rest of the regions are marked by *Pinus sylvestris*.

The olive tree is very representative of Mediterranean sclerophyll woodland in regions of hot (C_{sa}) summers. The Mediterranean proper, it is known from classical Greek literature, was formerly forested with live oaks (*Q. rotundifolia)*, pines, cedars, wild carob and wild olive.

Throughout the world, the Mediterranean biome is characterised by shrubs. In most regions these shrubs are evergreen and have small, leathery (sclerophyllous) leaves with thick cuticles. Sometimes the leaves are so reduced that their appearance is needle-like. Many typical members of the shrub flora are aromatic (e.g., sage, rosemary, thyme, and oregano) and contain highly flammable oils.

In the mountain regions of the Mediterranean, a distinction must be made between:

- the humid altitudinal belts in the mountains on the northern margins of the Mediterranean zone, in which, with increasing altitude, not only does the temperature decrease, but the dry season also disappears, and
- the arid altitudinal belt with a summer drought, extending up to the alpine region.

4.1.2 Mediterranean agricultural ecosystem description

Humans have long impacted upon Mediterranean regions, especially through the use of fire and the grazing of livestock. Since some of our most ancient civilisations originated in the Mediterranean region, the zonal vegetation was long ago forced to give way to cultivation. The slopes have been deforested and used for grazing, with resultant soil erosion, so that nowadays only soils exhibiting varying stages of degradation remain.

An agricultural ecosystem results from human influence on a natural system to adjust it for her own needs. To develop agricultural systems, the human being causes alterations such as:

- changes in the composition of the ecosystem species,
- genetic improvement of the species,
- soil disturbances by adding fertilisers and other elements to improve the agricultural potential of the soils,
- alterations to the soil moisture with artificial irrigation,
- greenhouse's agriculture,
- intensive and extensive livestock.

The most important limitation to crop production in the Mediterranean region is without doubt water. This limitation may be the result of insufficient rainfall, or even, when water is sufficient, the incapacity of the soil to replenish moisture.

Characteristics of the cropping systems

The diversity of the farming and cropping systems in the region is enormous, each presenting very distinct characteristics which clearly show their adaptation to the environment in which they are found. It is possible to divide them into six major groups:

Pastoralims	Prevails in many of the areas of the region which are unable to support regular cropping due to the lack of rain. However, these areas also seasonally support a considerable amount of vegetation, especially shrubs and grassy species adapted to marginal growing conditions. Such conditions have led to the development of systems of livestock husbandry characteristic of nomadic and transhuman life.
Mixed rain-fed farming	These systems may be very diverse comprising a wide variety of trees, vines and annual crops for both human and animal consumption. A wide diversity of crops, a high degree of dispersion of landholdings, and their

	considerable fragmentation frequently characterise them.
Annual rain-fed cropping	These are the most important and diverse systems within the Mediterranean. When the annual rainfall rate is about 400 mm, rain-fed annual cropping prevails over perennial crops and pastures. When the annual rainfall rate is about 500 mm, crops and rotations patterns tend to be more diverse and complex. Such systems include legumes, different forages and oil crops in rotation with cereals.
Irrigated agriculture	Irrigation constitutes the single most important factor for increase in crop yield and reduction of risk under Mediterranean conditions. There are two traditional ways of using water: in winter crop, water is often needed only to supplement rainfall. The second method entails the use of permanent irrigation systems like dams or rivers, or aquifers that supply water throughout the year.
Multiple cropping	This is a form of cropping found within irrigated systems, where mild climate together with irrigation allow the introduction of several crop cycles in the same year.
Covered cropping	Crop covered with plastics constitute an important, perfectly differentiated cropping system with their own technological and economic characteristics. In maritime areas with mild winter climate, covered cropping has developed especially in the last 20 years. The objective has been to exploit winter sunlight and mild weather, using simple coverings to produce flowers, fruits or vegetables that are not in season and thus have a high export demand and high prices. Covered cropping in the Mediterranean constitutes about 40 % of the area of greenhouse and tunnel cropping in the world.

Benefits of cover crops are [Agüero *et al.*, 1996]:

- Soil quality improvements – Soil tilth is improved whenever a plant establishes roots and grows into compacted areas. Water infiltration is improved as well. When a field lays fallow for a period of time, the surface tends to seal and water will run off. Cover crops protect the soil surface and reduce sealing. Also, beneficial organisms in the soil, such as earthworms, thrive when fresh plant material is decomposing. Organic matter levels tend to improve with the addition of cover crops.
- Erosion control – Cover crops reduce wind and water erosion on all types of soils. By having the soil held in place by cover crops during the fall, winter, and early spring, loss of soil via erosion is greatly reduced.
- Fertility improvements – Legumes can add substantial amounts of available nitrogen to the soil. Non-legumes can be used to take up excess nitrogen from previous crops and recycle the nitrogen as well as available phosphorus and potassium to the following crop. This is very important after manure application, because cover crops can reduce leaching of nutrients.
- Suppression weeds – A dense stand of winter rye or other cover crop can suppress weeds by soil shading. Allelochemicals from cover crops suppress the growth of other plants.
- Insect control – Beneficial insects, such as ladybugs (*Coccinella sp.*) or ground beetles, may be encouraged by planting cover crops.

Typical crops in a Mediterranean agricultural ecosystem

Table 1 has been elaborated from a literature review [MAPYA, 1991]. Groups of crops and species names are included in the table.

Table 4-1 Typical crops in a Mediterranean agricultural ecosystem.

TYPE OF VEGETAL	SPECIES NAME		
	Common Spanish	Common English	Latin Name
Citric Fruits	Limón	Lemon	*Citrus limonis*
	Mandarina	Mandarin	*Citrus reticulata*
	Naranja	Orange	*Citrus sinensis/aurantium*
	Pomelo	Grapefruit	*Citrus paradisi*
Non Citric Fruits	Manzana	Apple	*Pyrus malus*
	Nispero	Persimmon	*Eriobotrya japonica*
	Pera	Pear	*Pyrus communis*
	Albaricoque	Apricot	*Prunus armeniaca*
	Cereza y guinda	Cherries	*Prunus avium/cerasus*
	Ciruela	Plum	*Prunus domestica*
	Melocotón	Peach	*Prunus persica*
	Aguacate	Avocado pear	*Persea americana*
	Chirimoya	Anona soursop	*Annona cherimolia*
Dry fruits	Almendra	Almond	*Prunus amigdalus*
	Avellana	Hazelnut	*Corylus avellana*
Other Fruits	Aceituna	Olive	*Olea europaea*
	Uvas	Grape	*Vitis vinifera*
Root Vegetables	Patata	Potatoe	*Solanum tuberosum*
	Zanahoria	Carrot	*Daucus carota*
	Ajo	Garlic	*Allium sativum*
	Cebolla	Onion	*Allium cepa*
	Puerro	Leek	*Allium porrum*
	Espárrago blanco	Asparagus	*Asparagus officinalis*
Leafy Vegetables	Acelga	Beet	*Beta vulgaris*
	Apio	Celery	*Apium graveolens*
	Col-repollo	Cabbage	*Brassica oleracea*
	Col de bruselas	Brussels sprouts	*Brassica oleracea*
	Endibia	Chicory	*Cichorium intybus*
	Escarola	Endive	*Cichorium endivia*
	Espinaca	Spinach	*Spinacia oleracea*
	Lechuga	Lettuce	*Lactuca sativa*
	Alcachofa	Globe artichoke	*Cynara scolymus*
	Coliflor	Cauliflower	*Brassica oleracea*
Fruit Vegetables	Berenjena	Eggplant	*Solanum melongena*
	Calabaza	Pumpkin	*Cucurbita maxima*
	Calabacin	Tender pumpkin	*Cucurbita pepo*
	Fresa	Strawberry	*Fragaria sp.*
	Melón	Melon	*Cucumis melo*
	Pepino	Cucumber	*Cucumis sativus*
	Pimiento	Fresch peppers	*Capsicum annuum*
	Sandía	Watermelon	*Citrullus vulgaris*
	Tomate	Tomato	*Lycopersicum esculentum*
Legumes	Guisante	Peas	*Pisum sativum*
	Haba	Beans	*Vicia faba*
	Judía	Beans	*Phaseolus vulgaris*
	Lenteja	Lentil	*Lens esculenta*
	Garbanzo	Chickpeas	*Cicer arietinum*
Cereals	Trigo	Wheat	*Triticum sativum*
	Maíz	Maize	*Zea mays*
	Cebada	Barley	*Hordeum vulgare*
	Centeno	Rye	*Secale cereale*
	Arroz	Rice	*Oriza sativa*

Crop requirements

Table 4-2 Crop requirements for growth in specific conditions.

Crop	Soil Type (pH)	Water Request (m^3/ha)	Optimal Temperature	Variety
Citric Fruits				
Lemon	wide spectrum	large 9 000-12 000	warm, no freeze	Verna Fino Eureka
Mandarin	sand, sand-loam (6–7)	large 9 000–12 000	warm, no freeze	Satsumas Clementinas
Orange	sand, sand-loam (6–7)	large 9 000–12 000	warm, no freeze	Navelina Washington Navel Navelate Valenca Late Salustiana Verna Sanguinas
Grapefruit	sand, sand-loam (6–7)	large 9 000–12 000	warm, no freeze	Duncan Marsh Burgundy Ruby
Non Citric Fruit				
Apple	silica-clay	medium	no high temp.	Golden Red Starking Reineta Verde Doncella
Persimmon	well drained (6–8)	medium	> 10 ºC	Tanaka Algerie
Pear	silica-clay	medium	no high temp	William's Limonera Blanquilla De Roma
Apricot	wide spectrum	medium	temperate	Bulida Canino Paviot Moniqui
Cherries	well developed	medium	no high temp	Napoleon Ambrunesa Burlat
Plum	wide spectrum	10 000–12 000	no high temp	Golden-Japan Santa Rosa Reina Claudia Verde R. Claudia Oulling
Peach	wide spectrum	large	temperate	Springcrest Spring Lady Stark Redgold Snow Queen
Avocado pear	well drained (5,5–7)	medium	temperate	Hass Fuerte
Anona soursop	fertiles (6,5–7,5)	large	warm, no freeze	Fino de Jete Campa
Dry Fruits				
Almond	sandy sandy-loam	3 000	warm, no freeze	Marcona Demayo Largueta
Hazelnut	well developed	medium-large	temperate	Negret Morrell
Other Fruits				
Olive	wide spectrum	moderate	-10–35 ºC	Manzanilla Gordal Picual

Table 4-2 (continued) Crop requirements for growth in specific conditions.

Crop	Soil Type (pH)	Water Request (m^3/ha)	Optimal Temperature	Variety
				Arbequina Hojiblanca Picudo Cornicabra
Grape	wide spectrum	moderate	temperate	Verdejo Albariño Parellada Airen Moscatel Garnacha Tempranillo
Root vegetables				
Potatoe	sandy, sandy-loam (5.5–7)	12 000–15 000	temperate	Baraka Kind Edward Red Pontiac Jaerla
Carrot	clay (5.8–7)	large	temperate	Tip-Top Nantesa
Garlic	wide spectrum	low	wide spectrum	Morado Blanco
Onion	sandy	medium-low	temperate	Grano de oro Babosa Liria
Leek	sandy	medium	temperate	Helvetia Amarillo de Poiton Ecotipo-Varea
Asparagus	sandy-loam, loam (7.5–8)	medium-low	warm, no freeze	Darbonne Lorella Argenteuil Mary Washington
Leafy Vegetables				
Beet	clay, well developed (5.5–8)	large	temperate	Amarilla de Lyon Verde Bresanne
Celery	well developed	medium-large	temperate	Golden Spartan Pascal
Cabbage	loam, clay	medium	low	Brunswick Milan Lombarda Morada Corazon de Buey
Brussels sprouts	loam, clay	medium	low	Jade Cross Long Island Oliver
Chicory	sandy	medium	temperate	Zoom Wolfero
Endive	loam-clay (6–7)	medium	low	Francesa Gigante Rizada del Prat Cabello de Angel
Spinach	well developed	medium	temperate (5–15 ºC)	Viroflay Viroflex
Lettuce	sandy (6.7–7.4)	medium	moderate (-6–30 ºC)	Romana Trocadero Iceberg
Globe artichoke	sandy	low-moderate	warm no freeze	Blanca de Tudela Violeta de Provenza
Cauliflower	loam, clay	medium	low	Snow-Ball Catalina Matra
Fruit vegetables				
Eggplant	well developed (7–8.5)	medium	warm-hot	Bonica Midnite Jaspeada
Pumpkin & P. Tender	loam	medium	warm-hot	Elite

Table 4-2 (continued) Crop requirements for growth in specific conditions.

Crop	Soil Type (pH)	Water Request (m^3/ha)	Optimal Temperature	Variety
	(5.6–6.8)			Diamante President
Strawberry	wide spectrum (6–7)	large	low-hot (-20–55 ºC)	Camarosa Tudla Oso Grande Cartuno Carisma
Melon	well developed (6–7)	large	warm-hot	Piel de Sapo Amarillos Cantalupos Tendral
Cucumber	sandy	large	warm, no freeze	Pepinex Saticoy Corona
Fresh peppers	sandy-loam (5.5–7)	medium	warm, no freeze	Dulce-italiano Morron Piquillo
Watermelon	well developed	large	warm	Grimson Sweet Panonia Sugar Baby Resistent
Tomato	wide spectrum	medium	warm, no freeze	Dombo Nancy Muchamel Rio Grande
Legumes				
Peas	loam (7.3–8.2)	medium	Warm, no freeze	Tristar Lincoln Tirabeque
Beans (*V. faba*)	clay, well develop.	low-medium	temperate	Muchamiel Granadina Aguadulce
Beans (*Ph. vulgaris*)	loam (6–7.5)	medium	warm-humid (12–15 ºC)	Mocha Kora Strike Brasileña Garrafal
Lentil	well developed	low	temperate	Pardina Castellana
Chickpeas	loam-clay (6–9)	low	temperate	Kabuli Desi Gulabi
Cereals				
Wheat	well developed	low	temperate	of large cycle of short cycle alternatives
Maize	wide spectrum (6–7)	large	warm	
Barley	wide spectrum	moderate	temperate	unirrigated irrigated
Rye	sandy (< 7)	low	low	Giganton Petkus Galma
Rice	------	flooded soil	warm-hot	

Common farm animals in a Mediterranean ecosystem

The common animal races in a Mediterranean farm system are summarised in Table 4-3. In any case, the animal species are the same in any farmland system.

Table 4-3 Common farm animals in a Mediterranean ecosystem.

TYPE OF ANIMAL	SPECIES NAME Common Spanish	Common English	Latin Name
Bovine	Vaca	Cow	Box taurus
Ovine	Oveja	Sheep	*Ovis sp* races: Churra Merina Castellana Ojalada Aragonesa rasa Lacha Manchega Carranzana Frizona
Caprine	Cabra	Goat	*Capra hircus* races: Blanca andaluza Blanca celtibérica Canaria Alpina Payoya Saanen Murciana-granadina Boer Negra serrana Pigmea Malageña Verata Nubian Del Guadarrama Retinta extremaña
Porcines	Cerdo	Pig	*Sus sp* races: Iberica Chester blanca Duroc Berkshire Yorkshire Hampshire Landrace Poland china Spotted
Lagomorph	Conejo	Rabbit	*Oryctolagus cuniculus*
Barnyard fowl	Gallina Pato Oca Pavo Perdiz Codorniz Faisan	Hen Duck Goose Turkey Partridge Quail Pheasant	*Gallus gallinaceus* *Anas sp* *Anser sp* *Meleagris sp* *Perdix sp* *Colinus sp* *Phasianus sp*
Equidae	Caballo Asno	Horse Donkey	*Equus caballus* *Equus asinus*

Animal feedings

The feeding data in Tables 4-4 and 4-5 are normally used in dose calculation models.

Table 4-4 Animal feeding [Müller & Pröhl, 1993].

Animal	Feedstuff	Ingestion rate ($Kg\ d^{-1}$) fresh weight
Lactating cow	Grass	70[a]
Lactating Sheep	Grass	9[a]
Lactating goat	Grass	13[a]
Beef cattle	Maize silage	28
Calf	Milk substitute	2.9
Pig	Winter barley	3.0
Lamb	Grass	5[a]
Hen, chicken	Winter wheat	0.09
Roe deer	Grass extensive	4[a]

a Values given are for the vegetation period; during the winter an equivalent dry matter intake with hay or silage is assumed.

Table 4-5 Animal ingestion of water and dry matter and the fraction of the year that animals consume fresh pasture [IAEA, 2000].

Parameter	Default value	Units	Ref.
Q_w (milk, large animal)[a]	0.06	m^3/d	TRS-364 (IAEA, 1994)
Q_w (meat, large animal)[a]	0.04	m^3/d	TRS-364 (IAEA, 1994)
Q_m (milk, large animal)	16	Kg/d (dry weight)	GROGAN, 1989
Q_f (milk, large animal)	12	Kg/d (dry weight)	GROGAN, 1989
f_p	0.7	unitless	---

a For small animals such as goats and sheep 1/10th of these quantities should be used as a default.

These values are very conservative in assuming that livestock only eat natural pastures and cereals; actually animals are nourished with industrial by-products, concentrates, minerals and vitamins supplements also.

In general, feeds are classified into one of the following categories:

- forages,
- concentrates (energy and protein feeds),
- minerals and vitamins.

This classification is a convenient way to group feeds, but it is somewhat arbitrary.

Forages

In general, forages are the vegetative parts of grasses or legume plants containing a high proportion of fibre. Usually, forages are grown on the farm. They may be grazed directly or harvested and preserved as hay or silage. From a nutritional standpoint, forages may range from very good feeds (lush young grass, legumes at a vegetative stage of maturity) to very poor feeds (straw, browse). In addition, also considered as forage are several crop residues, which are the parts of the plants that remain in the field after harvesting the primary crop (e.g. corn stover, cereal straw, sugar cane bagasse).

Concentrates

There is not an accepted definition for the word concentrate, but a concentrate usually includes some of the following products:

- cereal grains (barley, corn, sorghum, rice, wheat),
- corn gluten meal and corn gluten feed,
- brewing and distilling by-products of cereal grains and malt sprouts,
- roots and tubers (carrots, cassava, beets, potatoes, turnips),
- by-products of the sugar industry (molasses, sugar beet pulp),
- oilseed meals (soybean, groundnut, cotton, sunflower),
- seed of legumes (beans, chickpeas, cowpeas),
- proteins of animal origin (meat and bone meal, fish meal).

4.1.3 Linkage to other ecosystems

The agricultural ecosystems are completely linked with the semi-natural pasture ecosystems. The animal feed is based on grazing and special agricultural products or by-products. As an example of this relation it is possible to cite the nomadism and the transhumance that are two different forms of pastoral life. Transhuman people usually move along established migration paths from some type of permanent base, where a small degree of cropping may be practised. Forms of transhumance include movements from winter quarters in valleys and plains to summer grazing on the mountains.

4.2 Exposure pathways and approaches used to select reference organisms

Transfer of radionuclides from soil to plants and from plants to grazing animals in an agricultural ecosystem depends on several factors. Different transfer processes become more or less important depending on the type of nuclide and means of contamination, season and time elapsed. In the initial phase of an aerial contamination the most important transfer processes are interception of radionuclides by plant surfaces, translocation to internal parts of the plant and leaching of radionuclides from the plant surfaces to the soil by weathering processes. Radionuclides will be incorporated in the cycling of nutrients in the agricultural landscape. This cycling can lead to partial removal of nuclides from the soil during growth periods and returning of nuclides to soil when plants or plant parts deteriorate. Weathering processes can also contribute to increased concentrations of nuclides in the soil. Further on, radionuclides in an agricultural ecosystem can enter other food chains through grazing animals.

The factors for soil to plant transfer are shown in Table 4-6. As can be deduced from these numbers transfer to agricultural plants are highest for strontium. This is not surprising since strontium usually shows more mobility in soils and is more readily taken up by plants than for example caesium. For strontium the transfer is highest to green vegetables. Generally the transfer for different nuclides is highest to pasture. The rooting depth of a plant has been considered as a key factor in the differences in ^{137}Cs levels [Römmelt, 1990; Guillitte, 1994; Nimis *et al*., 1994; Scheglov, 1998; Fesenko *et al*., 2001a]. The reason for this is the different content of ^{137}Cs in different soil layers.

When it comes to grazing animals in an agricultural ecosystem, it seams that caesium is concentrated in animals to a larger degree than strontium (Table 4-6). This is true for soft tissues where caesium is present in its largest concentrations, yet strontium tends to concentrate in bone (transfer to bone has not been reported here). Therefore the corresponding transfer factor for strontium in bone would be expected to be higher than for strontium in meat.

4.2.1 The Mediterranean ecosystem

Soil to plant transfer factors and equilibrium transfer factors for animals have been obtained for the selected radionuclides from literature reviews. The factors are shown in Tables 4-6 and 4-7. Underlined values have been used in the illustrative calculations (see Tables 4-9–4-16)

Table 4-6 Soil to plant transfer factor (Bq/kg dry weight veg.)/(Bq/kg dry weight soil), or *Soil to plant transfer factor (Bq/kg wet weight veg.)/(Bq/kg dry weight soil).*

	Green veg.	Cereal grain	Roots	Tubers, Potatoe	Pasture	Peas, Beans	Rice	Fodder	Crops	Fruits	Berries
Cs											
*	*7.0E-3*	*1.0E-2*	*5.0E-3*	*7.0E-3*	*3.0E-2*						
**											
pH=6	1.8E-1	1.0E-2	4.0E-2	7.0E-2	1.1E-1	1.7E-2	A5.0E-3	1.7E-2			
pH=5	4.6E-1	2.6E-2	1.1E-2	1.7E-1	2.4E-1	9.4E-2	B2.0E+	2.9E-1			
pH=4	2.6E-1	8.3E-2		2.7E-1	5.3E-1		1	3.0E-1			
***								1.0	*0.04*		
****	*2.0E-2*	*2.0E-2*	*1.0E-2*	*1.0E-2*	*5.0E-2*	*1.0E-2*		*2.0E-2*		*2.0E-2*	*2.0E-2*
Sr											
*	*3.0E-1*	*2.0E-1*	*1.0E-1*	*5.0E-2*	*5.0E-2*						
**											
pH=6	2.7	1.2E-1	1.1	1.5E-1	1.1	1.3		1.9E-1			
pH=5	3.0	2.1E-1	1.4	2.6E-1	1.7	2.2		1.0			
pH=4	2.6E-1	2.0E-2		2.0E-2	3.4E-1						
***								10.0	*3.0E-1*		
****	*4.0E-1*	*2.0E-1*	*3.0E-1*	*5.0E-2*	*5.0E-1*	*2.0E-1*		*3.0E-1*		*1.0E-1*	*1.0E-1*
Tc											
*	*5.0*	*5.0*	*5.0*	*5.0*	*5.0*						
**	1.2E+1 2.0E+2 2.6E+3	7.3E-1	7.9E+1	2.4E-1	7.6E+1	4.3		8.1			
***								80.0	*5.0*		
Po											
*	*2.0E-4*	*2.0E-4*	*2.0E-4*	*2.0E-4*	*2.0E-4*						

Table 4-6 (continued) Soil to plant transfer factor (Bq/kg dry weight veg.)/(Bq/kg dry weight soil), or *soil to plant transfer factor (Bq/kg wet weight veg.)/(Bq/kg dry weight soil)*.

	Green veg.	Cereal grain	Roots	Tubers, Potatoe	Pasture	Peas, Beans	Rice	Fodder	Crops	Fruits	Berries
**	1.2E-3	2.3E-3		7.0E-3	9.0E-2						
***								1.0E-1	*2.0E-3*		
K	–	–	–	–	–	–	–	–	–	–	–
Pu											
*	*1.0E-5*	*2.0E-5*	*5.0E-5*	*5.0E-5*	*1.0E-4*						
**	7.3E-5	8.6E-6	9.1E-4	1.5E-4	3.4E-4	6.1E-5		8.0E-4 7.5E-5			
***								1.0E-1	*1.0E-3*		
****	*1.0E-4*	*1.0E-4*	*1.0E-4*	*1.0E-4*	*2.0E-4*	*1.0E-4*		*3.0E-1*		*1.0E-4*	*1.0E-4*
Am											
*	*5.0E-5*	*5.0E-5*	*8.0E-5*	*8.0E-5*	*1.0E-3*						
**	6.6E-4	2.2E-5	2.2E-3 1.4E-3 1.6E-4	2.0E-4	1.2E-3	3.9E-4		7.1E-4 2.7E-4			
***								1.0E-1	*2.0E-3*		
I											
*	*2.0E-2*	*2.0E-2*	*2.0E-2*	*2.0E-2*	*2.0E-2*						
**					3.4E-3						
***								1.0E-1	*2.0E-2*		
****	*1.0E-1*	*1.0E-1*	*1.0E-1*	*1.0E-1*	*1.0E-1*	*1.0E-1*		*1.0E-1*		*1.0E-1*	*1.0E-1*
Ra											
*	*1.0E-2*	*1.0E-3*	*1.0E-3*	*1.0E-3*	*1.0E-2*						
**	4.9E-2	1.2E-3	1.1E-2 2.1E-2	1.1E-3	8.0E-2	7.0E-3					
***								4.0E-1	*4.0E-2*		
H	–	–	–	–	–	–	–	–	–	–	–
C	–	–	–	–	–	–	–	–	–	–	–
U											
*	*1.0E-3*	*1.0E-3*	*1.0E-3*	*1.0E-3*	*1.0E-3*						
**	8.3E-3	1.3E-3	1.4E-2	1.1E-2	2.3E-2						
***								2.0E-1	*1.0E-2*		
Th											
*	*5.0E-4*	*5.0E-4*	*5.0E-4*	*5.0E-4*	*5.0E-4*						
**	1.8E-3	3.4E-5	3.0E-4 3.9E-2 6.2E-5	5.6E-5	1.1E-2	1.2E-4		7.5E-3			
***								1.0E-1	*1.0E-3*		
Cl											
*	*5.0*	*5.0*	*5.0*	*5.0*	*5.0*						
Nb											
*	*1.0E-2*	*1.0E-2*	*1.0E-2*	*1.0E-2*	*1.0E-2*						
**						1.8E-2					
***								2.0E-1	1.0E-2		
****	*2.0E-3*	*4.0E-3*	*5.0E-4*	*1.0E-3*	*4.0E-3*	*5.0E-4*		*6.0E-3*		*5.0E-4*	*5.0E-4*

Table 4-6 (continued) Soil to plant transfer factor (Bq/kg dry weight veg.)/(Bq/kg dry weight soil), or *soil to plant transfer factor (Bq/kg wet weight veg.)/(Bq/kg dry weight soil).*

	Green veg.	Cereal grain	Roots	Tubers, Potatoe	Pasture	Peas, Beans	Rice	Fodder	Crops	Fruits	Berries
Ni											
*	<u>*1.0E-2*</u>	<u>*1.0E-2*</u>	<u>*1.0E-2*</u>	*1.0E-2*	<u>*1.0E-2*</u>						
**		3.0E-2			1.8E-1			5.1E-1			
***								<u>1.0</u>	*3.0E-1*		
Pb											
*	<u>*1.0E-2*</u>	<u>*1.0E-2*</u>	<u>*1.0E-2*</u>	*1.0E-2*	<u>*1.0E-2*</u>						
**	1.0E-2	4.7E-3	2.0E-2	1.3E-3				1.1E-3			
***								<u>1.0E-1</u>	*2.0E-2*		
Np											
*	<u>*2.0E-3*</u>	<u>*2.0E-3*</u>	<u>*1.0E-3*</u>	*1.0E-3*	<u>*1.0E-2*</u>						
**	3.7E-2	2.7E-3	3.5E-2 2.6E-2 3.3E-2	6.7E-3	6.9E-2	1.8E-2		8.1E-3 2.2E-2			
***								<u>5.0E-1</u>	*4.0E-2*		
Ru											
*	*1.0E-2*	*1.0E-2*	*1.0E-2*	*1.0E-2*	*1.0E-2*						
**		5.0E-3			3.4E-4						
***								<u>2.0E-1</u>	*5.0E-2*		
****	<u>*1.0E-2*</u>	<u>*1.0E-2*</u>	<u>*1.0E-2*</u>	*1.0E-2*	<u>*2.0E-2*</u>	*1.0E-2*		*1.0E-2*		<u>*1.0E-2*</u>	*1.0E-2*
Cm											
*	<u>*5.0E-5*</u>	<u>*2.0E-5*</u>	<u>*3.0E-5*</u>	*3.0E-5*	<u>*1.0E-3*</u>						
**	7.7E-4	2.1E-5	1.3E-3	1.5E-4	1.1E-3	7.5E-4		2.1E-4			
***								<u>1.0E-1</u>	*1.0E-3*		

* [European Commission, 1995].
** [IAEA, 1994].
A (irr.) soil to plant
B (irr.) water to plant
*** [IAEA, 2000].
**** [Müller & Pröhl, 1993].
Underlined values have been selected for the screening calculations.

Table 4-7 Equilibrium transfer factors for grazing animals (d/L) or (d/kg).

	Cow milk	Cow meat	Cow liver	Sheep meat	Sheep liver	Sheep milk	Goat milk	Goat meat	Pork	Poultry	Eggs
Cs											
*	<u>5.0E-3</u>	3.0E-2	3.0E-2	<u>5.0E-1</u>	5.0E-1						
**	7.9E-3	[A]5.0E-2 [B]2.0E-1		[A]1.7E-1 [B]4.9E-1		<u>5.8E-2</u>	1.0E-1	2.3E-1	<u>2.4E-1</u>	<u>1.0E+1</u>	<u>4.0E-1</u>
***	1.0E-2	<u>5.0E-2</u>		3.0E-1		1.0E-1	1.0E-1				
****	3.0E-3	[A]1.0E-2 [B]3.5E-1		[B]5.0E-1		6.0E-2	6.0E-2		4.0E-1	4.5	3.0E-1

Table 4-7 (continued) Equilibrium transfer factors for grazing animals (d/L) or (d/kg).

	Cow milk	Cow meat	Cow liver	Sheep meat	Sheep liver	Sheep milk	Goat milk	Goat meat	Pork	Poultry	Eggs
Sr											
*	2.0E-3	3.0E-4	3.0E-4	3.0E-3	3.0E-3						
**	2.8E-3	[A]8.0E-3 [B]1.0E-1		[A]4.0E-2 [B]3.3E-1		5.6E-2	2.8E-2	2.8E-3	4.0E-2	8.0E-2	2.0E-1
***	3.0E-3	1.0E-2									
****	2.0E-3	[A]3.0E-4 [B]2.0E-3		[B]3.0E-3		1.4E-2	1.4E-2		2.0E-3	4.0E-2	2.0E-1
Tc											
*	1.0E-2	1.0E-2	4.0E-2	1.0E-1	3.0E-1						
**	2.3E-5	1.0E-4					1.1E-2	2.2E-4	1.5E-4	3.0E-2	3.0
***	1.0E-3	1.0E-3									
Po											
*	1.0E-4	3.0E-3	8.0E-2	5.0E-2	6.0E-1						
**	3.4E-4	5.0E-3									
***	3.0E-3	5.0E-3									
K											
**	7.2E-3	2.0E-2									1.0
Pu											
*	1.0E-6	1.0E-4	2.0E-2	4.0E-4	3.0E-2						
**	1.1E-6	[A]1.0E-5 [B]1.0E-3		[A]9.4E-5 [B]3.1E-3		9.4E-6	9.4E-6		8.0E-5	3.0E-3	5.0E-4
***	3.0E-6	2.0E-4									
****	6.0E-5	[A]6.0E-5 [B]2.0E-4		[B]7.0E-4		4.0E-4	4.0E-4		3.0E-4	2.0E-4	7.0E-3
Am											
*	1.0E-6	1.0E-4	2.0E-2	4.0E-4	3.0E-2						
**	1.5E-6	[A]4.0E-5 [B]1.0E-3		[A]2.0E-4 [B]4.1E-3		1.4E-5	1.4E-5		1.7E-4	6.0E-3	4.0E-3
***	2.0E-5	1.0E-4									
I											
*	5.0E-3	2.0E-3	2.0E-3	5.0E-2	5.0E-2						
**	1.0E-2	4.0E-2		3.0E-2		4.9E-1	4.3E-1		3.3E-3	1.0E-2	3.0
***	1.0E-2	5.0E-2				5.0E-1	5.0E-1				
****	3.0E-3	[A]1.0E-3 [B]3.0E-3		[B]1.0E-2		5.0E-1	5.0E.1		3.0E-3	1.0E-1	2.8
Ra											
*	4.0E-4	5.0E-4	5.0E-4	5.0E-3	5.0E-3						
**	1.3E-3	9.0E-4									
***	1.0E-3	5.0E-3									
H	–	–	–	–	–	–	–	–	–	–	–
C	–	–	–	–	–	–	–	–	–	–	–
U											
*	6.0E-4	2.0E-4	2.0E-4	2.0E-3	2.0E-3						
**	4.0E-4	3.0E-4							6.2E-2	1.0	1.0

Table 4-7 (continued) Equilibrium transfer factors for grazing animals (d/L) or (d/kg).

	Cow milk	Cow meat	Cow liver	Sheep meat	Sheep liver	Sheep milk	Goat milk	Goat meat	Pork	Poultry	Eggs
***	6.0E-4	3.0E-3									
Th											
*	5.0E-6	1.0E-4	1.0E-3	1.0E-3	1.0E-2						
***	5.0E-6	1.0E-4									
Cl											
**	1.7E-2	2.0E-2									
Nb											
*	1.0E-5	1.0E-5	1.0E-5	1.0E-4	1.0E-4						
**	4.1E-7	3.0E-7		3.0E-4			6.4E-6	6.0E-5	2.0E-4	3.0E-4	1.0E-3
***	4.0E-6	3.0E-6									
****	4.0E-7	^{A}3.0E-7 ^{B}1.0E-6		^{B}3.0E-6		6.0E-6	6.0E-6		2.0E-6	3.0E-4	1.0E-3
Ni											
**	1.6E-2	5.0E-3									
***	2.0E-1	5.0E-2									
Pb											
*	3.0E-4	1.0E-3	2.0E-3	1.0E-2	2.0E-2						
**		4.0E-4									
***	3.0E-4	7.0E-4									
Np											
*	1.0E-6	1.0E-4	2.0E-2	4.0E-4	3.0E-2						
**	5.0E-6	1.0E-3					1.0E-4				
***	5.0E-5	1.0E-2									
Ru											
*	1.0E-6	1.0E-3	1.0E-3	1.0E-2	1.0E-2						
**	3.3E-6	^{A}5.0E-2 ^{B}4.0E-1		^{A}2.6E-1 ^{B}1.5					6.6E-1	8.0	5.0E-3
***	3.0E-5	5.0E-2									
****	1.0E-4	^{A}1.0E-3 ^{B}2.0E-3		^{B}1.0E-2		1.0E-3	1.0E-3		5.0E-3	7.0E-3	6.0E-3
Cm											
*	1.0E-6	1.0E-4	2.0E-2	4.0E-4	3.0E-2						
***	2.0E-6	2.0E-5									

A) Animal adult
B) Veal or Lamb
* [European Commission, 1995]
** [IAEA, 1994]
A (irr.) soil to plant
B (irr.) water to plant
*** [IAEA, 2000]
**** [Müller & Pröhl, 1993].
Underlined values have been selected for the screening calculations.

4.2.2 Modelling approaches

The interaction matrix and conceptual model

The interaction matrix was initially applied to the joint SKI/SKB scenario development project. At the time when the approach of ‘Rock Engineering System’ (RES) was presented, it became evident that this methodology could be applied to scenario development and analysis for assessment of radiological impact.

The methodology used can be named as top-down, in order to ensure that all aspects of the problem are being covered. The method used the interaction matrix as a visual and graphical device or tool for the representation of the ‘components’ and ‘interactions’ among them.

The main factors are identified and listed along the leading diagonal elements (LDEs) of a square matrix. The interactions between the factors occur in the off-diagonal elements (ODEs). Clockwise for the influence direction was adopted for convention [Stephansson & Hudson, 1993].

Interaction matrix construction for agricultural ecosystems

The matrix construction begins with:

1. definition of the Leading Diagonal Elements (LDEs),
2. definition of Off-Diagonal Elements (ODEs).

LDEs represent the main components of the system. ODEs represent interaction processes between LDEs.

The interaction matrix for a generic agricultural ecosystem is shown in Figure 4-1. Including transfer processes with different significance, these will be screening and from this analysis features and processes will be considered or not within the conceptual and mathematical models.

Source term	Release of gaseous effluent	Release of liquid effluent					
	Atmos-phere	Dry and wet deposition		Rainfall Snowfall Dry and wet deposition	Rainfall Snowfall Dry and wet deposition Interception External irradiation	Inhalation External irradiation	
		River water	Sedimentation	Irrigation	Irrigation	Ingestion	Outflow
		Resuspension	River sediment				Burial
	Resuspension			Agricultural soil	Resuspension Root uptake	Ingestion External irradiation	Erosion Percolation
	Transpiration			Weathering	Crops	Ingestion	Harvesting
	Exhalation			Excretion		Domestic animals	Slaughtering
							Sink

Figure 4-1 Matrix representation of the migration of radionuclides in an agricultural ecosystem. The leading diagonal elements (LDEs) correspond to the various components of the system that have been identified as being relevant conceptual model objects in the representation of the contaminant migration within the ecosystem. The off-diagonal elements (ODEs) are interactions between LDEs (transfer processes between components). In order to identify the transfer processes the matrix should be read clockwise.

Conceptual model development

From the screening analysis, the conceptual model developed for an agricultural ecosystem is shown in Figure 4-2. It includes four compartments representing environmental media (atmosphere, soil, water and sediment), two compartments representing concentrations in biota (crop concentration and animal concentration) and two biota final receptors receiving doses (crop total dose and animal total dose).

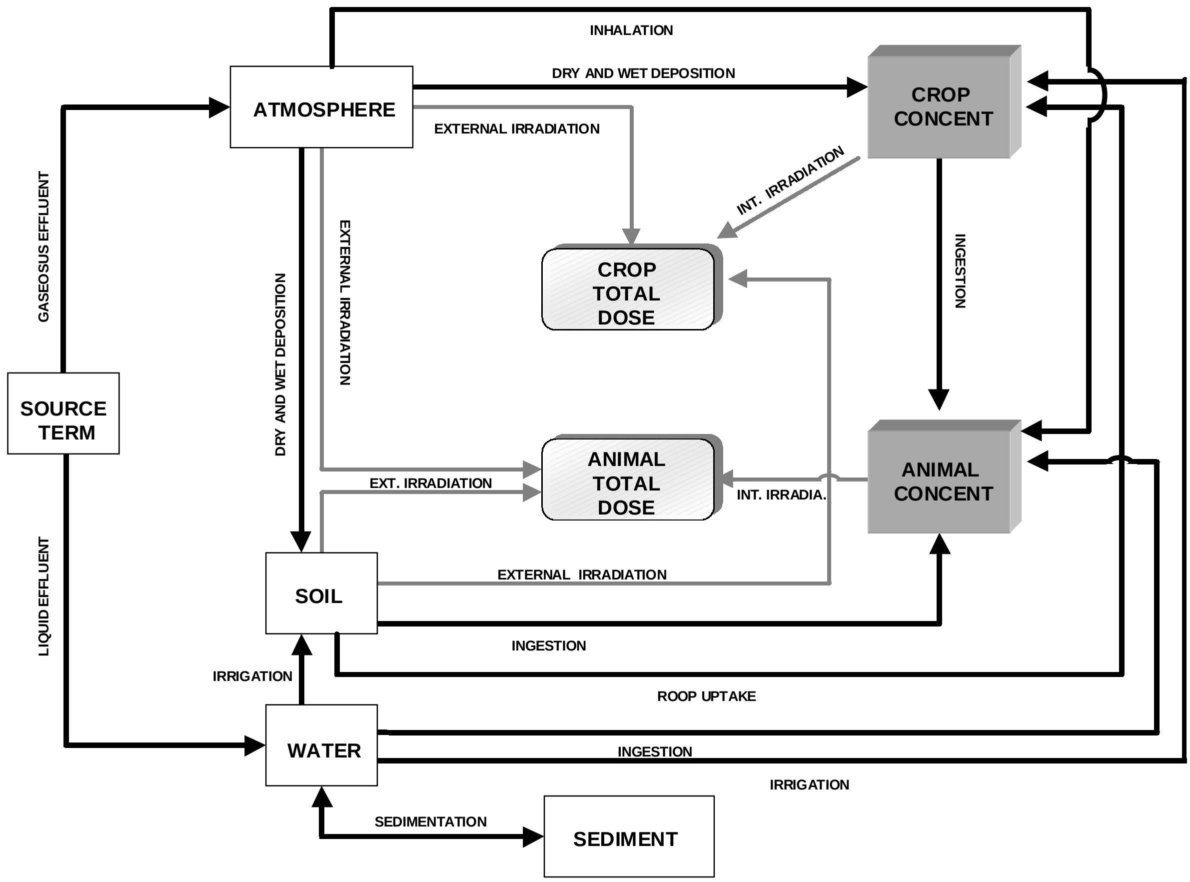

Figure 4-2 Conceptual model developed for agricultural ecosystems.

Mathematical model: Activity concentration in vegetation

Radionuclides intercepted by and retained on vegetation may result from fallout, washout, rainout, irrigation with contaminated water or deposition of resuspended matter. External deposits can be taken up by foliar absorption into plants. Radionuclides may also be incorporated by uptake from the soil via roots, followed by internal redistribution of radionuclides within the plant. Processes that can lead to the reduction of radionuclide concentration in vegetation include radioactive decay, growth dilution, wash-off of externally deposited radionuclides, leaching and soil fixation. Further removal of radioactive material from vegetation can occur due to grazing, harvesting, etc.

Starting from the estimated concentrations in air, water and soil, the assessment concentration for each radionuclide is made using a common mathematical expression for every vegetable type. The parameter's specificity determine the radionuclide concentration of different crop types (root vegetables, fruit vegetables, leafy vegetables, cereals and fruits). The general expression is [Robles *et al*., 2000]:

$$C_{i,veg,hum} = \left[\frac{\dot{d}_i\ \alpha_1 (1-\exp(-\lambda_{ief}^{v}\ t_e))}{\lambda_{ief}^{v}} + \frac{\dot{d}_i\ (1-\exp(-\lambda_{ief}^{s}\ t_b))\ F_{1vi}}{\rho\ \lambda_{ief}^{s}} \right] \exp(-\lambda_i\ t_h)$$

$$\lambda_{ief}^{v} = \lambda_i + \lambda_w$$

$$\lambda_{ief}^{s} = \lambda_i + \lambda_s$$

$$\dot{d}_i = d_i + C_{i,w}\ I_w$$

Where:

Symbol	**Units**	**Description**
$C_{i,veg,hum}$	Bq kg^{-1}	Radionuclide vegetable concentration
$Ci,_{w}$	Bq m^{-3}	Radionuclide water concentration
d_i	Bq m^{-2} d^{-1}	Atmospheric deposition rate
$\dot{d}_i$	Bq m^{-2} d^{-1}	Deposition rate modify for irrigation
F_{1vi}	---	Soil to plant transfer factor
I_w	m^3 m^{-2} d^{-1}	Irrigation rate
t_b	d	Duration of the discharge of radioactive material
t_e	d	Time period that crops are exposed during the growing season
t_h	d	Delay time between harvest and food consumption
α_1	m^2 kg^{-1}	Interception factor
λ_i	s^{-1}	Constant for radioactive decay
λ_{ief}^{s}	s^{-1}	Effective rate constant for reduction of the activity concentration in the root zone of soils
λ_{ief}^{v}	s^{-1}	Effective rate constant for reduction of the activity concentration from crops
λ_{si}	s^{-1}	Rate constant for reduction of the concentration of material deposited in the root zone of soils due to processes other than radioactive decay
λ_{wi}	s^{-1}	Rate constant for reduction of the concentration of material deposited on the plant surface due to processes other than radioactive decay
ρ	kg m^{-2}	Surface density for the effective root zone in soils

Mathematical model: Activity concentration in animals

The intake of radionuclides by animals depends on animal species, mass, age and growth rate of the animal, the digestibility of feed, and, in the case of lactating animals, the milk yield.

The radionuclide concentrations in meat, milk and eggs are calculated using the following expressions:

$$C_{i,meat} = \left[\sum (F_{i,veg,meat,anim}\, C_{i,veg,anim}\, M_{veg,anim}) + F_{i,water,meat}\, C_{i,w}\, V_{w,anim} \right] \exp(-\lambda_i\, t_{meat})$$

$$C_{i,milk} = \left[\sum (F_{i,veg,milk,anim}\, C_{i,veg,anim}\, M_{veg,anim}) + F_{i,water,milk} C_{i,w}\, V_{w,anim} \right] \exp(-\lambda_i\, t_{milk})$$

$$C_{i,egg} = \left[\sum (F_{i,veg,egg,anim}\, C_{i,veg,anim}\, M_{veg,anim}) + F_{i,water,egg}\, C_{i,w}\, V_{w,anim} \right] \exp(-\lambda_i\, t_{egg})$$

Where:

Symbol	**Units**	**Description**
$C_{i,meat}$	Bq kg^{-1}	Radionuclide concentration in meat
$C_{i,milk}$	Bq L^{-1}	Radionuclide concentration in milk
$C_{i,egg}$	Bq kg^{-1}	Radionuclide concentration in eggs
$C_{i,veg,anim}$	Bq kg^{-1}	Radionuclide concentration in vegetables
C_{iw}	Bq m^{-3}	Radionuclide concentration in water
d_i	Bq m^{-2} d^{-1}	Atmospheric deposition rate
$\dot{d}_i$	Bq m^{-2} d^{-1}	Deposition rate modify for irrigation
$F_{i,water,meat}$	d kg^{-1}	Fraction of the animal's water intake of a radionuclide that appear sin meat at equilibrium
$F_{i,water,milk}$	d L^{-1}	Fraction of the animal's water intake of a radionuclide that appears in milk at equilibrium
$F_{i,water,egg}$	d kg^{-1}	Fraction of the animal's water intake of a radionuclide that appears in eggs at equilibrium
$F_{i,veg,meat,anim}$	d kg^{-1}	Fraction of the animal's daily intake of a radionuclide that appears in meat at equilibrium
$F_{i,veg,milk,anim}$	d L^{-1}	Fraction of the animal's daily intake of a radionuclide that appears in milk at equilibrium
$F_{i,veg,egg,anim}$	d kg^{-1}	Fraction of the animal's daily intake of a radionuclide that appears in eggs at equilibrium
F_{2vi}	---	Concentration factor for uptake of the radionuclide from soil by edible parts of crops
I_w	m^3 m^{-2} d^{-1}	Irrigation rate

$M_{veg,anim}$	kg d^{-1}	Amount of feed consumed by the animal per day
$V_{w,anim}$		Amount of water consumed by the animal per day
t_b	d	Duration of the discharge of radioactive material
t_e	d	Time period that crops are exposed during the growing season
t_h	d	Delay time between harvest and vegetable consumption
t_{meat}	d	Average time between slaughter and consumption
t_{milk}	d	Average time between collection and consumption of milk
t_{egg}	d	Average time between collection and consumption of eggs
α_2	m^2 kg^{-1}	Interception factor
λ_i	s^{-1}	Constant for radioactive decay
λ_{ief}^{s}	s^{-1}	Effective rate constant for reduction of the activity concentration in the root zone of soils
λ_{ief}^{v}	s^{-1}	Effective rate constant for reduction of the activity concentration from corps
λ_{si}	s^{-1}	Rate constant for reduction of the concentration of material deposited in the root zone of soils due to processes other than radioactive decay
λ_{wi}	s^{-1}	Rate constant for reduction of the concentration of material deposited on the plant surface due to processes other than radioactive decay
ρ	kg m^{-2}	Surface density for the effective root zone in soils

4.2.3 Illustrative cases

The illustrative calculations carried out from atmospheric and aquatic releases, starting from a ten years constant source term to ensure the equilibrium. The radionuclides, which constitute the source term, are included in the suggested list that appears in Table 1-1 in FASSET Deliverable 1, ‘Identification of candidate reference organisms from a radiation exposure pathways perspective’. The relative importance of different exposure pathways and radionuclides was determined for units released (1 Bq/m^3 in air and water respectively) of individual radioisotopes.

Two different release scenarios have been accomplished, one of them considering an atmospheric release and the second one has been realised taking into account an aquatic release. Tables 4-9 and 4-13 show the selected radionuclides for the atmospheric and aquatic release scenario respectively.

In the atmospheric scenario, the effects of the wet and dry deposition have been considered to calculate soil concentration. The activity concentration of radionuclides on ground has been calculated using a total deposition coefficient of 1.0E+03 m/d, this value is normally recommended for screening purposes [IAEA, 2000].

In the aquatic scenario, two pathways have been taken into account, the irrigation for plants and the water ingestion for animals.

Soil to plant transfer factors values are shown in Table 4-6, the selected values used in these assessments are underlined in this table. In the same way the equilibrium transfer factors for animal are showed in Table 4-7. In Table 4-8 the diets used to calculate the concentration in different animals are presented.

Table 4-8 Ingestion rates used in the illustrative calculations.

Animal	Ingestion rates (kg d^{-1} or L d^{-1})					
	Water	Fodder (*)	Pasture	Grain	Root	Milk
Meat cow	40	8	30			
Milk cow	60	10	30			
Pork	5			3	2	3
Meat sheep	6		8			
Milk sheep	6		8			
Poultry-Eggs	0.1			0.1		

(*) Dry weight .

The results presented here are activity concentrations in crops and farm animals and have been calculated using the CROM code (CROM V1.0) developed by CIEMAT [Suañes & Robles, 1998], following the IAEA methodology [IAEA, 2000].

Case 1 – Atmospheric scenario

The source term implemented in the code for each radionuclide is show in Table 4-9. Air concentration is 1 Bq/m^3 and water concentration is taken as zero.

Table 4-9 Source term.

Radionuclide	Air concentration (Bq/m^3)	Water concentr. (Bq/m^3)
Am-241	1.0E+00	0.0E+00
Cm-242	1.0E+00	0.0E+00
Cm-244	1.0E+00	0.0E+00
Cs-134	1.0E+00	0.0E+00
Cs-137	1.0E+00	0.0E+00
I-129	1.0E+00	0.0E+00
I-131	1.0E+00	0.0E+00
Ni-59	1.0E+00	0.0E+00
Ni-63	1.0E+00	0.0E+00
Np-237	1.0E+00	0.0E+00
Pb-210	1.0E+00	0.0E+00
Pu-238	1.0E+00	0.0E+00
Pu-239	1.0E+00	0.0E+00
Pu-240	1.0E+00	0.0E+00
Pu-241	1.0E+00	0.0E+00
Ra-226	1.0E+00	0.0E+00
Ru-106	1.0E+00	0.0E+00
Sr-89	1.0E+00	0.0E+00
Sr-90	1.0E+00	0.0E+00
Tc-99	1.0E+00	0.0E+00
Th-228	1.0E+00	0.0E+00
Th-230	1.0E+00	0.0E+00
Th-232	1.0E+00	0.0E+00
U-234	1.0E+00	0.0E+00
U-235	1.0E+00	0.0E+00
U-238	1.0E+00	0.0E+00

Results for the atmospheric scenario

The activity concentrations have been obtained for the atmospheric releases in different types of crops and they are shown in the Table 4-10.

Table 4-11 summarises the activity concentration in animal feed. These results have been used to estimate the activity concentration in the different animal species according to the mathematical model described in Section 4.3.3. The animal concentrations are shown in Table 4-12.

Table 4-10 Vegetable concentrations (Bq/kg).

Radionuclide	Leafy veg.	Fruit veg.	Roots	Fruits	Grain
Am-241	5.701E+03	5.933E+03	5.933E+03	6.012E+03	5.985E+03
Cm-242	5.293E+03	1.824E+04	5.464E+03	5.498E+03	5.498E+03
Cm-244	5.691E+03	5.921E+03	5.921E+03	5.983E+03	5.972E+03
Cs-134	5.679E+03	5.896E+03	5.861E+03	5.943E+03	5.943E+03
Cs-137	5.893E+03	6.124E+03	6.024E+03	6.175E+03	6.175E+03
I-129	7.105E+03	7.337E+03	7.337E+03	7.388E+03	7.388E+03
I-131	2.024E+03	2.024E+03	2.024E+03	2.024E+03	2.024E+03
Ni-59	5.841E+03	6.073E+03	6.073E+03	1.019E+04	6.125E+03
Ni-63	5.834E+03	6.066E+03	6.066E+03	1.004E+04	6.118E+03
Np-237	5.729E+03	5.961E+03	5.947E+03	6.546E+03	6.013E+03
Pb-210	5.813E+03	6.043E+03	6.043E+03	6.215E+03	6.095E+03
Pu-238	5.700E+03	5.932E+03	5.932E+03	5.983E+03	5.983E+03
Pu-239	5.702E+03	5.934E+03	5.934E+03	5.986E+03	5.986E+03
Pu-240	5.702E+03	5.934E+03	5.934E+03	5.986E+03	5.986E+03
Pu-241	5.694E+03	5.925E+03	5.925E+03	5.976E+03	5.976E+03
Ra-226	5.841E+03	6.073E+03	5.947E+03	6.545E+03	5.998E+03
Ru-106	5.535E+03	5.737E+03	5.737E+03	5.780E+03	5.780E+03
Sr-89	4.651E+03	4.737E+03	4.710E+03	4.668E+03	4.696E+03
Sr-90	9.646E+03	9.877E+03	8.889E+03	6.964E+03	7.952E+03
Tc-99	7.589E+04	7.612E+04	7.612E+04	7.617E+04	7.617E+04
Th-228	5.608E+03	5.824E+03	5.824E+03	5.873E+03	5.889E+03
Th-230	5.708E+03	5.940E+03	5.940E+03	5.999E+03	5.992E+03
Th-232	5.708E+03	5.940E+03	5.940E+03	5.999E+03	5.992E+03
U-234	5.715E+03	5.947E+03	5.947E+03	6.125E+03	5.999E+03
U-235	5.715E+03	5.947E+03	5.947E+03	6.125E+03	5.999E+03
U-238	5.715E+03	5.947E+03	5.947E+03	6.125E+03	5.999E+03

Table 4-11 Animal feed concentrations (Bq/kg).

Radionuclide	Fodder (*)	Pasture	Roots	Grain
Am-241	6.069E+04	4.688E+03	5.931E+03	5.982E+03
Cm-242	3.748E+04	4.426E+03	3.742E+03	3.765E+03
Cm-244	5.981E+04	4.678E+03	5.866E+03	5.916E+03
Cs-134	5.693E+04	4.962E+03	5.399E+03	5.475E+03
Cs-137	6.877E+04	5.648E+03	5.991E+03	6.140E+03
I-129	6.073E+04	7.468E+03	7.337E+03	7.388E+03
I-131	9.402E+00	1.994E+03	0.941E+00	0.941E+00
Ni-59	7.337E+04	4.941E+03	6.073E+03	6.125E+03
Ni-63	7.272E+04	4.930E+03	6.055E+03	6.107E+03
Np-237	6.635E+04	4.941E+03	5.947E+03	6.013E+03
Pb-210	5.998E+04	4.897E+03	5.998E+03	6.049E+03
Pu-238	6.323E+04	4.665E+03	5.920E+03	5.972E+03
Pu-239	6.354E+04	4.666E+03	5.934E+03	5.986E+03
Pu-240	6.354E+04	4.666E+03	5.934E+03	5.986E+03
Pu-241	6.246E+04	4.661E+03	5.884E+03	5.935E+03
Ra-226	6.492E+04	4.941E+03	5.946E+03	5.998E+03
Ru-106	4.871E+04	4.636E+03	4.853E+03	4.889E+03
Sr-89	1.444E+04	4.230E+03	1.388E+03	1.384E+03
Sr-90	1.571E+05	1.453E+04	8.837E+03	7.906E+03
Tc-99	1.182E+06	1.450E+05	7.612E+04	7.617E+04
Th-228	5.388E+04	4.610E+03	5.353E+03	5.397E+03
Th-230	6.073E+04	4.675E+03	5.940E+03	5.992E+03
Th-232	6.073E+04	4.675E+03	5.940E+03	5.992E+03
U-234	6.214E+04	4.689E+03	5.947E+03	5.999E+03
U-235	6.214E+04	4.689E+03	5.947E+03	5.999E+03
U-238	6.214E+04	4.689E+03	5.947E+03	5.999E+03

(*) Bq/kg dry weight.

Table 4-12 Animal concentrations (Bq/kg).

Radio-nuclide	Cattle meat	Sheep meat	Pork	Poultry	Cow milk	Sheep milk	Eggs
Am-241	6.262E+01	1.500E+01	5.815E+00	3.589E+00	0.747E+00	0.525E+00	2.393E+00
Cm-242	8.617E+00	1.410E+01	---	---	0.505E+02	---	---
Cm-244	1.237E+01	1.496E+01	---	---	0.738E+00	---	---
Cs-134	3.018E+04	1.983E+04	1.011E+04	5.470E+03	3.587E+03	2.300E+03	2.188E+02
Cs-137	3.598E+04	2.259E+04	1.158E+04	6.140E+03	4.286E+03	2.620E+03	2.456E+02
I-129	3.549E+04	2.987E+03	4.278E+03	7.388E+00	4.157E+03	2.927E+04	2.216E+03
I-131	2.748E+03	7.319E+02	2.522E+02	8.641E-04	2.749E+02	7.173E+03	0.259E+00
Ni-59	3.676E+04	---	---	---	1.411E+04	---	---
Ni-63	3.648E+04	---	---	---	1.400E+04	---	---
Np-237	6.798E+03	1.581E+01	---	---	0.811E+00	---	---
Pb-210	4.387E+02	3.917E+02	---	---	2.240E+02	---	---
Pu-238	1.291E+02	1.492E+01	3.152E+00	1.791E+00	0.772E+00	0.350E+00	0.298E+00
Pu-239	1.296E+02	1.493E+01	3.161E+00	1.795E+00	0.775E+00	0.350E+00	0.299E+00
Pu-240	1.296E+02	1.493E+01	3.163E+00	1.795E+00	0.775E+00	0.350E+00	0.299E+00
Pu-241	1.279E+02	1.491E+01	3.130E+00	1.780E+00	0.764E+00	0.350E+00	0.296E+00
Ra-226	3.338E+03	1.976E+02	---	---	3.190E+02	---	---
Ru-106	2.638E+04	3.701E+02	1.605E+04	3.904E+03	0.625E+00	3.701E+01	2.404E+00
Sr-89	2.392E+03	1.001E+02	8.015E+02	1.092E+01	5.354E+02	1.869E+03	2.730E+01
Sr-90	1.693E+04	3.488E+02	5.670E+03	6.325E+01	4.014E+03	6.511E+03	1.581E+02
Tc-99	1.381E+04	1.160E+05	1.618E+05	2.285E+02	1,617E+05	---	2.285E+04
Th-228	5.688E+01	3.685E+01	---	---	3.382E+00	---	---
Th-230	6.261E+01	3.740E+01	---	---	3.738E+00	---	---
Th-232	6.261E+01	3.740E+01	---	---	3.738E+00	---	---
U-234	1.913E+03	7.502E+01	2.310E+03	5.999E+02	4.572E+02	---	5.999E+02
U-235	1.913E+03	7.502E+01	2.310E+03	5.999E+02	4.572E+02	---	5.999E+02
U-238	1.913E+03	7.502E+01	2.310E+03	5.999E+02	4.572E+02	---	5.999E+02

--- Transfer factor values not available.

Case 2 – Aquatic scenario

In this case only the water concentration is taken as source term (see Table 4-13).

Table 4-13 Source term.

Radionuclide	Air concentration (Bq/m^3)	Water concentr. (Bq/m^3)
Am-241	0.0E+00	1.0E+00
Cm-242	0.0E+00	1.0E+00
Cm-244	0.0E+00	1.0E+00
Cs-134	0.0E+00	1.0E+00
Cs-137	0.0E+00	1.0E+00
I-129	0.0E+00	1.0E+00
I-131	0.0E+00	1.0E+00
Ni-59	0.0E+00	1.0E+00
Ni-63	0.0E+00	1.0E+00
Np-237	0.0E+00	1.0E+00
Pb-210	0.0E+00	1.0E+00
Pu-238	0.0E+00	1.0E+00
Pu-239	0.0E+00	1.0E+00
Pu-240	0.0E+00	1.0E+00
Pu-241	0.0E+00	1.0E+00
Ra-226	0.0E+00	1.0E+00
Ru-106	0.0E+00	1.0E+00
Sr-89	0.0E+00	1.0E+00
Sr-90	0.0E+00	1.0E+00
Tc-99	0.0E+00	1.0E+00
Th-228	0.0E+00	1.0E+00
Th-230	0.0E+00	1.0E+00
Th-232	0.0E+00	1.0E+00
U-234	0.0E+00	1.0E+00
U-235	0.0E+00	1.0E+00
U-238	0.0E+00	1.0E+00

Results for the aquatic scenario

Table 4-14 summarises the activity concentration in leafy vegetables, fruit vegetables, roots, fruits and grain.

Table 4-15 shows the results for the animal feed concentrations due only to the irrigation process. Pastures are not irrigated, so the concentration is zero in this case.

From the model shown in Section 4.3.3, using the intake rates in Table 4-8, flesh animals concentrations are calculated (see Table 4-16).

Table 4-14 Vegetable concentrations (Bq/kg).

Radionuclide	Leafy veg.	Fruit veg.	Roots	Fruits	Grain
Am-241	1.710E-02	1.780E-02	1.780E-02	1.804E-02	1.197E-02
Cm-242	1.588E-02	5.474E-02	1.639E-02	1.650E-02	1.100E-02
Cm-244	1.707E-02	1.776E-02	1.776E-02	1.795E-02	1.194E-02
Cs-134	1.704E-02	1.769E-02	1.758E-02	1.783E.02	1.189E.02
Cs-137	1.768E-02	1.837E-02	1.807E-02	1.853E-02	1.235E-02
I-129	2.132E-02	2.201E-02	2.201E-02	2.217E-02	1.478E-02
I-131	6.072E-03	6.074E-03	6.074E-03	6.074E-03	4.049E-03
Ni-59	1.752E-02	1.822E-02	1.822E-02	3.059E-02	1.225E-02
Ni-63	1.750E-02	1.820E-02	1.820E-02	3.014E-02	1.124E-02
Np-237	1.719E-02	1.788E-02	1.784E-02	1.964E-02	1.203E-02
Pb-210	1.744E-02	1.813E-02	1.813E-02	1.865E-02	1.219E-02
Pu-238	1.710E-02	1.780E-02	1.780E-02	1.795E-02	1.197E-02
Pu-239	1.711E-02	1.780E-02	1.780E-02	1.796E-02	1.197E-02
Pu-240	1.711E-02	1.780E-02	1.780E-02	1.796E-02	1.197E-02
Pu-241	1.708E-02	1.778E-02	1.778E-02	1.793E-02	1.195E-02
Ra-226	1.752E-02	1.822E-02	1.784E-02	1.964E-02	1.200E-02
Ru-106	1.661E-02	1.721E-02	1.721E-02	1.734E-02	1.156E-02
Sr-89	1.395E-02	1.421E-02	1.413E-02	1.401E-02	9.392E-03
Sr-90	2.894E-02	2.963E-02	2.667E-02	2.089E-02	1.591E-02
Tc-99	2.277E-01	2.284E-01	2.284E-01	2.285E-01	1.524E-01
Th-228	1.682E-02	1.747E-02	1.747E-02	1.762E-02	1.178E-02
Th-230	1.712E-02	1.782E-02	1.782E-02	1.800E-02	1.198E-02
Th-232	1.712E-02	1.782E-02	1.782E-02	1.800E-02	1.198E-02
U-234	1.715E-02	1.784E-02	1.784E-02	1.838E-02	1.200E-02
U-235	1.715E-02	1.784E-02	1.784E-02	1.838E-02	1.200E-02
U-238	1.715E-02	1.784E-02	1.784E-02	1.838E-02	1.200E-02

Table 4-15 Animal feed concentrations (Bq/kg).

Radionuclide	Fodder (*)	Pasture	Roots	Grain
Am-241	0.188E+00	0.000E+00	1.779E-02	1.196E-02
Cm-242	0.112E+00	0.000E+00	1.122E-02	7.531E-03
Cm-244	0.179E+00	0.000E+00	1.759E-02	1.183E-02
Cs-134	1.170E+00	0.000E+00	1.619E-02	1.095E-02
Cs-137	0.206E+00	0.000E+00	1.797E-02	1.228E-02
I-129	0.182E+00	0.000E+00	2.201E-02	1.447E-02
I-131	2.820E-05	0.000E+00	2.825E-06	1.883E-06
Ni-59	0.220E+00	0.000E+00	1.822E-02	1.225E-02
Ni-63	0.218E+00	0.000E+00	1.816E-02	1.221E-02
Np-237	1.199E+00	0.000E+00	1.784E-02	1,202E-02
Pb-210	0.179E+00	0.000E+00	1.799E-02	1.209E-02
Pu-238	1.189E+00	0.000E+00	1.776E-02	1.194E-02
Pu-239	0.190E-00	0.000E+00	1.780E-02	1.197E-02
Pu-240	0.190E-00	0.000E+00	1.780E-02	1.197E-02
Pu-241	1.874E-01	0.000E+00	1.765E-02	1.187E-02
Ra-226	0.194E+00	0.000E+00	1.783E-02	1.199E-02
Ru-106	0.146E+00	0.000E+00	1.456E-02	9.779E-03
Sr-89	4.334E-02	0.000E+00	4.165E-03	2.768E-03
Sr-90	0.471E+00	0.000E+00	2.651E-02	1.581E-02
Tc-99	3.547E+00	0.000E+00	0.228E+00	0.152E+00
Th-228	0.161E+00	0.000E+00	1.606E-02	1.079E-02
Th-230	0.182E+00	0.000E+00	1.782E-02	1.198E-02
Th-232	0.182E+00	0.000E+00	1.782E-02	1.198E-02
U-234	0.186E+00	0.000E+00	1,784E-02	1.199E-02
U-235	0.186E+00	0.000E+00	1,784E-02	1.199E-02
U-238	0.186E+00	0.000E+00	1,784E-02	1.199E-02

(*) Bq/kg dry weight.

Table 4-16 Animal concentrations (Bq/kg).

Radio-nuclide	Cattle meat	Sheep meat	Pork	Poultry	Cow milk	Sheep milk	Eggs
Am-241	1.536E-02	2.399E-06	1.579E-05	7.779E-06	1.940E-06	8.399E-08	5.186E-06
Cm-242	1.951E-05	2.389E-06	---	---	1.239E-06	---	---
Cm-244	3.030E-05	2.399E-06	---	---	1.914E-06	---	---
Cs-134	7.225E-02	2.997E-03	2.716E-02	1.194E-02	9.131E-03	3.476E-04	4.776E-04
Cs-137	8.653E-02	2.999E-03	3.078E-02	1.328E-02	1.091E-02	3.479E-04	5.312E-04
I-129	7.688E-02	3.000E-04	1.003E-02	1.577E-05	9.710E-03	2.940E-03	4.733E-03
I-131	3.679E-03	2.752E-04	5.364E-04	9.191E-07	5.517E-04	2.697E-03	2.757E-04
Ni-59	9.204E-02	---	---	---	3.713E-02	---	---
Ni-63	9.126E-02	---	---	---	3.688E-02	---	---
Np-237	1.672E-02	2.399E-06	---	---	2.110E-06	---	---
Pb-210	1.063E-03	5.999E-05	---	---	5.757E-04	---	---
Pu-238	3.195E-04	2.399E-06	8.525E-06	3.883E-06	2.017E-06	5.639E-08	6.472E-07
Pu-239	3.210E-04	2.399E-06	8.548E-06	3.891E-06	2.026E-06	5.639E-08	6.486E-07
Pu-240	3.209E-04	2.399E-06	8.548E-06	3.891E-06	2.026E-06	5.639E-08	6.486E-07
Pu-241	3.158E-04	2.400E-06	8.466E-06	3.861E-06	1.994E-06	5.640E-08	6.435E-07
Ra-226	8.191E-03	2.999E-05	---	---	8.271E-04	---	---
Ru-106	6.233E-02	5.988E-05	4.510E-02	8.607E-03	1.578E-06	5.988E-06	5.379E-06
Sr-89	4.209E-03	1.775E-05	2.127E-03	2.973E-05	1.091E-03	3.314E-04	7.734E-05
Sr-90	3.850E-02	1.799E-05	1.408E-02	1.345E-02	9.667E-03	3.359E-04	3.362E-04
Tc-99	2.845E-02	5.999E-04	0.356E+00	4.600E-04	0.355E+00	---	4.600E-02
Th-228	1.371E-04	5.994E-06	---	---	8.674E-06	---	---
Th-230	1.537E-04	5.999E-06	---	---	9.710E-06	---	---
Th-232	1.537E-04	6.000E-06	---	---	9.710E-06	---	---
U-234	4.714E-03	1.199E-05	6.254E-03	1.299E-03	1.190E-03	---	1.299E-03
U-235	4.714E-03	1.200E-05	6.254E-03	1.299E-03	1.190E-03	---	1.299E-03
U-238	4.714E-03	1.200E-05	6.254E-03	1.299E-03	1.190E-03	---	1.299E-03

--- Transfer factor values not available.

Discussion of results

It can be pointed out that the atmospheric contribution (deposition) is clearly more important than the aquatic pathway (irrigation). The later contribution depends largely on the agricultural practices, and climatic and pedologic conditions.

The most important radionuclide for both release scenarios (atmospheric and aquatic) is Tc-99 for vegetables as well as animals. Other radionuclides with high contribution to the total activity concentration are Sr-90 and I-129 for vegetables and Sr-90, Ru-106, Ni-59, Ni-63, I-129, Cs-134 and Cs-137 for animals.

There are no significant differences among vegetable types. Only fruit shows values slightly higher than the other types.

The activity concentration values in animals must be interpreted with caution, since they are largely dependent of the selected feeding for each of them. In this assessment some very simplified and general feeding data (Table 4-8) have been used. To obtain more realistic concentration values it will be necessary to improve the knowledge in this area.

5. Wetlands

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5.1 Introduction

5.1.1 Scope of this review

This report provides an overview of the types of wetland and processes pertinent to exposure to radionuclides. A more detailed description is provided for typical Fenno-Scandinavian wetlands, i.e. fens and bogs with examples from Sweden. There are several other types of wetland of importance, which are not treated here. There is an overlap concerning coastal marshes, lakes and forests with other chapters in this report and they are mainly treated in those chapters.

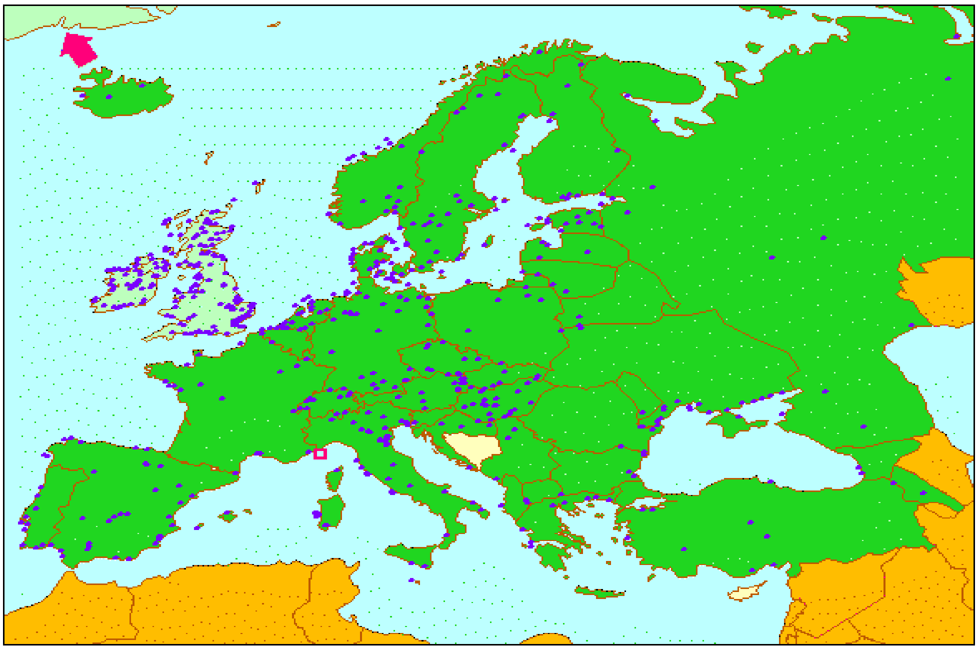

Figure 5-1 Wetlands in Europe listed on the RAMSAR Convention list. The Convention on Wetlands, signed in Ramsar, Iran, in 1971, is an intergovernmental treaty, which provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources.

5.1.2 Features of wetlands

Wetlands are among the most important ecosystems on Earth and have been for millions of years. Wetlands are valuable resources due to their extensive food chains and as a supply of fuel. Wetlands in general are widespread over Europe (Figure 5-1). They are situated at the interface between land and water, both in marine as well as freshwater areas. In inland situations they can cover large areas without any open water. Wetlands are often situated in areas where groundwater comes to the surface or downstream in the watershed. That means that large amounts of potentially contaminated groundwater pass through wetlands. In combination with the high capacity of the organic material to adsorb many radionuclides and the water seeping through the wetlands, they are efficient traps for many radionuclides. Wetlands have recently been appreciated for their ability to retain contaminants as well as stabilising the hydrological cycle and have sometimes been called the kidneys of the landscape. Moreover, wetlands are, in many cases, inhabited with a unique fauna and flora of endangered species. Thus many wetlands are designated as nature reserves and are accorded a high level of protection by international and national regulations and conventions. In spite of this, the wetland ecosystem is a widely threatened habitat, that has been exploited and destroyed by humans for thousands of years.

Wetlands, especially peat bogs, received scientific attention in the 1930's when profiles with pollen diagrams of several large peat bogs in Sweden were described and the past geology and climate of the bogs was discussed [e.g. Granlund, 1932; Du Rietz, 1949]. Starting during the Second World War and being revived in the 1970's, peat was regarded as an alternative to oil and coal fuel in Sweden and large inventories were made. Later, in the 1980's, the primary focus on wetlands related to their endangered status and in the 1990's attention focused on the wetland's role as the kidney of the landscape; as a trap of pollutants, nutrients and for flooding prevention [e.g. Folke, 1991; Brown *et al.*, 2000]. In this period, attempts were made to utilise wetlands as biological sewage treatment plants. Over the last five years, wetlands have received attention as potential sinks of carbon in the greenhouse discussion. Thus, there is a large amount of literature, from over almost a century, covering different aspects of wetlands [see reviews in Reid *et al.*, 1994; Lucisano & Bozkurt, 1998; Brown *et al.*, 2000; Mitsch & Gosselink, 2000]. However, there has been no real attempt to synthesise this knowledge in a structured way, combining the information on structure (e.g. stratigraphy, geographical extension and size) with the dynamic and underlying processes (e.g. growth, uptake and photosynthesis).

The levels and sources of natural and anthropogenic radionuclides in peat has also been investigated as well as dose assessments from the utilisation of these peat resources [Evans *et al.*, 1982; Fredriksson *et al.*, 1984; Landström & Sundblad, 1986; Wijk & Jensen, 1990; Amiro *et al.*, 1996] . Moreover, an assessment of the fate of radionuclides in wetlands has been conducted in safety assessments for radionuclide waste [Knight, 1990; Reid *et al.*, 1994; Bergström *et al.*, 1999; Nordlinder *et al.*, 1999; Karlsson *et al.*, 2001; Bergman, *in preparation*].

However, in general, the radiological assessment of wetlands has been limited in scope due to the lack of easily identifiable direct pathways to humans. This means that there is a scarcity of data.

5.2 Wetlands: definition and types

There are many definitions and many different types of wetland. With respect to the Nordic countries, some definitions are given below.

5.2.1 Definitions

Mitsch & Gosselink [2000] have conducted a long discussion on different definitions of wetlands which has changed with time. A very broad definition is that from the RAMSAR Convention (see Figure 5-1 for explanation of the Convention):

‘Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt including areas of marine water, the depth of which at low tide does not exceed 6 meters.’

The definition from the US National Academy of Sciences is:

‘A wetland is an ecosystem that depends on constant or recurrent, shallow inundation or saturation at or near the surface of the substrate. The minimum essential characteristics of a wetland are recurrent, sustained inundation or saturation at or near the surface and the presence of physical, chemical, and biological features reflective or recurrent, sustained inundation or saturation. Common diagnostic features of wetlands are hydric soils and hydrophytic vegetation. These features will be present except where specific physiochemical, biotic or anthropogenic factors have removed them or prevented their development.’

5.2.2 Coastal marshes

The coastal wetlands are one of the most important ecosystems due to their high productivity and the fact that they cover large parts of the coastal areas in the world. A dominant type is mangrove forest, which is not existent in Europe. However, coastal marshes and river deltas are present along the European coast. Marshes do not accumulate peat and are thus based on mineral soil.

Tidal marshes

Tidal marshes are grassy, wet, periodically inundated shallow areas. The predominant vegetation is rushes, reeds, reedgrasses and sedges and in open water areas, submerged and floating plants [Reid *et al.*, 1994]. Along the Baltic Sea there are several large coastal marshes (e.g. Matsalu bay).

River deltas

There are several wetlands in the river deltas of the non-glaciated parts of Europe. The delta of the Rhone River, Carmargue, is one of the largest. It has very rich fauna and flora with many species exhibiting their only occurrence in Europe, e.g. flamingos. The Ebro delta is another large area of diversity. The Rhine River delta is highly managed and constitutes a large part of the Netherlands. It consists also of a large part of the tidally influenced mudflats of the Wadden See.

5.2.3 Mires

Mires are wetlands with a vegetation which usually forms peat [Rydin *et al.*, 1999]. Peat formation is possible where the water table is close to the surface for most of the time. More than 20 % of Sweden's total area is covered by peat soil (depth > 0.3 m) and half of this area consists of natural to semi-natural mires [Rydin *et al.*, 1999].

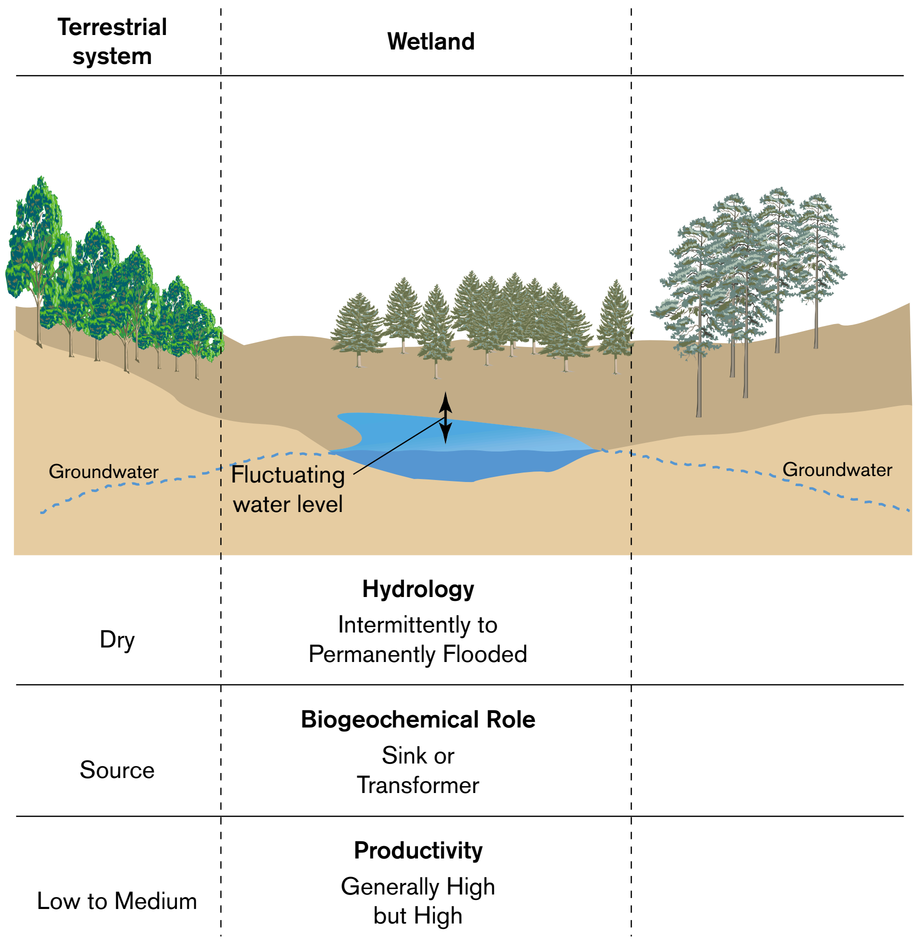

Figure 5-2 The processes in mires; the important hydrological, biogeochemical and productivity states and processes. Mires accumulate peat. 'Groundwater fed mires' are fens while 'rainwater fed mires' are raised bogs. Modified from Rydin et al. [1999] and Mitsch & Gosselink [2000].

Bogs

Bogs are mires where the upper peat layer and the local water table is above the water table of the surrounding mineral soil (Figure 5-6E). Thus, all water, nutrients and contaminants are supplied by rainfall and air deposition. They are often called ombrotrophic (rainfed) mires or raised bogs. The Sphagnum moss and the rainwater usually contribute to the low pH of bogs.

Fens

Fens are mires that receive at least *some* water that has passed through the mineral soil, i.e. groundwater (Figure 5-2). Thus they are richer in cations such as calcium, potassium and iron. That means that they are also a potential receiver of radionuclides (both natural and from e.g. a below ground repository).

5.2.4 Riparian wetlands

Riparian wetlands are defined as ecosystems in which soils are influenced by adjacent stream rivers or lakes [Mitsch & Gosselink, 2000]. Flooding can periodically influence them. The major subdivisions are shown in Figure 5-3.

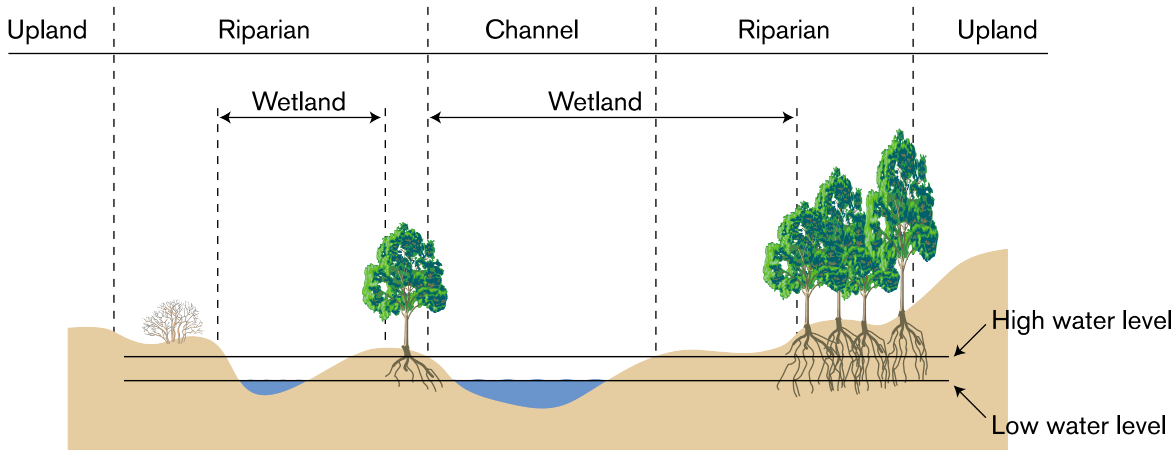

Figure 5-3 Riparian wetlands. Wetland periodically flooded by adjacent rivers or lakes, cf. Figure 5-2. Modified from Mitsch & Gosselink [2000].

5.2.5 Swamps

Swamps are wooded or reed dominated wetlands that seasonally, or for longer periods, have slow flowing or standing water at the surface. They have no accumulation of peat (cf. mires above).

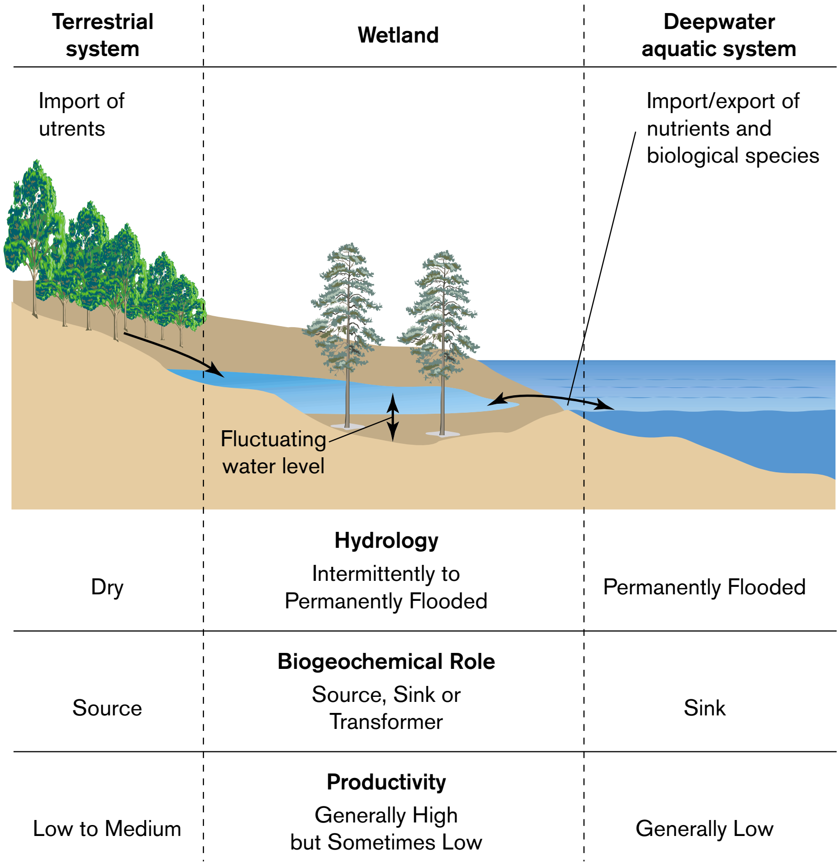

Figure 5-4 Swamps, wooded or reed dominated wetlands that periodically have an open surface of water, cf. Figure 5-2. Modified from Mitsch & Gosselink [2000].

5.3 Processes in wetlands

The processes in wetlands and their normal operating ranges are extensively described in Mitsch & Gosselink [2000] and Reid *et al.* [1994]. Below, an overview is presented.

5.3.1 Short term processes in wetlands

The major factors affecting wetlands are shown in Figure 5-5. External factors are the physical structure (geomorphology) and climate. The hydrology of the wetland, physico-chemical and biological factors interact internally. The physical structure and the climate affect the hydrology by affecting the basin volume (cf. long term processes), runoff, transpiration and gradient in the system. The hydrology strongly affects the physical and chemical environment mainly by the transport or restriction of transport of important nutrients or oxygen and carbon dioxide. This affects the biological environment by providing substrates only to organisms that are good competitors in this environment. The biological processes, such as photosynthesis, decomposition and exudation, affect the physico-chemical environment, e.g. oxygen depletion, pH, organic acids, as well as the hydrology by changing conductivity, evaporation and gradients.

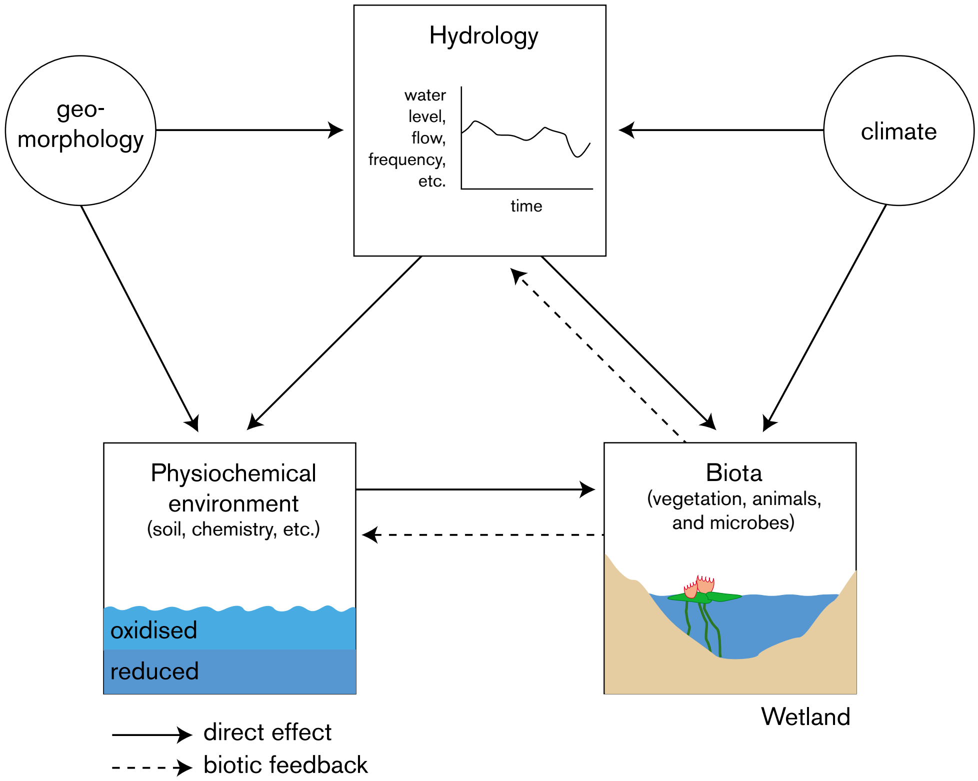

Figure 5-5 The interaction between important processes characterising wetlands. Modified from Mitsch & Gosselink [2000].

5.3.2 Long term processes – natural succession of mires

Sedimentation, erosion and peat accumulation processes usually affect wetlands. For this reason wetlands are more or less temporal in the time-scales of some 1 000 years, i.e. in the time-scale of performance assessment. An example is the development of a coastal area in Sweden where the land-rise process isolates bays from the sea which gradually then develops from lakes to mire as shown in Figure 5-6.

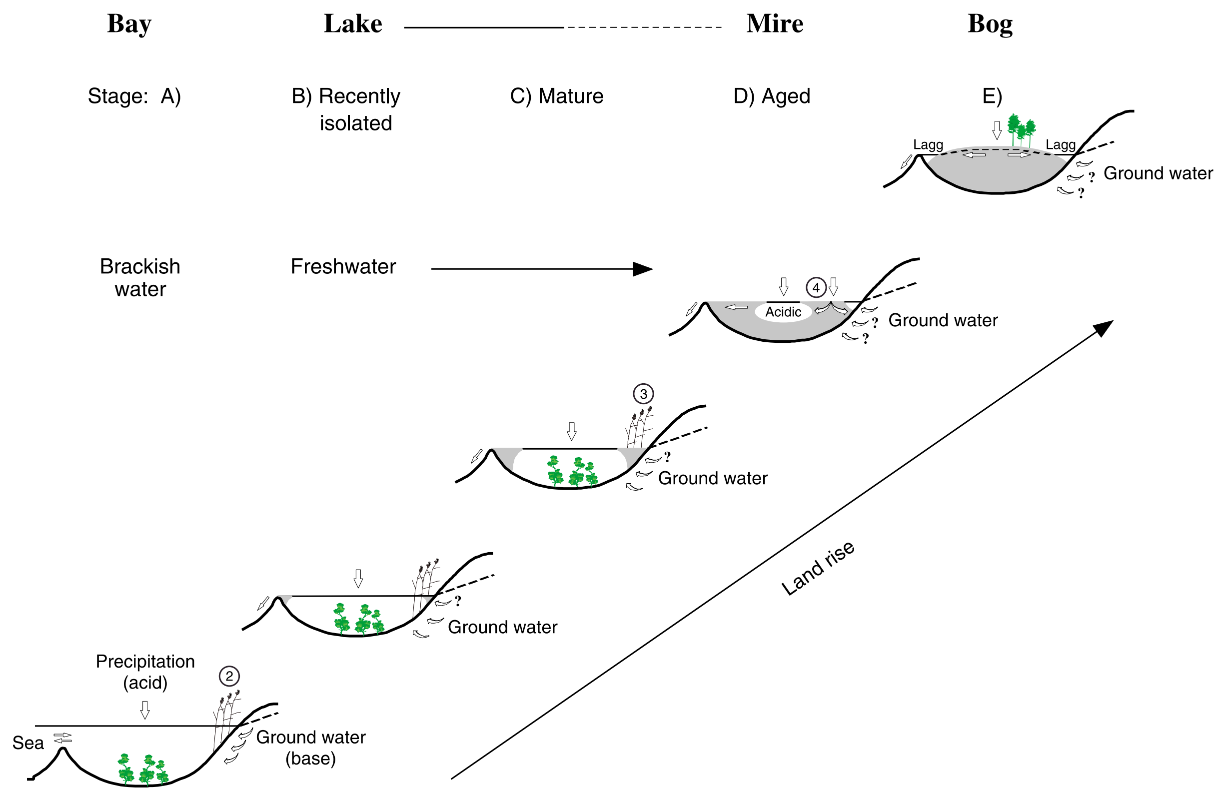

Figure 5-6 Suggested ontogeny of the oligotrophic hardwater lakes in the Forsmark area, from Brunberg & Blomqvist [2000]. Arrows indicate main water flows. The numbers in the figure represent different major components of the ecosystem: 1 = Chara meadow, 2 = Phragmites littoral, 3 = mire/floating-mat littoral, 4 = Sphagnum littoral.

5.4 Major and critical organisms

Wetlands are usually in the transitional zone between waterbodies (rivers, lakes or seas) and land and thus share many of the major and critical organisms of these ecosystems (e.g lake, marine and brackish waters, pasture and heathlands). In this section, only those organisms that are associated with mires are covered.

Usually mires are almost lacking in fauna, because of the oxygen depletion, the low pH and the fact that peat is antiseptic in nature. In open water, however, insects (e.g. mosquitoes and blackflies), and sometimes amphibians (mainly frogs) can survive. The bird fauna is diverse, however they are dependent on food mainly from the vegetation in the mires (e.g. insects, fruits and seeds). Moreover they are migratory and spend a limited time on mires and other parts of the year in mudflats or swamps in other countries. Mammals on mires are sparse; rodents (e.g. voles) can live in dryer parts. Moose and roe deer frequently pass through mires but also utilise surrounding areas.

Vegetation is usually dominated by mosses (especially *Spaghnum* mosses), grasses, shrubs and trees. In calcium rich fens the vegetation can have a higher diversity including orchids and other plants.

Thus, the organisms that can be subjected to high doses and are sensitive to radioactivity are probably amphibians, mosses, reeds and grasses, while rodents and other mammals are general for other ecosystems (e.g. forest) as are birds (e.g. marshes).

5.5 Radionuclides in wetlands

Many radionuclides have the ability to accumulate in wetlands due to the fact that water transport is channelled through this environment with sharp chemical gradients and that large amounts of organic matter are produced or accumulated.

Table 5-1 Contents of some radionuclides in natural peat [Wijk & Jensen, 1990, * from Evans *et al.*, 1982].

Peat	Bq/kg dw		Ash	Bq/kg ash
	max	mean	max	mean
U	4 500	50*	90 000	1 000
238-U				160
235-U				8,2
Ra	328 000	21*	6 600 000	
226-Ra		6,4		117
228-Ra		2,5		46
Th	30	8	560	160
228-Th		2,1		44
210-Pb		84		970
210-Po		60		1 200
137-Cs	2 200	500	43 500	10 000

5.5.1 Natural accumulation

In Swedish mires the natural concentration of uranium can be up to 7 500 ppm in ashed peat, while the average concentration is 70 ppm with a median concentration of 34 ppm, which corresponds to a maximum of 90 kBq/kg ash [Wijk & Jensen, 1990]. The content of some radionuclides in Swedish and Finnish peatbogs is listed in Table 5-1.

5.5.2 Assessments of a mire model

A mire model, among many other ecosystems, was developed for the safety assessment of high-level and intermediate level waste repositories in Sweden [SKB, 1999a; 1999b]. The model is described in Bergström *et al.* [1999] and site specific data model conversion factors are calculated for about 40 radionuclides at three sites [Nordlinder *et al.*, 1999]. The main purpose was to assess doses to humans but some of the major, albeit simplified pathways were used. In the safety assessment of SFR, the sub-seabed repository for low- and intermediate-level operational waste at the Forsmark NPP, the models were reviewed primarily to account for time dependent discharges [Karlsson *et al.*, 2001]. The structure of the mire model is shown in Figure 5-7. In the assessment, the mire is assumed to be exploited and a fraction of the contaminated peat is used as fuel in a household. Another fraction is drained and used for farming purposes. The crop grown on the peat is then assumed to be used as food for humans as well as cattle. Thus no exposure to other biota is estimated, but the concentrations in the two compartments are calculated [Karlsson, *in preparation*].

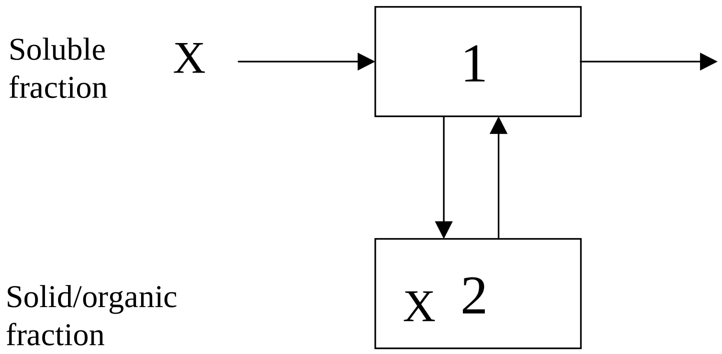

Figure 5-7 Structure of the mire model. The model consists of two compartments; one for the soluble fraction and one for the solid/organic fraction. Transfer of radionuclides within the system is marked with arrows. The crosses mark the possible sources of radionuclides from Karlsson [in preparation].

The structure of the mire model is very simple, consisting of only two compartments that represent the two physical phases present in this kind of ecosystem. It may be argued that the conditions are not homogeneous but there is no data available to support a more detailed model structure.

5.5.3 Transfer between solid and soluble fraction

The distribution of elements between dissolved and solid phases in peat is described by one parameter (here called K_d) although it involves chemical, biological and physical processes. This is a time-dependent process and therefore a parameter for the half-time to reach sorption equilibrium is used. If the reaction velocity is known, the transfer coefficients from dissolved to solid fraction can be given by the expression:

$$TC = \frac{K_d \cdot \ln(2)}{T_k} \cdot \frac{D_p \cdot \rho_p}{\varepsilon_p \cdot D_p} = \frac{K_d \cdot \ln(2)}{T_k} \cdot \frac{\rho_p}{\varepsilon_p}$$

and from particulate to dissolved fraction by:

$$TC = \frac{\ln(2)}{T_k}$$

where

K_d = Distribution coefficient, concentration of the element on solids relative to dissolved [m^3/kg], see Table 5-2

T_k = Half-time to reach sorption equilibrium [year]

D_p = Depth of peat in mire [m]

ρ_p = Density of peat [kg/m^3]

ε_p = Porosity in peat [m^3/m^3]

5.5.4 Outflow of soluble fraction

The transfer coefficient for the horizontal flow is based on water balance and becomes:

$$TC = \frac{R}{\varepsilon_p \cdot D_p}$$

where

R = Runoff [m^3/(m^2·year)]

ε_p = Porosity of peat [m^3/m^3]

D_p = Depth of peat in mire [m]

5.5.5 Uptake in biota and exposure pathways

The concentration of radionuclides within the compartment for the solid fraction of the mire is used when calculating the concentration in crops grown in the peat in the assessment. The concentrations of radionuclides were calculated for the solid and water phase. The concentrations in grass, vegetables, root crops and cereals, which may be grown on drained mire, were estimated with species-specific root uptake factors. No estimates were made for biota living on the mire before draining, except for external exposure from the mire to humans. However, since the concentrations in the compartments are available this can easily be done.

5.5.6 Major features of the mire models

The general result from this modelling is that there is a large difference in time to reach equilibrium for elements with different mobility. For very mobile elements it takes less than hundred years for equilibrium to establish whereas it takes eight to nine thousand years for sorbing elements such as plutonium. This means that the processes of the natural succession interacts with the accumulation of radionuclides for sorbing elements and needs to be considered.

Table 5-2 Element specific distribution coefficients (K_d) for peat, concentration in solid matter/concentration in solution ([Bq/kg d.w.]/[Bq/m^3]). Table from Karlsson [*in preparation*].

Element	K_d Peat (m^3/kg) B.E.	Distr	Min	Max	Reference
H	–	–	–	–	1)
C	7E-2	LT	7E-1	7E-3	[Davis *et al.*, 1993]
Cl	1E-2	LT	1E-1	1E-3	2)
Ni	1E+0	LT	7E+0	2E-1	[IAEA, 1994]
Sr	2E-1	LT	6E+0	4E-3	[IAEA, 1994]
Nb	2E+0	LT	2E+1	2E-1	[IAEA, 1994]
Mo	3E-2	LT	3E-1	3E-3	[IAEA, 1994]
Tc	2E-3	LT	6E-2	4E-5	[IAEA, 1994]
Ru	7E+1	LT	1E+2	4E+1	[IAEA, 1994]
I	3E-2	LT	3E-1	3E-3	[IAEA, 1994]
Cs	3E-1	LT	3E+0	1E-1	[IAEA, 1994]
Pb	2E+1	LT	6E+1	8E+0	[IAEA, 1994]
Po	7E+0	LT	7E+2	7E-1	[IAEA, 1994]
Ra	2E+0	LT	2E+1	2E-1	[IAEA, 1994]
Th	9E+1	LT	9E+2	9E+0	[IAEA, 1994]
U	4E-1	LT	4E+0	3E-3	[IAEA, 1994]
Np	1E+0	LT	3E+0	5E-1	[IAEA, 1994]
Pu	2E+0	LT	2E+1	2E-1	[IAEA, 1994]
Am	1E+2	LT	1E+3	1E+1	[IAEA, 1994]

1) Hydrogen does not sorb to particle matter and therefore no effort has been put on finding K_d-values since they are missing in e.g. IAEA [1994].

2) Assuming that chlorine behaves like the elements in the same column in the periodic table a K_d-value close to that used for iodine has been used.

6. References

6.1 Forest ecosystems

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