FASSET

Framework for Assessment of Environmental Impact

Deliverable 1: Appendix 2

Ecological characteristics of European aquatic 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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Contractors:

Swedish Radiation Protection Authority	SSI
Swedish Nuclear Fuel and Waste Management Co.	SKB
Environment Agency of England and Wales	EA
German Federal Office for Radiation Protection	BfS
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 radiations 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 exposure 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, 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 2 to Deliverable 1) focuses on aquatic 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 31candidate 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 reflecting the different criteria for selection being used within the FASSET project.

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

2. Freshwater ecosystems

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2.1 Introduction

Freshwater ecosystems include lakes, ponds and rivers. Inland waters cover less than 2 per cent of the earth's surface, approximately 2.5 x 10^6 km^2 . Most lakes are formed by glacial, volcanic, or tectonic processes, lakes of glacial origin being far more numerous than lakes formed by other processes. As a result many lakes are located in lake districts in which large number of lake basins are concentrated. A large number of lakes are found in the Northern hemisphere while numerous small, shallow basins are found throughout the arctic, subarctic, and northern temperate zones *(Wetzel, 1983)*. More recently, man has created a large number of reservoirs and ponds. The majority of lakes are small and relatively shallow, usually < 20 m in depth. In Sweden, Norway and Finland the number of lakes is considerably higher than in many other European countries *(Niemi et al., 2001)* (Table 2-1).

Table 2-1 Number of lakes and reservoirs in some European countries *(Rask et al., 2001, Thyssen , 1999)*.

Country	Surface area km^2				
	0.01-0.1	0.1-1	1-10	10-100	>100
Austria	9000		17	7	2
Bulgaria	53	175	288	14	0
Denmark	365	269	69	6	0
Finland	40309	13114	2283	279	47
France*	24068	2011	201	25	2
Germany**	~4700	~1300	~250	~24	2
Greece	-	-	-	>16	1
Ireland	~5500		~100	14	3
Italy	-	>168	>82	13	5
The Netherlands	>100	>100	100	10	2
Norway	116218	16417	2039	164	7
Portugal***	-	30	40	15	0
Spain	482	330	247	63	2
Sweden	71693	20124	3512	369	23
Switzerland	-	111	40	13	5
United Kingdom	478	197	146	27	2

Notes: * Includes lakes and reservoirs
** only natural lakes
*** reservoirs

2.2 Ecological niches and habitats

The shape of the lake basin often dictates its productivity. Steepsided U- or V-shaped basins are usually deep and relatively unproductive. In such lakes a proportionally smaller volume of water is contiguous with sediments. Shallow depressions with a greater percentage contact of water with the sediments generally exhibit intermediate to high productivity.

The littoral zone of the basin is the shallow water in which light can penetrate to the bed sediments, permitting the growth of vascular (rooted) plants. Extremely productive wetland-littoral areas lie at the interface between the terrestrial drainage basin and open-water zone of the lake. These complex wetland-littoral areas are exceedingly important in regulating lake metabolism. Since a majority of the lakes are small and relatively shallow, the metabolically active wet-land and littoral components regulate the productivity of most lakes of the world.
Lakes are classified according to their primary productivity into eutrophic (nutrient rich) and oligotrophic (nutrient poor) lakes. The classification is based mostly on the amount of phytoplankton, which is responsible of the primary productivity of lakes. Either type of lake can be dystrophic (containing substantial amounts of undecayed vegetable matter, e.g peat), although this is more common in oligotrophic lakes. The brown colour of dystrophic lakes is due to humic substances, which come from the bogs and forests of the surrounding drainage basin. According to the amount of humic substances in the lake dystrophic lakes are oligo-, meso- or polyhumosis lakes.
The morphology of a lake basin has profound effects on nearly all physical, chemical, and biological properties of lakes. Morphometry of lake basins and geological substrates of the drainage basin influence sediment-water interactions and lake productivity, especially extremely productive littoral communities. Greater productivity of large lakes is usually correlated with higher water-sediment interface area per water volume i.e. lower mean depth.

2.3 Linkage to other ecosystems

Fresh water ecosystems are linked to other ecosystems through the catchment areas of the lakes. The inputs of radionuclides with nutrients and organic matter (both dissolved and particulate) from the catchment area to the lake is important and depends on the type of the catchment *(Wetzel, 1983)*. Therefore, freshwater ecosystems are closely linked to wet area ecosystems, forest ecosystems, and agricultural ecosystems. At least in Nordic countries, large areas of the drainage basins of the lakes are covered by forests *(Niemi et al., 2001).*

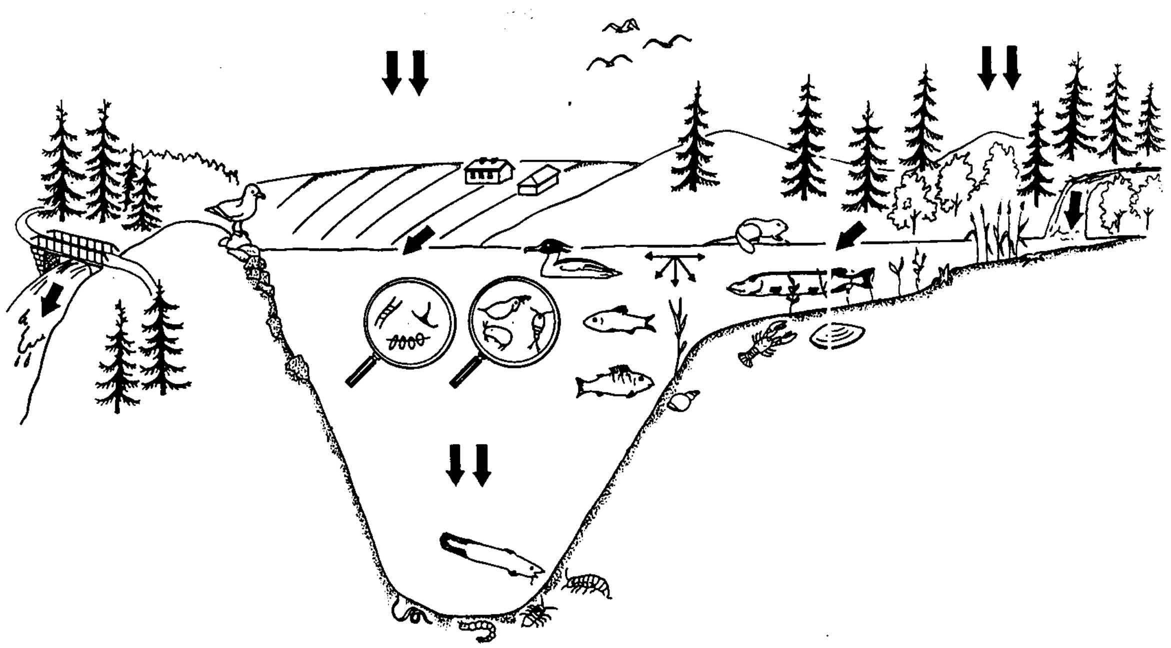

Fig. 2-1 Schematic description of a lake ecosystem with typical species of organisms. (Fig made by E. Ilus 1999).

2.4 Foodwebs and typical species

Lakes, ponds and rivers form ecosystems with a diversity of organisms. The littoral zone of lakes is an important habitat for many types of organisms. Organisms exist in complex foodwebs, in which nutrients and energy of one trophic level are utilized by organisms from several different trophic levels. The number of species in a community increases with the complexity of food webs and with the niche overlap. *Phytoplankton or algae*, the basic part of the productivity chain, forms the lowest trophic level in freshwaters. It consists of an assemblage of small plants whose powers of locomotion are zero or very limited ; they are therefore more or less subject to distribution by water movements. *Zooplankton* form the next step in the chain. Many types of *bottom animals* are living on the bottom sediment or inside the bottom sediment. These include many species of invertebrate including insect larvae together with molluscan and crustacean shellfish. Bottom animals and some species of *fishes* such as vendace, roach and other roach related fishes feed on zooplankton. The top of the productivity chain is formed by *predatory fishes* such as pike, pike-perch, burbot, etc. which feed on fishes which in turn eat bottom animals and zooplankton.

The main *fish* species in oligotrophic lakes are vendace *(Coregonus albula)*, perch *(Perca fluviatilis)* and pike *(Esox lucius)*. The main species in eutrophic lakes are: roach *(Rutilus rutilus)*, perch, pike, bream *(Abramis brama)* and pike-perch *(Stizostedion lucioperca)*. Alpine and sub-alpine species like brown trout *(Salmo trutta m. lacustris)* and Arctic char *(Salvelinus alpinus)* are the prevailing fish species in Norwegian lakes. In Sweden and in Finland, typical northern and boreal species such as perch, pike, roach and burbot *(Lota lota)* are prevailing. Some fish species live their whole life span in the littoral zone.

In rivers with high water discharge there exist fewer species of organisms than in lakes with lower water flow. In the flowing waters there live some *mammals* such as otter, beaver, and European mink. Many plant species (*macrophytes*) grow abundantly only in eutrophic lakes. *Amphibians* live in the littoral zone.

Sewage from population centres and increased use of fertilizers in agriculture and forestry have caused eutrophication of lakes. Input of additional nutrients to the water can increase populations of plants and animals, but cause, on the other hand, some disadvantages including the elimination of species such as vendace which flourish in oligotrophic conditions.

2.5 Exposure pathways relevant to the selection of reference organisms

Radionuclides enter freshwater ecosystems mainly from the athmosphere by deposition, both directly to water course and from the catchment with runoff waters or by erosion. The larger part of radionuclides in the water ultimately enter the bottom sediment *(Crusius et al., 1995, Saxén et al., 1994)*. Some nuclides concentrate in the epilimnic layer (surface layer). Focussing of radionuclides in sediment also happens due to horizontal fluctuations in the lakes *(Hilton et al., 1986, Kansanen et al., 1991)*. Radionuclides are transferred through the foodchains from water and bottom sediment to various trophic levels of plants and animals. Roots of aquatic plants extract radionuclides from the sediment as well as fauna living both on and in the sediment. Proportions of the radionuclides entering the bottom sediment are resuspended back into the water *(Kansanen and Seppälä, 1992).* A certain amount of the radionuclides are removed from the lake via outflow. Many interactions between biota and their surroundings influence the uptake and transfer of radionuclides. Accumulation of radionuclides by aquatic organisms is a complicated process depending on many factors leading to great variability in radionuclide concentration factors in various lakes and water courses. The main factors influencing accumulation of radionuclides are the physico-chemical form of the radionuclide, the chemical characteristics of the water (amount of nutrients conductivity, humic status, pH), type and quality of the catchment and the morphology of the lake concerned *(Kolehmainen et al., 1966, Hilton et al., 1993)*.

Chemical form affects the availability and uptake of radionuclides by organisms. Water soluble and exchangeable forms are available for uptake, while forms associated with particles which are insoluble in water radionuclides are biologically unavailable *(Konoplev and Bobovnikova, 1990).* Many transuranic elements exhibit a range of chemical forms depending on their oxidation state. For instance plutonium is water soluble in higher oxidation states and may then be more biologically available than in lower oxidation states.

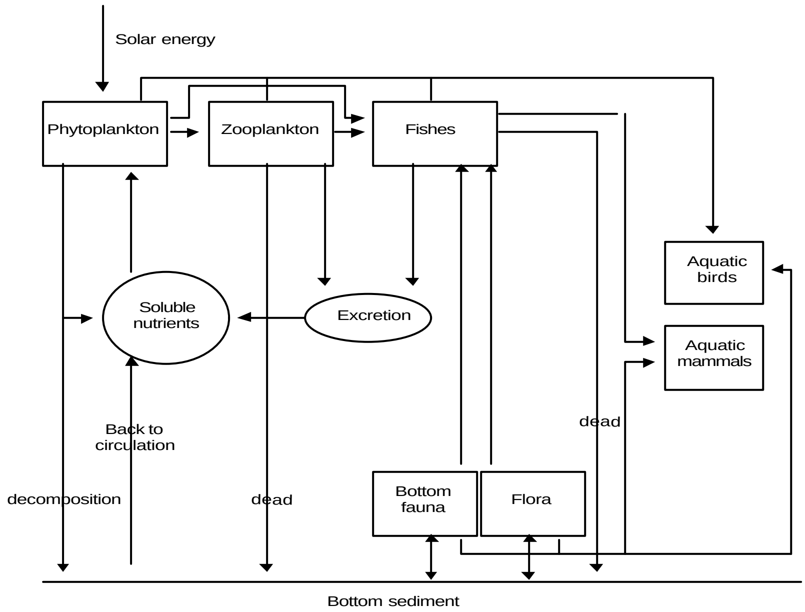

Fig. 2-2 Transfer pathways of radionuclides in freshwater ecosystem.

The behaviour of radionuclides is very strongly influenced by the partition in the water column between dissolved phase and suspended sediments, and the subsequent incorporation of sediment sorbed radionuclides into bed sediment. The quantification of radionuclide adsorption to particulate material in freshwater bodies is usually expressed in terms of a distribution coefficient K_d, which is defined as the ratio of the concentration of a radionuclide adsorbed to solid particles to the concentration of that radionuclide left in solution (K_d = Bq kg^{-1} dry or wet sediment per Bq l^{-1}). Average K_d values for various radionuclides are given in Table 2-2. The correct applicability of the K_d includes that equilibrium should exist between the compartments concerned *(Onishi et al., 1981)*. However, equilibrium rarely occurs in natural freshwater ecosystems. The reversilibity of the sorption process is another factor making the use of K_d values difficult.

Table 2-2 Gross average K_d values with emphasis on oxidizing conditions in aqueous systems (l/kg) (International Atomic Energy Agency (IAEA), *1994*).

	Fresh Water	
Elements	Expected	Range
P[a]	High	
Cr[a]	Low	$0 – 1 \times 10^3$
Mn[a]	1×10^3	$1 \times 10^2 – 1 \times 10^4$
Fe[a]	5×10^3	$1 \times 10^3 – 1 \times 10^4$
Co	5×10^3	$1 \times 10^3 – 7 \times 10^4$
Zn	5×10^2	$1 \times 10^2 – 1 \times 10^3$
Sr	1×10^3	$8 – 4 \times 10^3$
Zr	1×10^3	$1 \times 10^3 – 1 \times 10^4$
Tc[a]	5	$0 – 1 \times 10^2$
I	10	$0 – 8 \times 10^1$
Cs	1×10^3	$5 \times 10^1 – 8 \times 10^4$
Ra	5×10^2	$1 \times 10^2 – 1 \times 10^3$
Ce	1×10^4	$8 \times 10^3 – 1 \times 10^5$
Pm	5×10^3	$1 \times 10^3 – 1 \times 10^4$
Eu	5×10^2	$2 \times 10^2 – 8 \times 10^2$
Th	1×10^4	$1 \times 10^3 – 1 \times 10^6$
U[a]	5×10^1	$2 \times 10^1 – 1 \times 10^3$
Np[a]	10	$2 \times 10^{-1} – 1 \times 10^2$
Pu	1×10^5	$1 \times 10^2 – 1 \times 10^7$
Am	5×10^3	$9 \times 10^1 – 4 \times 10^4$
Cm	5×10^3	$10 – 7 \times 10^4$

[a] Dependent on oxidation-reduction conditions

The range of applicable K_d values for any individual nuclide reflects the influence of variables such as sediment type (mineralogy, particle size, organic content) and water chemistry (pH, eH, concentrations of trace organics such as humic acids) on sorption.

However, it is apparent that almost all radionuclides will be incorporated into sediments at activity concentrations higher than those which prevail in the overlying water column. Sediments therefore represent an important 'reservoir' of radionuclides in freshwater ecosystems. As such, they can be an important source of radiation exposure for organisms which live primarily in, or on, the bed sediments.

2.6 Uptake of radionuclides by biota

Some radionuclides are biologically more available than others. Their behaviour has similarities with some essential elements for plants and animals. Cs is a natural analogue with K and both Sr and Ra with Ca, both being essential elements for plants and animals. These relationships can lead to preferential uptake into plants and animals.

Transfer of radionuclides released to a freshwater ecosystem can be estimated by means of concentration factors between various compartments of the system. Concentration factors (CF) relate the concentrations of radionuclides in water to the concentrations of radionuclides in biota. They can be used to identify those radionuclides and organisms that are of greatest significance. A concentration factor approach to estimate the uptake of radionuclides by aquatic organisms can best be used in case of continuous discharge. In accidental situations a more dynamic approach is necessary, as the concentration factor approach tends to overestimate the concentration in the first period after the initial contamination *(*International Atomic Energy Agency (IAEA), *2000)*.

CF = Bq/kg in biota / Bq/kg in water

Most foodchain studies of the freshwater environment concern transfer and accumulation of ^{137}Cs. The literature on other radionuclides and on biota other than fish is much sparser. Very little or no data on radionuclide transfer into amphibians, aquatic plants, water birds, aquatic mammals and invertebrates in freshwater environment has so far been found in the literature. Nonetheless, some general features of radionuclide uptake can be described.

2.6.1 Phytoplankton and zooplankton

Phytoplankton are known to display high concentration factors for a range of radionuclides including ^{137}Cs *(*International Atomic Energy Agency (IAEA), *2000)*, ^{210}Po *(Shaheed et al., 1997)*, ^{226}Ra *(Hameed et al., 1997)*, ^{239,240}Pu and ^{241}Am *(Emery and Klopfer, 1975)*. Data on zooplankton is more limited, although in many of the studies referred to above it is not clear whether zooplankton have been excluded in the samples taken of phytoplankton. Nonetheless, zooplankton have also been shown to concentrate nuclides such as ^{210}Po in the marine environment *(Kharkar et al. 1976)* and similar behaviour may be expected in the freshwater environment.

2.6.2 Benthic invertebrates

Organisms living in or on the sediment appear prone to concentrate the more particle-reactive radionuclides such as ^{210}Po *(Shaheed et al., 1997)*, ^{239,240}Pu and ^{241}Am *(Emery and Klopfer, 1975)*. Thus organisms such as insect larvae, gastropod molluscs, bivalve molluscs and crustacea all show high concentration factors for a range of sediment-associated radionuclides.

2.6.3 Vascular plants

Plants show variable ability to concentrate radionuclides depending on the plant species and environmental conditions. Uptake may be lower than that for fish and invertebrates for some radionuclides, e.g. ^{137}Cs *(Hinton et al., 1999)*, ^{210}Po *(Shaheed et al., 1997)*, $^{239/240}$Pu *(Emery and Klofer, 1975)* but higher for others, e.g ^{226}Ra *(Mirka et al., 1996; Hameed et al., 1997)*. Concentrations of radionuclides in the roots or rhizome are commonly higher than those in foliage.

2.6.4 Fish

The uptake of radionuclides by freshwater fish has been extensively studied, because of the importance of fish as a potential pathway for the exposure of humans. Following the Chernobyl accident, contamination of the freshwater environment by ^{137}Cs has resulted in an extensive literature *(*International Atomic Energy Agency (IAEA), *2000)*, particularly from the Nordic countries where deposition was high *(Saxén et al, 1996)*.

The IAEA has produced a compilation of concentration factor values for fish *(*International Atomic Energy Agency (IAEA), *1994)* as summarised in Table 2-3.

Table 2-3 Concentration factors (CF) for edible portions of freshwater fish (l/kg) *(*International Atomic Energy Agency (IAEA), *1994)*.

Element	Expected	Range
^{3}H	1	6×10^{-1} - 1
He	1	
Be	1×10^{2}	
C	5×10^{4}	$5 \times 10^{3} – 5 \times 10^{4}$
N	2×10^{5}	
O	1	
Na	2×10^{1}	$2 \times 10^{1} – 1 \times 10^{2}$
P	5×10^{4}	$3 \times 10^{3} – 1 \times 10^{5}$
S	8×10^{2}	
Sc	1×10^{2}	$2 – 1 \times 10^{2}$
Cr	2×10^{2}	$4 \times 10^{1} – 2 \times 10^{3}$
Mn	4×10^{2}	$5 \times 10^{1} – 5 \times 10^{2}$
Fe	2×10^{2}	$5 \times 10^{1} – 2 \times 10^{3}$
Co	3×10^{2}	$10 – 3 \times 10^{2}$
Ni	1×10^{2}	
Cu	2×10^{2}	$5 \times 10^{1} – 2 \times 10^{2}$
Zn	1×10^{3}	$1 \times 10^{2} – 3 \times 10^{3}$
Br	4×10^{2}	
Rb	2×10^{3}	$2 \times 10^{2} – 9 \times 10^{3}$
Sr	6×10^{1}	$1 – 1 \times 10^{3}$
Y	3×10^{1}	
Zr	3×10^{2}	$3 – 3 \times 10^{2}$
Nb	3×10^{2}	$1 \times 10^{2} – 3 \times 10^{4}$
Mo	10	
Tc	2×10^{1}	$2 – 8 \times 10^{1}$
Ru	10	$10 – 2 \times 10^{2}$
Rh	10	
Ag	5	$2 \times 10^{-1} – 10$
Sn	3×10^{3}	
Sb	1×10^{2}	$1 – 2 \times 10^{2}$
Te	4×10^{2}	$4 \times 10^{2} – 1 \times 10^{3}$
I	4×10^{1}	$2 \times 10^{1} – 6 \times 10^{2}$
Cs	2×10^{3}	$3 \times 10^{1} – 3 \times 10^{3}$
Ba	4	$4 – 2 \times 10^{2}$
La	3×10^{1}	
Ce	3×10^{1}	$3 \times 10^{1} – 5 \times 10^{2}$
Pr	1×10^{2}	$3 \times 10^{1} – 1 \times 10^{2}$
Nd	1×10^{2}	$3 \times 10^{1} – 1 \times 10^{2}$
Pm	3×10^{1}	$10 – 2 \times 10^{2}$
Eu	5×10^{1}	$10 – 2 \times 10^{2}$
Ta	1×10^{2}	$1 \times 10^{2} – 3 \times 10^{4}$
W	10	$10 – 1 \times 10^{3}$
Hg	1×10^{3}	
Pb	3×10^{2}	$1 \times 10^{2} – 3 \times 10^{2}$
Bi	10	
Po	5×10^{1}	$10 – 5 \times 10^{2}$
Ra	5×10^{1}	$10 – 2 \times 10^{2}$

Element	Expected	Range
Th	1×10^2	$3 \times 10^1 – 1 \times 10^4$
Pa	10	
U	10	$2 – 5 \times 10^1$
Np	3×10^1	$10 – 3 \times 10^3$
Pu	3×10^1	$4 – 3 \times 10^2$
Am	3×10^1	$3 \times 10^1 – 3 \times 10^2$
Cm	3×10^1	$3 \times 10^1 – 3 \times 10^2$

^{137}Cs is a particularly important nuclide because it combines a high concentration factor (c.2000) with a moderate K_d value (c. 1000); this means that concentration within the organism is likely to be higher than that in bed sediments. There is also some indication that ^{137}Cs displays 'bio-magnification', that is concentrations increase in organisms higher up the food-chain, as indicated in Table 2-4:

Table 2-4 Averages of concentration factors, CF = (Bq kg^{-1} in fish / Bq kg^{-1} in water), for ^{137}Cs in predatory (pike, pike-perch, burbot), non-predatory (vendace, roach and roach-related) and intermediate fishes (perch, white fish) twelve years after the deposition *(Saxén & Koskelainen, 2001)*.

	Predatory fish	Intermediate fish	Non-predatory fish
Average	7040±870	6160±2600	2420±540

2.6.5 Water birds

Very little data is available in the literature on the uptake of radionulcides by freshwater birds. However there is an indication that significant concentration factors relative to water may be observed for ^{210}Po and ^{226}Ra *(Clulow et al., 1992; Martin et al., 1998)*.

2.6.6 Amphibians

We have not so far identified any data on the uptake of radionuclides by amphibia. Amphibians have a low metabolic rate and may therefore exhibit slow clearance rates for internally incorporated radionuclides. Some data are available for aquatic reptiles *(Martin et al., 1998)*, albeit in a sub-tropical environment, which indicate significant concentration factors for ^{210}Po and ^{226}Ra.

2.6.7 Aquatic mammals

We have not identified any data on the uptake of predatory aquatic mammals such as otter, which represent the top of the aquatic food-chain. Mirka *et al.* (1996) report significant concentration factors relative to water for ^{226}Ra uptake by the muskrat, which is an omnivorous rodent inhabiting lakes in the Canadian shield. However the reported concentration factors relate to bone, rather than soft tissue.

3. Marine ecosystem

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3.1 Marine ecosystem description

Before describing the ecology of the marine system under study, it is necessary to briefly describe the scope of our analyses. The FASSET project has been designed to be generic in nature but should not loose realism. For this reason actual species inhabiting European marine environment should be identified and utilised in the process of selecting reference organism types. The European marine system can be defined as an area bounded by the coast of Egypt and Israel in the south-east, by the Azores in the south-west and by the east coast of Greenland and Novaya Zemlya in the North-west and north East respectively. In terms of sea areas, we are therefore interested in the North-Eastern section of the Atlantic Ocean and its marginal seas including the Mediterranean Sea, Greenland Sea, the Irish Sea, North Sea, Norwegian Sea, Skagerrak, Kattegat and Barents Sea (note that the Baltic Sea is analysed under the section on Brackish waters).

The marine areas of interest in this study are shown in Figure 3-1 with concomitant bathymetry information.

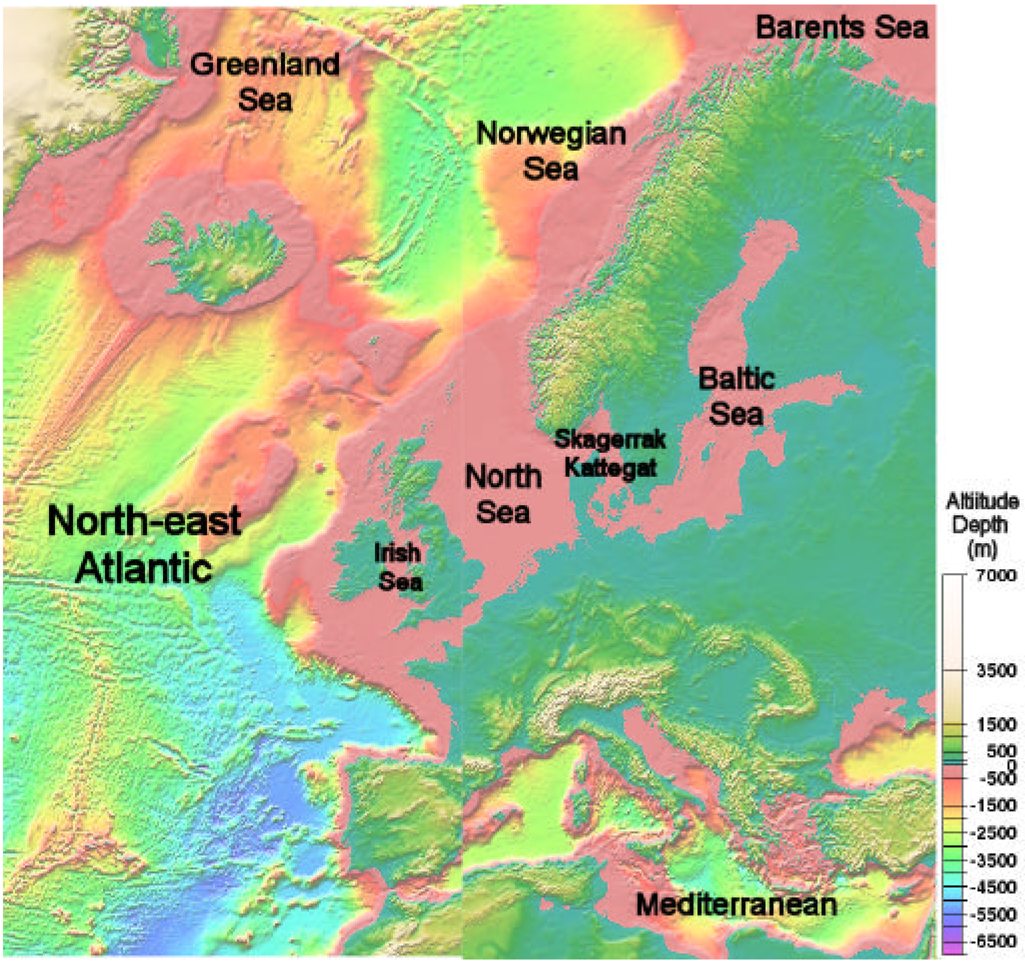

Figure 3-1: Bathymetry for European marine waters derived from a global map of predicted seafloor depth (Smith & Sandwell, 1997) and elevation from GTPO-30.

The depth at which a biological community is found has a significant bearing on factors such as biomass, feeding strategies and, most importantly in the context of this project, species type.

Reference to figure 3-1 illustrates that the study area includes a wide range of water depths and encompasses all numerous pelagic zones (as discussed below). It is necessary, therefore, to consider all oceanic depths during our subsequent deliberations relating to the choice of reference organisms. Nonetheless, it should be appreciated that the discussions are likely to be biased towards the continental shelf seas and surface waters purely because the data we have, whether derived from life history or contaminant uptake studies, are far more numerous for biota residing in these habitats.

3.1.1 Typical species

Phytoplankton consists of freely floating flora, often with minute dimensions (< 1mm) that drift with marine surface currents and form the main primary producers in the seas. Phytoplankton is composed of several main groups including diatoms, flagellates (specifically phytoflagellate) dinoflagellates, and coccolithophores. In more detail, dinoflagellates are any of numerous one-celled, aquatic organisms bearing two dissimilar flagellae and having characteristics of both plants and animals. Most are microscopic and marine. Botanists place them in the algal class Dinophyceae of the division Pyrrophyta, and zoologists claim them as members of the protozoan order Dinoflagellida. Dinoflagellates range in size from about 5 to 2,000 micrometres. Nutrition among dinoflagellates is plantlike, animal-like, or mixed; many species are parasitic or commensal. The group is an important component of phytoplankton in all but the colder seas and is an important link in the food chain.

Specific examples of species found in northern marine waters include *Ceratium bucephalum, C. focus, C. furca, C. tripos, Dinophysis acuta, D. rotundata, D. norvegica, Glenodinium lenticula, Goniaulax triacanta, Protoperidinum depressum, P. divergens, P. ovatum, P. pallidum, P. pellucidum.* Diatoms are any member of the algal division or phylum Bacillariophyta (about 16,000 species world-wide) found floating in all the waters of the Earth. Diatoms may be either unicellular or colonial. The silicified cell wall forms a pillbox-like shell (frustule) composed of overlapping halves that contain intricate and delicate markings. Specific examples of species found in northern marine waters include *Bacterosira fragilis, Chaetoceros atlanticus, C. convolutes, C. furcellatus, C. laciniosus, C. teres, Corethron hystrix, Leptocylindrus danicusMelosira nummuloides,Rhizosolenia hebetata, R. Semispina,R. Styliformis,Sceletonema costatum,Thalassiosira anguste-lineata, T. Gravida, T. nordenskioelddi, Asterionella bleakeleyi, Navicula pelagica, N. vanhoeffenii, Nitzschia grunowii, N. longissima, N. seriata.* The taxonomy of coccolithophores is not yet established. Zoologists place them either in the protozoan order Chrysomonadida or in their own order Coccolithophorida; botanists include them in the algal division Chrysophyta because they contain yellow to brown chromatophores. Specific examples of species found in northern marine waters include *Dictyocha speculum, Dinobryon balticum. Phaeocystis pouchetii*.

Macroalgae are normally split into the 3 categories brown, green and red algae. Brown seaweeds are members of the phylum or division Phaeophyta, and comprise of about 1,500 species world-wide. The English Channel alone has ca. 700 species of benthic macroalgae (Tappin & Reid, 2000). Common types of brown seaweeds in European marine environments include *Laminaria*, a type of kelp, and members of the genera Fucus, examples of specific species include *Fucus spiralis, F. vesiculosus, F. serratus*. Seaweeds of the genera *Fucus* and *Laminaria* are widely distributed in colder waters because they cannot reproduce in temperatures above 18° C. Red algae are members of the division *Rhodophyta* (about 3,000 species world-wide), are more typical of tropical and subtropical marine environments. Nonetheless certain species are found in northern marine waters and example being Irish moss (*Chondrus crispus*) and members of the genus *Porphyra*. *Ulva* species, commonly called sea lettuce, are among the relatively few green algae that occur as seaweed in European coastal waters. Size and form of seaweeds can vary considerably from the long, flat blades of up to several metres common for kelp to small, tufted seaweed with thin fronds from 5 to 25 cm common for Irish moss.

Zooplankton which are greater than 0.05 millimetre in size, are divided into two general categories: **meroplankton**, which spend only a part of their life cycle, usually the larval or juvenile stage, as plankton, and **holoplankton**, which exist as plankton all their lives. Examples of holoplankton include the larvae of sea urchins, intertidal snails, crabs, lobsters, and fish. Important holoplanktonic animals include such lobster-like crustaceans as the *copepods*, *cladocerans*, and *euphausids* (krill). In order to provide some idea as to the number of species that might be considered, it should be noted that Copepods alone include (globally) over 14 000 species, 2 280 genera and 210 families. Examples of copepod species common in northern Atlantic waters include *Calanus finmarchicus* and *Oithona similes*. Holoplankton will, of course include the larvae of many of the species discussed below.

Benthos or **Benthic invertebrates** form a very broad category that includes not just a single class of fauna but classes from several phyla including, amongst others, *Annelida*, *Porifera Arthropoda*, *Echinodermata* and *Mollusca*. An important subphylum under arthropoda is crustacea of which there are about 39 000 known species worldwide, although not all are benthic and/or marine species. Specific examples of benthic marine crustaceans that inhabit northern European marine waters include European Lobsters (*Homarus gammarus*), Spiny lobsters (*Palinurus elephas*), Edible Crabs (*Cancer pagarus*), Kamtchaka crabs (*Paralithodes camtschaticus*), European shore crabs (*Carcinus maenas*), Spider crabs (*Macropodia rostrata*).

Molluscs form one of the largest phyla of invertebrate animals comprising more than 50,000 living mollusk species when one includes all terrestrial and aquatic types. Mollusks are soft-bodied, and most have a prominent shell. The members of this highly successful and diverse phylum are mostly aquatic and in the benthic environments of European marine areas include the familiar organism types winkles (*Littorina littorea*), clams (e.g. soft shelled clam - *Mya arenaria*), oysters (e.g. Native oyster - *Ostrea edulis*), mussels (Common Mussel - *Mytilus edulis*) and whelks (e.g. common whelk - *Buccinum undatum*).

The rimmed reef platforms that are found in the southeast Mediterranean are formed by the gregarious sessile gastropod *Dendropoma petraeum* (Monterasto) and *Vermetus triqueter* (Bivona) that are endemic to the Mediterranean (Herut & Galil, 2000). Examples of Echinoderms include organisms of the class Asteroidea (Sea stars) and the Brittle star (*Ophiohpholis aculeata*). In the southeast Mediterranean, species from the Red Sea (including crustaceans, mollusks and echinoderms) have migrated into the area and compete with native species, sometimes, although rarely, replacing them. An example, is the indigenous Mediterranean sea star *Asterina gibbosa* that appears to have been out-competed by the Red Sea congener *A. wega* (Herut & Galil, 2000). The Demospongiae (Phylum *Porifera*) include the carnivorous Mediterranean sponge, which is classified as *Asbestopluma hypogea*. Species from other phyla may also be included in the category benthic invertebrates, including *Platyhelminthes* (flat worms), *Cnidaria* (specifically corals – although Coral reefs are generally found within 30 ^{0} N and 30 ^{0} S latitudes, various species of corals are found in all oceans of the world, from the tropics to the polar regions.), *Priapulida*, *Bryozoa*, *Phoronida*, *Brachiopoda*, *Sipuncula*, *Echiurida*, *Hemichordata* and *Pogonophora*.

There are nearly 1000 species of **cartilaginous fish (Chondrichthyes)** world-wide, including sharks, rays, and chimaeras, or ratfish. **The bony fishes (Osteichthyes)** encompass by far the largest diversity of fish, with about 24,000 species inhabiting nearly every body of water on the earth. They are divided into two groups—the lobe-finned fish and the ray-finned fish. In European marine waters the number of species is in the order of hundreds. For example**,** about 224 species of bony fish and sharks are found in the North Sea, alone, ranging in size from 5 cm gobies (*Pomatoschistus sp*.) to the 10 m basking shark (*Cetorhinus maximus*) (Ducrotoy *et al*., 2000).

Further selected examples of Osteichthyes + Chondrichthyes found in the north-eastern Atlantic and adjacent seas include Atlantic cod (*Gadus morhua*), Pollack (*Pollachius pollachius*), Plaice (*Pleuronectes platessa*), Flounder (*Platichthys flesus*), Sole (*Solea solea*), Herring (*Clupea harengus*), Salmon (*Salmo salar*), Conger Eel (*Conger conger*), Bass (*Dicentrarchus labrax*), Red mullet (*Mullus surmuletus*), Red Sea Bream (*Pagellus bogaraveo),* Ballan Wrasse (*Labrus bergylta*), Greater Sandeel (*Hyperoplus lanceolatus*), Mackerel (*Scomber scombrus*), Broadbill Swordfish (*Xiphias gladius*), White Marlin (*Tetrapturus albidus*), Common Goby (*Pomatoschistus microps*), Eelpout (*Zoarces viviparous*), Thick-lipped grey Mullet (*Chelon labrosus*), Garfish (*Belone belone*), Sea-Horse (Long-snouted) (*Hippocampus ramulosus*), John Dory (*Zeus faber*), Ten-spined stickleback (*Pungitius pungitius*), Angler Fish (*Lophius piscatorius*), Hammerhead shark (*Sphyrna zygaena),* Blue shark (*Prionace glauca*), Greater spotted Dogfish (*Scyliorhinus stellaris*), Monkfish (*Squatina squatina*), Common skate (*Raja batis*) and Thornback Ray (*Raja clavata*).

As is the case for benthos, the south-east Mediterranean has been invaded by a number of Red Sea immigrants, examples include the goldband goat fish (*Upeneus moluccensis*) and the brushtooth lizardfish (*Saurida undosquamis*) (Herut & Galil, 2000).

Amphibia and reptilia are not common in northern European marine environments mainly owing to the colder climatic conditions found there. In more southerly latitudes both reptiles and amphibia are present. For example in the Greek marine environment contains 16 species of amphibian and 28 species of reptile (Dassenakis et al., 2000). Examples include terrapins (e.g. *Emys arbicularis* and *Mauremys caspica*) water snakes of the genus *Natrix* and sea turtles (e.g. *Caretta caretta*). A number of species are under severe pressure from human activity (Dassenakis et al., 2000).

Seabirds are ubiquitous along the coastline of Europe and are dependent upon the sea for nutrition and the coast for breeding space. It is therefore important to include them in our marine assessment. 28 species of seabird breed and a further 6 species feed in the North Sea alone (Ducrotoy et al., 2000). The Arctic (including the European part) supports some of the world's largest seabird populations. Key nesting areas can be found along the coasts of Iceland, Svalbard and Northern Norway (Murray, 1998a). Selected examples of species found in European marine environments include Atlantic guillemot (*Curia aalge*), Atlantic puffin (*Fratercula arctica*), Black-legged kittiwake (*Rissa trydactyl*), Brent goose (*Branta bernicula*), Common eider (*Somateria mollissima*), Little auk (*Alle alle*), Herring gull (*Larus argentatus*), great black-backed gulls (*Larus marinus*), great skuas (*Catharacta skua*), Iceland gull (*Larus gaucoides*), Long-tailed jaeger (*Sterocorarius longicaudus*), Northern fulmar (*Fulmaris glacialis*), Northern gannet (*Sula bassanus*), Osprey (*Pandion haliaetus*),Pintail (*Anas acuta*), Razorbill (*Alca torda*), Red-breasted merganser (*Mergus serrator*), Water pipit

(*Anthus spinoletta*), White-tailed sea-eagle (*Haliaeetus albicilla*). Numerous other species can be found periodically over European marine waters, at locations such as the Straits of Gibraltar where many species congregate on migration routes between the European and African continents.

The two main groups of European marine **Mammals** are the whales (cetaceans) and seals (pinnipeds). This can be extend to a group of 3 if one includes the European Arctic and the Polar bear (*Ursus maritimus*). Harbour Seal (*Phoca vitulina*) and Grey Seal (*Halichoerus grypus*) breed along the coast of the North Sea (Ducrotoy *et al*. 2000) The Greek coast of the Adriatic sea is of prime importance for the endangered Monk Seal (*Monachus monachus*) (Dassenakis *et al*., 2000) and the European Arctic contains the largest member of the seal family namely the Atlantic Walrus (*Odobenus rosmarus rosmarus*) (Murray, 1998a). As regards the cetaceans, the harbour porpoise (Phocoena phocoena) is most common in the North Sea and white beaked dolphins (*Lagenorhynchus albirostris*), bottle-nosed dolphins (Tursiops truncates) and Minke whale (Balaenoptera acutorostrata) are sighted regulary (Ducrotoy *et al*. 2000). This group of mammals also includes some of the largest organisms found in European Marine waters. For example, blue whales (*Balaenoptera musculus*) may reach 30 m (100 ft) in length

The various broad categories listed above contain most, but not all the species found in European marine ecosystems. Examples of miscellaneous organism types and species that are not strictly included in the above groups include pelagic and demersal members of the phylum *Cnidaria* (e.g. jellyfish, hydras, Portugese-men of war), the phylum *Ctenophora* (Comb jellies), the phylum *Mollusca* (examples include squid (e.g. *Loligo vulgaris*), cuttlefish (e.g. The Common Cuttlefish *Sepia officinalis*) octopus (e.g. The long-armed octopus *Octopus salutii*)), the phylum *Nemertea* (Ribbonworms) and a number of parasitic organisms belonging to phyla such as *Mesozoa*, *Acanthocephala* (Spiny headed worms), *Nemertea* (Ribbonworms), *Asceliminthes* (round worms including nematodes), *Tardigrada* and *Chaetognatha*.

3.1.2 Ecological niches and habitats

Oceanic conditions in **coastal waters** differ in many respects to the open sea. Some of the factors causing these differences are the presence of the coast as a boundary to flow, the shallowness of the water over the continental shelf, river runoff and precipitation and the effects of continental air masses flowing out over the sea. In particular, variations in water properties and motions with position and time are larger than those generally associated with the open ocean (Pickard & Emery, 1982). The shelf areas of the European seas are dynamic in nature, prone to wind-driven circulation (Millero, 1996) and the action of the tides. The latter is caused due to the coast blocking the horizontal flows generated by periodically varying astronomical forces and in turn causes water to pile up against the coast during the flood and fall away during the ebb (Pickard & Emery, 1982). Some North Atlantic coastal waters, notably the Bristol Channel and the Bay of Biscay, have large tidal ranges in the order of several metres whereas other areas, such as large parts of the Mediterranean coast have much smaller tidal ranges in the order of centimetres. In terms of habitat several distinct zones can be identified. The **intertidal zone/shore,** which includes mud flats, estuaries and rocky coastal margins, is inundated periodically by the sea (diurnally or semidiurnally depending on the tide) and is prone to high-energy wave action. This is a habitat where many macroalgae are found along with numerous benthic organisms including molluscs (e.g. mussels, winkles) and polychaete worms (phylum Annelida). Bird species, in particular the waders (e.g. oyster catchers and lapwings) are reliant on the invertebrates associated with exposed intertidal sediments. Coastal cliffs and skerries provide suitable nesting areas for many seabirds including gulls, guillemots and puffins. Marine mammals use coastal areas as resting areas and for raising young.

Away from the coast we move to truly **pelagic ecosystems**, which can be split into several different zones. The **euphotic or epipelagic zone** forms the surface layer of open ocean water. In this layer, with a depth of 150-200m, sunlight penetrates the water column sufficiently for photosynthesis to occur (International Atomic Energy Agency (IAEA), 1988). The euphotic zone supports all marine trophic levels from primary producers to high trophic level carnivores. Phytoplankton, zooplanton, fish/sharks and ray are residents of the epipelagic zone . These surface layers of the ocean may be prone to significant seasonal variations in physical conditions including fluctuations in temperature and light levels. European marine surface water/epipelagic zones encompass a wide range of physical conditions including:

- Temperature: from high latitude regions such as the Barents Sea and Greenland seas where sea temperatures rarely exceed 10^0C and can fall below 0^0C in the winter (Murray, 1998b) to the southeast Mediterranean where surface water temperatures fall in the approximate range 16-28 ^{0}C (Herut & Galil, 2000).
- Salinity: from the high salinity waters observed in parts of the Mediterranean, e.g. Aegean Sea where salinities of up to 39.2 ‰ have been recorded (Dassenakis et al., 2000) to the low salinity waters (25 ‰) observed in the Skagerrak and Kattegat
- Seasonality in light regime: from the temperate conditions in the Mediterranean where day length varies little throughout the year to the Arctic where there is a marked seasonal distribution in the level of sunlight leading to cyclic annual productivity.

Directly below the euphotic zone lies the **mesopelagic zone**. Light penetrates this layer but at intensities insufficient to allow photosynthesis. The mesopelagic zone is a transition region between the highly variable epipelagic zone and the stable deep sea environment. The zone underlies the thermocline where temperature drops below 10 ^{0}C even in tropical and subtropical waters. The presence of an oxygen deficient layer, brought about by bacterial decay, is common at these depths. The zone is rich in biota and includes fish species (an example might be Lantern fish –Myctophids) and zooplankton that migrate vertically to feed nocturnally on epipelagic biota. Beyond the continental shelves, which are typically 100-200 m in depth and fall therefore within the epipelagic zone, the seafloor drops abruptly forming the continental slope before merging with a section of the ocean floor with shallower gradient, called the continental rise at a depth of roughly 4,000 to 5,000 m. At depths below 1000 m but above 4000 m, termed the **bathypelagic zone**, the biota are quite distinctive and largely consist of coprophagous (faeces-consuming) biota and carnivorous animals (e.g. fish) preying on these biota. The conditions in this zone and below are very stable, temperatures are low, < 5^0C and there is no sunlight.The numbers of bathypelagic organisms decrease exponentially with depth and below 3000 m biota densities are very low indeed. Below depths of 4000-6000 m, the **abyssopelagic zone**, the characteristics of the organisms change once again.

Close to the bottom of the ocean we reach the **benthopelagic zone**, arbitrarily defined as a layer of about 100 m above the sea floor. In all areas of the ocean, **benthic** organisms feeding on the sediment surface, epifauna, or within the sediment, infauna, can be found. The density and biomass of the benthic community decreases exponentially with water depth and a high proportion of the global oceanic biomass of benthos is distributed on the continental shelves (International Atomic Energy Agency (IAEA), 1988). Contrary to this observation, the diversity of macrofaunal and megafaunal benthos has been shown to increase with depth below the continental shelf reaching a peak near depths of 2000-3000 m (International Atomic Energy Agency (IAEA), 1988). On reaching this peak the diversity of macrofauna and megafauna decrease markedly on decent to abyssal depths. In terms of habitat benthic invertebrates can be associated with a wide range of sediment types/sea bottoms including fine-grained clay and muds, coarse-grained sands and rocky outcrops. The types of bottom material will of course have an influence on the associated level of contamination (actual or potential).

For the Northern shelf areas in the European marine environment, i.e. English Channel, Irish Sea, North Sea, Skaggerak and Kattegat, Norwegian Sea and Barents Sea, the water depth is consistently below 500 m, which means that the species occupy a habitat that is epipelagic in nature. This specific zone can also be termed ‘neritic’ - a subdivision of the pelagic zone. Many fish species, for example cod, rays etc. can be found intermittently or continuously near the seafloor and can be termed **demersal**. The shelf seas are important spawning and feeding areas for numerous fish species.

Finally, parasitic organisms, should be considered as having a ‘habitat’ that is effectively the organ of a host animal.

A simplified summary of the types of habitats found in the marine ecosystem is illustrated in Figure 3-2.

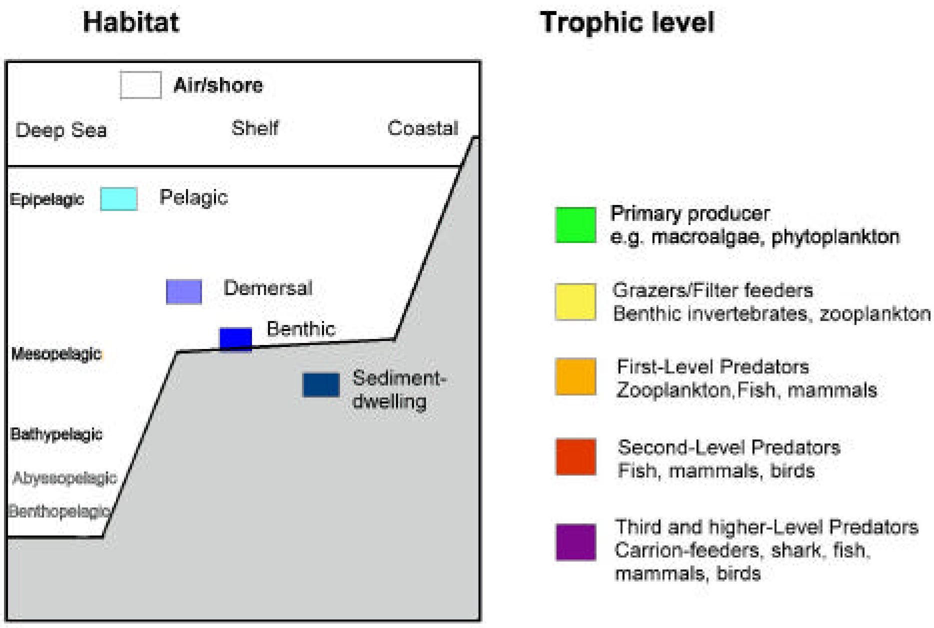

Figure 3-2 A classification of habitat and trophic in European marine environments.

3.1.3 Foodwebs

Marine food chains consist of numerous trophic levels often with 3 or more predator levels lying above the zooplankton and filter-feeding fauna. Examples of some of the broad organism groups that can be categorised under various food-chain levels are presented in Figure 3-2.

In its simplest form the offshore marine ecosystem can be split into pelagic and benthic components. Oceanic phytoplankton is the primary food source, directly or indirectly, of organisms in both parts of the marine ecosystem. Like land vegetation, phytoplankton uses carbon dioxide, releases oxygen, and converts minerals to a form animals can use.

Food chains in marine waters are generally regulated by nutrient concentrations. The availability of these elements (examples of biolimiting elements include Si, N and P) largely controls the biological activity in surface waters (Henderson, 1982). It was originally thought that phytoplankton in the 5–100 µm size range were responsible for most of the primary production in the seas and that grazers such as copepods controlled the numbers of phytoplankton. The current view in the field of marine ecology introduces a more complex picture, however, and considers that most primary production in marine waters is accomplished by single-celled 0.5- to 10 µm phototrophs (bacteria and protists). Moreover, heterotrophic protists (phagotrophic protists) are now viewed as the dominant controllers of both bacteria and primary production in the sea (Encyclopoedia Brittanica, 2001).

The phototrophs provide food for the primary consumers, such as protozoa and zooplankton, which are in turn consumed by higher trophic level organisms.

In the pelagic component of the ecosystem, nekton, i.e. freely-swimming organisms, fill successively higher predatory trophic levels. The vast majority of nekton are vertebrates (e.g., fishes, reptiles, and mammals), mollusks, and crustaceans. Nekton are found at all depths and latitudes of marine waters. Whales, seals, and fish including cod and redfish are common in polar waters. Lantern fish (family *Myctophidae*) are common in the aphotic zone along with whalefish (family *Cetomimidae*) and others. It should be noted that the distinction between nekton and plankton is not always sharp. Many large marine animals, such as cod, spend the larval stage of their lives as plankton and their adult stage as large and active members of the nekton. Other organisms such as krill are referred to as both micronekton and macrozooplankton.

Fish, such as capelin (*Mallotus villosus*), herring (*Clupea harengus*) and sandeels (e.g.Hyperoplus lanceolatus) and baleen whales such as blue, humpback, Fin whales, are examples of the biota types consuming zooplankton. They are effectively 1^{st}-2^{nd} level predators. At the next highest trophic level (2^{nd}-3^{rd} level predators), a variety of fish and mammals may prey upon a number of these organisms (with the exception of large baleen whales). The actual species types filling this role will be dependent on depth and latitude. For example in the north-eastern North Atlantic surface waters , e.g. Barents Sea, Cod (*Gadus morhua*) feed on populations of capelin and Arctic cod (*Boreogadus saida*). In more temperate locations more common predator-prey species filling the same trophic levels might be herring and blue-finned tuna (Thunnus thynnus). Complexity, of course is introduced by the fact that many species have varied diets from several different trophic levels. Grey Seal (*Halichoerus grypus*) are known to feed on a variety of fish along with cephalopods and crustaceans. The largest carnivores that consume large prey include the toothed whales (order *Odontoceti*—e.g. killer whales, *Orcinus orca*) and great white sharks (*Carcharodon carcharias*). In coastal areas, seabirds will also form part of the foodweb including 1^{st}-2nd level predators feeding off plankton and fish (e.g. Atlantic puffin, *Fratercula arctica*) and top predators feeding off fish and other seabirds (e.g. White-tailed sea-eagle, *Haliaeetus albicilla*). An example of a pelagic foodchain is illustrated in Figure 3-3.

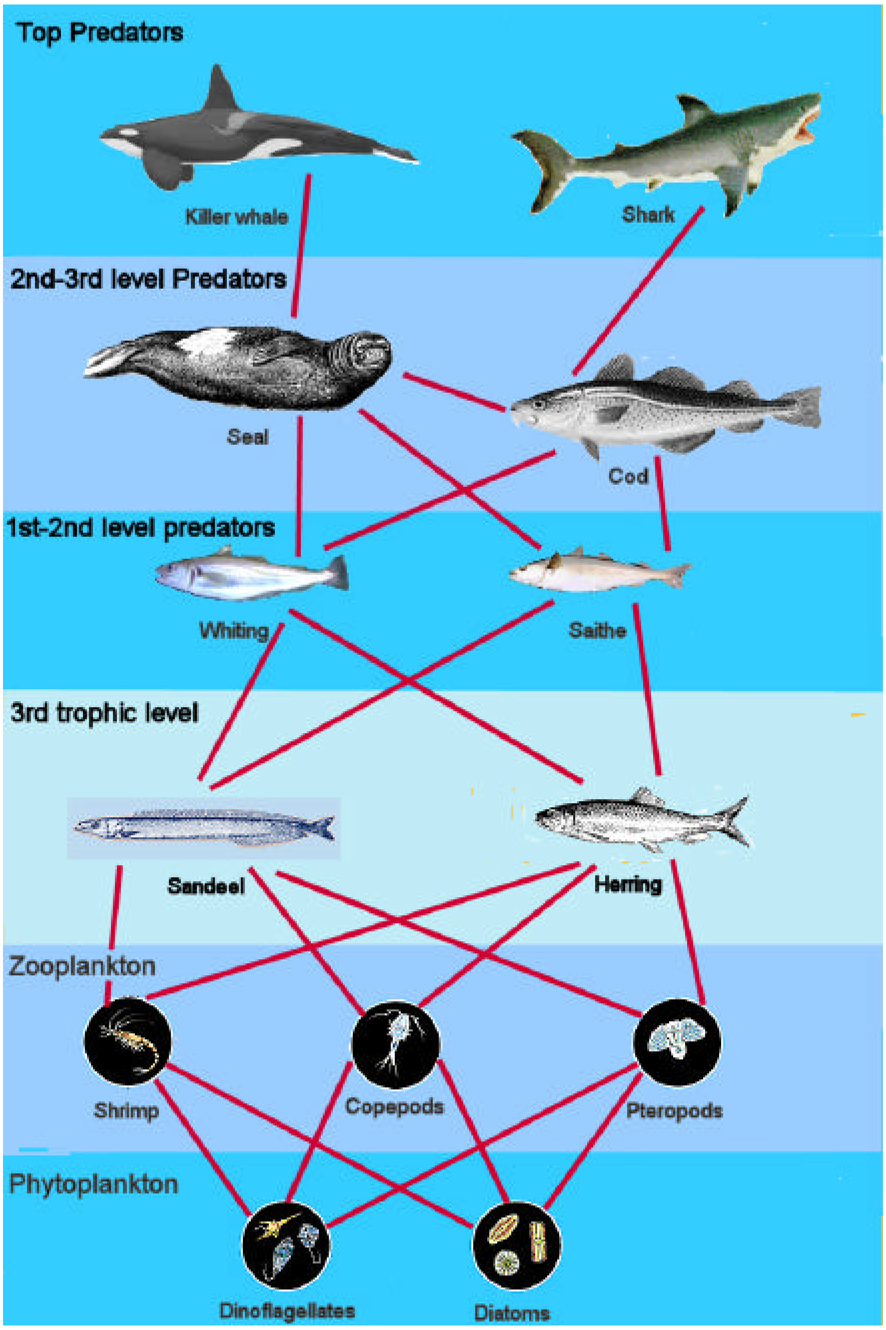

Figure 3-3 An example of a pelagic marine foodweb.

Pelagic plankton is an important source of food for organisms in the benthic component of the ecosystem. Suspension feeders such as anemones and barnacles filter living and dead particles from the surrounding water while detritus feeders graze on the accumulation of particulate material raining from the water column above. The molts of crustaceans, plankton faeces, dead plankton, and marine snow all contribute to organic matter inputs from the pelagic environment to the ocean bottom. There also is variation in the rate of fallout of plankton according to seasonal cycles of production. This variation can create seasonality in the lower pelagic zones where there is little or no variation in temperature or light. The photosynthetic organisms that are normally present in benthic communities inhabiting coastal environments and sediments in the euphotic zone include, macroalgae and microscopic benthic diatoms. Microbenthos, or those organisms smaller than 1 millimetre form components of the lower levels of many foodchains and include organisms such as diatoms, bacteria, and ciliates. Macrobenthos, i.e. organisms > 1mm, can be grouped into 3 components:

- deposit feeders - those consuming organic material in sediments (examples include holothurians, echinoids, gastropods)
- suspension feeders – those consuming plankton floating in the water column (e.g., bivalves, ophiuroids, crinoids).
- predators - those that consume other fauna in the benthic assemblage (e.g., starfish, gastropods).

However, meiobenthos, i.e. organisms between 0.1 and 1 mm, which include foraminiferans, turbellarians, and polychaetes, frequently dominate benthic food chains, filling the roles of nutrient recycler, decomposer, primary producer, and predator (Encyclopoedia Brittanica, 2001).

In coastal areas, birds that feed on benthic invertebrates (intertidal or shallow coastal) will form higher trophic levels of the food-chain (examples include the bearded seal, *Erignathus barbatus* and the Eider duck, Somateria spp.).

At the top of marine trophic levels we also find:

(i) Carrion feeding arthropods feeding on the remains of all animals including top level predators, and
(ii) parasitic organisms living within the bodies of all animals including top level predators

A simplified summary of the foodwebs discussed above is provided in the form of Figure 3-4. This particular example is for an Arctic (shelf sea) coastal marine environment.

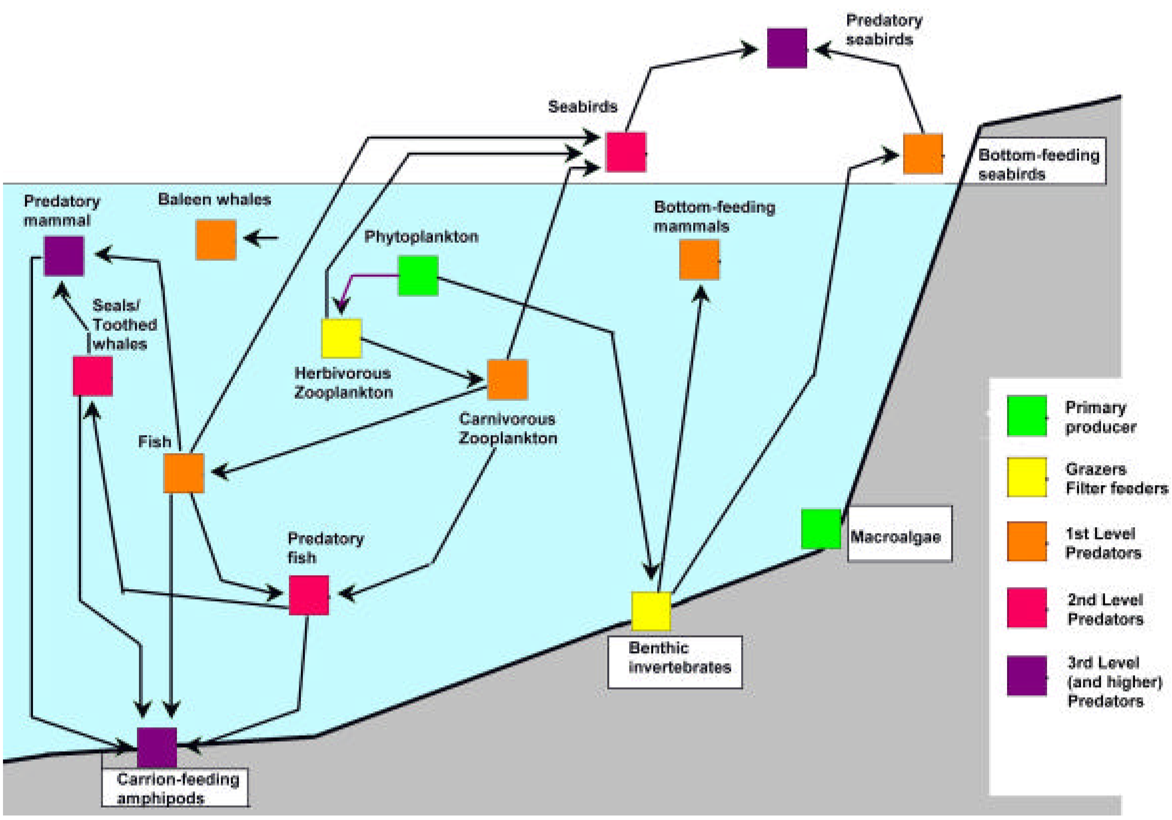

Figure 3-4 Simplified food-web in a northern marine ecosystem.

3.1.3 Linkage to other systems

It is necessary to avoid considering any one ecosystem in isolation. In the case of the marine environment 2 clear zones of interaction with other ecosystem types can be identified. The first is the boundary with the **freshwater ecosystem** and can be termed the Estuarine environment. Estuaries are defined as semi-enclosed, coastal bodies of water having free connection with the open ocean and within which seawater is measurably diluted by freshwater derived from land drainage (Pritchard, 1967). These are zones where euryhaline species, capable of living in low salinity environments, can be found. In addition certain fish including salmon (*Salmo salar*) and eels (*Anguilidae*) migrate from the marine environment to freshwater environment to spawn. Elements and compounds originating in terrestrial ecosystems are washed from the land into rivers where they are transported and finally introduced to the marine system. The second zone of interaction is with the ecosystem defined in FASSET as **wetlands ecosystem**. These are partly defined as areas on the margins of coasts and estuaries and represent areas that may be periodically flooded by the sea or at least be influenced by the presence of saline waters.

3.2 Methods used in the selection of reference organisms

3.2.1 Literature review and transfer databases

Radiological assessments in the marine environment are often based upon the use of distribution/partition coefficients –(K_d) and concentrations factors (CF). K_ds are used to describe the equilibrium balance between dissolved and particulate phases (normally for the pelagic environment), a tacit assumption being that exchanges of radionuclides between particulate phases and water are wholly reversible. Concentration factors also describe the equilibrium balance between environmental compartments. For coastal sediment the sediment water distribution coefficient is referred to in the text as a sediment water CF defined as the (activity) concentration per unit mass of deposited sediment relative to that in ambient seawater (Bq l^{-1} or Bq kg^{-1}).

In the case of flora and fauna, a CF can be defined as the activity concentration in biota (i.e. Bq kg^{-1}) relative to that of the ambient seawater (Bq kg^{-1}). For practical purposes the activity concentrations in seawater are often defined as Bq per unit volume (Bq l^{-1}) but this makes little difference to the final derivation of the CF. It should also be noted that the CF term simply relates the activity concentration of biota to the medium within which it lives. The implication is that uptake does not necessarily occur via direct uptake from the water column only. Other pathways such as ingestion and assimilation of radionuclides associated with food may also be important, if not the dominant factor, in affecting the activity concentrations associated with a selected organism. K_ds and CFs have been used as the basis for the following assessment.

An extensive literature review for the purpose of compiling information on distribution coefficients and concentration factors has not been conducted as part of this work as this has been extensively considered elsewhere (International Atomic Energy Agency (IAEA), 1985; Fisher et al., 1999). Instead literature has been selected and cited where supporting evidence for the assessment/selection procedure is required.

The procedure has been to consider each of the selected radionuclides in turn and to assess by the process of presenting a biogeochemical description, using the K_d and CF approach described above and simple conceptual models, which biota and which habitats in the marine environment are likely to exhibit the highest concentrations of each radionuclide following an input/release to the system.

3.2.2 Modelling approaches and expert judgement

Features that are general to all radionuclides (conceptual model)

Following the release/input of a radionuclide suite to the surface waters of a marine system several immediate processes are likely to occur. There will be an interaction of the radionuclide with suspended particulate material and uptake by biota. The degree of interaction and uptake depends on the radionuclide in question and its physico-chemical form. Those radionuclides that are analogues/radioisotopes of metals important in enzyme systems, for example, will be actively taken up. Passive uptake may occur following the adsorption of radionuclides/heavy metals on organic particles following interaction with surface groups, e.g. carboxilic and phenolic groups (Millero, 1996). Radionuclides exhibiting low reactivity will remain primarily in the dissolved form and are termed conservative.

Those radionuclides that are assimilated to a significant degree by biota in the epipelagic zone will enter the pelagic food-chain. Bioconcentration may occur in some instances whereby increasing activity concentrations are associated with successively higher trophic levels. In contrast those radionuclides that have no natural, biologically important analogues may be actively selected against and exhibit reduced activity concentrations at higher trophic levels.

Radionuclides can be removed from the water column and transferred to sediments by several processes including:

(i) Direct uptake of the radionuclide at the sediment-water interface.
(ii) Sedimentation with organic matter either after assimilation or adsorption onto cell surfaces
(iii) Adsorption onto inorganic compounds (e.g. clays, carbonates) or scavenging from solution by iron-manganese oxy-hydroxides.
(iv) Sedimentation with humic matter

The physico-chemical properties of the sediments along with the physico-chemical form of the radionuclide will regulate the magnitude of each possible mechanism. Grain-size often strongly influences the activity concentration of radionuclides in marine sediments (Hetherington & Jefferies, 1974; Bonnett et al., 1988; Assinder et al., 1993; Clifton et al., 1997).

In marine environments, conservative radionuclide introduced at the sea surface will migrate downwards by the process of diffusion and advection (Bowen et al., 1980). The vertical flux of water will influence how quickly the radionuclides will come into contact with bottom sediments. In the deep ocean advection in the vertical plane may be limited. An example can be found in the Arctic Sea where stratification prevents winter convection and deepening of the polar mixed layer reducing the flow of contaminants to deeper layers (Gregor et al., 1998). In the European shelf seas where water depths are < 500m and the epipelagic layer is prone to the turbulent action of winds and tides the whole water column often becomes well mixed. Radionuclides that were originally introduced at the surface will be mixed throughout the water column after a relatively short period (months ➔ years). Evidence for this can be derived from the fact that “conservative” radionuclides introduced as global fallout or from

W-European reprocessing plants can be observed at enhanced levels in deeper shelf waters (e.g. see Kershaw & Baxter, 1995).

The settling velocity of the particles will also influence the removal rate of particle-reactive radionuclides from the water column. Large particles will of course fall rapidly. An important consideration relating to removal mechanisms and rates is the nature of the inorganic and organic components of the suspended matter. It is apparent that the dominant mass flux of matter to the seafloor is by the rapid transport of rare large particles. These are faecal pellets from grazing zooplankton containing organic matter, skeletal material and minor amounts of clay minerals. Sinking rates of approximately 100 m day^{-1} are required to account for the material caught in traps set at abyssal depths (Sholkovitz, 1983).

Once incorporated into sediments, radionuclides will be exposed to early diagenetic reactions (hardening to solid rock). This will either result in the virtually irreversible binding of the radionuclide to robust phases of the sediment matrix or to redissolution and return of radionuclide to the water column. Geochemical phase association studies for example via sequential extractions of sediments provide information with regards to these processes.

The process of physical disturbance and bioturbation leads to the mixing of radionuclides in the surface layer of the sediment over short time periods. In the north East Irish Sea for example, mixing of surface sediment (< 13 cm) occurs on a time-scale of ca. 1 year (Mackenzie et al., 1998). Biological activity in this area is both extensive and heterogeneous and is probably responsible for the great variety of vertical profiles of Sellafield radionuclides which have been observed in cores taken from the Irish Sea (Kershaw et al., 1992). The sedimentation of particulate material will also lead to the burial of contamination. The net sedimentation rate is approximately 0.1 mm y^{-1} for the NE Irish Sea (Kershaw et al., 1992) but in the order of mm per year (Brown et al., 1999a) for coastal/estuarine environments in the same area. Sedimentation rates in the open ocean may be a thousand times lower than for coastal deposits (Gregor et al.,1998). A third process leading to the redistribution in sediment is the dissolution and vertical migration of radionuclides via pore waters.

In addition to the physical-chemical processes, direct uptake through assimilation by biota on bottom sediments may also occur. Suspension feeders extract particles from the water column and in so doing also ingest contaminants. Radionuclides will also be introduced to the benthic foodchain via direct uptake to benthic primary producers such as macroalgae and benthic diatoms and via the ingestion of contaminated sediments by deposit and suspension feeders. The amount of a radionuclide available for uptake will depend, to a large extent, on the fraction of the radionuclide in soluble form, either in the water column or in pore waters. Sequential extraction experiments on sediments can provide ancillary, although often indirect, information on the inventory of a particular radionuclide in the sediment that can act as a reservoir for potential transfer to biota. A new dimension is introduced into the assessment in the sense that geochemical phase association data can be used to predict the fraction of a given radionuclide in the sediment which may be released if environmental conditions (Eh, pH etc.) change. The fraction of the radionuclide inventory that is unavailable for biological uptake, over long time periods, can also be considered by this method.

K-40
Potassium is an element essential for life and forms a major (elemental) component of seawater. The radioisotope ^{40}K forms a small fraction of K found on earth with an isotopic abundance of 0.018 % (Henderson , 1982). In view of the long half-life of ^{40}K it is reasonable to assume that there is no isotopic fractionation between stable isotopes and radio-isotopes of K and therefore activity levels of ^{40}K will reflect the relatively constant, homogeneous concentrations of stable K in (filtered) seawater observed throughout the worlds oceans. With respect to biological function, potassium is important for normal muscle and nerve responsiveness, heart rhythm, to balance the metabolism of nitrogen compounds and, in particular, intracellular fluid pressure in mammals. For other organisms K forms an important elemental component of the osmotic system. A small fraction of ^{40}K, corresponding to the isotopic abundance, will be assimilated within the body of marine organisms during the uptake of stable potassium.

The concentration of ^{40}K in surface seawater is reported as being 18 Bq l^{-1} (International Atomic Energy Agency (IAEA), 1988). The ^{40}K content of sediment will vary according to mineralogy and sedimentology. Sediments with a high proportion of potassium rich clay minerals, e.g. illite, might be expected to exhibit enhanced ^{40}K levels. Activity concentrations are typically 100 Bq kg^{-1} for beach sands (International Atomic Energy Agency (IAEA), 1988) to > 500 Bq kg^{-1} for fine-grained sediments rich in micaceous minerals (e.g. Brown, 1997).

Potassium-40 activity concentrations in phytoplankton, zooplankton, molluscs, crustacea and the muscle of teleost fish are consistently in the range 90-110 Bq kg^{-1} (International Atomic Energy Agency (IAEA), 1988; Woodhead 1973).

Benthic plants, infauna and epifauna, especially those living over/within sediments rich in potassium are likely to be exposed to the highest levels of radiation arising from ^{40}K. Organisms of small size will be prone to a significant external exposure by β radiation The concomitant emission of a medium energy γ-photon at 1460 keV, albeit at low yield, leads to irradiation of all sizes of organisms. Small differences in the internal ^{40}K body burden of different animals and plants removes the need to refine our reference flora and fauna selection beyond the level of 'benthic organism'.

Cs-137
Caesium, an alkali metal, forms monovalent Cs^{+} ions in aqueous solutions. Caesium-137 is defined as conservative radionuclide behaving as a tracer of marine currents although it has an intermediate concentration factor of around 3 x 10^{3} for coastal sediments/waters (International Atomic Energy Agency (IAEA), 1985), which means that interaction with the solid phase is significant. In the specific case of surface water discharges to the shallow Irish Sea from the reprocessing plant at Sellafield, it appears that approximately 10 % of caesium is initially incorporated into the offshore deposits of silt and mud (Miller et al., 1982, Jones et al., 1988; Cook et al., 1997).

Hydrous manganese oxides appear to be a poor scavenger for removing ^{137}Cs from seawater (Om Vir Singh & Tandon, 1977) although other components of suspended matter, e.g. clays, may remove Cs from the aquatic phase. Caesium-137 is strongly and preferentially adsorbed to the clay mineral illite (Murdock *et al.*, 1993). Caesium$^+$ ions are sorbed on specific sites located within the wedge zones of illitic crystallites. Caesium fixation (i.e. interlayer collapse of the clay lattice) can then take place leading to the virtually irreversible fixing of the caesium ion within the clay mineral matrix (see discussion in Hird *et al*., 1996 relating to soils). In laboratory studies concerned with the diagenesis and mobility of radionuclides in near-shore environments, Sholkovitz *et al*. (1983) concluded that ^{137}Cs was not involved in diagenetic processes such as redox reactions, decomposition of organic matter and production of nutrients. Although ^{137}Cs did not appear to be involved in these reactions, it was suggested that pore water concentrations could increase by the process of ion exchange.

The exchange of caesium with NH_4^+ at interlayer sites of illitic clay minerals and the relatively high mobility of caesium mean that remobilisation of caesium from marine sediments may be an important process (Hunt & Kershaw, 1990). Investigations into the geochemical phase association of caesium have shown the predominant association of caesium with the irreversibly bound fraction (which corresponds to association with primary and secondary minerals, including illite) in marine-estuarine environments (Brown et al., 1997). It is important to remember that kinetics do play a role in this situation. For studies on soils dosed with concentrations of ^{137}Cs, Vidal *et al*., (1995) observed a significant decrease in the available fraction and a corresponding increase in the 'fixed' fraction over a period of 6 years. In a similar fashion, ^{137}Cs (aqueous) that become associated with sediment directly after release to the aquatic environment will originally be configured in phases weighted towards exchangeable and easily-available forms. After time the association will progress to a more strongly bound configuration. In the case of ^{137}Cs associated with sediments having measurable micaceous clay contents, a large fraction of the radionuclide (after a period of a few years) is likely to be associated with the residual fractions. Caesium-137 associated with this phase will not be available for biota in the short-term and even over long time periods (years, decades) may remain excluded from biological interaction.

The quasi-conservative behaviour of radiocaesium in the marine environment necessitates the consideration of both pelagic and benthic components of the ecosystem.

Recent studies, using coccolithophorid, non-calcareous and diatom species of phytoplankton have shown that the uptake of ^{137}Cs is often low or even negligible (Heldal et al., in press). These results indicate that phytoplankton is unlikely to influence the Cs build-up in marine food chains and Cs flux to deep waters. Concentration factors for ^{137}Cs to the brown seaweed *Fucus* in the north-east Atlantic and adjacent seas (Norwegian coastal, Faroe Islands, Scottish coastal) appear to fall within the range 150-200 (Dahlgaard et al., 1997) which essentially indicates that uptake is low. The derivation of concentration factors for the copepod *Calanus finmarchius* based on field expeditions in Norwegian coastal waters, primarily the Barents Sea, yielded values of approximately 50 (Brown, 2000). The fact that this value was higher than the tertiary consumers, i.e. those organisms preying upon *Calanus*, in this ecosystem suggested that biomagnification was not occurring at this level in the foodchain although measurement uncertainties necessitated caution in the interpretation of results.

Uptake of radiocaesium to macrobenthos appears to be fairly limited. Fisher et al. (1999) present CF values in the range of 52-63 for crustaceans, echinoderms and molluscs species including benthic types.

Studies concerning the accumulation of caesium radioisotopes by fish species common in the Irish Sea including plaice (*Pleuronectes platessa*) and thornback ray (*Raja clavata*) (e.g. Mauchline & Taylor, 1964, Jefferies & Hewett, 1971, Pentreath et al., 1973) have shown that radiocaesium is accumulated in the muscles of these organisms. Other studies concerning the activity concentrations and uptake of ^{137}Cs to fish in northern marine environment suggest that activity concentrations in species from higher trophic levels, e.g. cod (*Gadus morhua*) may be slightly higher than for organisms at lower trophic levels, e.g. herring (*Clupea harengus*) and certain crustaceans (Brown, 2000; Brungot et al., 1999) however such a conclusion based on a limited number of data is tentative and is confounded by observations made in other monitoring programmes/studies where a trophic level effect for fish species appears to be absent (FSA & SEPA, 2000; Rissanen et al., 1997).

Data collated on ^{137}Cs activity concentrations in high trophic level fish, e.g. cod (*Gadus morhua*) and organisms, including fish, at lower trophic levels, e.g. herring (*Clupea harengus*) and certain crustaceans inhabiting northern marine environments (Brown, 2000; Brungot et al., 1999; FSA & SEPA, 2000; Rissanen et al., 1997) do not provide evidence for any clear trophic-level effect. Fish living as 1st level predators often exhibit ^{137}Cs activity concentrations commensurate with those fish species living higher up the food-chain at the same location. It should be conceded, however, that a comparison of fish species from the same marine area might not allow suitably robust conclusions to be drawn owing to the migrant nature of many species. Sampling location and feeding areas may not coincide.

Activity concentration/CF data for seabirds and mammals are scarce. Those data that are reported for seabirds in the literature (e.g. Rissanen et al., 1997, Fisher et al., 1999) suggest that bioaccumulation of ^{137}Cs may be occurring for seabirds, especially those living at high trophic levels such as skuas and gulls. Data for sea mammals, primarily whales and seals, (e.g. Rissanen et al., 1997, Fisher et al., 1999; Strand et al., 1998 , Brown, 2000) provide no clear evidence for any trophic level effect, i.e. elevated body concentrations over those observed in prey species.

An overview of recommended ^{137}Cs concentration factors for selected generic marine organism types is given in Table 3-1. These data suggest that a degree of bioaccumulation occurs over a generic pelagic foodchain seen as a whole, i.e. from phytoplankton ➔ fish.

Table 3-1 International Atomic Energy Agency (IAEA) (1985) recommended values of element uptake to various generic marine organisms.

Element	Phytoplankton	Macroalgae	Zooplankton	Mollusca*	Crustaceans	Fish
Cs	2×10^{1}	5×10^{1}	3×10^{1}	3×10^{1}	3×10^{1}	1×10^{2}
Tc	5×10^{0}	1×10^{3}	1×10^{2}	1×10^{3}	1×10^{3}	3×10^{1}
Sr	3×10^{0}	5×10^{0}	1×10^{0}	1×10^{0}	2×10^{0}	2×10^{0}
U	2×10^{1}	1×10^{2}	5×10^{0}	3×10^{1}	1×10^{1}	1×10^{0}
Th	2×10^{4}	2×10^{2}	1×10^{4}	1×10^{3}	1×10^{3}	6×10^{2}
Pu	1×10^{5}	1×10^{3}	1×10^{3}	3×10^{3}	3×10^{2}	4×10^{1}
Am	2×10^{5}	2×10^{3}	2×10^{3}	2×10^{4}	5×10^{2}	5×10^{1}
Cm	3×10^{5}	8×10^{3}	2×10^{3}	3×10^{4}	5×10^{2}	5×10^{1}
Np	1×10^{2}	5×10^{1}	1×10^{2}	4×10^{2}	1×10^{2}	1×10^{1}
Ra	2×10^{3}	1×10^{2}	1×10^{2}	1×10^{3}	1×10^{2}	5×10^{2}
Pb	7×10^{3}	1×10^{3}	1×10^{3}	1×10^{3}	1×10^{3}	2×10^{2}
Po	3×10^{4}	1×10^{3}	3×10^{4}	1×10^{4}	5×10^{4}	2×10^{3}
C	9×10^{3}	1×10^{4}	2×10^{4}	2×10^{4}	2×10^{4}	2×10^{4}
H	1×10^{0}	1×10^{0}	1×10^{0}	1×10^{0}	1×10^{0}	1×10^{0}
Nb	1×10^{3}	3×10^{3}	2×10^{4}	1×10^{3}	2×10^{2}	3×10^{1}
Ni	3×10^{3}	2×10^{3}	1×10^{3}	2×10^{3}	1×10^{3}	1×10^{3}
Ru	2×10^{5}	2×10^{3}	3×10^{4}	2×10^{3}	1×10^{2}	2×10^{0}
I	1×10^{3}	1×10^{3}	3×10^{3}	1×10^{1}	1×10^{1}	1×10^{1}
Cl	1×10^{0}	5×10^{-2}	1×10^{0}	5×10^{-2}	5×10^{-2}	5×10^{-2}

*excluding cephalopods
Italicised values are best estimates

In conclusion the ambient activity concentrations of ^{137}Cs in sediments are likely to become higher than those observed in seawater following a release of this radionuclide to coastal waters although a major fraction of the ^{137}Cs inventory may remain in the aqueous phase. Uptake and transfer through the foodchain occurs to a limited extent and evidence for significant bioaccumulation is not compelling. Once radiocaesium becomes associated with bottom sediments, the bioavailable fraction tends to be reduced. A major exposure pathway is therefore likely to be the irradiation of benthic organisms from contaminated sediments. Those benthic organisms residing near the top of the foodchain e.g. Atlantic halibut (*Hippoglosus hippoglossus*), Plaice (*Pleuronectes platessa*), carrion-feeding crustaceans, skates and rays, may receive an extra internal exposure from elevated ^{137}Cs body burdens, compared to organisms residing at lower levels in the food-chain, and can be identified as organisms that are most vulnerable to inputs of radiocaesium to the marine system. Seabirds, especially those that are categorised as top predators, e.g. great black-backed gulls (*Larus marinus*), great skuas (*Catharacta skua*), may also be prone to elevated ^{137}Cs exposure via an ingestion pathway. It should be noted that for deep oceanic systems, the intermediate half life of ^{137}Cs (30 yrs) may prevent substantial amounts of the radionuclide from ever reaching the seabed and therefore a high trophic level pelagic organism may be more vulnerable to inputs of this radionuclide.

Sr-89,90

Strontium forms divalent Sr^{2+} ions in aqueous solutions. Strontium-90 is a conservative radionuclide having a K_d of 1×10^3 l kg^{-1} in coastal marine sediments (International Atomic Energy Agency (IAEA), 1985). Sediment water partition coefficients for some Arctic marine environments have been reported at a level 10-fold lower than this value (Fisher et al., 1999).

Sequential experiments conducted on marine sediments from northern marine waters (e.g. Strand et al., 1997, Børretzen et al., 1995) illustrate that radiostrontium is often associated with easily exchangeable geochemical phases. Sequential data suggest that sediments may play an important role in the long-term transfer of radiostrontium to biota because a large fraction of the radionuclide is available in easily exchangeable fractions. Over time marine sediments can behave as a 'reservoir' slowly releasing radiostrontium into aqueous phases where it may then be available for uptake.

Strontium can behave as an analogue for calcium (Blaylock, 1982) which means that it may be involved in biological reactions such as the uptake and formation of exoskeletons in invertebrates - in other words strontium will often become associated with biogenic calcium carbonate prior to precipitation and incorporation in bottom sediments.

A major difference with ^{137}Cs accumulation in fish is related to the biological deposition of ^{90}Sr. Whereas potassium, the nutrient analogue for Cs, is a major component of the osmotic system; calcium, the nutrient analogue for Sr, forms an important constituent of bones. As a result ^{90}Sr tends to approach equilibrium in fish (bone) much more slowly than ^{137}Cs (primarily in flesh) (Whicker *et al.*, 1972). In a study of the behaviour of radiostrontium in a Canadian lake ecosystem, Ophel (1963) suggested that when fish are exposed to radiostrontium, equilibrium (and concentration factor) can be reached quickly with the exchangeable fraction of fish tissue. However most of the radiostrontium in adult fish is retained in the skeleton, a considerable fraction of which is non-exchangeable. The 'true' equilibrium and concentration factor will therefore only be attained if the fish have formed all their bone in the contaminated environment. Even if water concentrations are uniform this equilibrium may take several year to be established. The fact that ^{90}Sr will be incorporated into the minerals of skeletons, in the case of teleosts this is primarily in the mineralogical form of apatite (Odum, 1957), suggests that depuration rates/biological half-times will be correspondingly protracted.

In a study considering Mediterranean species of fish, crustaceans, molluscs and algae from field and laboratory experiments (Cancio et al., 1973), concentration factors rarely exceeded 100 for any organism type and uptake appeared to be highest for the mollusc group. Activity concentration data from monitoring reports (e.g. FSA & SEPA, 2000, British Nuclear Fuel (British Nuclear Fuels Limited (BNFL), 1994) suggest that molluscs may concentrate ^{90}Sr to a higher degree than either fish or crustaceans. Data collated from Arctic marine environments (Fisher et al., 1999), report concentration factors of 182 ± 48, 4.1 ± 2.4 and 0.4-1.2 for brown macroalgae, fish muscle and seal muscle respectively.

An overview of recommended ^{90}Sr concentration factors for selected generic marine organism types is given in Table 3-1. Sr CFs appear to be lower for higher trophic levels species, i.e. successive trophic level concentration is < 1 in agreement with Whicker & Schultz (1982). Although Sr behaves conservatively in marine environments, under equilibrium conditions, activity concentrations per unit mass of sediment will be higher than those per unit mass of water. Target organisms, defined as those types most vulnerable to exposure from elevated levels of Sr in their habitat, are benthic infauna and epifauna. Of the benthic organisms considered above, macroalgae and possibly molluscs appear to accumulate the highest body burdens of radiostrontium and might therefore form suitable reference flora and fauna.

Tc-99

The most stable form of Tc in oxygenated seawater is pertechnetate (TcO_4^-), the negatively charged, high valence (VII), oxyanion (Beasley & Lorz, 1986). The pertechnetate ion is soluble with distribution coefficients (K_ds - activity per unit mass solid/ activity per unit mass liquid) for sediments low in organic matter rarely exceeding 1-4 (Beasley & Lorz, 1986). In contrast, K_d values for technetium for sediments high in organic matter can be appreciable - values of up to 1500 have been reported for this type of sediment (Masson *et al*., 1989). Tc-99 discharged from the western reprocessing plant at Sellafield appears to behave conservatively in the oxygen rich water masses of the North Sea and is transported by prevailing marine currents (Brown et al., 1999b).

Uptake of Tc by macroalgae can be significant. Concentration factors from a collation of data in the open literature (Dahlgaard et al., 1997, Smith et al., 1997, Brown et al., 1999b) suggest that values in the order of 10^5 are appropriate for the brown seaweed Fucus although higher CFs have been observed (Swift et al., 1989). Information for other species is scarce but the studies that have been conducted suggest that uptake of ^{99}Tc by red and green seaweeds is much lower (McCartney & Rajendran, 1997). Tc CFs for phytoplankton is also apparently much lower than for brown seaweeds (see International Atomic Energy Agency (IAEA), 1985).

Certain species of benthic crustaceans, e.g. lobster (*Homarus gammarus*) and Norwegian lobster (*Nephrops norvegicus*) are known to accumulate ^{99}Tc to a high degree (Busby et al., 1997, Brown et al., 1999b). Elevated concentrations in lobster are often found in digestive glands (Swift 1985) and differences between crustacean types, crabs having a much lower affinity for Tc, may be related to physiological specialisation (Swift, 1989). Uptake by some species of Benthic molluscs, notably the Mesogastropoda *Littorina littorea* can also be substantial with CFs exceeding 10^4 in some cases (Swift, 1989).

Information for zooplankton is not well documented although the fact that the meroplankton community may contain juvenile forms of species that are known to have a high affinity for Tc (e.g. crabs and lobsters) means that zooplankton may need to be considered as potentially vulnerable organism types.

Monitoring data that are available for northern European waters (e.g. British Nuclear Fuels Limited (BNFL), 1994, FSA & SEPA, 2000) lead us to the suggestion that fish concentrate ^{99}Tc to a much lower degree than many benthic invertebrate species. Schulte & Scoppa (1987) made the observation that low accumulation of Tc from water and food and short biological half-lives for Tc in marine organism prevents much Tc from passing to higher trophic levels. In the absence of data concerning Tc uptake to marine birds and mammals, the conclusion made in this earlier work lead us to the tentative suggestion that ^{99}Tc CFs for these organism groups is likely to be low.

An overview of recommended ^{99}Tc concentration factors for selected generic marine organism types is given in Table 3-1.

Although Tc has a low affinity for sediment, K_d/CF data suggest that where equilibrium conditions exists i.e. coastal sediments in prolonged contact with contaminated water masses, activity concentrations per unit mass in the sediment will be higher than those observed in the aqueous phase. Slightly higher external exposures may therefore be observed for benthic epifauna and infauna, compared to pelagic organisms, although the fact that ^{99}Tc is a soft β-emitter ($E_{\beta max}$=293 keV) suggests that exposures from the surrounding habitat will only be significant for small organisms with correspondingly thin cuticle/shell surfaces. From a basic consideration of the CF data reported in the open literature, 3 organism types can be identified as potentially vulnerable to exposure from coastal input of ^{99}Tc. These are brown seaweeds, benthic molluscs, in particular from the class Gastropoda, and crustaceans, in particular from the order Decapoda.

U & Th

The weathering of igneous rocks gives rise to ^{232}Th, ^{235}U and ^{238}U, long-lived radionuclides remaining from the primordial nuclesynthesis. The long-lived radioisotopes are the parents of of decay chains that contain raidisotopes of other elements including radium, radon, actinium, protactinium, polonium and lead. In igneous rocks, both thorium and uranium exist in the 4+ oxidation state. Uranium, unlike thorium, however, can be oxidized to the 5+ and 6+ oxidation states in the near-surface environment. The 6+ state is the most stable chemically and forms soluble uranyl ions (UO_2^{2+}) (Plater *et al.*, 1988). This ion can form soluble complexes with common anions in natural waters such as CO_3^{2-}, SO_4^{2-} and Cl^-. Moderate concentrations of complexing anions may inhibit the adsorption of the uranyl ion onto humic materials and iron oxyhydroxides, which are capable of high enrichment factors. This may account for the observed uranium mobility in natural waters (Ivanovich, 1994). In the 4+ oxidation state, uranium is almost chemically immobile in the near surface environment (Gascoyne, 1982). This results in the fact that U concentrations are very sensitive to reduction/oxidation conditions and the observation of large concentration variations (10^{-2} to 10^2 μg kg^{-1}) in groundwaters (Ivanovich, 1994).

Thorium has a tendency to hydrolyse to insoluble forms (Ivanovich ,1994). As a result of its relative insolubility in most natural waters (Langmuir & Herman, 1978), thorium is almost entirely transported in particulate form. The mobility of Th may be controlled by colloidal and polymeric species (Dearlove *et al*., 1991).

Kershaw & Young (1988) reported K_d values in the eastern Irish Sea for ^{238}U ranging from 0.75 x 10^3 -3 x 10^3, indicating the relatively conservative nature of uranium. In the same study K_d values for ^{234}Th were seen to range from 1.2 x 10^6 -4.5 x 10^6, indicating the particle-reactive nature of thorium.

The source of U, whether natural or technologically enhanced is likely to affect the geochemical phases association with sediments. For an estuary in south-west Spain receiving inputs of (technologically-enhanced) uranium and thorium series radionuclides, Martinez-Aguirre & Garcia-Leon (1994) reported that uranium was mainly associated with 'specifically adsorbed' and 'coprecipitated with oxyhydroxides' phases whereas for uncontaminated rivers that (i.e. no technologically enhanced levels of U and Th), draining into the Wash in east England Plater *et al.* (1992) showed uranium was predominantly associated with the residual geochemical phase. Th tends to be found in residual phases irrespective of whether the radionuclide is derived from natural or anthropogenic sources (Plater et al., 1992, Martinez-Aguirre & Garcia-Leon, 1994).

Activity concentrations in surface seawater have been reported as 44 mBq l^{-1} ^{238}U, 48 mBq l^{-1} ^{234}U, (0.2-5.2) x 10^{-2} mBq l^{-1} ^{230}Th, (0.4-20) x 10^{-3} mBq l^{-1} ^{232}Th, (0.7-12) x 10^{-2} mBq l^{-1} ^{228}Th and 1.9 mBq l^{-1} ^{235}U (Woodhead, 1973, International Atomic Energy Agency (IAEA), 1988). Activity concentrations per unit mass in sediments are generally much higher than in water. Levels in the range 5.6-63 Bq kg^{-1} ^{238}U (and ^{234}U, ^{230}Th in secular equilibrium), 4.4-96 Bq kg^{-1} ^{232}Th (and ^{228}Th in secular equilibrium) and 0.4-3 Bq kg^{-1} ^{235}U have been reported for selected sediment and rock types (International Atomic Energy Agency (IAEA), 1988). Fractionation or disequilibrium between members of the uranium-thorium decay chains in nature is brought about by a number of processes, including selective leaching of more soluble daughters in the decay series and the diffusive escape of radon gas which is a short-lived daughter in the uranium-thorium decay series. Alpha recoil and recoil-induced vulnerability can also lead to disequilibrium (Ivanovich, 1994).

Radioisotopes of uranium are not highly concentrated by the soft tissues of marine organisms (International Atomic Energy Agency (IAEA), 1988). However, uranium is incorporated into the skeletal materials and marked differences occur in such incorporation among different phyla (International Atomic Energy Agency (IAEA), 1988). Biogenic incorporation in addition to scavenging by non biogenic particles, e.g. clays are important processes leading to the ultimate transport of U to the seabed (International Atomic Energy Agency (IAEA), 1988).

For fish, apart from accumulation by the bones and scales, highest concentrations are found in the liver at levels of the same magnitude as those observed in seawater (International Atomic Energy Agency (IAEA), 1988). Molluscs appear to concentrate U to the highest degree of all marine organism types and express CFs in the range 30-100 (International Atomic Energy Agency (IAEA), 1985, Hodge et al., 1979).

Concentration factor data for Th are fairly limited and are often restricted to ‘not greater than’ values. Highest CFs appear to be associated with phytoplankton (Table 3-1), probably reflecting concomitant large surface areas where adsorption can take place.

Using the summary of CFs for different organism types (Table 3-1) and a knowledge of the biogeochemical behaviour of U and Th it is apparent that biota living in proximity to and within sediments are likely to be exposed to the highest external exposures of radiation. In view of the fact that isotopes of U and Th decay primarily by $\alpha-$ and $\beta-$emissions (low yield γ-photons and X-rays are also emitted) microbenthos as oppose to macrobenthos will be especially susceptible to external irradiation. In consideration of internal body loadings, (benthic) molluscs are notable for their enhanced accumulation of U. For Th, phytoplankton (benthic in shallow areas, pelagic in deep sea area) is selected as a potentially highly exposed biota group.

Ra, Po and Pb

Two radioisotopes of Ra are readily detectable in seawater, namely ^{226}Ra and ^{228}Ra. ^{226}Ra is an intermediate in the ^{238}U decay chain and ^{228}Ra an intermediate of the ^{232}Th decay chain.Their occurrence in seawater is mainly by diffusion from sediments via interstitial waters. Ra-226 is often depleted in surface water owing to biological accumulation and ^{228}Ra concentrations are observed to decrease with increasing distance from the sediment water interface (International Atomic Energy Agency (IAEA), 1988). A fraction of gaseous ^{222}Rn generated from the decay of non-volatile ^{226}Ra diffuses to the atmosphere where it decays, through a series of short-lived radionuclides to ^{210}Pb and returns to earth by precipitation or dry deposition. Aerosol ^{210}Pb falling directly on surface waters is subsequently removed to sediments where it is subsequently buried by accumulation. The final distribution of excess ^{210}Pb above that in secular equilibrium with *in situ* ^{226}Ra is governed by the rates of sedimentation, mixing (though physical disturbance and bioturbation) and radioactive decay (Hamilton et al., 1994). Pb-210 decays to ^{210}Po that is, despite its short half-life of 138 days, frequently out of equilibrium with ^{210}Pb. There is a preferential uptake of ^{210}Po by plankton and it is possible that ^{210}Po is recycled in surface waters (International Atomic Energy Agency (IAEA), 1988). Sediments also contain a supported component of all the radioisotopes mentioned above in equilibrium with the decay chain parents ^{238}U and ^{232}Th. Sediment-water concentration factor for coastal sediments are recommended as 5 x 10^3, 2 x 10^5 and 2 x 10^7 for Ra, Pb and Po respectively by International Atomic Energy Agency (IAEA) (1985). Phytoplankton is a significant accumulator of the radioisotopes of all 3 elements (see Table 3-1). Po-210 concentrations in different tissues of marine organisms vary enormously but on a whole body basis CF values in both pelagic and benthic food chains are thought to be similar at about 10^4 greatly exceeding that of ^{210}Pb at 10^2 (Heyraud & Cherry, 1979). Variations in ^{210}Po concentrations do not appear to be generally related to the trophic level in fishes, although large pelagic carnivores are often the highest concentrators (Pentreath, 1977). Species vulnerable to high habitat and internal/surficial concentrations of radioisotopes of Ra, Pb and Po include phytoplankton and benthic organisms, in particular crustaceans because of their tendency to accumulate ^{210}Po to a high degree.

Artifical actinides (Pu, Am, Cm and Np)

Plutonium exhibits four oxidation states (III, IV, V, VI) in the natural environment; americium predominantly exists in the (III) oxidation state. Data reported by Pentreath *et al.*, (1986) show that Pu (V) predominates in Irish Sea water. The International Atomic Energy Agency (IAEA) (1985) recommended K_d value, in coastal waters, for plutonium is 1 x 10^5 and for americium is 2 x 10^6. For nearshore Cumbrian waters (< 1.5 km from coast), McKay & Walker (1990) found K_d values of (2.1 ± 0.1) x 10^5 for ^{239,240}Pu and (1.0 ± 0.1) x 10^6 for ^{241}Am, consistent with IAEA recommended values. The plutonium oxidation states have a major influence on its K_d values, i.e. Pu (III & IV) = 1 x 10^6 and Pu (V & VI) = 1 x 10^4. Field measurements represent the properties of the particular mixture of oxidation states present (Nelson & Lovett, 1978).

If Pu and Am adsorption is dependent on the availability of reaction surfaces and equilibrium is reached quickly, a relationship between K_d and particle size should be established. Plutonium distribution coefficients do appear to be strongly dependent on particle size. The picture for ^{241}Am is more complicated (McKay & Pattenden, 1993). As an aside it might be worth considering that chemical composition, particle size distribution and suspended concentrations of particles in coastal waters can have large spatial and temporal variations in response to hydrographically-controlled resuspension and deposition. This will influence the K_d. The K_d term provides a static picture of a dynamic set of complex and not necessarily interrelated processes.

Plutonium can react in many different ways including hydrolysis, formation of polynuclear species and complexes with inorganic and organic ligands and polymerisation, i.e. colloidal formation (Sholkovitz, 1983). Sequential extraction experiments (Aston *et al.* , 1981; Aston & Stanners, 1981; Cook *et al.*, 1984; Wilkins *et al.*, 1984; McDonald *et al.* , 1990; McDonald *et al.*, 1992; Murdock *et al*., 1993) have shown the importance of the organic and sesquioxide fractions in the geochemical phase association of plutonium in sediments and soils. The concentration and speciation of plutonium appears to be related to the redox chemistry of iron (Sholkovitz, 1983).

The phase association of americium is less clearly defined. Wilkins *et al*. (1984) reported that the carbonate fraction was important in an estuarine sample but that the organic phase was dominant for a saltmarsh sample. Murdock *et al*. (1993) reported that a large fraction of americium was associated with the organic fraction in freshwater sediments from a stream near Drigg in Cumbria, UK. Although ^{241}Am exists primarily as a trivalent ion in solution, it may still form complexed species which behave analogously to plutonium (Murdock *et al.*, 1993).

The particle reactive nature of the actinides plutonium and americium means that the transport of contaminated sediment is often an important process leading to the local dilution and dispersion of these radionuclides in the marine environment (see McDonald *et al*., 1990; Brown et al., 1999a). Over greater distances (thousands of km), Pu is often transported as dissolved species (Kershaw & Baxter, 1995). There is no evidence to suggest the irreversible binding of plutonium and americium to sediments. Some 8 TBq of Pu(α) and 2.5 TBq of ^{241}Am are estimated to have been remobilised between 1979 and 1987 for the Sellafield offshore area in the Irish Sea (Hunt & Kershaw, 1990). Cook *et al.* (1997) recently reported low levels of plutonium redissolution for Irish Sea sediments.

Upon delivery to the ocean surface (of global fallout derived radionuclides), ^{239,240}Pu is rapidly (i.e. within a few years) removed onto biogenic particles. Within this suite of particles, certain types, upon sinking release Pu back to the subsurface water to form a maximum. This rapid release implies that a large proportion of Pu is held in labile fractions of the biogenic material which undergo decomposition and mineralisation at shallow depths. The existence of high Pu concentrations in bottom waters also indicates that a certain fraction of Pu is associated with large particles capable of rapid descent (Sholkovitz, 1983).

In view of the large surface area to volume ratio it is likely that many elements, in particular particle-reactive actinides, are accumulated by phytoplankton via adsorptive processes. This is reflected in the relatively high CFs derived for these organisms (Fisher et al., 1983). Detailed studies of the body burdens of actinides in plaice in the 1970s (Pentreath, & Lovett, 1976, 1977). Am-241 appears to be more available to plaice than Pu. It has generally been concluded that neither Pu nor Am derived from Windscale/Sellafield, is highly accumulated by benthic or pelagic fish (Pentreath et al., 1979). Rissanen et al. (1997) calculated a CF of 1 x 10^3 for ^{239,240}Pu for a ray sampled from the Barents Sea although values of <0.3 x 10^3 were derived for bony fish. In contrast the uptake to invertebrates and algae can be significant and the consumption of these marine-derived organisms is considered to be an important dose-forming pathway for man for Sellafield-derived radioactivity (Kershaw et al., 1992). Data from a sampling study in 1980 (Pentreath et al., 1982) were used to calculate CFs for various seaweeds and molluscs. Pu-239,240 CFs were generally above 1 x 10^3 for both organism types and CFs for ^{241}Am were generally slightly below and significantly over 1 x 10^4 for seaweeds and molluscs respectively. However it should be noted that the authors drew attention to the fact that the values may not have represented equilibrium conditions. Limited data on the activity concentration of Pu in marine mammals from northern European seas suggest that transfer to these organism types is very low (Brown, 2000). An overview of recommended Pu and Am concentration factors for selected generic marine organism types is given in Table 3-1. The data are consistent with the observation of Whicker & Schultz (1982) that successive trophic level concentration is low.

Curium exhibits multiple oxidation states, although Cm (III) is the most common. Radioisotopes of Cm are highly particle reactive expressing a sediment water concentration factor of 2 x 10^6 in coastal environments (International Atomic Energy Agency (IAEA), 1985). This tendency to adsorb to suspended particulate material probably accounts for the relatively high CFs observed for phytoplankton and organisms feeding on water-borne suspended particulates, e.g. certain molluscs. Recommended CFs for selected organism groups (Table 3-1) show that Cm, similar in its biogeochemical behaviour to Am, appears to be transferred to a limited extent along pelagic and benthic foodchains. Following uptake by mammals, the radioisotopes of Cm tend to accumulate in bone marrow.

Neptunium exhibits multiple oxidation states. Np appears to be less particle–reactive than the other artificial actinides expressing a sediment water concentration factor of 5 x 10^3 in coastal environments (International Atomic Energy Agency (IAEA), 1985). The limited data on the uptake of radioisotopes of Np by marine organism (Table 3-1) suggest that molluscs accumulate Np to the greatest extent under equilibrium conditions.

From the brief overview of the biogeochemical behaviour of Pu, Am, Cm and Np in marine systems it is apparent that for coastal areas, sediments will be a primary reservoir for these radionuclides. For this reason benthic organisms, especially those with concomitantly high concentration factors, examples include brown seaweeds and molluscs, might be considered vulnerable to (external and internal) exposure. Phytoplankton appears to be especially prone to the accumulation of high Pu, Am and Cm levels and therefore pelagic groups, in the open ocean, and benthic groups in the near coastal zone, should be included in the reference organism list.

C and H

Hydrogen is one of the few elements for which the sediment water concentration factor is < 1. The high carbonate, and in some cases organic, content of certain types of marine muds and shales has led to the derivation of a coastal sediment-water concentration factor of 2 x 10^3 for carbon (International Atomic Energy Agency (IAEA), 1985).

C-14 and ^3H interact with the marine ecosystem through their involvement in the hydrological cycle. C and H are key elemental components for life, and their radioisotopes are completely assimilated by living organisms (Whicker & Schultz, 1982). The high water content of marine flora and fauna leads to the simple assumption that hydrogen concentrations within the biota are equal to those in the ambient seawater. Concentration factors recommended by IAEA are therefore unity for all animal and plant groups. The wet weight tissue concentrations for C vary for marine organism ranging from 45 g kg^{-1} for phytoplankton to 95 g kg^{-1} for fish. Derived CFs reflect this variation in the C content of marine biota. It should be noted that for C, CFs relate to the organic component of C in seawater only as opposed to additional inorganic forms that would necessitate the inclusion of carbonate, bicarbonate and CO_2 concentrations in the calculation(International Atomic Energy Agency (IAEA), 1985).

All types of pelagic marine organisms would be exposed to similar levels of radiation following an input of ^3H to oceanic surface waters. The basic sediment-water concentration factor data suggest that sediment may act as a sink for ^{14}C over long time periods and that benthic organism might be vulnerable to the highest exposures from this radionuclide. Benthic fish, molluscs and crustaceans have similar tissue concentrations of C and therefore might be expected to experience similar levels of internal exposure following the equilibration of ^{14}C in the system.

Cl and I
Chlorine generally forms highly soluble salts in solution and is present as chloride (Cl^-) ions in seawater. Interaction with the sedimentary material is negligible and IAEA recommend the application of a sediment water CF for coastal sediments of < 3 x 10^{-2}. Chloride ions are essential for certain biological functions but recommended CFs are low (< 1) for the organisms listed in Table 3-1 reflecting the high ambient stable Cl concentrations.

Iodine is mainly found in the form of iodate (IO_3^-) in seawater and exhibits a nutrient type distribution with depth whereby a depletion of the element is observed at the surface of the water column and en enrichment is observed at depth (Millero, 1996).. Iodine is more particle-reactive than chlorine and expresses a recommended sediment water CF for coastal sediments of 2 x 10^1 (International Atomic Energy Agency (IAEA), 1985). Iodine is an essential element for life forming, for example an elemental component of thyroxine, which is important in the process of carbohydrate metabolism and protein synthesis and breakdown in humans. The highest accumulation of iodine occurs at lower marine trophic levels.

With respect to the environmental transfer and fate of Cl radionuclides, we are primarily concerned with the pelagic ecosystem. In view of the fact that ^{36}Cl decays by electron capture with the concomitant emission of β^+ and β^- particles, micro-organisms (e.g. phytoplankton and zooplankton) are likely to be those biota types most vulnerable to exposure in the marine ecosystem. Under equilibrium conditions, benthic organisms are likely to be exposed to higher concentrations of iodine in their habitat than pelagic organisms. Brown seaweeds are known to accumulate iodine and may be considered as a suitable reference organism.

Nb, Ni and Ru

Niobium forms multiple oxidation states but is normally present as Nb (V). It expresses a relatively high sediment water concentration factor of 5 x 10^5 for coastal sediments (International Atomic Energy Agency (IAEA), 1985). Data concerning the accumulation of Nb in marine organisms are scarce. In the derivation of CFs by International Atomic Energy Agency (IAEA) (1985), zirconium data are often used, in conjunction with ^{95}Nb/^{95}Zr ratios where appropriate. Recommended biota CF values for Nb illustrate that transfer to high trophic levels is limited and that zooplankton appear to accumulate radioisotopes of this element to the greatest degree although the derivation of the value reported in International Atomic Energy Agency (IAEA) (1985) for zooplankton is a best estimate only (Table 3-1).

Stable nickel forms a nutrient-type profile in open oceanic water. The element is depleted in surface waters and enriched at depth. The element is removed from surface waters by plankton or biologically-produced particulate matter. Ni is then regenerated in deep waters when organic particulate material is oxidised by bacteria (Millero, 1996). Ni is particle reactive and expresses a sediment water concentration factor of 1 x 10^5 for coastal sediments. Nickel is an essential trace element for many species. The nickel-dependent metalloenzyme urease is found in a wide array of different organisms, it has been isolated from various bacteria, fungi, and higher plants. Urease has a wide variety of functions. The primary environmental role is to allow organisms to use externally and internally generated urea as a nitrogen source. The element appears to be important in maintaining the healthy function of the liver in birds and mammals. The important role in the biological functioning for many species is reflected in CF values that although expressing a wide range appear to be generally similar for most organism groups (Table 3-1).

Ruthenium expresses a relatively low sediment-water concentration factor of 3 x 10^2 for coastal sediments. Some species of benthic macroalgae are known to concentrate ^{106}Ru to a high degree, notably *Porphyra umbilicalis* (Kershaw et al., 1992). It is of interest to note that in contrast to Tc, for example, green algae accumulate more Ru than brown algae (International Atomic Energy Agency (IAEA), 1985). Plankton (zoo- and phyto-) accumulate Ru to the highest degree of all marine organisms (Table 3-1).

In summary, the benthic habitat will be prone to the highest levels of accumulation of Nb, Ni and Ru. Owing to the relatively high accumulation of Nb and Ru by plankton, these organism types should also be considered vulnerable to exposure.

3.3 Summary

In summary of the discussion on marine ecosystems benthic organisms, phyto- and zooplankton and seabirds have been suggested for reference organisms. Benthic organisms like macroalgae, benthic fish, crustaceans, and molluscs are mentioned for their sensitivity or ability to concentrate radionuclides of different kinds. Table 3-2 summarizes the proposed reference organisms for the marine ecosystem.

Although the discussion on the marine ecosystem has shown that benthic organisms are the most likely species to be adversely affected by radioactive contamination, one must not rule out the possibility of other types of species being candidate reference organisms. An example of this is biomagnification in pelagic fish.

Table 3-2 Summary table of proposed reference organisms for marine ecosystem.

Element	Proposed reference organism
K-40	Benthic organisms (plants, epifauna and infauna)
Cs-137	Benthic organisms (Atlantic halibut, Plaice, crustaceans, skates, rays), Seabirds (black-backed gulls, great skuas)
Sr-89/90	Benthic in- and epifauna (molluscs, macroalgae)
Tc-99	Brown seaweed, benthic crustaceans (lobster), and benthic molluscs
U, Th	Benthic organisms (microbenthos), benthic molluscs (U), phytoplankton (Th)
Ra, Po, Pb	Phytoplankton, benthic organisms (particularly crustaceans)
Pu, Am, Cm, Np	Benthic organisms (brown seaweed, molluscs), phytoplankton
C, H	Benthic fish, molluscs and crustaceans (C), pelagic species (H)
Cl, I	Pelagic (phytoplankton, zooplankton) (Cl), Brown seaweed (I)
Nb, Ni, Ru	Benthic organisms (macroalgae), plankton

4. Brackish water (Baltic Sea)

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4.1 Ecosystem description

The Baltic Sea is a brackish, non-tidal, shallow sea in a formerly glaciated area (Håkansson, 1991). It was formed after the icecaps of the last ice age, some 12 000 years ago, first depressed the bedrock and then thawed. Today, the area of the sea is 365000 km^2, which makes it one of the largest brackish water areas in the world. The mean depth is only 60 meters while the maximum depth is 459 m. Since the sea is shallow, the volume is small and its dilution capacity (compared to oceanic environments) is small with respect to radioactive discharges and fallout. The dilution capacity is also reduced due to the slow exchange of water and strong vertical stratification (thermal and salinity). The theoretical retention time of Baltic Sea water is 22.5 years. The Baltic Sea is divided into several sill-separated sub-basins with limited water exchange in between. The compartmentalisation, combined with differences in fresh water influx, results in a north to south salinity gradient. The salinity increases from 2 ‰ in the northernmost parts to over 20 ‰ in the Skagerrak.

The organisms living in the area are immigrants from either marine or freshwater systems. Due to its short history, there are only a few endemic brackish-water species in the Baltic Sea. Thus, many organisms existing in the intermediate salinity of the Baltic Sea live under constant physiological stress and are therefore extra sensitive to additional stress such as from pollutants (Tedengren et al., 1988). The harsh environment and the relative youth of the Baltic Sea result in low species richness. In the Skagerack area there are 1500 species of macrofauna compared to 50 in the Bothnian Bay. This low species diversity increases the risk that whole functional groups can be lost (Kautsky & Kautsky, 2000). For example, the blue mussel, *Mytilus edulis*, is the major filter feeder in the Baltic Sea. If this species should be lost following an environmental perturbation of some sort, there will be no other species that can play the same functional role.

The large amounts of river discharges and precipitation are the reasons for the brackish-water character of the Baltic Sea and for its salinity gradient. In many respects the Baltic Sea resembles a large estuary, with out-flowing fresh surface waters and inflowing saline near-bottom waters. Besides the small number of species, the gradually decreasing salinity of water also causes considerable reduction in size of the marine organisms in the northern parts of the Baltic Sea (e.g. herring, blue mussel, bladderwrack). The low salinity also influences the metabolism of organisms, e.g. the uptake of many radionuclides is faster at low salinities.

The Baltic Sea covers an area corresponding to one thousandth of the world's total ocean surface. In 1985 it yielded 1 million tons of seafood, or 1 % of the world catch. The catchment area of the Baltic Sea is four times as large as the sea; it houses 10 % of Europe's population (90 million people in 2000) and accounts for nearly 15 % of the world-wide industrial production (Jansson et al. 1995). Human activities in the area combined with the low water exchange rates make the Baltic Sea one of the most polluted and otherwise perturbed marine ecosystems in the world. The main problems are: 1) eutrophication, which favours growth of nuisance algae and causes a reduction of oxygen contents in the sediments 2) over-fishing of cod and herring, which has caused significant ecological and socio-economic repercussions and 3) pollution from metals and organochlorine compounds, which has caused reproductive failure in several wildlife species (Kautsky & Kautsky, 2000).

The Baltic Sea is situated at high latitudes (N 54°-N 66°). The northern parts are covered with ice during the period December to May. In the spring, when ice and snow-melts and the ground thaws, pulses of nutrients, contaminants and fresh water are washed out into the Baltic Sea. The nutrient dynamics are thus tightly coupled to temperature. In the spring, when light conditions become favourable and nutrient levels are high, massive phytoplankton blooms occur. These spring blooms, consisting primarily of heavy diatoms, sink down to the deeper benthic habitats. Bottom-living organisms receive a significant part of their yearly energy requirements from these spring bloom events. During the summer, zooplankton dominate the pelagic environment and nutrients assimilated by pelagic organisms sink down to greater depths. Therefore, in the late summer (Aug-Sep) when nitrogen levels are low and limit phytoplankton growth, cyanobacteria (bluegreen algae), capable of converting inorganic nitrogen from the atmosphere into organic nitrogen, can give rise to massive blooms that can be poisonous and foul smelling.

4.1.1 Ecological niches and habitats

The various habitats in the Baltic Sea may be subdivided into three main zones: (a) the coastal zone with two kinds of phytobenthic communities (soft and hard bottom community), (b) the pelagic zone (the open water mass) and (c) the deep soft bottoms.

The coastal zone in the Baltic Sea is typically shallow with lots of skerries and small islands. This area harbours the highest biological diversity and also exhibits the highest productivity. Many organisms that live most of their lifetime in other habitats depend on the coastal area for spawning grounds. Herring, for example, come in to shallow archipelago areas to reproduce. The coastal zone is also highly valued and utilised by humans for recreational purposes such as boating, fishing and diving.
The rocky hard bottom community, which is one of the phytobenthic habitats in the coastal zone, are dominated by macroalgae, primarily bladderwrack (*Fucus vesiculosus*), together with blue mussel *(Mytilus edulis*) and is the habitat that sustains the highest species diversity in the Baltic Sea (Kautsky, 1991). The second type of habitat, the shallow sandy and soft bottoms are also extremely productive and biodiverse habitats, occupied by often dense populations of aquatic phanerogams e.g. *Potamogeton* spp., eelgrass *(Zostera marina*) and reed *(Phragmites spp.*), and are important spawning grounds for several species of fish (e.g. flounder, turbot).

The pelagic zone is the free water mass in both coastal and open sea areas. The energy flux in the pelagic food web is driven by photosynthesising phytoplankton and bacteria. Energy assimilated by the various phytoplankton subsequently sustains other grazing (zooplankton), and predatory species (zooplanktivore and piscivore fish) at higher trophic levels. Primary production (photosynthesis) is nutrient limited and is largely dependent on nutrient influx from the coastal area or from atmospheric deposition. The major energy flow passes through pelagic bacteria.

The deep softbottom zone is the soft bottoms below the light compensation depth (i.e. no primary production occurs). Owing to the distance from the surface and the layering of waters above, the temperature is quite constant between 2-6°C. Energy input to this habitat comes from particle fallout from the coastal and pelagic systems. The sediments vary in composition and character due to differences in sedimentation rates and water movements. An environmental factor, which to a large extent governs the structure and function of the benthic food web, is the oxygen concentration. In areas where the deposition of organic matter is high, oxygen levels often become low, since oxygen is used in the bacterial break-down of organic matter. Extended areas of the deeper soft bottoms are anoxic, and are therefore devoid of macroscopic life (only anaerobic bacteria exist there). The benthic food web is dominated by decomposers and detritus feeders, but also predatory invertebrates and fish as well as occasional filter feeders can be found. The benthic community is often dominated by the Baltic Sea mussel, *Macoma balthica*. Other burrowing fauna of interest are the amphipods *Monoporeia affinis* and *Pontoporeia femorata,* the isopod *Saduria entomon* along with some ostracods, copepods and nematodes (Figure 4-1).

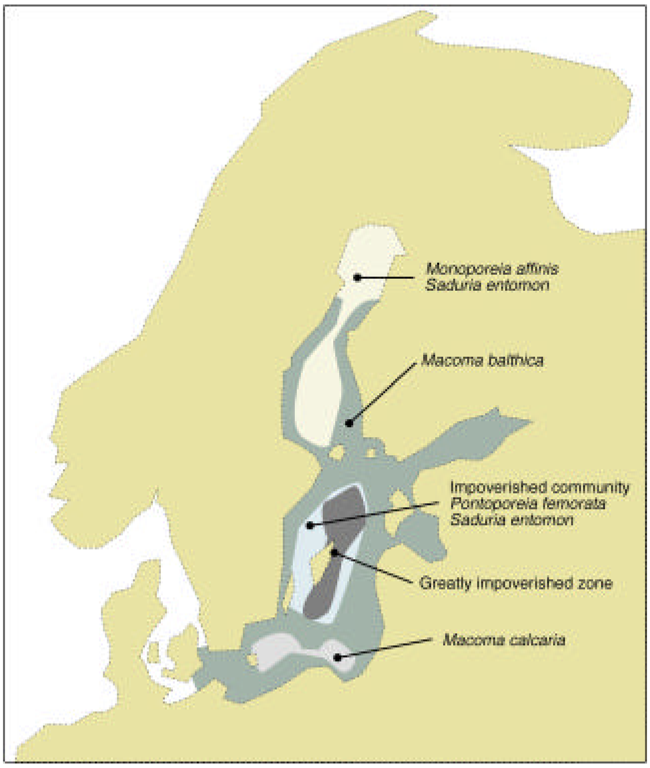

Figure 4-1 Dominating species in the benthic zone of the Baltic Sea (after Zenkevitsch, 1963).

4.1.2 Typical species

Benthic plants

The bladderwrack *(Fucus vesiculosus*) is a large, kelp-like, marine brown alga that forms dense belts on shallow (1-10 m) rocky surfaces and is of great ecological importance in the Baltic Sea area up to the Åland archipelago and the SW Finnish coast. With its rigid structure held buoyant by vesicles, it provides shelter, food and egg laying surfaces for several organisms. During the last decades, the depth distribution of the bladderwrack has decreased (Kautsky et al., 1986). The main factor responsible for this is eutrophication, since increased nutrient levels enhances overgrowth of the bladderwrack by annual fast growing filamentous algae and a decreased light compensation depth. A rich fauna of mussels, snails, crustaceans, bryozoans and even insect larvae inhabits the bladderwrack community. This is the most species rich system in the Baltic Sea, containing some ten species of algae and 30 species of macroscopic animals (Kautsky, 1991). By being the only large, long-lived, belt-forming algae on Baltic rocky bottoms, bladderwrack is a key species in these habitats, and in fact also in the whole Baltic Sea (Kautsky, 1991).

Aquatic phanerogams such as *Potamogeton* spp. and eelgrass *(Zostera marina)* grow on unexposed soft bottoms in the coastal zone down to a depth of approximately 3 meters. These 'meadows' of submerged plants can become quite dense and provide an important habitat for a multitude of other organisms. For example, shallow soft bottoms are spawning grounds for several fish species.

Filamentous algae are commonly found in the phytobenthic zone both on hard substates and as epiphytes on larger species. For instance, the filamentous green alga *Cladophora glomerata* typically occupies the shallow littoral zone (0-1 m) on rocky shores along the whole Baltic Sea coast. *Ectocarpus spp.* and *Pilayella spp.* are two very common filamentous brown algae that often grow as epiphytes on the bladderwrack. The red algae *Ceramium spp.* is also a characteristic filamentous alga in the Baltic Sea and is abundant along shallow (0-5 m) rocky coasts. These species are especially interesting from the point of view of environmental impact assessment, since *Ceramium* is one of the few indigenous Baltic Sea macrophytes for which a standardised toxicity test has been developed (Eklund, 1998).

Plankton

There are three major types of plankton found in the Baltic Sea, cyanobacteria, phytoplankton and zooplankton. The pelagic phytoplankton and cyanobacteria (blue green algae) are all primary producers and different taxa dominate during separate periods of the year. The spring bloom starts in late February or early March mainly by growth of diatoms (e.g. *Thalossisera spp.* and *Skeletonema costatum*) and dinoflagellates (e.g. *Dinophysis spp.* and *Ceratium spp.*). Small forms of monads and dinoflagellates often dominate during the summer and larger dinoflagellates, green algae and diatoms dominate the autumn bloom. The bluegreen algae species *Aphanizomenon spp.* and *Nodularia spp.* are responsible for the pelagic blue green algal blooms, typically occurring in late summer and fall. *Nodularia* can be poisonous and every year domestic animals die after drinking water with high concentrations of this alga. Zooplankton species commonly occurring in the Baltic Sea are, for example, calanoid copepods e.g. *Acartia tonsa*, cladocerans e.g. *Bosmina spp.*, rotatorians e.g. *Synchaeta spp.* and the jellyfish *Aurelia aurita.* Zooplankton may graze upon the phyto- and bacterioplankton or function as carnivorous plankton, consuming smaller zooplankton. In the water mass there are also larger amounts of other bacteria than the cyanobacteria, which decompose pelagic detritus. Some of these bacteria are also able to utilize dissolved organic matter as a carbon source.

Benthic invertebrates

The blue mussel (*Mytilus edulis*) dominates the animal biomass (up to 90 % and 15 kg m^{-2}) on shallow rocky surfaces (2-20 m). The blue mussel is a filter feeder living attached to hard substrates in large, dense populations. Since it captures small organic particles (down to 1 µm) which otherwise would have stayed in suspension, the blue mussel acts as a link between pelagic production and benthic consumers. Apart from funnelling carbon and nutrients to the bottom, the blue mussel has also been shown to increase the deposition of contaminants bound to particles by up to 50 % (Gilek et al., 1997). The blue mussel is also an important food source for several bird species e.g. the eider duck (*Somateria molissima*) and the long-tailed duck (*Clangula hyemalis*). Due to the low salinity of water the blue mussel is smaller than in fully marine areas and does not exist in the Bothnian Bay and Eastern Gulf of Finland.
The crustaceans *Idotea balthica* and *Gammarus spp.* are common in the vegetation zone of the Baltic Sea up to the Kvarken area and are important grazers on bladderwrack (Malm, 1999). These species also constitute an important food-source for small-sized and juvenile fishes.
The Baltic Sea mussel, *Macoma balthica*, is a predominantly deposit feeding bivalve but may also under certain condition filter feed. The Baltic Sea mussel is present in the whole Baltic Sea and dominates the biomass of most soft-bottom communities. In the same community, the crustaceans *Monoporeia affinis* (detritivore on deep soft-bottoms), *Pontoporeia femorata* (detritivore on deep soft-bottoms), *Saduria entomon* (predator and scavenger on deep soft-bottoms) are common. There are a few polychaete worms present in the Baltic Sea. The most common is the predatory *Harmothoe sarsi*, which inhabit soft bottoms and the omnivorous *Nereis diversicolor*, which can be found on shallow soft and rocky bottoms. On shallow bottoms the grazing and detritivorous gastropods *Hydrobia spp.*, *Potamopyrgus spp.*, *Theodoxus fluviatilis, Bithynia tentaculata* and *Lymnea spp.* may be found in large numbers on vegetation or on the benthic microalgae.

Fish

Cod (*Gadus morrhua*) is a pelagic predatory fish, which may reach sizes of up to 30 kg. It is of marine origin and is intensively harvested in commercial fisheries. Population numbers have declined over the last decade due to low reproductive success and over-fishing. Pike (*Esox lucius*) is of limnic origin and is a ferocious predator in the coastal zone. It typically stalks its prey from hidden positions in the benthic vegetation. It reaches sizes of up to 20 kg and has cannibalistic habits. The meat has low fat content. Pike has no commercial value but is one of the most popular species in recreational fishing. Perch (*Perca fluviatilis*) is an omnivorous, warm water species of limnic origin with lean white meat. The perch is a popular food and recreational fish (but is not commercially fished). Salmon (*Salmo salar*) and other salmonid fishes are carnivorous, cold-water fishes with a high fat content of the meat. Salmon migrates annually between marine and freshwater environments to spawn and is of intermediate commercial interest. Since the meat has a high fat content, salmon is known to accumulate organic compounds such as PCBs). It may reach a size of up to 25 kg. Baltic herring and sprat (*Clupea harengus* and *Sprattus sprattus*) are pelagic, planktivorous fish, which form large schools. These species are intensively harvested in commercial fisheries and are mostly used for domestic animal feed and aquaculture feed. Due to the high fat content of the meat, high concentrations of various organic compounds such as PCBs, may be found. Commercially, bottom-living fishes such as flounder *(Platichtyes flesus*) and turbot (*Scopthalmus rhombus*) are common in the Baltic proper and are dependent on shallow soft bottoms for spawning and juvenile growth. Other common fishes in the Baltic Sea are for instance various gobides (e.g. *Gobius minutus* and *Gobius niger*), burbot (*Lota lota*) eel-pout *(Zoarces viviparus*), lumpfish *(Cyclopterus lumpus*), lavaret (*Coregonus pidschian*) and common bream (*Abramis brama*).

Birds
Sea birds dependent on food produced in Baltic Sea are, for instance, the eider duck *(Somateria molissima*), long-tailed duck (*Clangula hyemalis*), white-tailed eagle *(Haliaeetus albicilla*), cormorant *(Phalacrocorax carbo*), and Razorbill (*Alca torda*).

Marine mammals
Three different seal species can be found in the Baltic Sea. The Grey seal *(Halichoerus grypsus)* can be found in the whole area but is rare in the southern part of the sea. Harbour seal *(Phoca vitulin*a) is common in the southern most part of the Baltic Sea and the third species, ringed seal *(Pusa hispidia*), has its main area of distribution in the northern part.
Mink, *Mustela vison,* a species introduced by fur farming, is frequently encountered along the Baltic Sea coast. Its larger relative, the Eurasian Otter (*Lutra lutra*) is today very rarely seen.

4.1.3 Food webs

Since the Baltic Sea is species poor it has been possible to study species interactions to a large extent (Figure 4-2). Both interactions and flow of energy and matter are well described, e.g. by Kautsky & Kautsky (2000).

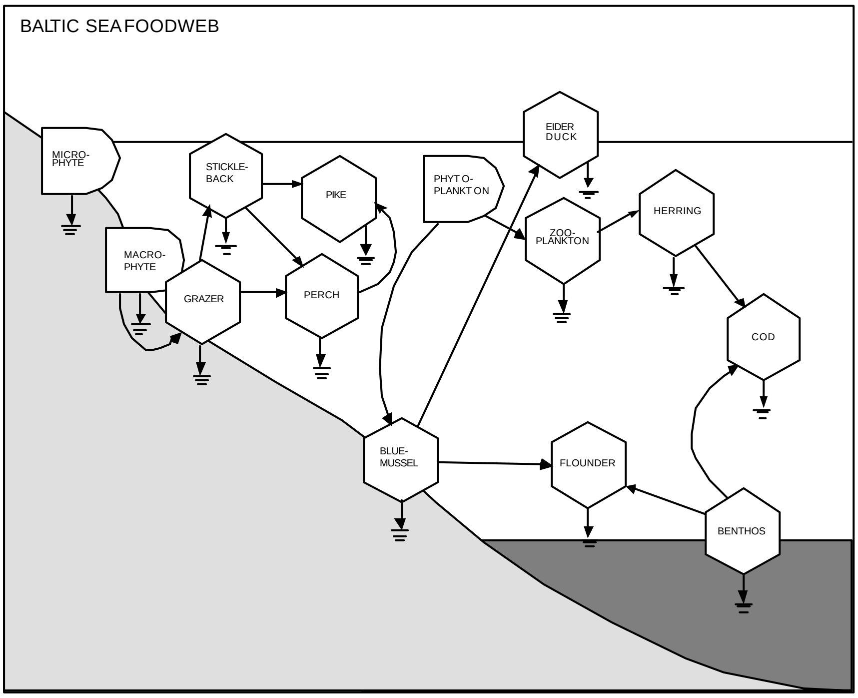

Figure 4-2. Typical food webs for the coastal area, pelagic area and the benthic area of the Baltic Sea.

4.1.4 Linkages within and to other ecosystems

Import of matter, goods and services
The Baltic Sea is mainly linked to terrestrial ecosystems by rainwater runoff and river discharges. The human society in the catchment area contributes to the import of substances to the Baltic Sea water with, for example, wastewaters, industrial effluents, atmospheric transport of contaminants and nutrients. Especially agricultural activities especially leak various chemicals, e.g. pesticides and nutrients, partly due to the low retention capacity of some soils in the area and extensive drainage from agricultural land.

Export of matter, goods and services
The main export to human society from the Baltic Sea is fish. A large fraction of the annual Baltic Sea fish production is harvested every year by commercial fisheries. Another important mobile link (organisms with ability to transport substances within and between ecosystems) are the sea birds. In the Baltic Sea, seagulls *(Larus spp.)* and other colony forming species (e.g. *Phalacrocorax spp.* and *Alca*) can catch fish and by defecating concentrate substances in the proximity of the colony, e.g. in terrestrial environments. Another aspect of ecosystem linkage is the landrise phenomenon in the Baltic Sea area. When the land rises, the seabeds will transform into soil in a terrestrial system. The reclaimed land will be fertile and ideal for agricultural activity, enhancing the potential for transport of matter to the human society. This can be seen as a link in a longer time perspective.

Internal processing
The coastal zone often acts as spawning and nursery areas for many pelagic species of fish. This contributes to a considerable transport of biomass between communities. The filter feeding activity of blue mussels constitutes an important link between the pelagic zone and the benthic organisms, by funnelling energy, nutrients and contaminants to coastal benthic communities. The filter feeding by the blue mussels may also contribute to upholding primary production during the summer months, e.g. by nutrient cycling.

4.2 Exposure pathways for radionuclides in the Baltic Sea

Since the Baltic Sea is a semi-enclosed and shallow sea with long turnover time, substances will remain for a relatively long period compared to other seas, which may result in greater accumulation of hazardous substances (e.g. radioactive materials) in biota and abiotic compartments than in other seas. Also, the influence of rivers in the catchments area is relatively large, which is of importance for the long-term balance of radionuclides following accidental releases (Holm, 1996).

The transfer of radionuclides present within aquatic environments is affected by both living and nonliving components and their environmental behaviour and turnover is largely determined by transport of the dissolved and solid phases, chemical interactions between phases and biological cycling (Warner & Harrison, 1993). Since many radionuclides tend to accumulate to particulate material, Baltic Sea soft bottoms often act as an effective sink for those substances. Eventually, the sediments may become sources of radioactivity if (1) the bottoms are oxic and dwelled by benthic organisms, if (2) the discontinuity layer is only slightly or not developed or if (3) the water current at the sea bottom is high enough to transport sediment particles (Weiss, 1989).

Aquatic organisms can accumulate radionuclides directly from the water or receive them by ingestion of contaminated food (Grimås, 1991). The ability of different organisms to actively accumulate and concentrate radionuclides varies greatly. Some radionuclides are readily absorbed since they are essential for the specific organism while others are accumulated since they show a chemical similarity to other elements, which take active part in the exchange processes with the surrounding medium. Furthermore, some radionuclides adsorb to the cell surfaces, and the element content of the organism is in those cases merely a function of the surface-to-volume ratio (Evans, 1991). The concentration and turnover within the organism is largely controlled by structural and dynamic properties of the food web, such as trophic level, body size and metabolic rate (Meili, 1991).

There are several introduction pathways for radionuclides into the Baltic Sea food web. One entrance pathway is via macroalgae, which have been shown to be able to accumulate extensive amounts of dissolved radionuclides from seawater (e.g. Carlsson, 1990; Holm, 1996). Another introduction pathway is via bottom dwelling organisms, such as the deposit-feeding Baltic Sea mussel (*Macoma balthica*), which for instance demonstrated increased Cs-137 concentration after the Chernobyl fallout (Evans, 1991). The concentrations were three times higher in the mussel compared to the sediment. A third entrance pathway into biota is diffusion of radionuclides directly through body integuments, e.g. over gill membranes.

Many radionuclides have a tendency to be enriched while being transferred upwards in the food web, i.e. from primary producers at the lower level of an ecosystem to the consumers (predators). However, the tendency of various radionuclides to be transported through the food web varies greatly depending on chemical properties of the radioactive substance. A reduction of the number of radionuclides is often shown when moving from primary producers up to first or secondary consumers. In a Baltic Sea study by Evans (1991), a substantial discrimination in radionuclide uptake was observed along food chains, where Co-60, Zr-95, Nb-95, Ru-103, Ru-106, Ag-110m, I-131, Cs-134 and Cs-137 were recovered in primary consumers like gastropods, bivalves and crustaceans, while Cs-134 and Cs-137 were the only nuclides frequently observed in fish muscle tissue. However, the concentration of radionuclides generally decreases when going from vegetation to fish. Going from primary producers to consumers at various trophic levels of the ecosystem, the concentrations of various radionuclides are regulated by factors like feeding habits and the physiology of the organisms. Transport through the food chains takes time, i.e. there is a delay in time for each step taken in the food chain. This general time lag in the ecosystem has been demonstrated in many investigations of Swedish waters (e.g. Evans, 1988).

Since radionuclide accumlation due to food consumption is one entrance pathway into biota, transfer through the food web will follow if the organism does not have the ability to actively excrete the substance. In Figure 4-3, possible pathways for radionuclides through the Baltic Sea food web are shown.

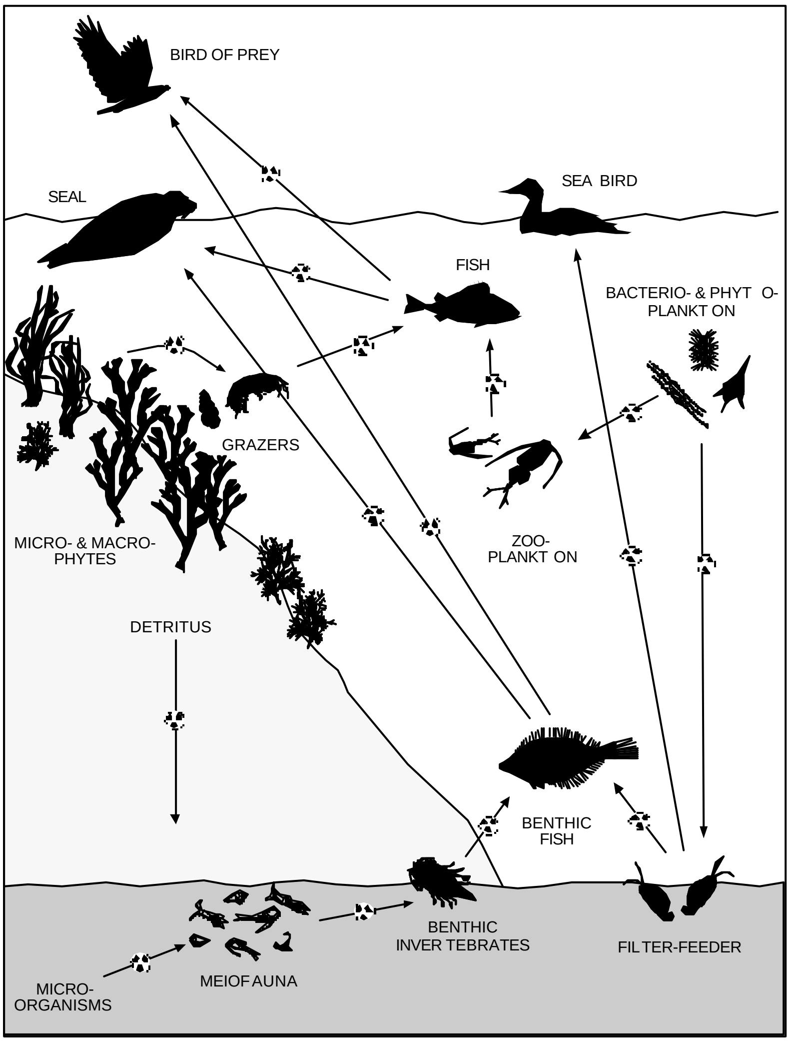

Figure 4-3 Some possible pathways for radionuclides through the Baltic Sea food web.

4.3 Approaches to select reference organisms

The approach in using reference organisms, is a way to enable and facilitate assessment of environmental impact of radionuclides. The authors believe that the appropriate choices of reference organisms only can be made if they are based on the context of the goals of the assessment (e.g. generic/site specific) and the particular ecosystem and impact of interest. Furthermore, it is believed that the most important criteria for selecting organisms are their ecological and societal importance.

It is anticipated that a satisfactory way to do this would be to identify possible transfer pathways for the radionuclides in the food web of the ecosystem as a first step. When the pathways are identified, the next step may be to pinpoint the different target organism groups (functional groups) and select relevant species from these groups according to the following criteria:

- The most abundant species (e.g. *Macoma balthica*).
- A representative species from each functional group (e.g. primary producer, filter feeder).
- A representative species for each morphologically distinct organism group in the system (e.g. filamentous algae, perennial fanerogame).
- A representative species from each trophic level in the system (e.g. primary producer, grazer, herbivorous fish, predatory fish).
- A keystone species (e.g *Mytilus edulis*).
- A sensitive species
- An important species in terms of human interest in the system (e.g. cod).
- Organisms with great difference in body size (e.g. crustacean plankton ⇒ salmon).
- Species with different fat content (e.g. pike and salmon).

For instance, it is important that the species should be abundant and/or a keystone species in the ecosystem (e.g *Macoma balthica, Mytilus edulis*) and that they are sensitive to radioactivity or make up a food source for radiosensitive organisms. It is also of importance to select representative species from each functional group (e.g. primary producer, filter feeder) and each trophic level in the ecosystem (e.g. primary producer, grazer, herbivorous fish, predatory fish) as well as representative species for each morphologically distinct organism group in the system (e.g. filamentous algae, perennial fanerogame). The selected species should also be of importance in terms of human interest in the system (e.g. cod). To summarise, the authors believe (i) that endpoint selection should be based on the goals of the assessment and what it is desired to protect, and (ii) that there is a danger of having an over-rigid set of criteria built into the regulatory framework, since the optimal set of reference organisms will vary considerable depending on objectives, and specific ecosystem and impact of interest. This topic will be further discussed and analysed in WP 4.

In 1985, HELCOM (Helsinki commission) established a group of experts (EC MORS) for the monitoring of radioactive substances in the Baltic Sea, which has carried out a comprehensive investigation and monitoring of the Baltic Sea brackish water environment. Numerous measurements have been made on seawater, sediment, and various species of biota, which are reported in *Radioactivity in the Baltic Sea 1984-1991* (HELCOM, 1995). The biota samples in the HELCOM-project were selected according to three criteria:

1. Organisms directly of importance for human consumption,
2. Organisms of importance in food webs ultimately leading to human consumption and
3. Organisms useful as bioindicators for radionuclides.

4.4 Modelling approaches and expert judgement

The transfer of radionuclides within aquatic food webs is often modelled by using bioaccumulation factors that describe the ratio of the radionuclide concentration in the organism and the surrounding water. Since bioaccumulation factors often vary widely between different organisms and environments the model predictions may contain significant uncertainties.

In traditional environmental dispersion models, chemistry and hydrology are often in focus, few consider biological transport processes. Alternative modelling approaches are to express the bioaccumulation factors as functions of known environmental variables (e.g. Rowan & Rasmussen, 1994), or to use mechanistic models where for instance the dynamics of uptake and loss of radionuclides in both biotic and abiotic compartments are incorporated. Because of the complexity of ecosystems, such models tend to be large and site specific and often require a lot of input data.

4.4.1 Examples of existing dispersion models for radionuclides in the Baltic Sea

A box model for the calculation of the collective dose commitment from radioactive waterborne releases to the Baltic Sea was published by Evans (Evans, 1985). The model calculates radionuclide concentrations in 25 boxes for discharges into given compartments and takes into consideration the dispersion rates of the water masses, the sediment-water interaction and radioactive decay. The calculated concentrations in water and sediment at steady-state are then used to evaluate individual and collective intakes of activity and external exposures by various organisms. This model is an example of a large-scale approach (the whole Baltic Sea area) where the main contributing abiotic factors influencing the transport and fate of radionuclides are identified and used. The concentration of the radionuclides in the abiotic compartments, water and sediment are calculated and these values are multiplied by site specific transfer factors for each box to yield the concentration of radionuclides in biota (Figure 4-4). Evans concludes that this model can be used as a screening tool to study controlling mechanisms for the dispersal of radioactive materials in the marine environment.

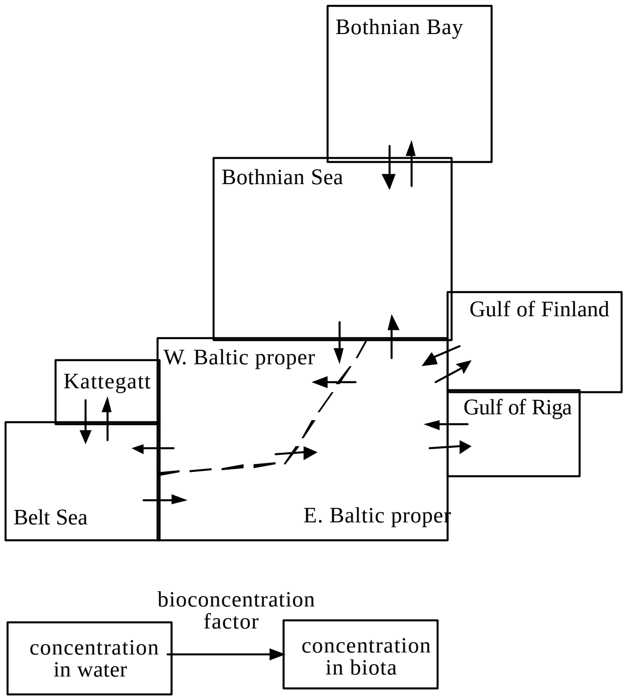

Figure 4-4 Schematic overview of the Baltic Sea compartment model developed by Evans (1985). The western and eastern Baltic proper were driven in two separate water layers. The radionuclide concentrations were derived by using bioconcentation-factors.

For the final repository for radioactive operational waste (SFR) in Forsmark, Sweden, two different dose assessment models are being developed. These models predict the transport, distribution and uptake by biota of relevant radionuclides after a hypothetical release into the Baltic Sea bay above the repository (Karlsson *et al*. in press, Kumblad *et al*. in press).

The model platform in preparation by Karlsson *et al.* [in press], accounts for the transport and distribution of the selected radionuclides in various ecosystem types, their uptake by biota, and radiation doses from a multitude of exposure pathways. The model is based on element specific distribution coefficients (K_d) and bioaccumulation factors obtained from the literature (Figure 4-5). This model is described in detail in a Technical Report that soon will be published in SKB's (Swedish Nuclear and Waste Management Co) Technical Report series.

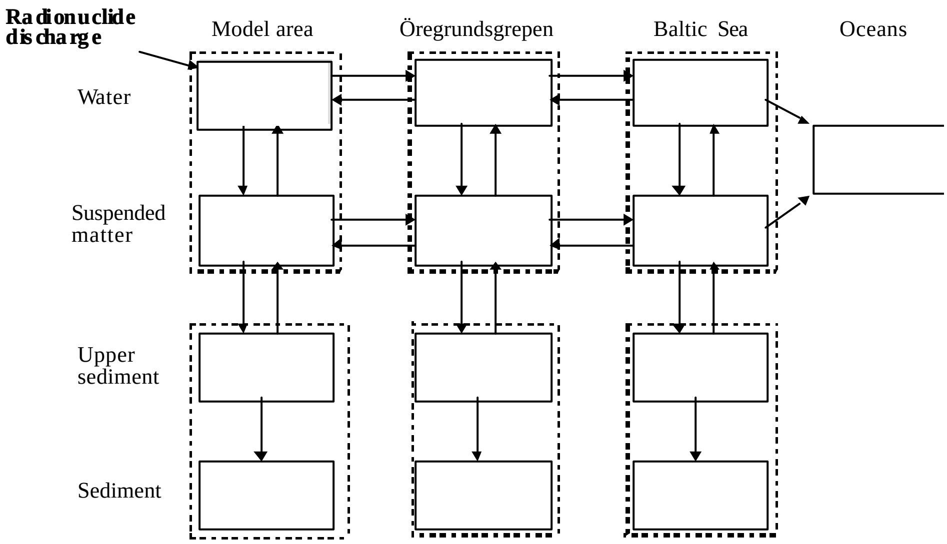

Figure 4-5 Structure of the coastal model used in the study by Karlsson et al. (in preparation). Three levels of scale are considered: the model area (small, close and dynamic), Öregundsgrepen (intermediate) and the Baltic Sea (large, remote and less dynamic).

In Kumblad *et al* (2001 in press) the environmental transport and fate of the hypothetical discharge from the repository was investigated with an ecological food-web model. This approach involves identification, quantification and modelling of the flows and storages of energy (carbon) in the system both in the physical environment and in the food-web. The radionuclide transfer is in the model driven by the metabolic rates (gC/day) of the various functional groups of organisms and turnover rates for some abiotic compartments, which enables prediction of the concentration (Bq/gC or Bq/m^3) of radionuclides in the compartments (Figure 4-6). The developed model was also used to evaluate implications of various assumptions concerning the route of radionuclide entry in the food-web and the rate of water exchange in the studied ecosystem.

In both models (Karlsson *et al.* in press and Kumblad *et al.* in press) calculated ecosystem specific dose conversion factors (EDFs) were of the same order of magnitude for comparative simulations. Model results from Kumblad *et al* [in press], gained in simulations with varying assumptions for uptake pathway and water retention time, varies several orders of magnitude demonstrating the necessity of understanding ecosystem functioning and the importance of mechanistically describing the fate of released pollutants.

The dynamic model developed by Kumblad and co-workers is site-specific and so far only valid for C-14. However, the authors intend to adapt the model, within the FASSET-framework, to other radionuclides of interest and to make it generic and thus useful for more general assessments of radionuclide transfer in aquatic environments.

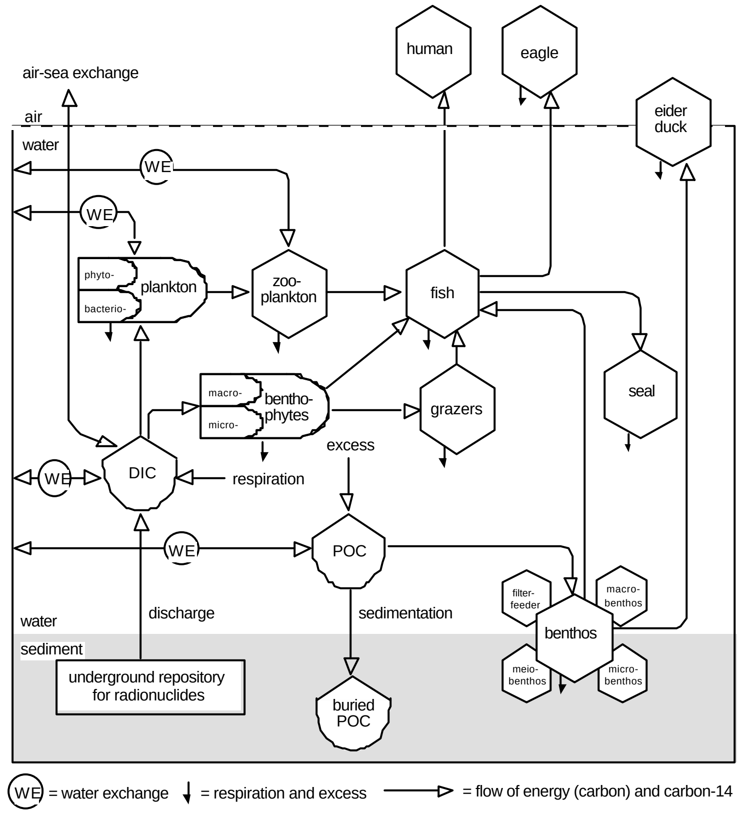

Figure 4-6 Conceptual framework used in the model by Kumblad et al. (in press) The radionuclide is bioaccumulated in proportion to consumption or uptake by plants, thus no bioaccumulation factors are used.

4.5 Distributions of radionuclides in flora and fauna in the Baltic Sea

There are many differences between radionuclides concerning their ability to accumulate in aquatic organisms, e.g. the caesium isotopes have a tendency to be enriched in fish flesh, whereas strontium mainly accumulates in bones (e.g., Kanisch et al., 1995). In the case of caesium it is obvious that the salinity of the water affects the bioaccumulation; the uptake being stronger at low salinities (e.g. Agnedal, 1986).

In a study of radioactive caesium and plutonium in water, sediment and macroalgae, it was concluded that the outflow from rivers was of importance for the overall radionuclide balance in the Baltic Sea (Holm, 1996). Moreover, it was stated that the ecological residence time was long and further investigations were needed to be able to develop a model for the turnover of radionuclides, which is important for the estimation of the radiological impact in the Baltic Sea and its catchment area. It was also stated that a significant fraction of radiocaesium from rivers entering the Baltic Sea is in soluble form, or is dissolved in the interface between fresh and brackish water, which influences the radionuclide distribution in the ecosystem (Holm, *op. cit*.). Since the Chernobyl accident, there has been a net outflow of radiocaesium to the North Sea by the Baltic Sea surface water, which has higher concentrations than the more saline, inflowing near-bottom waters, while plutonium is mainly trapped into the sediments.

The Baltic Sea was the marine area most affected by the Chernobyl accident, because the first radioactive clouds from Chernobyl travelled north and caused high deposition in the Baltic Sea region. Due to the semi-enclosed nature of the Baltic Sea and its small exchange of water with the North Sea, the levels of ^{137}Cs there have remained the highest in marine environment in Europe (Povinec et al., 1996). In the initial phase the fallout was very unevenly distributed due to the different strength of rainfall in various regions. The highest deposition values of Chernobyl-derived ^{137}Cs in the catchment area of the Baltic Sea occurred in the areas surrounding the Bothnian Sea and the Gulf of Finland. A substantial difference was also observed between the coastal areas and the open sea in that the concentrations were highest near the coasts (Ilus, 1999). During the years after the accident, the distribution pattern has changed due to sea currents and mixing of water masses. The Gulf of Finland has been cleansed more rapidly of fallout nuclides than the Bothnian Sea due to its efficient exchange of water with the Baltic Proper. At the same time, the concentrations of fallout nuclides have increased in the Baltic Proper (Ilus, *op.cit*.).

Since 1986, the levels of ^{137}Cs in fish samples caught from the Baltic Sea were largely explained by the varying amounts of initial fallout from Chernobyl in different subregions (Kanisch et al., 1995). The slow counterclockwise circulation of surface water in the Gulf of Bothnia, Gulf of Finland and Baltic Proper, and the net southwards transport of contaminated water masses along the Swedish coast have had a clear impact on time trends found in fish samples caught from different areas.

Subareas with relatively high levels of Chernobyl fallout (Bothnian Sea, Gulf of Finland, Åland Sea, Archipelago Sea and northern Baltic Proper) were characterized by the presence of maximal ^{137}Cs levels in most fish species in 1986 and 1987, and an even decrease in concentrations during the following years (Kanisch et al., 1995). This pattern was typical for some species, e.g. Baltic herring and cod. The trends in ^{137}Cs activities in pike showed a slower increase, attaining a maximum only during 1988-1990. This is due to the position of pike as a top predator that attains maximal values later than the prey. Likewise, Baltic herring from the Bothnian Bay showed slower increase in caesium levels and reached a maximum during 1989-1990. This was a result of slow riverine inflow of caesium from the drainage area, as well as transport of caesium from more contaminated areas in the south (Kanisch et al., 1995). The highest activity concentrations of ^{137}Cs in Baltic Sea fish after the Chernobyl fallout were about 300 Bq kg^{-1} wet weight in fillets of pike and cod caught from the Bothnian Sea. In the southern subareas of the Baltic Sea the maximum concentrations in cod were about one order of magnitude lower.

The bladderwrack, *Fucus vesiculosus*, has been widely used as an indicator of radioactivity in the Baltic Sea. In 1987, the highest activity concentrations of ^{137}Cs occurred in the eastern part of the Finnish south coast (Ilus et al., 1988). After that, the decrease in ^{137}Cs activity in *Fucus* has been more rapid in the Gulf of Finland than in the Bothnian Sea (Ilus, 1999). Since 1991, the highest concentrations have occurred in the Åland archipelago (Figure 7-7), and the concentrations on the Swedish coast of the Baltic Proper have increased. Recently, *Fucus* has also been used in tracer studies of ^{99}Tc in Nordic Sea areas. Small amounts of ^{99}Tc were observed in all the samples collected along the Finnish coast. Global fallout from nuclear weapons tests was certainly the most important source of ^{99}Tc detected in these samples (Ilus et al., submitted).

Distribution of radionuclides has been surveyed in flora and fauna in the discharge area of the Loviisa nuclear power plant, Finland. The Loviisa power plant is located on the north coast of the Gulf of Finland in an area, where the salinity of water is very low (about 4.5‰). Local discharge nuclides (^{60}Co, ^{54}Mn, ^{110m}Ag) were detected almost exclusively only at the low levels of the food web (macroalgae, periphyton, other aquatic plants, benthic animals, phytoplankton and zooplankton), whereas the highest concentrations of Chernobyl-derived ^{137}Cs and ^{134}Cs occurred in the inner organs of fish, waterfowl and seals (Ilus *et al.*, 1992).

The highest ^{137}Cs and ^{134}Cs concentrations occurred in fish flesh of perch (*Perca fluviatilis*), burbot (*Lota lota*) and pike (*Esox lucius*). The concentrations in fish liver, other entrails, milt and spawn were somewhat lower. Furthermore in seals (*Phoca hispida*), the highest ^{137}Cs and ^{134}Cs concentrations were found in the muscle tissue, but the concentrations were considerably lower in kidney and liver (in the seal train the concentrations were very low). On the other hand, in waterfowl (*Larus canus, Mergus merganser*) the ^{137}Cs and ^{134}Cs concentrations in liver and stomach were at least as high as in the muscle tissue; in many cases even higher. The concentrations in eggshells, egg entrails and embryos were clearly lower than in the birds themselves (Ilus *et al.*, 1992).

The highest concentrations of ^{60}Co occurred in the seaweed (*Fucus vesiculosus*), in periphyton and in vascular plants (*Myriophyllum spicatum* and *Potamogeton pectinatus*). In benthic animals (*Macoma balthica, Saduria entomon, Marenzelleria viridis* and *oligochaetes*), zooplankton and phytoplankton the concentrations were clearly lower. The uptake of caesium by aquatic plants and invertebrates was strongest in periphyton, *Fucus vesiculosus, Saduria entomon* and in insect larvae, i.e. *Chironomidae* (Ilus *et al.*, 1992).

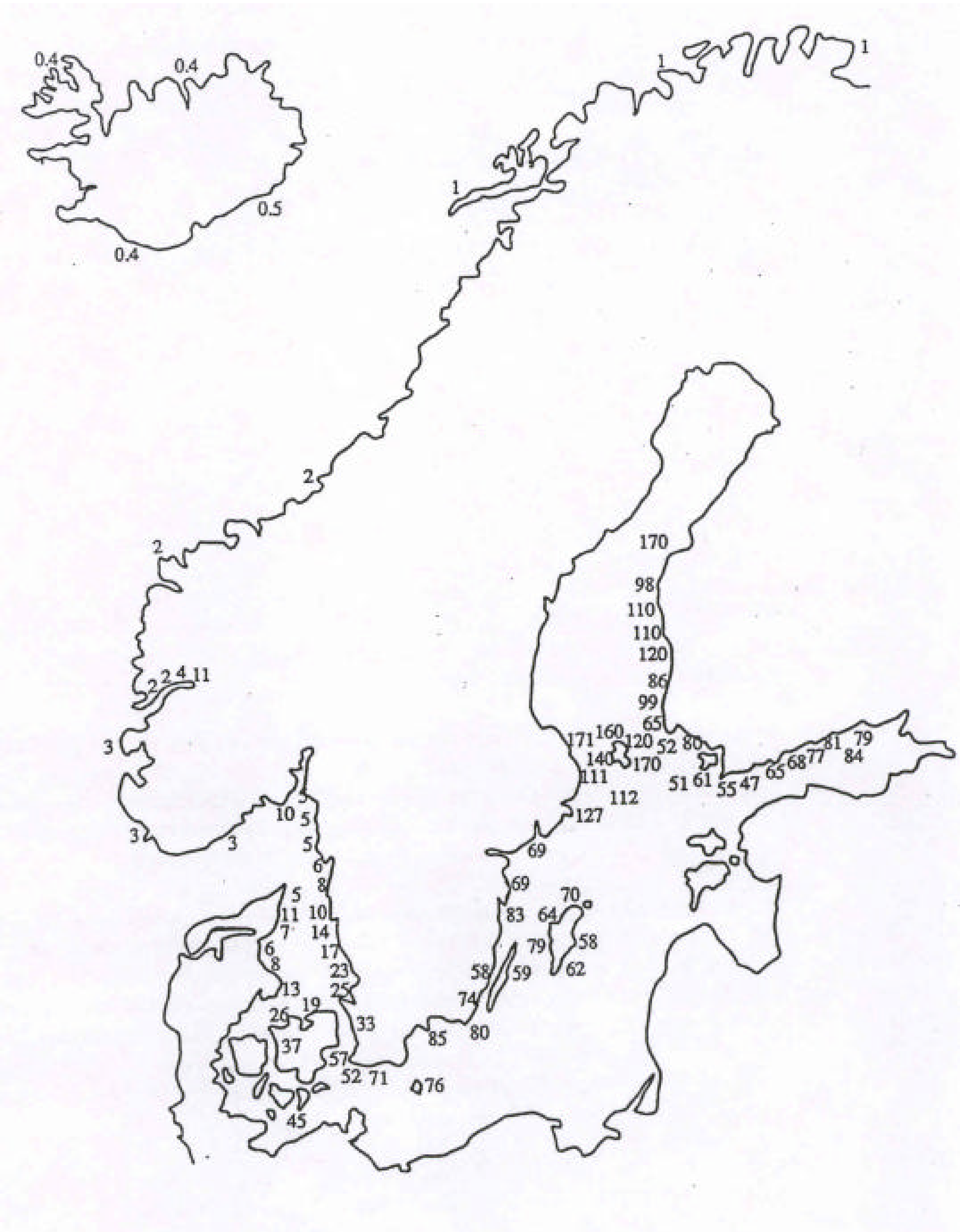

Figure 4-7 Activity concentration of ^{137}Cs in Fucus vesiculosus (Bq kg^{-1} dry wt) along the Nordic coasts in 1991 (Carlson et al., 1992).

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