FASSET

Framework for Assessment of Environmental Impact

Deliverable 5: Appendix 2

Underpinning scientific information (Life history sheets, empirical data and models)

Handbook for Assessment of the Exposure of Biota to Ionising Radiation from Radionuclides in the Environment

October 2003

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A project within the EC 5th Framework Programme

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1.1 Forests; semi-natural pasture and heathlands

1.1.1 Creeping bent (*Agrostis stolonifera*)

Classification

Kingdom: *Plantae*
Division: *Magnoliopsida*
Class: *Lilidae*
Order: Glumiflorae
Family: *Poaceae*
Genus: *Agrostis*

Distribution, habitat

Agrositis stolonifera is a stoloniferous perennial grass native to Eurasia and North Africa occurring in many different lowland habitats including woodland, shrubland and many semi-natural (meadows, salt-marshes and roadsides etc.) or managed grasslands such as golf courses. It prefers a moist site based on loam, clay-loam, or sand but can survive periods of drought.

Growth patterns (including size, rooting depth, longevity, seeding time)

Agrositis stolonifera can grow from seed in the same year but will also reproduce by producing stolons quickly becoming established on disturbed ground. It produces a substantial seedbank in meadows where the seeds can remain viable for more than a year. The stem is about 0.4-1m long, the leaves generally flat and about 2-10 mm wide and 2-10 cm long and the flowering stems (produced from June onwards) can grow to be as much as 40cm high. The roots are fine, fibrous, multibranched and are usually no deeper than 400mm; they are generally all replaced in spring. At soil temperatures above 27°C the roots stop growing and begin to die and they can also be attacked by the fungi *Gaeumannomyces graminis which* is a parasite of the *Poaceae* family.

Plant and animal associates

Creeping bent can be grazed quite heavily due to its prostrate growth and is thus an important forage plant; it is also used commercially on golf courses. In the semi-natural environment it provides cover for upland game birds, waterfowl, ground nesting birds and many small mammals and insects. As it is so widespread it occurs with many trees (especially willow spp.), shrubs (including *Juniperus* spp.) and other plants including red clover (*Trifolium pratense*), white clover (*Trifolium repens*), broadleaf plantain (*Plantago major*) tufted hairgrass (*Deschampsia cespitosa*) and timothy (*Phleum pratense*).

Sources of information

http://www.fs.fed.us/database/feis/plants/graminoid/agrsto/index.html
http://www.thepowerof2.co.uk/download/Turfax%20May%20-%20June%201999%2050Kb.pdf

Picture credits

http://www.uib.es/depart/dba/botanica/herbari/generes/Agrostis/stolonifera/index.htm

1.1.2 Heather (*Calluna vulgaris*)

Classification

Kingdom: *Plantae*
Division: *Magnoliopsida*
Class: *Magnoliidae*
Order: *Ericales*
Family: *Ericaceae*
Genus: *Calluna*

Distribution, habitat

Calluna vulgaris is widespread throughout Europe and can be found within many habitats including heathlands, boggy ground and clearings within woodland. It prefers nutrient poor acidic soils in open areas with little shade.

Growth patterns (including size, rooting depth, longevity, seeding time)

Heather is a slow-growing, predominantly bushy, evergreen shrub that varies in height depending upon its habitat but can grow to be up to one metre tall. The leaves are 0.2-0.4cm long produced on thin stems. Young plants produce a taproot which is lost as the lateral roots develop. These roots are usually matted within the top 10cm of the soil although they can extend down as far as 20cm on poorly drained soils. Reproduction occurs from both seed and from the base of the stem. The seeds require light to germinate and the germination rate is better in mineral soil; they will not germinate in waterlogged sites. Large seedbanks are formed, normally in the top 5cm of the soil where they can remain viable for >100 years. New growth from the stem base usually occurs in plants that are between 6-10 years of age although when the plant is >15 years old layering becomes the more common form of reproduction. On dry heathland *Calluna vulgaris* usually has a life span of about 30-40 years. In this type of habitat there are four distinct phases of growth: (i) pioneer stage (0-6 years) when the plant becomes established and grows upwards and has a relatively low biomass (ii) building stage (6-14 years) when the plant begins to become more ‘bushy’ and thus has a higher biomass (iii) mature stage (14-25 years) when growth ceases which in turn leads on to the final (iv)degenerate stage (25 or more years) when the central part of the plant dies. These latter two stages also have a high biomass but a notable decline in productivity. In wet environments these phases of growth are not followed as sphagnum mosses (*Sphagnum* spp.) constantly grow over the stems leaving only the shoots visible above the ground. Plants rarely exceed 22 years of age and are of an uneven-age structure within the ‘stand’.

Plant and animal associates

Calluna vulgaris is a very important food source for rock ptarmigan (*Lagopus mutus*) and grouse (*Lagopus lagopus*) in Scotland and Denmark and also forms a large part of the diet of hill sheep which are farmed in many upland environments. It responds well to light grazing pressure, preferably during the winter. In some areas it is actively managed by burning to increase its productivity for these commercially important animals rapidly regenerating from seed and the surviving basal stems. Browsing by red deer (*Cervus elaphus*) and mountain hare (*Lepus timidus*) is common in some areas and it provides good cover for many upland birds and small mammals. Vegetation found within the same habitat as *Calluna* spp. include the Scots pine (*Pinus sylvestris*), birch (*Betula* spp.), other shrubs including *Erica* spp. and *Vaccinium* species, sedges, grasses e.g. *Molinia* spp., bracken (*Pteridium aquilinum*), reindeer lichens (*Cladonia* spp.), and numerous mosses including *Sphagnum* spp. and *Polytrichum* spp.

Sources of information

U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Available from: http://www.fs.fed.us/database/feis/plants/lichen/claspp/all.html

Picture credits

http://fr.freefoto.com/preview.jsp?id=12-68-1

1.1.3 Reindeer lichen (*Cladonia rangiferina*)

Classification

Kingdom: *Fungi*
Division: *Bryophyta, Pteridophyta*
Class: *Lycopsida*
Order: *Lecanorale*
Family: *Cladoniaceae*
Genus: *Cladonia or Cladina*

Distribution, habitat

Cladonia spp. has a circumpolar distribution adapted to a cool, moist climate with little or no shade. It occurs in submontane to alpine environments in the open, in 'open' canopy forest and in tundra on shallow humus layers or dry peat and on moist to very dry, sandy, nitrogen-poor soils with low pH values ranging between 4.5 and 5.5, thus avoiding calcareous soils. As they are able to take up moisture from the air, the underlying soil is not as important a source of moisture so they can colonize and become dominant on rocks, logs and soils too shallow or sterile to support higher plants, provided that humidity is sufficiently high for lichen growth and temperature is sufficiently low to inhibit competitors. *Cladonia rangiferina* can survive in a broad range of habitats and is thus more common than the other species of lichen in less favourable habitats such as wet bogs and shaded woods.

Growth patterns (including size, rooting depth, longevity, seeding time)

Cladonia spp. are slow-growing (*C. rangiferina* had an average annual growth rate of approximately four mm per year), long-lived, densely branched ground lichens often forming clumps or mats which have a high surface to volume ratio. Wind is the most important dispersal agent. As they have no roots they absorb their nutrients from the atmosphere. They grow vegetatively by producing new growth annually at the top, passing through three growth stages: (i) growth-accumulation period, which lasts an average of 10 years but can vary from 5 to 25 years. (ii) growth-renewal period (which has the highest growth rate) where the base dies off at a rate equal to the growth, this stage often exceeds 100 years and (iii) degeneration period, where the base dies off at a greater rate than it grows from the top, this stage may also exceed 100 years. Factors that contribute to the variation in lichen growth include: plant age; disturbance by animals; substrate, drainage and exposure. After events such as fire, the first reindeer lichen to become established is *Cladonia Mitis* followied by *C. alpestris*, *C. rangiferina* or *C. arbuscula*.

Plant and animal associates

Cladonia spp. are commonly associated with: whortleberry (*Vaccinium myrtillus*), rock cranberry (*V. vitris-idaea*), bog blueberry (*V.* uliginosum), lowbush blueberry (*V.* angustifolium), bog birch, sheep laurel (*Kalmia angustifolia*), common bearberry (*Arctostaphylos uva-ursi*), black crowberry (*Empetrum nigrum*), *Stereocaulon paschale*, and Schreber's moss (*Pleurozium schreberi*). They are important in the winter diet of reindeer (*Rangifer tarandus*) but as their growth rate is so slow they cannot tolerate re-grazing for 2 to 5 years if the grazing is moderate, or 10 to 15 years if the area is heavy grazed.

Sources of information

http://www.fs.fed.us/database/feis/plants/lichen/claspp/all.html

Picture credits

http://biology.clc.uc.edu/graphics/taxonomy/fungi/lichens/reindeer%20moss/

1.1.4 Cep (*Boletus edulis*)

Classification

Kingdom: *Fungi*
Division: *Eumycota*
Class: *Hymenomycetes*
Order: *Agaricales*
Family: *Boletaceae*
Genus: *Boletus*

Distribution, habitat

Boletus edulis is found in many countries but in Europe it occurs from Northern Scandinavia to the southern tips of Greece and Italy. It can be found growing in coniferous, broad leaved or mixed woodland and grows in association with trees (usually beech, but also with others) hence it is often near to their bases. It occurs in greater numbers in mature woodland where grazing animals are absent but is also found in less developed, stocked, woodland.

Growth patterns (including size, fruiting time)

Boletus edulis is a Mycorrhizal species (i.e. one that lives mutualistically (symbiotically) with the roots of a higher plant (tree). The presence of fungal mycelia which grow both throughout the soil and around the host trees' root tips, some also penetrating into the root itself, increases the absorption of essential nutrients thus improving the trees vigour and disease resistance. They appear at the surface as fruiting bodies (defined as the above ground organ of the plant (i.e. the reproductive part of the organism)). The usual fruiting time is summer to late autumn dependant upon local weather conditions. For a good crop there is a requirement for a period of rain followed by some warmer weather. They are killed by even a slight frost. The species has small, round pores (i.e. looks like a 'sponge' underneath the cap) rather than gills (like those of the commercially grown field mushroom). The cap itself has a chestnut brown outer face whilst the flesh within is off white in colour; it grows to be between 8-30cm in diameter. The stem can be between 3-23 cm high and 3-7cm in diameter broadening towards the base and is dark cream. The pale olive brown spores it produces to reproduce measure 14-17 x 4.5.-5.5 µm.

Plant and animal associates

As it lives symbiotically with the roots of trees it is obviously dependant upon them. Where grazing animals (e.g. deer) are present within the woodland it will be consumed. Its main use within Europe is as one of the most important edible species often seen for sale in a fresh state in continental markets. Most commercially available supplies originate from Western Europe, Eastern Europe (eg, Poland), China, Indonesia, USA, Canada and India although the fSU and Morocco also supply small quantities; and is generally sold dried (and used as a flavouring for soup), frozen, canned or pickled in brine thus guaranteeing a year round supply of a predictable quality. It is sold in many countries such as France, Italy, Germany and North America and, in limited quantities, to New Zealand and the United Kingdom. From official statistics, more than one thousand tonnes were sold in France and Germany in 1987 and >2000 in Italy a year later but much more will have been collected and consumed non commercially within these and other countries.

Sources of information and Picture credits

Philips, R. 1994. *Mushrooms and other fungi of Great Britain and Europe*. Macmillan Reference.
http://www.crop.cri.nz/psp/broadshe/boletus.htm
http://www.bioimages.org.uk

1.1.5 Scots pine (*Pinus sylvestris*)

Classification

Kingdom: *Plantae*
Division: *Coniferophyta*
Class: *Pinopsida*
Order: *Pinales*
Family: *Pinacea*
Genus: *Pinus*

Distribution, habitat

Scots pine is an exotic, medium-sized, two-needle conifer and is the most widely distributed pine in the world. It is found from sea level to 2440m and grows on a wide variety of soils, although growth is best on well-drained mineral soils with a pH range of between 4.5 and 6.0. Its native range includes Scotland, Scandinavia (excluding Denmark), northern Europe, and northern Asia. High light intensities are required for good growth, its performance varying with site and seed source.

Growth patterns (including size, rooting depth, longevity, seeding time)

Scots pine is long-lived; ages of 200 and 400 years are common in Scandinavia and individuals of nearly 1,000 years of age occur in northern Sweden. Height at maturity usually ranges from 15-30 meters. Needles range from 4.5-9.0cm in length and normally remain on the tree for 2-3 years before they are shed. The bark is relatively thin in a young tree but can be up to 5cm thick in a mature tree. A taproot with a depth of which ranges from 1.5-3.0 meters is frequently developed only on sandy soils, but in general most of its roots are horizontal and within 20cm of the soil surface. It reproduces by seed which is dispersed by the wind; sexual maturity usually being reached at between 10 to 15 years of age with viable seed being produced for up to 200 years. The pollen cones open from late May to early June and pollination occurs in early summer followed by fertilisation 12 months later. Seeds (which weigh 0.005g) mature and then the cones (of which each tree can produce 3000) ripen from September to October. Seed dispersal (which ranges 50-100m from the parent) occurs from December to March. Seed germination is good even at depths of up to 10 cm and if buried the seed will stand a good chance of surviving fire. Seedlings establish best with adequate moisture and some shade on acidic soils.

Plant and animal associates

Scots pine is commonly managed commercially for its wood but in the natural environment moose (*Alces alces*) browse it in Scandinavia and Russia and the pine grosbeak (*Pinicola enucleato*) feeds on the buds. It has relationships with many plants, insects, birds and animals, some of these live on the pine itself, particularly epiphytic lichens and mosses and it has mycorrhizal associations with many species of fungi. The shade provided by the canopy of mature Scots pines provides a good habitat for bilberries (*Vaccinium myrtillus*) and cowberries (*Vaccinium vitis-idaea*). The species also plays a successional role in the development of the hummocks which are commonly found in the pinewoods.

Sources of information

http://www.fs.fed.us/database/feis/plants/lichen/claspp/all.html
http://www.treesforlife.org.uk/tfl.scpine.html

Picture credits

http://treehelp.com/trees/pine/pine-types-scotch.asp (G. Lumis)

1.1.6 Common oak (*Quercus robur*)

Classification

Kingdom: *Plantae*
Division: *Magnoliophyta*
Class: *Magnoliopsida*
Order: *Fagales*
Family: *Fagaceae*
Genus: *Quercus*

Distribution, habitat

The common oak is a large deciduous tree that is native to Europe, western Asia and northern Africa and is commonly found in mixed deciduous forests to an altitude of 300-400m. It thrives in full sun and prefers fertile soils with a range in pH of 4.5 - 7.5 although it will tolerate a wide range of environmental and soil conditions (including wet soil and dry clay).

Growth patterns (including size, rooting depth, longevity, seeding time)

The tree can reach a height of 30-40m, has an average growth rate of 25cm per year (in the first 10-20 years) and can live for more than 1000 years. It has a short sturdy trunk, wide spreading crown with small deciduous leaves which are approximately 7.5-12cm and do not drop until late winter producing a significant amount of litter. Flowers are not generally formed until the tree is 20-30 years old and appear with the emerging leaves in March to April as hanging catkins which are 5-7.5 cm long. The fruits (acorns) appear in October and are oval, slightly elongated and are about 2.5cm long, with a cap that covers 1/3 of the nut. They are borne singly or in clusters of 2-5 which dangle on a long stalk and mature in one growing season. They do not achieve full production until they are between 80 and 120 years old. Propagation is by seed and there can be as many as 110 - 450 seeds per kg. A period of chilling is required for the acorn to germinate. The root system develops over a period of time, actively growing between April and November. In the first 10 years of life it produces very few horizontal roots relying upon a tap root (which can remain until the tree is up to 30 years old) with more substantial branch roots not staring to develop until later in its life. The system of development is modified dependant upon conditions but in general it is a deeply rooting tree.

Plant and animal associates

Deciduous woodlands in general support a wide variety of fungi (but no one particular genus), numerous lichen and bryophytes and a wide variety of insects and birds. Acorns attract squirrels and other small mammals and wild boar (*Sus scrofa*) is a major consumer as can be the brown bear (*Ursus arctos*). Various species of deer will browse the leaves but no one species is wholly dependant upon the tree. Many plants of the forest floor have underground energy-storing organs (e.g. bulbs, corms, or rhizomes) that allow them to grow rapidly in spring thus taking advantage of the short, warm, shade-free period since once the canopy of the oak has formed it almost completely suppresses ground vegetation. The strong, durable wood has many uses e.g. household furniture and to make casks for storing wines and spirits during aging.

Sources of information

Morris, M.G. & Perring, F.H. 1974. The British Oak. E.W. Classey Ltd. Berkshire.
http://www.floridata.com/ref/Q/quer_rob.cfm
http://www.british-trees.com/guide/commonoak.htm
http://hort.ifas.ufl.edu/trees/QUEROBA.pdf

Picture credits

http://www.cnr.vt.edu/dendro/dendrology/Syllabus2/qrobur.htm

1.1.7 Earthworm (*Lumbricus terrestris*)

Classification

Kingdom: *Animalia*
Phylum: *Annelida*
Class: *Oligochaeta*
Order: *Haplotaxida*
Family: *Lumbricidae*
Genus: *Lumbricus*

General habits, habitat and home range

The main function of the earthworm is to aerate the soil thus improving its productivity, in addition their castings fertilise the soil. Earthworms require adequate moisture for growth and survival and will live and breed at temperatures up to 38°C. They will die in freezing temperatures but protect themselves as much as is possible by moving to lower depths in soil, burrowing to a maximum depth of up to 2.5m. They do not hibernate and can be found in most soils with a pH range of about 4.2 - >8.0 with as many as 50 to 500 per m^2, the colonies spreading about 3-5 m per year.

Dietary habits

The earthworm is a detritivor; its main food consists of decaying organisms. As they eat they also ingest soil, sand, and tiny pebbles and so can ingest and discard their own weight (0.006kg) in food and soil every day. In an average adult worm, digestion takes about twenty-four hours.

Birth weight, gestation period and general life cycle

Earthworms are usually not self-mating although they are hermaphroditic. They will only mate in warm conditions; the eggs develop within cocoon and the baby worms (usually between two to twenty, although four is more normal) will hatch from it after about three weeks. Each worm is capable of producing 38 cocoons per year. The young reach maturity in about one year and have a life expectancy of up to six years. Worms have an ability to maintain optimum population size according to the available food and space. While conditions are right, they will breed until the optimum food and space ratio is reached and will cease breeding until more food and/or space is provided. Their main predator is the mole (*Talpa europaea*) and many species of birds.

Sources of information

Lee, K.E. 1985, Earthworms their ecology and relationships with soils and land use. Academic Press London.
Encyclopaedia Britannica 2002
http://edis.ifas.ufl.edu/IN047

Picture credits

Jack Kelly Clark http://www.sarep.ucdavis.edu.

1.1.8 Woodlouse / Pillbug (*Armadillidium vulgare*)

Classification

Kingdom: *Animalia*
Phylum: *Arthropoda*
Subphylum: *Mandibulata*
Class: *Crustacea*
Subclass: *Malacostraca*
Order: *Isopoda*
Family: *Armadillidiidae*
Genus: *Armadillidium*

General habits, habitat and home range

Woodlice are detritivorous invertebrates. Adequate moisture is essential for their survival and they group together to reduce water loss. On a hot day, they remain under objects on the damp ground and only move at night during the lower temperatures and more humid conditions. They become inactive during the winter months.

Dietary habits and main prey species

Woodlice are plant eaters, feeding on decaying organic matter and occasionally young plants and their roots. They do occasionally eat flesh although they do not catch and kill their prey, but consume flesh if they come across dead and decaying animals. One species is an exception, *Tylos latereilei,* which is a Mediterranean coastal isopod, will also eat other woodlice that have died or those going through the moult - a time when their bodies are very soft.

Birth weight, gestation period and general life cycle

Woodlice mate throughout the year although most activity is in the spring. The female carries the eggs, numbering from 7 to 200, in a brood pouch on the underside of her body. The eggs hatch in three to seven weeks and they remain in the brood pouch for six to eight weeks. There may be one to two generations per year, with an average life expectancy of 3.5 years, depending on weather conditions and location.

Sources of information and picture credits

http://ohioline.osu.edu/hyg-fact/2000/2072.html
http://www.biology4kids.com/misc/isopoda.html

1.1.9 Wood ant (*Formica aquilonia*)

Classification

Kingdom: *Animalia*
Phylum: *Arthropoda*
Class: *Insecta*
Order: *Hymenoptera*
Family: *Formicidae*
Genus: *Formica*

General habits, habitat and home range

The wood ant (*Formica aquilonia*) is a social insect that is approximately 5-12mm long (dependant upon their sex). It is primarily active during the spring and summer and lives in 'colonies' (which contain up to half a million insects) found in coniferous (and occasionally broadleaved) woodland throughout many countries of Europe (and also Arctic Norway); their local distribution is determined by the availability of suitable nest sites, a favourable microclimate, and a good food supply. They prefer open woodland with little understory vegetation and limited disturbance by both grazing animals and man. Their nests are usually built on well drained sites and are made from woodland debris, in particular pine needles and can be over one metre high and two meters in diameter having a series of underground chambers and galleries which provide protection both from their predators and the elements as well as a place in which to rear their young. Their home range is quite large in comparison to their size and can be between 270m^2 and 1600m^2. The ants are classified by the International Union for the Conservation of Nature (IUCN) as lower risk, near threatened, but they are considered essential within forest ecosystems as they control the numbers of herbivorous insects which have the potential to defoliate and kill trees.

Dietary habits and main prey species

The ants spend approximately 90% of their time foraging in the canopy of trees. They are omnivorous, consuming the needles of scots pine (*Pinus sylvestris*) and other insects, such as the caterpillars of the sawfly (*Neodiprion sertifer*) and the pine looper moth (*Bupalus piniaria*). They detect food mainly by vibration and scent (although they can see up to about 10cm) each one bringing back about one and a half times its own weight in food every day; mostly for the larvae within the nest which rely on the liquid food that is provided by them. Wood ants also have a symbiotic relationship with aphids (e.g. *Symydobius oblongus*) wherby the aphids are 'milked' by the ants so that they release their waste product 'honeydew', which, to the ants, is a rich valuable food source. A similar relationship is formed with the earthworm which is protected from the ants' sting by its mucus coating. Large numbers of worms are commonly found within the ant nests as they provide a very comfortable 'home' and a plentiful supply of food (microbes and decomposing litter). In return the worms prevent the build of moulds and fungi within the nests. Badgers (*Meles meles*) are known to eat wood ant pupae.

Birth weight, gestation period and general life cycle

A nest can contain many queen ants, each about 12mm long; following mating (which occurs only once) each one lays millions of eggs that are reared to adulthood. The eggs hatch into larvae which moult several times before going through a period of pupation, eventually emerging as distinctive red and black coloured adults. If by chance a female 'worker' ant lays any eggs (unlikely as they are not fully developed and hence should not reproduce) then her eggs are used as food. Queen ants have the longest lifespan, which can be more than fifteen years; 'workers' (which are all female) survive for about a year and the males (who's only purpose is to mate with the queen) are even more short lived, dying after their mating flight in the spring.

Sources of information and Picture credits

http://213.121.208.4/pdfs/education/woodants.pdf
http://www.treesforlife.org.uk/tfl.woodants.html
http://www.wcrl.ars.usda.gov/cec/formica.htm
http://www.ukbap.org.uk/asp/UKPlans.asp?UKListID=306#1 & http://centralpets.com/

1.1.10 Ground-nesting bird egg - Red Grouse (*Lagopus lagopus scoticus*)

General information

Red grouse nest in a hollow in the ground, generally in open moorland. The eggs are yellowish white with rich dark chocolate or red brown blotches. The hen usually lays about six eggs, but can lay up to 17 and will spend approximately 20 days incubating them. Crows (*Corvus corone corone*) and stoats (*Mustela erminea*) are the most common predators of the eggs.

Sources of information

http://www.environmental-entomology.co.uk/birdsam3.html
Witherby, H.F., Jourdain, Rev. F.V.R., Ticehurst, N.F. and Tucker, B.W. The handbook of British Birds. Volume V. London: H.F. Witherby Ltd.

Picture credits

http://www.asken.co.uk/Practical/4_shooting_management.htm

1.1.11 Ground-nesting bird egg - Skylark (*Alauda arvensis*)

General information

The skylark can be found in open grasslands, plains, meadows, marshes, peat bogs or sand dunes. They nest on the ground, sometimes with little or no cover and usually in grass. The hen usually lays between 3-4 eggs (with a maximum of 7) and will spend 11 days incubating them. Foxes (*Vulpes vulpes*), carrion crow (*Corvus corone corone*), stoats (*Mustela erminea)*, rats (*Rattus norvegicus*), hedgehogs (*Erinaceus europaeus*) and badgers (*Meles meles*) are common predators of nests.

Sources of information

http://www.environmental-entomology.co.uk/birdsam3.html
Witherby, H.F., Jourdain, Rev. F.V.R., Ticehurst, N.F. and Tucker, B.W. The handbook of British Birds. Volume VI. London: H.F. Witherby Ltd.

1.1.12 Mole (*Talpa europaea*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Subphylum: *Vertebrata*
Class: *Mammalia*
Order: *Insectivora*
Family: *Talpidae*
Genus: *Talpa*

General habits, habitat and home range

Moles are burrowing mammals. They are active both day and night. They move about in a network of "main galleries" which are used by many individuals. Part of this network is semi-permanent and used by successive generations of mole. In general, there are no molehills where these galleries are located as the layout is old and there is no more earth to excavate. Temporary "hunting galleries" are dug out and used by a single individual and are not reused, it is here where the earth is excavated and expelled as debris in the form of molehills. Within these "hunting galleries" there is usually a resting place situated at the junction of several galleries and some "secondary chambers" which can be stocked with food reserves. Males and females are solitary for most of the year, occupying exclusive territories. Males enlarge their territories at the start of the breeding season, tunneling over large areas in search of females. Their burrows are usually between 0.15 - 0.50m, the entire network being between 100 to 200m in length. They can create a burrow about 20m long in a day.

Dietary habits and main prey species

Earthworms are the most important component of the mole's diet although they also eat many insect larvae, particularly in the summer. An 80g mole needs 50g fresh weight of earthworms per day so they sometimes collect and store them alive in "secondary chambers" (470 worms have been recorded in one chamber). Moles rarely forage on the surface; their food is either actively dug out of the soil by the mole itself or, more often, collected from the floor of the tunnel as many soil animals fall through into them.

Birth weight, gestation period and general life cycle

The female normally gives birth to one litter per year (occasionally two) after a 28 day gestation period. The average litter size is between two and seven young, each with a birth weight of approximately 3.5g. Their average life span is around three years.

Sources of information

Van den Brink, F.H. 1967. Field guide to the mammals of Britain and Europe. London: Collins
Encyclopaedia Britannica 2002
http://www.borealforest.org/world/mammals/european_mole.htm
http://www.abdn.ac.uk/mammal/mole.htm
http://www.press.jhu.edu/books/walker/insectivora.talpidae.talpa.html
http://tim.rawle.org/moles/genral.htm

1.1.13 Rabbit (*Oryctolagus cuniculus*)

Classification

Kingdom: *Animalia*
Subkingdom: *Metazoa*
Phylum: *Chordata*
Subphylum: *Vertebrata*
Superclass: *Tetrapoda*
Class: *Mammalia*
Subclass: *Theria*
Order: *Lagomorpha*
Family: *Leporidae*
Genus:*Oryctolagus*

General habits, habitat and home range

Rabbits are herbivorous burrowing mammals which do not hibernate. Tunneling is undertaken predominantly by the female and the depth of the burrow depends upon the nature of the soil and the height of the water table but can be up to 3m in depth. Their home range varies in size with population density and food abundance, but can be as much as 8000m^2; males' home ranges are on average twice as large as those of females and overlap with those of several females. The random network of tunnels, dens and bolt holes is known as a warren and they use runs to and from their feeding areas. Social groups vary from a single pair to up to 30 rabbits using the same warren. They are normally nocturnal but will come out in daylight if undisturbed, especially in the summer. Their preferred habitat is dry areas near sea level with soft, sandy soil (for easy burrowing). A rabbit spends approximately 50 % of its time underground.

Dietary habits and main prey species

Rabbits eat a wide range of plants including grasses, cereal crops, root vegetables, young shoots of meadow and garden plants and will also eat the bark from trees especially when snow covers other food sources. Adult food consumption is around 5-100g per day and water consumption about 5-10ml. They graze closer to the ground than other animals, often preventing pasture seedlings emerging or eliminating species altogether, thus allowing weeds to grow. They are a vital food source for a wide variety of carnivores, mainly foxes (*Vulpes* spp.) and also many birds of prey.

Birth weight, gestation period and general life cycle

Healthy females can produce one litter of 3-7 young per month during the summer. The young are suckled in a special burrow, dug by the mother separate from the warren and lined with a nest of her own fur. The entrance to this burrow is plugged with earth when she leaves. The gestation period is approximately 30d and the young weigh 30-35g at birth. They rarely live for more than 3 years. Over 90% die in the first year of life, most of these in the first three months either due to attack by predators or through myxomatosis.

Sources of information

G.B. Corbet 1964. The identification of British mammals. London: British Museum (Natural History).
Van den Brink, F.H. 1967. Field guide to the mammals of Britain and Europe. London: Collins.
Encyclopedia Britannica 2002
http://www.abdn.ac.uk/mammal/rabbit.htm
http://animaldiversity.ummz.umich.edu/accounts/oryctolagus/o._cuniculus$narrative.html

1.1.14 Weasel (*Mustela nivalis*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Mammalia*
Order: *Carnivora*
Family: *Mustelidae*
Genus: *Mustela*

General habits, habitat and home range

The weasel is a fast moving (approximately 1.3km per hour) carnivorous mammal which does not hibernate. They can survive in a wide variety of habitats, including open forests, farmlands, meadows, prairies, steppe, and semi-deserts and are well adapted for the tundra; they avoid deep forests, sandy deserts and open spaces. Males and females live in separate territories; the male ranges over a larger area and in spring extends it further to seek mates. Home ranges vary in size (1-15ha) according to their sex, distribution and density of prey and their habitat, and usually contain several dens and resting places that are visited occasionally. They take over the dens and nests of their prey (which is usually the vole) and can be found from just below the soil surface up to a depth of approximately 0.5m. In cold climates these nests are often lined with fur from prey. A weasel spends approximately 50 % of its time underground.

Dietary habits and main prey species

The weasel is the smallest true carnivore and has, gram for gram, one of the highest metabolic rates of any mammal so they have to eat frequently. They require a third of their body weight (which is approximately 23-33g, fresh weight) in food each day, but, as they have very small stomachs, they can eat only 3g of food at a time. Their diet is dependant on their habitat, but it contains mainly small mammals, if these are scarce then they will eat birds' eggs and young chicks. Analysis of gut contents has been shown to contain 55% small rodents (voles) 19% rabbits, 15% birds, 3% rats, 2% insectivores and 6% others (e.g. lizards). The extreme northern populations will also eat the carcasses of brown lemmings. The weasels' small size enables it to search through the tunnels and runways of mice and voles so it can hunt at any time of the day or year even under deep snow. Their dens may contain the remains of food from several days' meals. Their main predators are various species of hawks, owls, kestrels and foxes (*Vulpes* spp.).

Birth weight, gestation period and general life cycle

Mating usually occurs in the spring, but can occur all year round. The gestation period is typically between 30-37 days, after which a litter of between 1-7 young is born weighing approximately 1.5g. It is possible for the female to give birth to 2 litters per year, but there is a higher mortality rate in the second litter. They generally live for up to 3 years.

Sources of information

G.B. Corbet 1964. The identification of British mammals. London: British Museum (Natural History).
Van den Brink, F.H. 1967. Field guide to the mammals of Britain and Europe. London: Collins.
Encyclopaedia Britannica 2002
http://www.abdn.ac.uk/mammal/weasel.htm
http://www.environmentyukon.gov.yk.ca/fishwild/weasel.shtml

1.1.15 Red fox (*Vulpes vulpes*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Mammalia*
Order: *Carnivora*
Family: *Canidae*
Genus: *Vulpes*

General habits, habitat and home range

The red fox is a carnivorous mainly solitary mammal, primarily active at dusk and at night. Its preferred habitat is mixed farmland and woodland. The size of their territories depends on their habitat ranging from 0.2km^2 in urban areas up to 40km^2 in upland areas. Each territory is usually occupied by a fox family group (containing several adults in areas where there is a plentiful supply of food) and they remain in the same home range for life. Individuals and family groups have main earthen dens and often other emergency burrows in the home range. The same den is often used over a number of generations. Pathways throughout the home range connect the main den with other resting sites, favoured hunting grounds and food storage areas. A fox spends approximately 10 % of its time underground.

Dietary habits and main prey species

Red foxes have a very varied diet, usually foraging alone. They are opportunist feeders, catching food surplus to their requirements. In lowland rural areas, small mammals (especially field voles (*Microtus agrestris*) and rabbits (*Oryctolagus cuniculus*) are the major source of food along with earthworms, beetles, fruit (particularly blackberries) and small birds. On salt marshes, they eat crabs and dead seabirds, whilst in upland regions carrion can be important, particularly during winter. Daily food consumption is between 0.5- 1 kg per day (fresh weight).

Birth weight, gestation period and general life cycle

Usually only one vixen in a family group produces cubs; once a year in the spring. The gestation period is generally around 52 days, but ranges between 49 and 56 days. Litter size varies from 1-13 pups, the average is 5 with a range in birth weight of 100-130g. A vixen stays in the earth with her cubs for the first two weeks of their lives. At about four weeks old, usually in late April or early May, the cubs begin to come into the open. Their average life expectancy is between 3 and 6 years.

Sources of information

G.B. Corbet 1964. The identification of British mammals. London: British Museum (Natural History).
Van den Brink, F.H. 1967. Field guide to the mammals of Britain and Europe. London: Collins.
Encyclopaedia Britannica 2002
http://www.abdn.ac.uk/mammal/redfox.htm
http://animaldiversity.ummz.umich.edu/accounts/vulpes/v._vulpes$narrative.html
http://www.gov.nf.ca/snp/Animals/redfox.htm

Picture credits

http://www.floodlight-findings.com/2redfox/redfox.html

1.1.16 Moose (*Alces alces*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Mammalia*
Order: *Artiodactyla*
Family: *Cervidae*
Genus: *Alces*

General habits, habitat and home range

The moose is an herbivorous mammal and is the largest member of the deer family. It is solitary for most of the year, gathering in high densities for the breeding season. They are generally found in forested areas where there is less snow cover in the winter, preferring moist conditions where there are lakes, ponds, and swamps. They have a relatively small home range; the entire summer may be spent in an area about 0.5km^2, but during the rut the males range over a much larger area. European populations will migrate more than 300km if necessary to reach favoured sites at different times of the year and will even swim short distances to reach them.

Dietary habits and main prey species

Moose require approximately 45kg (fresh weight) of food per day. During the autumn and winter, moose consume large quantities of willow (*Salix* spp.), birch (*Betula* spp.), and aspen (*Populus tremula*) twigs. In the spring they graze as well as browse, eating a variety of foods, particularly sedges, *Equisetum spp*., pond weeds, and grasses. During the summer they feed on vegetation growing in shallow ponds, forbs, and the leaves of birch, willow, and aspen. Their main predators are wolves (*Canis lupus*) and brown bears (*Ursos arctos*).

Birth weight, gestation period and general life cycle

Following a gestation of about 240 days the female gives birth to either one or two calves (twins are born 15-75% dependant upon their food supply) each with a birth weight of around 13-16 kg. At three weeks old the calf will follow its mother and browse for food. By 5 months they are completely weaned but will stay with their mother until the next calf is born. Their average lifespan is approximately 20 years.

Sources of information and picture credits

Van den Brink, F.H. 1967. Field guide to the mammals of Britain and Europe. London: Collins.
Encyclopaedia Britannica 2002
http://www.state.ak.us/local/akpages/FISH.GAME/notebook/biggame/moose.htm
http://animaldiversity.ummz.umich.edu/accounts/alces/a._alces$narrative.html
http://www.thebigzoo.com/Animals/Moose.asp
http://www.coastalsafari.com/MOOSE.htm

1.2 Freshwater ecosystems

1.2.1 Water milfoil (*Myriophyllum spicatum*)

Classification

Kingdom: *Plantae*
Phylum: *Magnoliophyta*
Class: *Magnoliopsida*
Order: *Haloragales*
Family: *Haloragaceae*
Genus: *Myriophyllum*
Species: *Myriophyllum spicatum*

Description

Water milfoil is a submerged aquatic perennial. Long underwater stems branch as they approach the surface, where they produce whorls of three or four finely divided greyish-green leaves.Its stems can ‘top out’ in 20 feet of water, but the plant is most often found in water 0.5 to 3.5 m deep. It grows rapidly and tends to form a dense canopy on the water surface.

Habitat and reproduction

Water milfoil seems to prefer lakes, ponds and slow-moving rivers and streams but can also grow in fast-moving water. It speads and reproduces mainly by regrowth of plant fragments and spreads locally by stolons; a slender stem growing along the ground producing roots and branches at nodes.

References

http://aquat1.ifas.ufl.edu/seagrant/myrspi2.html
http://www.fw.umn.edu/research/milfoil/milfoilbc.html

1.2.2 Freshwater clam (*Anodonta sp.*)

Classification

Kingdom: *Animalia*
Phylum: *Mollusca*
Class: *Bivalvia*
Order: *Eulamellibranchiata*
Family: *Unionidae*
Genus: *Anodonta*
Species: *Anodonta sp.*

Description

The shell of the freshwater clam is large, thin, nearly circular, smooth, yellow or yellowish green to dark brown, with fine green rays in some individuals. They can grow to a length of 17.8 cm. The average life time expectancy is 3-5 years, but can sometimes be up to 20 years.

Habitat

Ponds, lakes, or sluggish mud-bottomed pools of creeks and rivers.

References

http://www.inhs.uiuc.edu/cbd/musselmanual/page76_7.html
http://fly.hiwaay.net/dwills/mussels/alafw1.jpg

1.2.3 Gastropoda (*Lymnea sp.*)

Classification

Kingdom: *Animalia*
Phylum: *Mollusca*
Class: *Gastropoda*
Order: *Basommatophora*
Family: *Lymnaeidae*
Species: *Lymnea sp.*

Description of the species, life history, feeding habits

Gastropoda have an asymmetrical, univalved, usually spiral shell. Gastropod communities are often linked with the fish, macrophyte and periphyton communities in a complex way.Freshwater forms usually lay their eggs on plants or stones, and miniature adults emerge. Individuals of some species may live for several years. The life cycle is non-emergent. Freshwater species are mostly phytophagous scrapers that feed by way of the file-like radula. Many also feed on detritus or vascular plants. They may feed from the surface film. Many can digest cellulose directly. A few use their gills to filter feed, at least in part. Snails are important in many food chains and they are frequently used by fish and other wildlife as food.

References

http://www.esg.montana.edu/dlg/aim/mollusca/gastropd.html
Suomen eläimet, osa 5, pp. 152-155, Weilin+Göös, 1986, Espoo (In Finnish).

1.2.4 Freshwater isopod (*Asellus aquaticus*)

Classification

Kingdom: *Animalia*
Phylum: *Arthropoda*
Subphylum: *Crustacea*
Class: *Malacostraca*
Order: *Isopoda*
Family: *Asellidae*
Genus: *Asellus*
Species: *Asellus aquaticus*

Description of the species, habitat

Water slater *(Asellus aquaticus)* is a crustacean of about 1.5 cm, living as well in muddy and stony bottoms of the litoral zone of lakes. It eats mostly decomposing vegetation, detritus, benthos, and it is eaten by many fish species. For winter *Asellus aquaticus* goes to somewhat deeper water, in great lakes generally to about the depth of 3m.

Reproduction and life span

Asellus aquaticus carries its young on its stomach for 3 weeks and the juvenile stage 3 months. Egg production occurs once per month and moults every 2-3 weeks. The duration of moults is 1.5 days. In Nordic countries it has two generations per year. The life expectancy of *Asellus aquaticus* is 1 year.

References

http://www.uni-oldenburg.de/zoomorphology/Styx3.html
Suomen eläimet, osa 5, pp. 5,Weilin+Göös, Espoo,1986 (In Finnish).

1.2.5 Burbot (*Lota lota*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Osteichthyes*
Order: *Gadiformes*
Family: *Gadidae*
Genus: *Lota*
Species: *Lota lota*

Geographical distribution, habitat, life span

The burbot is the only representative of the cod (Gadidae) family in fresh waters. Burbots (benthic fishes) are distributed in cold fresh waters throughout North America and Eurasia southward to about 40 degrees north. They occupy deep, cold rivers and lakes, preferring to be near the bottom in areas of low light intensity. In summer burbot lives in water close to the bottom, often in deep basins where water is cool also in summer time. Burbot are relatively long-lived and slow-growing species with a lifespan of 10-15 years. Burbot older than 20 years are not common. It takes about six or seven years to reach an adult size of about 45 cm.

Dietary habits

Young burbot feed mainly on insects and other invertebrates. By the age of 5, burbot feed almost exclusively on fish. Adult burbot can appear sluggish, but they are voracious predators, feeding mostly at night. Small fish, burbots included, are common food items.

Reproduction

Most burbot spawn for the first time at the size of 45 cm. Burbot spawn under the ice in late winter before the ice melts (February to March). Spawning usually takes place at night with eggs scattered over a sand or gravel bottom. Eggs are small, and an individual burbot can produce over million eggs.

References

http://www.stake.ak.us/adfg/notebook/fish/burbot.htm
http://huntingfishing.tripod.com/burbot.htm
http://www.rook.org/earl/bwca/nature/fish/lota.html
Koli, Lauri. Suomen kalat, WSOY, Porvoo, Finland, 1998 (in Finnish).

1.2.6 Perch (*Perca fluviatilis*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Osteichthyes*
Order: *Perciformes*
Family: *Percidae*
Genus: *Perca*
Species: *Perca fluviatilis*

Geographical distribution, habitat

Perch, a pelagic fish, is very common, found throughout Europe and northern Asia. It inhabits a variety of water habitats, even quite barren environments. It can tolerate acidic water well.

Dietary habits

Young fish (not longer than 10 cm) live on zooplankton. Older individuals feed on small bottom-living animals and fishes.

Reproduction and life span

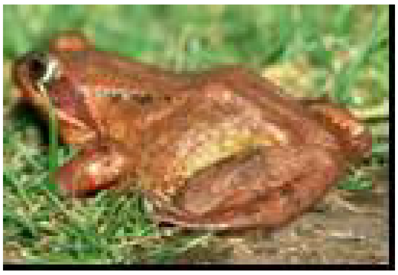
Perch are mature to spawn at the age 3-6 years, with males spawning a couple of years earlier than females. Reproduction is not possible at water temperatures below 15 ^{0}C. Spawning begins at a water temperature of 6-7 ^{0}C in spring. In large lakes, the spawning period lasts up to 3-4 weeks. The fry hatch 2-3 weeks after spawning and when they are approximately 6-7 mm in length. In the first autumn, the juvenile fish attain sizes of 6-7 cm in favourable conditions. Perch grow very slowly: 3-year-old perch are 11-13-cm, 9-year-old perch 22-24 cm and the fish only attain a mass of 1 kg at 15-18 years. The average size is in the range 12-25 cm (average concomitant mass 100-400 g), although some grow up to 50-55 cm (concomitant mass 2-3 kg). Perch have a lifespan of approximately 13 years.

References

http://www.ahven.net/opetusmateriaali/talouskalalajit/kalat
http://www.laajy.edu.hel.fi/comenius/kotisivu/englanti/Ahven.htm
http://cnet.windsor.ns.ca/Environment/Advocates/Anim/perch.html
Koli, Lauri. Suomen kalat, WSOY, Porvoo, Finland, 1998 (in Finnish).

1.2.7 Common frog (*Rana temporaria*)

Classification



Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Lissamphibia*
Order: *Anura*
Family: *Ranidae*
Genus: *Rana*
Species: *Rana temporaria*

Physical characteristics

Rana temporaria is a small animal that has a squat body and no tail. It has a wide, flat head connected to a short, solid body. The common frog is approximately 7.5-8 cm long and can reach a length of 10 cm. The average mass is 23 g. There is a lot of variation in colour, with grey, olive, even yellow or pink hues as well. Females are typically yellower and may have patches of red on their sides.

Geographical distribution and habitat

Rana temporaria is a common terrestrial frog in Great Britain, Europe and north-western Asia. It is resistant to cold climates and lives as far north as the Arctic Circle in Scandinavia, farther north than any other amphibian in the region. It is abundant in the British Isles and most of Central and Northern Europe, but rarer in Spain, Portugal, Italy and the Balkans where it is only found in mountainous areas.
Rana temporaria can be found in any damp habitat within its range, though they are more common in cooler upland forests and wet meadows and mountain lakes. It searches for food during the night or rainy days. Before the autumn frosts, frogs seek out good hiding places in earth holes and go into hibernation, although a few adult males may over-winter at the bottom of ponds.

Dietary habits

Rana temporaria eats insects, their larvae, wood lice, spiders, snails and worms. Eating habits are greatly influenced by the time of year.

Reproduction and life history

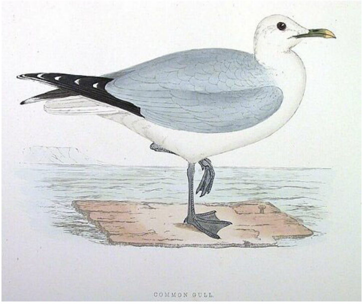
Rana temporaria breeds in warmer place in February-March and in the north as late as June. Frogs are explosive breeders with all spawn being laid over a ten day period at any one site. The eggs of *Rana temporaria* are in gelatinous envelopes (about 400 eggs) and are laid in thick groupings. It takes approximately 30-40 days for their eggs to hatch. During metamorphosis the tadpole grows legs – the hind legs are the first to grow. Its tail is absorbed into its body and it loses its gills and grows lungs. The structure of the digestive system, heart and skeleton change as well. Young frogs leave their ponds in June and quickly hide in the surrounding vegetation. *Rana temporaria* reaches sexual maturity in three years, returning to water where it originally metamorphosed to spawn. The average life time is estimated to be about 10 years.

References

http://animaldiversity.ummz.umich.edu/accounts/rana/r._temporaria.html
http://www.wartsoc.co.uk/rt.html
http://www.arclab.org/medlineupdates/abstract_8568500.html

1.2.8 Common gull (*Larus canus*)

Classification



Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Aves*
Order: *Charadriiformes*
Family: *Laridae*
Species: *Larus canus*

Geographical distribution

The Common gull is an aquatic bird, which has a wide global distribution. It is found all over Europe, easily located in Scandinavia and other parts of northern Europe. It is also abundant as a wintering bird in Western Europe, especially in coastal areas. In Europe, the Common Gull only breeds in northern regions. In Denmark, Germany and Poland, its breeding distribution is scattered and localised. In more northerly regions, such as Scotland and Scandinavia, Common Gull breeding distribution is more continuous.

Habitat

Outside the breeding season, Common Gulls remain at sea for long periods. They are commonly observed inland in parts of central Europe.

References

http://www.birdcheck.co.uk/main/previewpages/previewpage61.htm
http://www.birdguides.com/html/vidlib/species/larus_canus.htm

1.2.9 Muskrat (*Ondatra zibethicus*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Mammalia*
Order: *Rodentia*
Family: *Muridae*
Genus: *Ondatra*
Species: *Ondatra zibethicus*

Physical characteristics

Muskrat is a small furry rodent, which weighs in at an average 1.4 kg. Its total body length is twelve and a half inches.

Geographical distribution and habitat

Muskrats are distributed from northern North America to the Gulf coast. Early in the 20th century muskrats were introduced to northern Eurasia. It lives in most parts of Europe, Scandinavia, the former Soviet Union and China. Favouring an aquatic environment, *Ondatra zibethicus* fur consists of a thick waterproof layer of down with an overlayer of longer, glossy guard hairs in shades of light brown to almost black. Muskrats are found in wet environments, favouring locations with water depths of 15-60 cm. While muskrats are found in ponds, lakes, and swamps, their favourite locations are marshes, where the water level stays constant. They find shelter in bank burrows, which are tunnels excavated in the bank. Muskrats are good swimmers and can stay underwater for 12-17 minutes.

Dietary habits

Muskrats are mainly vegetarian but will eat animals as well. In summer muskrat favours a diet of shoots, roots, bulbs and leaves of aquatic plants and also cultivated crops such as carrots, corn etc. In winter they swim under the surface ice to get the plants. Occasionally, these adaptable creatures will eat clams, mussels or fish. Muskrats consume about one-third of their weight every day.

Reproduction

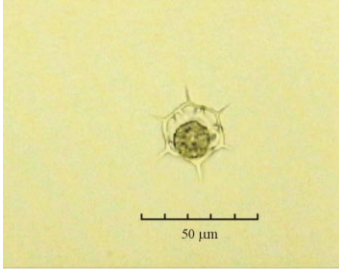
Northern populations breed only in the warmer months (March to August), but in southern regions year-round. The gestation period is 28-30 days. Northern animals will breed two or three times in the season. Young are born in grass-lined nests, with three to nine young per litter. When born, the muskrat has short dark fur, closed eyes, and weighs around 22 grams. They are able to swim at 10 days and by 21 days can eat green vegetation. In 30 days muskrats gain their independence and will reach adult size in 200 days. Muskrats are victims of many predators, e.g. marsh hawks, large owls, foxes and minks. Approximately 50% of the young will survive from birth to autumn and over winter survival varies from 19% to 68%. Life span is up to 4 years in the wild.

References

Baker, R.H. 1983. Michigan Mammals. Michigan State University Press. United States of America.
Http://animaldiversity.ummz.umich.edu/accounts/ondatra/o._zibethicus$narrative.html
http://www.fur.ca/archives/pubs/comm/comm07/34-e.asp
http://www.nsrl.ttu.edu/tmot1/images/ondazibe.jpg

1.3 Marine ecosystems

1.3.1 Phytoplankton



Phytoplanktons are the free drifting microscopic organisms that form the largest plant community in the oceans. Though normally existing in solitary form, they may form large chains or spherical shaped colonies, some large enough to see with the naked eye. These single celled organisms are the primary food source, directly or indirectly, of all sea organisms. Phytoplankton contain the pigment chlorophyll, which is used by plants for photosynthesis, in which sunlight is used as an energy source to fuse water molecules and carbon dioxide into carbohydrates—plant food. Because sunlight is most abundant at and near the sea surface, phytoplanktons remain at or near the surface.

Diatoms and Dinoflagellates form the main groups of phytoplankton inhabiting the Arctic Ocean, where the bottom surface of the ice, the ice-water interface and the water column form distinct habitats, which are colonized by different taxonomic assemblages. It is stated that pennate diatoms are dominant in the bottom ice, centric diatoms at the ice-water interface and flagellates in the ice-covered water column. The biomass and production in the sea ice are generally dominated by large algal cells (> 5µm) while the under-ice water column is dominated by small algal cells (0.7-5µm).

Phytoplankton varies seasonally in amount, increasing in spring and Autumn with favourable light, temperature, and minerals.

Phytoplankton’s reproduction occurs both sexually and asexually. While their asexual reproduction is based on binary fission, cell fusion forms the basis of sexual reproduction.

Sources of information

Image: http://www.nodc.noaa.gov/OC5/BARPLANK/WWW/HTML/allphoto.html

Encyclopædia Britannica 2003 Encyclopædia Britannica Online.
06 Feb, 2003 <http://search.eb.com/eb/article?eu=61384>.

http://www.marine.ie/scientific+services/monitoring/phytoplankton/

Gosselin M., M. Levasseur, P. A. Wheeler, R. A. Horner, B. C. Booth (1997). New measurements of phytoplankton and ice algal production in the Arctic Ocean. *Deep-Sea Res. II*, **44**, 1623-1644.

1.3.2 Macroalgae - Bladder wrack (*Fucus Vesiculosus*)

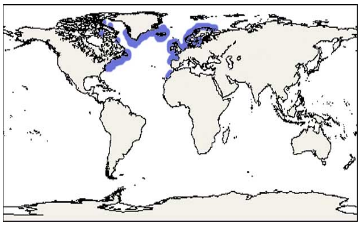
Classification

Kingdom: *Protista*
Phylum: *Chromophycota*
Class: *Phaeophyceae*
Order: *Fucales*
Family: *Fucacea*
Genus: *Fucus*

Physical characteristics

The bladder wrack, *Fucus vesciculosus*, is a large brown algae, common on the middle shore. It is found in high densities and fronds grow up to 2 metres long, living for about three years.

Geographical distribution and habitat



The species is found intertidally on rocky shores in a wide range of exposures. It provides substrate and shelter for herbivorous isopods and surface grazing snails.It is found in the Baltic Sea, Norway, Britain, Ireland, Atlantic coast of France, Spain and Morocco, Iceland, Greenland and the eastern shores of United States and Canada.

General information

The morphology of the plant varies in response to the environmental conditions leading to distinct varieties. Plants from exposed locations usually have no airbladders and are known as *Fucus vesiculosus* forma *linearis*. The loss of airbladders is thought to be because they increase a plant's drag, making them more vulnerable to being washed off by waves. Depth is not relevant as the plant is intertidal although it does occur at shallow depths in the Baltic.No conducting tissue is found in *Fucus spp*.; it is unnecessary as the plant is small enough to be able to manufacture food locally.

Reproduction

These brown algae have a gametic life cycle. That is, the products of meiosis are gametes. Gamete production takes place in specialised crypt-like structures called *conceptacles* which are borne in fertile, swollen areas at the tips of the plants called *receptacles*. Some species are monoecious with both sexes occurring on one plant; others are dioecious with each sex being found on different plants. Some monoecious species may have both sexes in one conceptacle whilst others may have them in separate conceptacles.The species is highly fecund often bearing more than 1000 receptacles on each plant, which may produce in excess of one million eggs. Development of the receptacles takes three months from initiation until when gametes are released. On British Shores receptacles are initiated around December and may be present on the plant till late summer. Gametes may be produced from mid winter until late summer with a peak of fertility in May and June. Eggs and sperm are released into the seawater and fertilised externally. Zygotes settle to the seabed and begin development wherever they fall. The egg becomes attached to the rock within a few hours of settlement and may adhere firmly enough to resist removal by the next returning tide. Size at maturity is 15-20 cm. *Fucus vesiculosus* is used in cosmetic preparations and in thalassotherapy.

Sources of information

Image: http://www.horta.uac.pt/species/Algae/Fucus_vesiculosus/Fucus_vesiculosus.htm

White, N., 2000. *Fucus vesiculosus*. Bladder wrack. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme* [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 06/08/02]. Available from: http://www.marlin.ac.uk

http://www.seaweed.ie/descriptions/fucves.html

1.3.3 Pelagic crustacean - Northern shrimp (*Pandalus borealis*)

Classification

Kingdom: *Animalia*
Phylum: *Crustacea*
Class: *Malacostraca*
Order: *Decapoda*
Family: *Pandalidae*
Genus: *Pandalus*

Geographical distribution and habitat

Pandalus borealis, the northern shrimp, is a very important commercial product. It is one of the most common and numerous of invertebrate species in the Atlantic, from the North Sea to Spitsbergen, Iceland, along the shores of Newfoundland and Greenland, and in the Pacific Ocean, from the Japan Sea and British Columbia to the Bering Sea.*P. borealis* is most common over a soft mud bottom. Its bathymetric range is from 9 to 1380 m but fishable concentrations normally occur between 54 and 400 m. There is a direct relationship between abundance of this shrimp and high organic content in sediment. This shrimp exhibits migratory behaviour, inshore-offshore migrations, which are related to seasonal and inshore-offshore temperature differences. Both the distribution and migratory behaviour of northern shrimp change with age. Adult shrimps tolerate water temperatures from -1.68 to 11.13˚C, whereas larvae may live at 14˚C. Both larvae and adults have been found at salinities from 25.9 to 35.7 per cent.

Feeding behaviour

The diet of *P. borealis* is obtained from the plankton as well as from the benthos. The shrimp feed on euphausiacea, copepods, mysids, decapod larvae, harpacticids, isopods, tanaidaceans, cumaceans and benthic amphipods. The polychaetes are second in importance to the crustaceans in terms of the number of species consumed. The spectrum of food organisms is determined essentially by the prey available, the time of day, and the developmental stage of the shrimp. Following stomach investigations it has been reported that the shrimps have a nocturnal activity phase during which they mainly feed on plankton. On the other hand, there is also a diurnal activity phase during which benthic species are consumed, and the stomachs are filled to a maximum degree in the afternoon. The males feed on plankton in the pelagic zone more actively than do females. In its habitat, *P. borealis* is eaten by large fish such as dogfish, Greenland halibut, turbot, and hake.

Sex change, spawning and hatching

Pandalus borealis is a protandric hermaphrodite, which reproduce first as male and subsequently changes into female and spawn as such for the rest of its life. Temperature plays a significant role in determining the time (age) of sex change. Over its geographic range, the northern shrimp has different seasons of spawning and hatching, and water temperature appears to be the controlling factor. In southern Norway, where mean annual bottom temperature is about 7˚C, spawning take place in October and November and hatching of eggs in March and April, for an ovigerous period of between five and six months. Upper north in Norway (Ofoten and Mist Fjords), where mean annual bottom temperature is about 5˚ C, spawning occurs in September and October and hatching in April and May. In the far northern areas (Spitsbergen, Jan Mayan, western Greenland), having mean temperatures of 1˚C or less, the ovigerous period (including spawning and hatching periods) may begin as early as July or August and last 10 to 12 months. The life span of *P. borealis* range from 3 to over 8 years in various locations in the Atlantic and its length can reach up to 120 mm or larger. In high latitudes and at colder ambient temperatures the growth rate is slower, the life span and ovigerous period longer and age at sex changes later.

Sources of information

Image: http://www.coastalimagery.com/gallery/okh52.htm

BUTLER, T. H., 1971. A review of the biology of the pink shrimp, *Pandalus borealis*. In: Proceedings – Conference on the Canadian Shrimp Fishery, St. John, N.B., Oct. 27-29, 1970. *Can. Fish. Rep*., No. **17**, 17-24.

SKÚLADÓTTIR, U. 1999. Defining populations of northern shrimp, *Pandalus borealis* (Krøyer 1838), in Icelandic waters using the maximum length and maturity ogive of females. *Rit Fiskideildar*, **16**, 247-262.

WIENBERG, R. 1981. On the food and feeding habits of *Pandalus borealis* (Krøyer 1838). *Arch. Fischereiwiss*. **31**, 123-137.

APOLLONIO, S., STEVENSON, D. K., DUNTON, E. E. 1986. Effects of temperature on the biology of the northern shrimp, *Pandalus borealis*, in the Gulf of Maine. *NOAA Technical report NMFS 42*.

IVANOV, B. G. 1969. Biology of the northern shrimp (*Pandalus borealis*) in the Gulf of Alaska and the Bering Sea. Tr. VNIRO, **65**: 392-416. *Trans. Fish. Res. Bd. Can., Transl. Ser.*, No. 1576, 1970.

1.3.4 Worm - Blow lug (*Arenicola marina*)

Classification

Kingdom: *Animalia*
Phylum: *Annelida*
Class: *Polychaeta*
Order: *Capitellida*
Family: *Arenicolidae*
Genus: *Arenicola*

Geographical distribution and habitat

Lugworms are burrow-dwelling annelid worms, and can reach densities as high as 100-150 per square metre in certain areas. They live in U or J-shaped burrows (20-40cm deep) with characteristic depressions at the head end (the 'blow hole') and a cast of defaecated sediment at the tail end. These worms can make up to 30% of the biomass of an average sandy beach, making them a very important part of the food web of these beaches. They bioturbate the sand and are food for a wide variety of other animals such as flatfish and wading birds, which may 'nip' off the tail as it deposits casts.Population density is correlated with mean particle size and organic content of the sediment. Arenicola marina is generally absent from sediments with a mean particle size of <80µm and abundance declines in sediments >200µm (fine sand) because they can not ingest large particles. Its absence from more fluid muddy sediments is probably because they do not produce large amounts of mucus with which to stabilise their burrows. Populations are greatest in sands of mean particle size of 100µm. Between 100-200µm the biomass of *Arenicola marina* increases with increasing organic content. However, juveniles prefer medium particle sizes (ca. 250 µm) over fine or coarse sand. Lugworms have a wide distribution and are found in shores of Western Europe, Spitzbergen, north Siberia, and Iceland. In the western Atlantic it has been recorded from Greenland, along the northern coast form the Bay of Fundy to Long Island. Its southern limit is about 40° N.

Feeding behaviour

It has been observed that the lugworms show a pronounced preference for small particles. This is ascribed to a difference in chance of adhesion of small and large particles to the papillae of the proboscis. *Arenicola marina* ingests small particles (<2mm) which stick to the proboscis papillae while larger particles are rejected. It feeds on micro-organisms (bacteria), meiofauna and benthic diatoms in the sediment and is also capable of absorbing dissolved organic matter (DOM) such as fatty acids through the body wall. Feeding, defaecation and burrow irrigation is cyclic. Each cycle takes about 42 minutes in large worms but 15 min in smaller worms, depending on individual. Each cycle consists of defaecation (worm mainly in the tail-shaft), followed by rapid irrigation and a longer period of feeding, after which the worm defaecates again and the cycle repeats.

Reproduction

Lugworms have separate sexes with external fertilization and an annual episodic breeding frequency; their spawning is highly synchronized and usually occurs on only one or two days a year over a two week period in October to November. The exact timing of spawning varies between locations and some populations demonstrate protracted spawning. *Arenicola marina* is sexually mature at 1-2 years. While the number of eggs can varies between 100,000-1,000,000 the average number of oocytes is reported to be 316,000 oocytes per female, with an average wet weight of 4 g.

Spawning occurs at low tide, and as the tide comes in, the viscous sperm puddles are washed, diluted and enter the burrows of the females. The sperm puddles contain inactive sperm, the addition of seawater triggers them to become active and begin swimming. Fertilization occurs in the female burrow and after four to five days the larva hatches, 0.24 mm long. The larvae undergo early development here, later moving to the surface to be transported by the tide to settle on firmer areas. They then develop in mucous tubes attached to the substratum. Once developed, the worms are carried by the tide to more sandy/mud sediments where they can burrow.

Adults reach between 120 -200mm in length and vary in colour from pink to dark pink, red, green, dark brown or black. The suggested life span is 5-10 years. *Arenicola marina* is used routinely as a standard bioassay organism for assessing the toxicity of marine sediments.

Sources of information

Image: Dr Matt Bentley (published on the MarLIN Web site)

Tyler-Walters, H., (2001). *Arenicola marina*. Blow lug. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme* [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 06/08/02]. Available from: <http://www.marlin.ac.uk>

http://www.marinebio.com

Newell, G. E., (1948). A contribution to our knowledge of the life history of *Arenicola marina.*

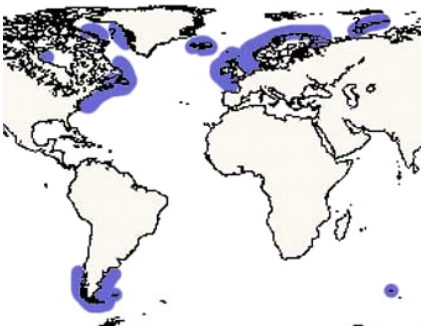
1.3.5 Bivalve mollusc - Blue mussel (*Mytilus edulis*)

Classification

Kingdom: *Animalia*
Phylum: *Mollusca*
Class: *Pelecypoda*
Order: *Mytiloida*
Family: *Mytilidae*
Genus: *Mytilus*

Geographical distribution and habitat

The blue mussel, *Mytilus edulis,* is a semi-sessile epibenthic bivalve that is anchored to a secure substrate, or attached to other mussels, with byssus threads secreted from glands in the animal's foot. As a gregarious organism it (at high densities) forms dense beds of one or more (up to 5 or 6) layers. It is found intertidally and subtidally, in estuarine and fully saline habitats.



M. edulis is widely distributed in the northern hemisphere; it occurs in European waters extending from the arctic waters of the White Sea and northern Norway southwards to as far south as the Atlantic coast of southern France. In the W. Atlantic it extends from the Canadian Maritimes south to North Carolina. It occurs on the coasts of Chile, Argentina, the Falkland Islands and the Kerguelen Isles. *Mytilus edulis* has been reported from Iceland.

Dietary habits and predators

The blue mussel, both as a planktotrophic larva and as an adult, is an active suspension feeder, deriving its nutrition by filtering organic particles from the water column. Phytoplankton cells are the dominant food source for all life stages. Attached bacteria are a major source of protein in detritus, and there is evidence that adult blue mussels can digest bacteria. Both larvae and adults use cilia to remove food particles from suspension. The mussel is capable of removing particles down to 2-3 µm with 80-100% efficiency and shows a great range of adaptations to changing conditions, including the ability of adjusting its filtration rate to maintain a complex balance between the amount of material filtered, the amount rejected as pseudofeces, and the amount ingested. Predation pressure on the blue mussel is highest during the 3 weeks when it is a planktonic larva, for it is then subject to grazing from a wide variety of species, ranging from jellyfish to larval and adult fishes. The vulnerability of mussels decreases as they grow and attain relatively large and thick shell (4-5 cm). They may then be preyed upon by predators such as large starfish, large crustaceans, and some birds.

Life history

Mussels generally produce gametes and are ready to spawn by the time they are one year old; however, when adverse environmental conditions (e.g., prolonged periods of exposure to air) cause a slow rate of growth, sexual maturity is sometimes not attained until the second year. Gametogenesis, spawning, and nutrient storage are linked in an integral process termed the reproductive cycle. This cycle in any blue mussel population is the result of a complex balance between exogenous factors such as food availability, temperature, salinity, and duration of exposure to air and endogenous factors such as nutrient reserves, hormonal cycle, and genotype. Thus, it is impossible to predict the timing of the reproductive cycle for any particular population except for environments in which variations in physical factors are not large. In general, mussels from the warmer more southerly waters of the northern hemisphere spawn earlier than those further north.

Fertilization is external. Fecundity and reproductive effort increase with age and size, young mussels divert energy to rapid growth rather than reproduction. An individual female (ca 7mm) can produce 7-8 million eggs, while larger individuals may produce as many as 40 million eggs.

Larval development

The stages of larval development and their durations are summarized in the Table below. It must be emphasized that the larval stage may last anywhere from 15 to 35 days and that the duration is dependent on prevailing environmental conditions.

Life stages and characteristics of the blue mussel (Bayne 1976b)

Stage	*Size (length)*	*Age and characteristics*
Fertilized egg	68-70 µm	0-5 h Non motile.
Trochophore	70-110 µm	5-24 h Ciliated and motile.
Veliger		Up to 35 days; Feeds and swims with ciliated velum.
Plantigrade	0.26-1.5 mm	Up to 6 months; temporarily attached to filamentous substrates.
Juvenile		Up to 2 years; sexually immature.
Adult	Up to 100 mm	Up to 20 years; sexually mature.

Sources of information

Image: Keith Hiscock (published on the MarLIN Web site)

Tyler-Walters, H., (2002). *Mytilus edulis*. Common mussel. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme* [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 06/08/02]. Available from: <http://www.marlin.ac.uk>

Seed, R. (1976). Ecology. In: Marine Mussels: Their Ecology and Physiology (B.L. Bayne, ed). Cambridge University Press.

The mussel Mytilus: ecology, physiology, genetics and culture. Developments in Aquaculture and Fisheries Science, volume 25. Gosling, E. (ed.), Elsevier, 171-222.

Newell, R. I. E. (1989). Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (North and Mid-Atlantic) blue mussel. U. S. Fish. Wildl. Serv. Biol. Rep. 82 (11. 102). U. S. Army Corps of Engineers. TR El-82-4. 25 pp.

1.3.6 Crustacean - European lobster (*Homarus Gammarus*)

Classification

Kingdom: *Animalia*
Phylum: *Arthropoda*
Class: *Malacostraca*
Order: *Pleocyemata*
Family: *Nephropidae*
Genus: *Homarus*

Geographical distribution and habitat

The European lobster is found in the eastern Atlantic from northern Norway (Lofotoen Islands) to south-eastern Sweden and Denmark, where it is apparently blocked from inhabiting the Baltic Sea by lowered salinity and temperature extremes. Its distribution extends southward along the mainland European coast and around the Great Britain and the Azores, to a southern limit of about 30˚ north latitude on the Atlantic coast of Morocco. This species also occurs, though less abundantly, in the north-western regions of the Black Sea and in the coastal and island areas of the Mediterranean Sea and its subseas. The European lobster generally selects or excavates shelter on rocky or stony bottoms where the substrate is sand or gravel. Juveniles and adults dig out hollows or tunnels under the boulders or stones with one or more openings, using the hollows as hiding places. It is found from very shallow water to the depth of 150 meters, but is more common at depth of 10-60 meters. Marking and recapturing experiments off Ireland, Scotland, and Norway suggest that the European lobster does not undertake extensive migrations alongshore or inshore-offshore. Maximum distances travelled were 8-12 km, with an average of approximately 2 km.

Dietary habits and predators

Reported observations indicate that the European lobster hides during the day within its shelter and forages for food at night. Like the American lobster, a larger percentage (60-70%) of the population leaves their shelter during the summer and fall than during the winter. Lobsters normally do not feed in the winter, but remain in their shelters when the water temperature falls below 5˚ C. Investigations of the feeding behaviour and diet of the European lobster off the west coast of Sweden indicated that the major food items consumed were crabs, gastropods, polychaetes, with mussels and starfish comprising a minor portion of the diet. During the molting season lobsters ate a lot of calcareous material. Berried females had the same feeding behaviour and diet as other lobsters. Small lobsters preferred polychaetes, small crabs, and gastropods. The major predators on the juvenile lobster are sculpin, cunner, tautog, black sea bass, and sea raven.

Life history

New lobster life begins as thousands (5000-20000) of fertilized eggs, about 1 mm in diameter, are pushed out of the female's oviducts. The embryos travel along the underside of their mother's abdomen until they reach the pleopods, where they attach and remain for the next nine to eleven months. The incubation period of the eggs is highly variable and temperature dependent. Fully developed embryos hatch as pre-larvae. On their way toward the surface waters, they molt into the first larval stage. Lobsters have three distinct planktonic larval stages with a total duration of 3-6 weeks. Metamorphosis from the larval to a postlarval stage occurs at the fourth molt. These postlarvae lobsters move downward to the sea bottom and start a benthic life.Adulthood is reached after five to eight years, depending largely on the water temperature. The lobsters mate when the female changes her shell. The spawning period for European lobster begins early in July and extends into September. The European lobster can exceed 5kg in weight and 1m in length. They are solitary creatures with a potential lifespan of 30 years.

Sources of information

Image: Erling Svensen (published on the *marinbi* Web site)

Wilson, E., (2002). *Homarus gammarus*. Common lobster. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme* [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 28/02/03]. Available from: <http://www.marlin.ac.uk>

The biology and management of lobsters. Edited by J. Stanley Cobb, Bruce F. Phillips. New York: Academic press, 1980.

http://www.tolgus.com/marinelife/lobster.htm

1.3.7 Benthic Fish - Plaice (*Pleuronectes platessa*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Actinopterygii*
Order: *Pleuronectiformes*
Family: *Pleuronectidae*
Genus: *Pleuronectes*

Geographical distribution and habitat

The plaice is one of the most economically important flatfish in Europe. It inhabits most of the shallow coastal waters of Northern Europe. It is found as far north as Iceland and The Faeroes and south along the coasts of France, Spain and Portugal with specimens being recorded in the Mediterranean Sea. They are found in the Irish Sea in the west and extend east to the North Sea, Baltic Sea and as far as the White Sea off the northwest coast of Russia. There appear to be distinct stocks of plaice near the Murman coast, Iceland, The Faeroes and in the Baltic Sea, each with its own migration pattern.

The plaice lives on sandy and muddy bottoms from the shoreline and to a depth of about 200 metres. Most adults are found at depths of 10-50 metres, while the young are almost exclusively found in shallower water. In the autumn, when the young are 7-12 cm, they move into deeper water to pass the winter. These observations have been expressed as a law wherein size increases with distance from shore (and depth), while numbers decline.

Dietary habits and main prey species

As larvae they feed on the microscopic larvae of worms and gastropods and as young they feed mainly on small worms and crustaceans, but with increasing age they start to take larger food animals; by autumn they are eating larger bristle-worms, sand hoppers and thinshelled bivalves, which also form the principal diet of the adults; they are then about 7-12 cm long and move slowly into deeper water for the winter. During the winter the plaice´s food consumption is reduced and it is not until spring that the young fish move back again to their feeding grounds in shallow water.

Reproduction and general life cycle

As with most fish, the breeding cycle of plaice is temperature dependent. Spawning takes place at temperatures of about 6°C, in the western Baltic from November to June in depths of 69-90 metres, in the North Sea from January until June in depths of 20-40 metres. The main spawning grounds for North Sea plaice are south of Dogger Bank. Off Iceland the plaice spawns in March-April, in the Barents Sea from March to May, partly in depths of 160-200 m at a temperature of only 2-2.5°C, and partly also in shallower water. The female, depending on her size, lays between 50000 and 520000 eggs. The eggs have a diameter of 1.6 mm and are shed and fertilized above the sea bed in areas of sufficient salinity for them to float. The eggs hatch in 2-4 weeks, according to the temperature, the colder the later. The newly hatched larvae are 6 mm long and look like normal round fish living in a pelagic state. 1-2 months later, when they are about 10 mm long, they start their transformation to a bottom living fish. The left eye wanders up over the head and the young start swimming on their left side. When they reach 12-14 mm in length they abandon their pelagic lifestyle and move to shallower coastal waters. It can take 2-3 years before they move to deeper water. In the North Sea the males reach sexual maturity in their 3rd- 4th year at a length of 18-26 cm, the females in the 6th year at a length of about 35 cm, but in the Barents Sea the males in their 6th-9th year (30-40 cm) and the females in their 7th-13th year (34-47 cm). Age at maturity depends on water temperature. The colder the water, the later the fish matures. Plaice can grow very large, up to 1m in length, average length 25-40 cm. The largest plaice caught weighed 7kg. They are also very long lived and can survive to 50 years of age. Females grow faster and live longer than males.

Sources of information

McKeown, B. A. (1984). Fish Migration

Muus, B. J. and Dahlstrøm, P. SEA FISHES of Britain and North- Western Europe

http://www.dainet.de/bfafi/ifo/Cuxhaven/body_plaice.htm
http://www.divernet.com/biolog/plaice0200.htm
http://web.ukonline.co.uk/aquarium/pages/plaice.html
http://www.marine.ie/industry+services/fisheries/fisheries+biology/plaice+biology.htm
http://www.vattenkikaren.gu.se/fakta/arter/chordata/teleoste/pleuplat/pleupl2e.html#ater

1.3.8 Pelagic Fish – Common Mackerel (*Scomber Scombrus*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Osteichthyes*
Order: *Perciformes*
Family: *Scombridae*
Genus: *Scomber*

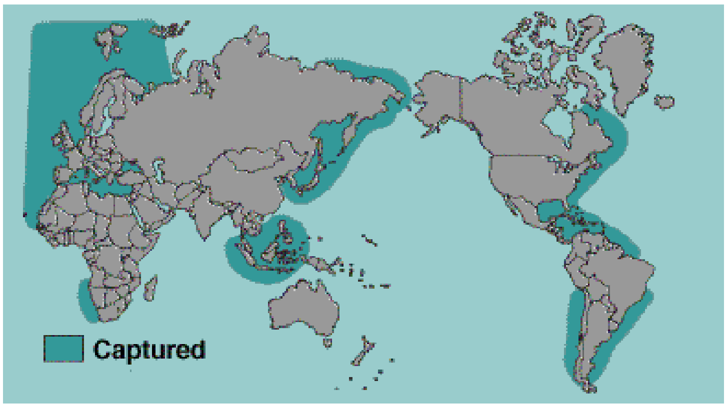
General information

The common mackerel is an abundant and economically important species. It is a pelagic fish and one of the smaller fishes of the family that includes tunas. All members are streamlined predatory fish, supremely designed to cleave through water, with the absolute minimum of protuberances to offer resistance to their swift passage. Its speed can reach 100 metres in 11 seconds. It averages about 30 cm in length and 0.675 kg in weight. One of this species' main distinguishing features is the lack of a swim bladder (air bladder), requiring the animal to swim continually so that it will not sink. Mackerel are, therefore, in almost continuous motion, their power of endurance being equal to the rapidity of their motions. The constant motion forces water through the gills where oxygen is extracted. The fish would suffocate if motionless.Mackerel travel in schools that often contain thousands of fish and migrate between deep and shallow waters. They swim actively in the upper 50 m of the water in the warmer months and then descend to as deep as 185 m during the winter. All individuals within a specific school tend to be the same size. Since cruising speed increases significantly with age and size, scientists believe that conformity of body size within a specific school is necessary to allow all fish to maintain identical swimming speeds.

Geographical distribution

The common mackerel occurs along both coasts of the North Atlantic Ocean, from the Canary Islands to the Orkneys, and from the Mediterranean and the Black Sea and the coasts of Norway to the United States.

In Europe, the mackerel family is divided into two main stocks: one spawning in the North Sea and Skagerrak, and the other spawning to the west of the British Isles. The mackerel fished in the North Sea, Skagerrak and the Norwegian Sea generally belong to the western stock of mackerel.



Dietary habits and main prey species

Towards the spring large schools approach the coasts, searching for food as in the warmer season the food is more abundant in the neighbourhood of land than in the open sea. Young mackerel feed on microscopic copepods. As they grow, they feed on progressively larger prey. Adults will eat any fish smaller than themselves, feeding heavily upon small herring, sand lance and young mackerel. They also consume a variety of invertebrates such as copepods, crab larvae, squid and shrimp.

Stomach content analysis of adult mackerel sampled in Loch Ewe and the Minch (Scottish seas) during the summer months indicates a variety of prey including small fish such as sand-eels, *Ammodytes spp*., small gadoids (cod family), and clupeoids (herrings), as well as filter feeding for small crustaceans such as Euphausids. In the latter case, small creatures exceeding 1.5 mm can be sieved from the water in close-set gill-rakers. This can occur when the fish swims at speeds in excess of 0.7 body lengths per second. Feeding ceases when the mackerel return to deeper water, during the winter.

Reproduction

Mackerel reproduce from spring through summer, with more northerly fish spawning later in the season. Mackerel spawn near the surface and along coastlines. Their eggs average 1 mm in diameter, are buoyant, and drift in the uppermost 30 feet of the water column. Many males and females reach sexual maturity at the age of 2 and all do so by 4. The fecundity of females increases as a function of age and size, with an individual female spawning 550,000 to 1,000,000 eggs per season. Surviving larvae at 3 mm long will feed on copepods. By the first autumn, the North Sea population of Mackerel will attain a length of 15 cm, reaching 25 cm after one year. After the spawning the schools break up into smaller companies which are much scattered. Mackerel may grow as large as 3.4 kg and have a maximum age of more than 20 years. Fish of over 25 years old have been caught in the North Sea.

Sources of information

"Mackerel" Encyclopædia Britannica. 2003. Encyclopædia Britannica online. 04 Aug. 2003 <http://search.eb.com/eb/article?eu=50996>
http://www.seafood.no/eff/eng/effandfacts.nsf
http://ourworld.compuserve.com/homepage/BMLSS/mackerel.htm
http://www.newmex.com/platinum/data/light/species/mackerel.html
http://www.encyclopedia.com/html/m1/mackerel.asp
http://www.cptdave.com/atlantic-mackerel.html

1.3.9 Seabird/duck- Common eider (*Somateria mollissima*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Aves*
Order: *Anseriformes*
Family: *Anatidae*
Genus: *Somateria Leach*

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General habits, habitat and migration

Common eiders are sea ducks found across the arctic and subarctic zones of the northern hemisphere. They are the largest duck in the northern hemisphere, and easily identified by the male's dramatic pattern of black and white plumage. The common eider frequents coastal headlands, offshore islands, skerries, and shoals, as far north as open water permits. The king eider *Somateria spectabilis*, spectacled eider *Somateria fischeri*, and common eider all belong to the same genus, and hybridization is known to occur between common and king eiders. Common eiders average about 1800 g in weight, but this varies considerably with race, sex, and time of year. There are four subspecies of common eiders in North America. Three other subspecies are found in other places: one in northwest Europe, one in Iceland and a third in the Faeroe Islands. In late Autumn, eiders begin to migrate south. The timing is largely influenced by the onset of ice formation and by the growth of pack ice, which occurs progressively later as one proceeds south and is a more important influence in the northwest Atlantic than elsewhere. There is generally a supplanting effect; birds breeding in an area are absent seasonally and their place is taken by others from farther north. Some Common Eiders are longdistance migrants, some are shortdistance migrants, and others are essentially sedentary. Spring migration of the common eider is rapid. Most birds fly along the coast, though some individuals are known to cross over significant tracts of land. Fall/winter migration is slower and more leisurely. During fall and winter, eiders rarely cross land, usually only projecting points of land or headlands, and then only under certain weather conditions such as snow and onshore winds. The eiders of the inner Gulf of St. Lawrence are an exception, many of them moving first southwest into the St. Lawrence estuary a short way downstream of Quebec City and then flying over much of the state of Maine. In summer, there are significant moult migrations of adult males and nonbreeders, which move often several hundred kilometres north from their breeding areas. The moult migration is undertaken so that the eiders can replace their worn-out plumage in an area where they will be protected from weather and predators. During the moult, as new feathers are replacing old ones, the eiders are unable to fly for a three- to four-week period. At this time eider ducks lose weight as energy is used to shed and grow new feathers. By mid-September the drakes have resumed flight and are ready to return to the wintering grounds. The adult females moult later, and young of the year and adult females are the first to arrive on wintering grounds.

Dietary habits and main prey species

Eider ducks are gregarious, travelling and feeding in flocks numbering from tens to thousands. Eiders feed during the day by diving to the bottom in waters from 3 to 20 m deep to take mussels, clams, scallops, sea urchins, starfish, and crabs, which are swallowed whole and crushed in the large gizzard. During the breeding season, they may also forage for algae, berries and seeds.In winter, when daylight is short, more than half the daytime hours are spent in feeding. After 15-30 minutes of intensive feeding, flocks move offshore to rest, preen, and digest the contents of the gullet. The feeding sequence is then repeated. At night, they gather in compact flocks, sometimes offshore, at other times in the protection of a headland or cove. It is known that in winter when the temperature drops eiders minimize their energy expenditure. They become inactive, stop feeding and, presumably to insulate themselves, gather in groups so dense that individual ducks cannot be counted. Losses to predation by gulls, jaegers, and foxes can occur during the incubation period and shortly after hatching.

Breeding and general life cycle

Common eiders breed on small offshore marine islands (and occasionally on islands in inland lakes) or isolated spits and points that are free of mammalian predators. they nest in early summer; nesting starts progressively later as one proceeds farther north. Eiders return to the breeding islands as soon as shore-fast ice or pack ice starts to dissipate. Many eider ducks are paired when they arrive on the breeding grounds, although some pairing occurs there. They nest in dense colonies of tens to ten thousand or more. During spring migration, and when the eider ducks have just arrived near their breeding places, much time is spent feeding, and the birds store fat. These reserves carry the hens and drakes through the incubation period. Unlike many ducks, the hen does not feed once she starts sitting on her eggs, nor does the drake feed during courtship and egg laying. Prospecting flights and visits to lakes on islands are made within a couple of weeks of arrival. After several days the nest site is prepared, and the female eider duck lines her nest with down and other available materials such as seaweed, lichens, moss, sticks and grasses. Egg laying commences shortly thereafter. Female eiders lay two to eight eggs each year; there are usually four or five eggs per nest. Incubation begins after the second or third egg has been laid and lasts for about 25 days (range, 24-30 days). Only the female eider ducks incubate the eggs. About 50% of the eggs hatch successfully. Broods are often raised by extended families and/or with the help of non-breeding females. Male molting begins in late June to early July with females molting much later after the brood are raised. Some female eiders may breed in their second year of life, but males do not breed until three years of age. They undergo major changes in appearance in the course of a year, so that in any flock of eiders many different plumages can be seen. Young first fly at 60 days of age. Generally, few survive to fly; many are lost to predators, exposure, or starvation in their first week of life. In good years, one duckling per adult pair may survive for the fall flight. On the other hand, adults are often longlived, up to 20 years, and estimated annual survival rates vary from 80 to 95%. In Norway, Iceland and Russia, eider ducks are farmed for their incredibly soft down. Farmers collect the first lining of down from eider nests, after which the female ducks reline their nests before laying their eggs.

Sources of information

http://collections.ic.gc.ca/arctic/species/ceiphoto.htm
http://www.cws-scf.ec.gc.ca/hww-fap/hww-fap.cfm?ID_species=12&lang=e
http://www.stpaul.gov/depts/parks/comopark/zoo/eiderduck.html
http://www.deltaduckhunter.com/waterfowl_id/common_eider.htm
http://www.state.ak.us/local/akpages/FISH.GAME/notebook/bird/eiders.htm

1.3.10 Marine mammal – Harp Seal (*Phoca groenlandica*)

Classification

Kingdom: *Animalia*
Phylum: *Chordata*
Class: *Mammalia*
Order: *Carnivorora*
Family: *Phocidae*
Genus: *Phoca*

General habits, habitat and home range

Harp seals are the third most abundant seal in the world and probably the most commercially important ones. They inhabit the North Atlantic and Arctic Oceans from northern Russia, to Newfoundland and the Gulf of St. Lawrence, Canada. Harp seals are separated into three populations based on where they breed; the White Sea north of the Russia, the "West Ice" near Jan Mayen Island southeast of Spitsbergen, Norway, and off Newfoundland. The last population is divided into two herds, one breeding on the southward drifting Arctic pack ice off Southern Labrador (called the "Front" sub-population) and the other breeding on ice in the Gulf of St. Lawrence near the Magdalen Islands (called the "Gulf" sub-population). In years of negligible ice in the Southern Gulf, some seals that would normally have whelped (given birth) there reproduced instead on the Labrador ice floes. In spite of the evidence of mixing between the sub-populations, there is a consistent difference of about 5 days in the dates of whelping between the two areas. From recent marking studies and blood protein analyses, it now seems likely that these sub-populations do interbreed. The difference in birth dates of pups between the two areas appears to be the result of environmental differences. The survival of a harp seal pup during its first two weeks depends upon the availability of stable habitat. At the Front, heavy Arctic ice provides this stability until late March or early April. In the Gulf, the ice usually begins to disappear by mid-March and for the pup to survive, it must be born earlier than at the Front. Harp seals are highly gregarious marine mammals, hauling themselves out of the water on to the ice in dense herds to bear their young, to mate and to moult. They also migrate and feed in loose herds of up to several hundred individuals. Harp seals are closely associated with pack ice. During spring, they migrate north following the receding pack ice. Herds that breed in the Gulf of St. Lawrence migrate north to Hudson Bay, Davis Strait, and Baffin Bay. The breeding population that congregates in the White Sea off the coast of Russia, and the population that pups mainly between Jan Mayan and Svalbard, move to ice patches north of the breeding areas which include the northern Barents and Kara Seas north of Svalbard, Franz Josef Land, and Severnaya Zemlya. Animals that reach the maximum extent of the range may migrate as far as 5000 km. The southward migration begins just ahead of the formation of new Arctic ice and involves all adults and most juveniles. Some immature seals spend much of the winter in the Arctic, as tagged seals have been recorded at West Greenland in all months. All three populations exhibit similar patterns of annual migration, although the timing of specific events such as pupping, varies slightly from place to place.

Dietary habits and main prey species

Harp seals consume a wide range of prey species and their diet appears to vary with age, season, location and year. They feed primarily on small marine fish and secondarily on crustacean macroplankton. Pups feed on crustaceans, mainly krill and amphipods of the genus *themisto*. The diet of older harp seals also comprises krill and *themisto*, but in addition, is characterized by substantial amounts of fish such as Arctic cod, Atlantic cod, capelin, and herring [AMAP, P. 133]. Young seals feed in the surface waters while adult harps dive deeper for cod and herring. They are reported to be capable of diving to depths of 100 to 150 fathoms (1 fathom = 1.83 m) and remain submerged for up to 15 minutes. One seal consumes about 450 kg of fish annually, cod being their most important food. Intense feeding occurs during summer and winter while less feeding occurs during spring and fall migration, whelping and moulting. The few predators that take harp seals are polar bears, killer whales, sharks and humans. Other causes of mortality are decreases in food by large scale capelin fisheries, discarded netting, and oil pollution.

Natural history (Birth weight, gestation period and general life cycle)

In late September when new Arctic ice is forming, the seals start their journey southward. During January and February seals disperse widely and feed intensively. Huge amounts of energy in the form of blubber are accumulated during this time. This is particularly important for pregnant females, for they need this energy to support the enormous demands of their rapidly growing offspring during lactation. Pregnant females give birth several days after they have hauled out onto the winter pack ice in late February or early March. Newborn pups are about 85 cm long, weigh about 11 kg and are yellowish in colour. In about 3 days the fur turns to a fluffy white from which the pups derive the name "whitecoats". Young harp seals rank among the fastest growing and most precocious of young mammals. They are nursed for about 12 days and then abandoned by their mothers. During this period pups nurse for periods of about 10 minutes six or seven times a day and they more than triple their weight on milk which contains up to 45% fat (compared to 4% for cow's milk). When weaned, pups weigh an average of 35 kg. More than half of this weight is fat in the form of blubber.After the pups are abandoned by their mothers, they begin to lose weight and to moult their white coats. After about 18 days this coat is completely shed and is replaced with a short silvery one. Harp seal pups fast for four to five weeks following weaning during which they lose about 10 kg of body weight. Most likely the fast is necessary to provide the pup with time to develop the behavioral and physical abilities that are necessary for efficient foraging by these young mammals.As soon as females have finished nursing but before they leave the "whelping patch", they are courted by males which have been waiting nearby in large herds. Mating appears to be promiscuous and may occur either in the water or on the ice. Males reach maturity at 7 or 8 years of age. The females come into breeding condition annually about two weeks after their pups are born, when nursing has ended. The gestation period is approximately 11.5 months. However, there is a period of about 3 months during which the development of the embryo is suspended. This delay in the growth of the embryo serves to ensure that pups are born at the same time each year. Usually only a single pup is born each year, but twins have been recorded. Females generally mature at between 4 and 6 years of age. Males are only slightly larger than females; the average length (from the nose to the tip of the tail) of adult males is 169 cm, and of adult females 162 cm. Weight ranges from 85 to 180 kg depending on time of year. Harp seals live for up to 40 years. Each year, beginning in early April, harp seals moult. Adult males and immatures, called "bedlamers", moult first, followed by adult females, which start to moult about the third week of April. During the moulting which lasts approximately 4 weeks, harp seals rarely feed and as a result lose more than 20 % of their body weight mainly in the form of fat. After they have moulted, adults and immatures migrate to their summer feeding grounds in the Arctic, thus completing their annual cycle.

Sources of information

AMAP (1998). Arctic Pollution Issues: *a State of the Arctic Environment Report.*
http://www.nwtwildlife.rwed.gov.nt.ca/NWTwildlife/seals/harpseal.htm
http://www.ncr.dfo.ca/zone/underwater_sous-marin/hseal/harpseal.htm
http://www.stemnet.nf.ca/CITE/harp_seals.htm

2 Approach for derivation of transfer factor in forest ecosystems

Before 1986 most studies of terrestrial radioecology dealt with agricultural ecosystems. As a result, there is a rather good knowledge of the migration of radionuclides in this type of ecosystems. When extrapolating this knowledge to forest ecosystems the particularities of the latter should be carefully considered. Forest ecosystems differ from agro-ecosystems in several important ways, the main ones being the heterogeneity of the soil profile, the plant biodiversity, the extent of mycorrhization and the animal diversity.

In agro-ecosystems soils are periodically ploughed and fertilised, while in forest ecosystems they exhibit a more or less clear subdivision into an upper, mainly organic horizon and a lower mineral horizon, differing in characteristics such as density, pH, clay content, moisture, nutrient status, biological activity, etc.

The agro-ecosystems are often monocultures, while forests are generally species rich. Forests have a much more complicated structure than agro-ecosystems, and a much wider range of ecological conditions. One of the main components of forest ecosystems, the tree, is a perennial plant. As a consequence, the migration of radionuclides in forests is a poly-cyclic process. Long-term cycles are composed of many annual cycles. Annual cycles involve the removal of radionuclides from the soil compartment during growth and the subsequent return of a portion of them through litter-fall, dead wood, etc.

In forest ecosystems the extent of mycorrhization is much higher than in agricultural ecosystems as most plants are in symbiosis with mycorrhizal fungi. This fact complicates the interpretation of uptake and transfer mechanisms from soil to plants via roots.

In agricultural ecosystems the animals usually live in conditions controlled by man, whereas forest animals have a free-ranging habitat. Wild forest animals often move over large distances within a short time span. This makes it difficult to measure their feed composition and to characterise the habitat conditions and accordingly estimate activity concentrations in individual animals.

The above mentioned features of the forest ecosystems complicate the derivation of transfer factors (TFs)[1], even if empirical data exists, which is not the case for most radionuclides, except for ^{137}Cs and to some extent ^{90}Sr. In this chapter we describe the approaches that we applied for filling some of the existing data gaps.

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[1] The Transfer Factors (TFs) for forest reference organisms are expressed in units of Bq/kg per Bq/m^2, either on a dry weight (for plants and fungi) or a wet weight (for animals) basis. This type of Transfer Factors is known as Aggregated Transfer Factors (Tag). It can be obtained from reported CR, by dividing the latter by a density of 1400 kg/m^3 and a soil thickness of 0.1 m, where most of the radionuclides are usually present.

2.1 Derivation of ^{239}Pu and ^{99}Tc TF to plants using dynamic models.

To derive soil-to-plant TF of ^{239}Pu and ^{99}Tc we used two ecosystem models that were developed with the purpsoe of studing the contamination dynamics in forest adjacent to Oak Ridge Site.

^{239}Pu

The Plutonium model (Garten *et al.*, 1978) is a six compartment models with 10 transfers, that was calibrated using biomass data and empirical data of radionuclide levels in various forest components.

To obtain the TF, included in Appendix 1, we simulated the accumulation of Pu in vegetative components of the forest starting from an initial condition of 1 Bq/m^2 distributed homogeneusly if the upper 10 cm of the soil and using the available site specific parameters. A steady state equilibriunm was obatined in approximately 100 years. After this time, more than 99.9 % of ^{239}Pu was in the soil. The roots and forest litter were the principal biological reservoirs of ^{239}Pu.

Probability distributions for the model parameters were not available, and therefore in order to obtain a range of variation of the TF we performed multiple simulations (1000) varing each parameter by 50 % around the existing site specific values. This is a reasonable assumption for parameters related with the biomass growth. However, the parameters describing the radionuclide uptake by roots and translocation in the vegetation may experience larger variability and therefore we have assumed that the estimated range of TF (Table 2.1) will not encompas more than 50 % of the possible range of variation.

Table 2-1 Range of TF of ^{239}Pu predicted with the model

Compartment	Minimun value of the TF Bq/kg per Bq/m^2	Maximum value of the TF Bq/kg per Bq/m^2
Tree roots	9.6E-5	6.7E-4
Litter	1.45E-4	1.0E-3
Tree wood	9.0E-8	8.1E-7
Tree leaves	8.8E-8	8.1E-7
Understory vegetation	4.0E-6	1.9E-5

^{99}Tc

The technectium model (Garten, 1987) includes five compartments: above ground trees (wood and leaves), the litter layer, an unaavailable mineral soil pool (reduced sorbed soecies) and an available pool (TcO_4^- in soil water).

Simulations where carried out for a continuous input until equilibrium was obtained, after about 30 years. The TF were derived by dividing the activity concentrations in different compartments by the total inventory in the system.

To estimate the range of variation of the derived TF, the most sensitivite model parameters, i.e. the input/export ratio and the reoxidation rate, were varied over 2 orders of magnitude, which is the expected range of variation for these parameters over a wide range of environmental conditions.

The obtained TF and the percentage distribution of ^{99}Tc in different media are shown in Table 2.2.

Table 2-2 Predicted TF and percentange distribution of ^{99}Tc in different media.

Compartment	Predicted range of TF Bq/kg per Bq/m^2	Predicted range of percentage distribution %
Tree leaves and wood	6.5E+0-2.9E2	0.6-27
Litter		0.04-2.0
Tree roots	7.8E-1-3.3E+1	0.26-11
Soil available		0.8-26
Soil unavailable		34-98.3

2.2 Derivation of TF to mammals using a kinetic-allometric approach

For derivation of the TF to mammals we used the FASTer (see chapter 5 in this appendix) kinetic-allometric model and parameters. We made predictions for two herbivores: roe deer and moose and for one carnivore: a fox feeding on roe deer.

The summer/autumn diet composition (see Table 2.3) typical for roe deer and moose living in boreal forests was considered (Cederlund *et al.*, 1980, Avila, 1998).

Table 2-3 Diet composition assumed in the simulations for derivation of TF to herbivores (roe deer and moose).

Diet component	Percentage contribution to the total daily food intake in summer-autumn %	
	Roe deer	Moose
Tree leaves/needles	17	50
Understorey plants	70	49.23
Fungi	13	0.77

The TF from soil to different diet components were set to follow a lognormal distribution, which is expected due to their multiplicative nature. The parameters of the distribution were estimated by assigning percentiles to the minimum and maximum values given in the look-up-tables (Appendix 1.1) as follows:

- If there was high confidence in the estimated TF, then the range was assumed to encompanse the 5 and 95 percentiles,
- If there was moderate (medium) confidence in the estimated TF, then the range was assumed to encompanse the 10 and 90 percentiles.
- If there was low confidence in the estimated TF, then the range was assumed to encompanse the 25 and 75 percentiles.

For estimating the TF to the fox we made the simplifying assumption that the fox is feeding exclusively on soft-tissues of roe deer.

All simulations were conducted using latin hypercube sampling. All other parameters were set constant. The 5 and 95 percentiles of the obtained probability distributions were used as minimum and maximum values in the look-up-tables included in the Appendix 1.1.

3 Collation of empirical data for reference organisms within semi-natural ecosystems

3.1 Data Collation and Review

A database of the transfer of the FASSET radionuclides (as selected within Strand *et al*. (2001)) from soil to reference organisms was generated predominantly from the following sources:

i) Literature review (using Web of Science[2]) of English language refereed publications and citied works within these;
ii) Data supplied by Institute of Radiation Hygiene (IRH) for areas with elevated natural radionuclides within the Komi Autonomous Republic of the Russian Federation (Litver *et al*. 1976; Pokarzhhevskii & Krivolutzkii 1997; RCSI 1974-1998; Troitskaya 1981; Verhovskaya 1972[3]) for the EPIC project (see Beresford *et al*. 2003);
iii) Data supplied by IRH (from published Russian language sources and in-house databases) on the transfer of a range of radionuclides to wildlife species from throughout European Russia (with an emphasis on Arctic regions and post Chernobyl studies in the Bryansk Oblast) for the EPIC project (see Beresford *et al*. 2003);
iv) Data for wildlife species within the Chernobyl Exclusion Zone (Gaschak *et al*., 2003);
v) The Arctic Monitoring and Assessment Programme database (see AMAP 1998).

More than 300 publications (refereed literature, books, institute reports and conference proceedings) were reviewed. Because of the scarcity of appropriate data all appropriate terrestrial wild species were considered with no differentiation between habitats (i.e. the database contains some values for wild species inhabiting forests and agricultural land). A considerable number of data were rejected from the review as the level of detail within the original publications was insufficient to enable its use with any degree of confidence (e.g. all collated Th data for grasses and herbs were rejected). Transfer to soil micro-organisms was not considered as it can be considered that absorbed doses for bacteria will be predominately defined by the activity concentrations in the surrounding medium (Pröhl 2003)[4].

The transfer of ^{3}H and ^{14}C from soil to biota was not considered: an approach to predicting the activity concentrations of these two radionuclides in reference organisms is described within Appendix 2, Section 4.

3.1.1 Data Manipulation

Radiostrontium and radiocaesium data collected during either the period of weapons fallout (assumed to be before 1970) or the year of the Chernobyl accident (1986) were not used to derive transfer parameter values to avoid surface contamination. Unfortunately, this removed a considerable amount of data from further consideration.

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[2] http://wos.mimas.ac.uk/
[3] Original references mostly in Russian – see also Maslov *et al*. (1966) for site description in English.
[4] Readers interested in the uptake of radionulcides by micro-organisms should refer to Keith-Roach & F.R. Livens (2002).

For the purpose of estimating internal doses of wild animals, we need transfer parameters to enable the prediction of whole body activity concentrations. However, much of the available data is reported for specific organs. Where possible whole body activity concentrations were generated from these organ specific data using published radionuclide tissue distributions (e.g. Coughtrey *et al.*, 1985; Morgan 1991; ICRP 1979) and proportions of total live-weight contributed by the given organ. For instance, it was assumed that 97 % of the body radiostrontium was present in bone (analyses of available data in the database) and that bone contributed 10 % of the live-weight of mammalian species (Beresford *et al.* 1997; Gaare & Staaland 1994) to generate whole-body activity concentrations on the basis of reported data for bone. As radiocaesium is known to be approximately homogenously distributed throughout body tissues (Coughtrey & Thorne 1983), reported transfer values to muscle were considered to be representative of whole-body transfer.

Collated data for plants were reported for a range of plant parts (e.g. data for many shrub species were dominated by berries); no differentiation between plant parts has been taken into account in the derivation of mean transfer values. All transfer data for plant species have been converted to a dry matter basis (DM) (from ash or fresh weights within original references not reporting as dry matter). Similarly all animal data has been transformed to fresh weight (FW).

Some sources present individual data values whilst others present mean transfer estimates; in the derivation of (arithmetic) mean transfer values, previously reported mean estimates have been treated as single data values.

Soil-biota transfer was expressed in (or calculated from) the original references as either:

- *concentration ratio* (CR; the ratio of the radionuclide activity concentration in an organism (Bq kg^{-1} FW or DM) to the radionuclide activity concentration in soil (Bq kg^{-1} DM)[5]; or
- *aggregated transfer coefficient* (Tag; defined as the ratio of the radionuclide activity concentration in an organism (Bq kg^{-1} FW or DM) to the radionuclide deposition in soil (Bq m^{-2}) and having units m^2 kg^{-1}).

For the purposes of this review values have been standardised as concentration ratios assuming a soil bulk density of 1.4 g DM cm^{-3} and a sampling depth of 10 cm (as quoted in IAEA (1994) for grasslands). The assumption of soil depth within such a conversion will be in error in some instances. However, the degree of error is likely to be less than one order of magnitude which is within the range of reported transfer values for most organism-radionuclide combinations. Indeed, it should be noted that the reported transfer values we have used here do not all represent the same soil sampling depth, when this information is given in the orginal reference (which is not always the case).

3.2 Soil–Reference Organism Transfer Parameters

A summary of the available transfer values for reference organisms and their representative species are presented in Tables 3-1–3-5.

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[5] Some CR values were available for stable elements.

The majority of available soil-biota transfer data are for ^{137}Cs and ^{90}Sr (predominantly from global fallout and the Chernobyl accident), and natural radionuclides from the uranium decay series (predominantly from sites in the Russian Komi Republic). Transfer values were available for the majority of reference organisms for these radionuclides. There are few available data for the actinide elements and no data expressing the transfer of some of the FASSET radionuclides from soil to reference organisms (see Tables 3-1-3-5). There is a predominance within the available data of CR values originating in northern Europe; comparatively few data having been published for Mediterranean semi-natural ecosystems.

Table 3-1 Summary of reference organism concentration ratios (DM) for lichen and bryophytes.

Radionuclide	Mean	Min	Max	n	References
Strontium	$1.16\times10^{+1}$	4.02×10^{-1}	$7.64\times10^{+1}$	356	1-6
Caesium	$1.35\times10^{+1}$	2.31×10^{-1}	$4.09\times10^{+1}$	388	1-6
Polonium	2.76×10^{-1}	1.33×10^{-2}	8.80×10^{-1}	5	2,7
Lead	$1.76\times10^{+1}$	6.06	$1.24\times10^{+2}$	45	2,4,5,10,11
Radium	8.33×10^{-1}	4.80×10^{-1}	1.39	6	8,9
Thorium	2.67×10^{-1}	1.61×10^{-1}	6.17×10^{-1}	6	8,9
Uranium	1.97×10^{-1}	-	-	1	8

Note: all data are for *Cladonia* spp.

References: 1. Miretsky *et al.* (1993); 2. RCSI (1974-1998); 3. Bakunov *et al.* (1998); 4. Balonov (1999); 5. Balonov (2000); 6. Matishov *et al.* (1994); 7. Mahon & Mathews (1983); 8. Verhovskaya (1972); 9. Litver *et al.* (1976); 10. Holtzman (1966); 11. Troitskaya (1981).

Table 3-2 Summary of reference organism concentration ratios (DM) for grasses and herbs.

Radionuclide		Grasses & herbs
Strontium	n	327
	Mean	6.92×10^{-1}
	Min	1.76×10^{-1}
	Max	3.64
	References	8, 9,12,15,16
Caesium	n	542
	Mean	2.30
	Min	1.20×10^{-2}
	Max	$2.37\times10^{+2}$
	References	1-15

References: 1. Howard *et al.* (2002); 2. Albers *et al.* (2000); 3. Anderson *et al.* (1992); 4. Bunzl & Kracke (1989); 5. Bunzl & Krake (1984); 6. Bunzl *et al.* (2000); 7. Copplestone *et al.* (1999); 8. Balonov (1999); 9. Balonov (2000); 10. Johanson *et al.* (1994); 11. Livens *et al.* (1991); 12. Miretsky *et al.* (1993); 13. Pálsson *et al.* (1994); 14. Pietrzak-Flis *et al.* (1996); 15. RCSI (1974-1998); 16. Gastberger (2000).

A considerable amount of available transfer data for herbivorous mammals originates from measurements of reindeer (see Table 3-5). The transfer of some radionuclides to reindeer is known to be comparatively high as a consequence of the importance of lichens in their diet (e.g. Howard *et al.* 1991). Therefore, to investigate any bias induced by the reindeer data, transfer estimates are summarised for *all herbivorous mammals* and *all herbivorous mammals excluding reindeer* in Table 3-5. For radiocaesium and radiostrontium, the exclusion of reindeer data results in a lower mean CR values by factors of *circa* 7 and 3 respectively. However, the overall range in collated values is similar. All ^{210}Pb and ^{210}Po data for herbivorous mammals were for reindeer.

No data were available describing the transfer of ^{210}Po and ^{210}Pb from soil-carnivorous mammals. However, Thomas *et al*. (1994) report values for the transfer of these radionuclides from reindeer to wolf muscle. Assuming distributions of ^{210}Po and ^{210}Pb in carnivorous mammals consistent with that in herbivores these were used to adapt the soil-herbivorous mammal transfer value in Table 3-5 to a value of transfer from soil-carnivorous mammal (Table 3-5).

No data were available describing the transfer of radionuclides from soil to herbivorous bird eggs. For U and Cs CR values of 2.0×10^{-3} and 6.4×10^{-2} respectively can be derived by comparison of the transfer of these radionuclides from the diet to meat and eggs of hens (IAEA, 1994) with soil-muscle CR values within the database collated during this work (collated data are predominantly for *Lagopus* spp.).

Some stable element data for Pb and Ni (predominantly from heavy metal pollution studies) were used to provide estimates of transfer from soil to burrowing mammals (Pb) and worms (Tables 3-4 to 3-5).

Table 3-3 Summary of reference organism concentration ratios (DM) for shrubs.

Radionuclide			Radionuclide		
Strontium	n	78	Caesium	n	637
	Mean	1.08		Mean	6.74
	Min	5.71×10^{-2}		Min	9.59×10^{-2}
	Max	$1.08\times10^{+1}$		Max	$3.53\times10^{+1}$
	References	1-3		References	1-11
Thorium	n	10	Radium	n	10
	Mean	8.81×10^{-2}		Mean	2.73
	Min	2.36×10^{-2}		Min	6.25×10^{-1}
	Max	1.65×10^{-1}		Max	7.64
	References	12		References	12
Lead	n	28	Uranium	n	10
	Mean	1.74		Mean	1.43×10^{-1}
	Min	2.00×10^{-2}		Min	3.38×10^{-2}
	Max	5.50		Max	7.50×10^{-1}
	References	3,6		References	12
Polonium	n	4			
	Mean	1.23			
	Min	1.92×10^{-1}			
	Max	3.17			
	References	3			

References: 1. Balonov (1999 & 2000); 2. Miretsky *et al*. (1993); 3. RCSI (1974-1998); 4. Anderson *et al*. (1992); 5. Howard *et al*. (2002); 6. Bunzl & Krake (1984); 7. Bunzl & Krake (1986); 8. Johanson *et a*l (1994); 9. Livens *et al*. (1992); 10. Matishov *et al.* (1994); 11. Palsson *et al*. (1994); 12. Verhovskaya (1972).

Where sufficient data enabled analysis, the distribution of available transfer values was highly skewed. Figure 3-1 illustrates this presenting radiocaesium CR values for herbivorous mammals (the reference organism-radionuclide combination for which most data were available). This suggests that it may have been more appropriate to express mean transfer values as geometric and not arithmetic means as presented in Tables 3-1 to 3-5. However, the use of arithmetic mean values as presented is likely to result in conservative CR values.

Table 3-4 Summary of reference organism concentration ratios (FW) for worms and detritivores.

Radionuclide		Worms	Detritivore
Nickel	n	32	-
	Mean	7.17×10^{-2}	
	Min	5.66×10^{-3}	
	Max	3.16×10^{-1}	
	References	7,11,12,14	
Caesium	n	12	6
	Mean	5.66×10^{-2}	8.49×10^{-2}
	Min	1.70×10^{-3}	2.46×10^{-2}
	Max	1.77×10^{-1}	1.39×10^{-1}
	References	1,3,4	1,15
Lead	n	89	-
	Mean	1.29×10^{-1}	
	Min	1.89×10^{-3}	
	Max	1.55	
	References	5-14	
Radium	n	4	12
	Mean	8.14×10^{-2}	1.90×10^{-1}
	Min	3.50×10^{-2}	5.16×10^{-3}
	Max	1.27×10^{-1}	4.66×10^{-1}
	References	2	2
Plutonium	n	-	4
	Mean		2.16×10^{-1}
	Min		3.47×10^{-2}
	Max		3.26×10^{-1}
	References		1
Americium	n	2	4
	Mean	1.30×10^{-1}	1.32×10^{-1}
	Min	8.31×10^{-2}	1.12×10^{-1}
	Max	1.70×10^{-1}	1.90×10^{-1}
	References	1	1

References: 1. Copplestone *et al.* (1999); 2. Pokarzhevskii & Krivoluzkii (1997); 3. Janssen *et al.* (1996a); 4. Janssen *et al.* (1996b); 5. Wei-Chun (1987); 6. Diercxsens *et al.* (1985); 7. Hendriks *et al.* (1995); 8. Ireland (1979); 9. Morgan & Morgan (1990); 10. Morris & Morgan (1986); 11. Nelson *et al.* (1982); 12. Pietz *et al.* (1984); 13. Spurgeon (1996); 14. Wei-chun (1982).15. Toal *et al.* (2002a).

Table 3-5 Summary of soil – reference organism concentration ratios (FW) for mammals.

Radionuclide		Herbivorous mammals *all species*	Herbivorous Mammals *excluding reindeer*	Carnivorous Mammals	Burrowing Mammals
Strontium	n	445	80	8	-
	Mean	5.18	1.96	1.30	
	Min	4.49×10^{-3}	9.20×10^{-3}	2.21×10^{-1}	
	Max	$1.43\times10^{+1}$	$1.11\times10^{+1}$	3.35	
	References	2-6,8,9	2-6,8,9	9	
Caesium	n	1257	412	12	-
	Mean	$1.26\times10^{+1}$	1.84	4.96	
	Min	1.92×10^{-2}	1.92×10^{-2}	1.76×10^{-1}	
	Max	$1.37\times10^{+2}$	$1.37\times10^{+2}$	$2.31\times10^{+1}$	
	References	1-9,16-23	3,4,6,8,9,16-23	9	
Polonium	n	42	-	3	-
	Mean	4.17		1.68	
	Min	3.97×10^{-1}		1.51	
	Max	$1.43\times10^{+1}$		1.85	
	References	8,10,11		See text	
Lead	n	53	-	3	17
	Mean	4.11		4.88×10^{-1}	7.56×10^{-2}
	Min	3.74×10^{-1}		2.71×10^{-1}	4.36×10^{-3}
	Max	$1.67\times10^{+1}$		6.58×10^{-1}	3.95×10^{-1}
	References	3-5,8,10,11		See text	24,25
Radium	n	49	33	17	34
	Mean	4.77×10^{-2}	4.13×10^{-2}	3.53×10^{-2}	6.01×10^{-2}
	Min	2.14×10^{-3}	2.14×10^{-3}	4.28×10^{-3}	4.28×10^{-3}
	Max	1.95×10^{-1}	1.95×10^{-1}	9.56×10^{-2}	1.95×10^{-1}
	References	8,12-14	12,13	12,13	12,13
Thorium	n	8	2	2	4
	Mean	6.39×10^{-1}	7.74×10^{-3}	5.52×10^{-3}	1.18×10^{-2}
	Min	2.14×10^{-3}	2.14×10^{-3}	1.04×10^{-3}	2.14×10^{-3}
	Max	4.66×10^{-1}	1.33×10^{-2}	1.00×10^{-2}	2.87×10^{-2}
	References	13,14	13	13	13
Uranium	n	-	3	1	4
	Mean		1.80×10^{-3}	7.09×10^{-4}	2.91×10^{-3}
	Min		1.22×10^{-4}	-	2.43×10^{-3}
	Max		2.84×10^{-3}	-	3.34×10^{-3}
	References		13	13	13
Plutonium	n	-	1	-	-
	Mean		1.82×10^{-3}		
	References		15		
Americium	n	-	1	-	-
	Mean		4.06×10^{-3}		
	References		15		

References: 1. AMAP (1998); 2. Bakunov *et al.* (1998); 3. Balonov (1999); 4. Balonov (2000); 5. Lubashevsky *et al.* (1993); 6. Miretsky *et al.* (1993); 7. Ramzaev (1967); 8. RCSI (1974-1998); 9. Gaschak *et al.* (2003); 10. Kauranen & Miettinen (1969); 11. Troitskaya (1981); 12. Pokarzhevskii & Krivolutzkii (1997); 13. Verhovskaya (1972); 14. Litver *et al.* (1976); 15. Copplestone *et al.* (1999); 16. Johanson & Bergstrom (1989); 17. Johanson & Bergstrom (1994); 18. Johanson *et al.* (1994); 19. Nelin (1995); 20. Rantavaara (1990); 21. Rantavaara (*pers. comm.*) in Johnanson (1994); 22. Cristaldi *et al.* (1991); 23. Avila *et al.* (1999); 24. Wei-chun (1987); 25. Read & Martin (1993).

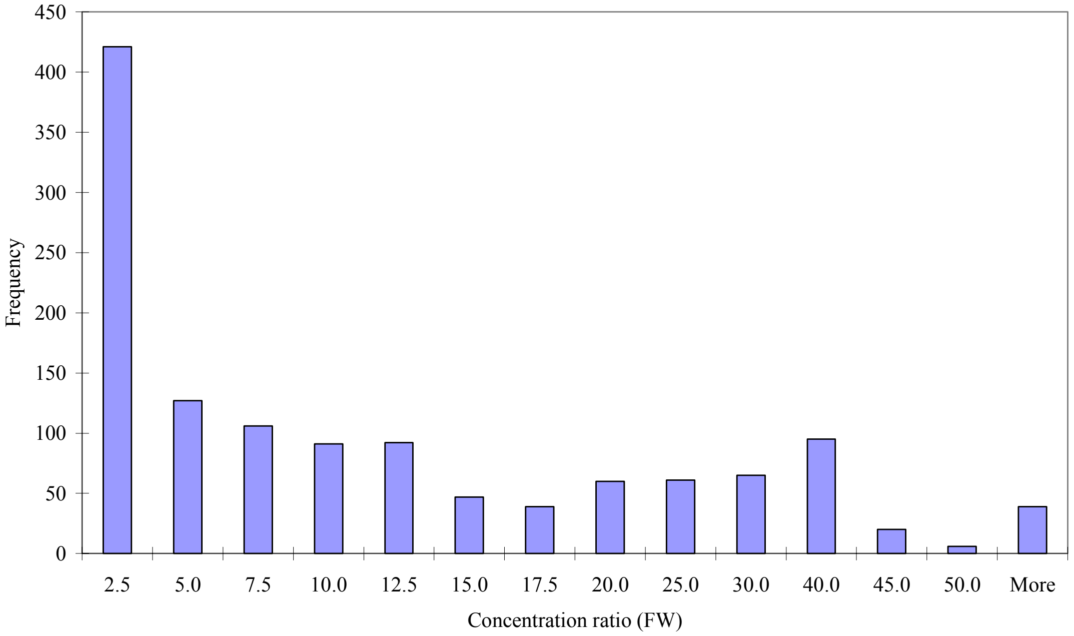

Figure 3-1 An example of the distribution of available transfer parameter data. Data shown are CR values (n=1257) describing the transfer of radiocaesium to all herbivorous mammals.

4 Approach to predict ^{3}H and ^{14}C transfer in semi-natural environments.

4.1 Introduction

Tritium and ^{14}C are radionuclides of macro-elements which are structural components of plant and animal tissues and, in the case of ^{3}H, water. Therefore, conventional modelling techniques for modelling radionuclide transfer are not appropriate. Also, as these radionuclides are primarily present in the environment as reversible gases ($^{14}CO_2$ and ^{3}HHO), we have adopted the common practice of expressing activity concentrations in biota relative to an annual average activity in air of 1 Bq m^{-3}.

4.2 Tritium

The simplest method of estimating ^{3}H activity concentrations in biota under equilibrium conditions is to use the specific activity approach (USEPA 1989). This assumes that ^{3}H is in equilibrium in any environmental compartment and that plant and animal activity concentrations of tritiated water (HTO) are derived from air moisture. Whilst full equilibrium is unlikely to be reached under field conditions (Spencer & Vereecken-Sheehan 1994), the specific activity approach is accepted within radiation protection for conservative assessments. The specific activity approach used here is based upon existing models (Galeriu 1994; Peterson & Davis 2002; Galeriu & Belot 2002). The prediction of both HTO and organically bound ^{3}H (OBT) activity concentrations in biota are allowed for in these models. Model inputs include precipitation, relative humidity and temperature. For adaptation to the Arctic, ^{3}H deposition with snow fall (Davis 1997), the time of snow melt, and the contribution of snow melt to soil ^{3}H (Galeriu 1993) and water have been included (see Beresford *et al.* 2003 for further details). The HTO and OBT activity concentrations in animal tissues are estimated using the approach published by Galeriu *et al.* (2001; 2002).

The tritium activity concentration in plant water is estimated by (Galeriu & Belot 2002; Belot *et al.* 1996):

$$C_{plantHTO} = 1.1\left(\frac{\rho_a}{\rho_v}\right)\cdot C_a + 1.17\left(1-\frac{\rho_a}{\rho_v}\right)\cdot C_s \qquad (4\text{-}1)$$

where: $C_{plantHTO}$ is the HTO concentration in leaf water [Bq L^{-1}];
C_a is the HTO concentration in air moisture [Bq L^{-1}]:
ρ_a is the water vapour mass per unit air volume (average value for summer is used; kg m^{-3}];
ρ_v is the saturated water vapour mass per unit volume at leaf temperature, average value for summer [kg m^{-3}]
C_s is the HTO concentration in the routing depth of soil [Bq L^{-1}]

The HTO concentration in air moisture is estimated as:

$$C_a = C_{av}/\rho_a \qquad (4\text{-}2)$$

where: C_{av} is the HTO concentration in air volume [Bq m^{-3}].

In practice, the average leaf temperature can be considered equal to the average air temperature and the ratio in Equation (4-2) is equal to the relative humidity (during the growing season).

If FD is the plant dry matter fraction, then the HTO concentration in edible plant parts is simply:

$$C_{fresh,HTO} = (1 - FD) \cdot C_{plantHTO} \tag{4-3}$$

The fresh weight OBT concentration in plants part is given by:

$$C_{OBT} = 0.6 \cdot FD \cdot C_{plant\,HTO} \tag{4-4}$$

The ^{3}H concentration in soil water C_s (rooting depth average) is estimated as the sum of wet and dry deposition:

$$C_s = D_w / I_r + 0.15 C_a \tag{4-5}$$

The wet deposition contribution (D_w/I_r) is derived from the average HTO concentration in rainwater during the vegetation growing period where D_w is the total wet deposition (Bq m^{-2}) during the growing period and I_r the average precipitation during the growing period (mm). D_w is given by:

$$D_w = C_{av} \cdot \lambda \cdot MH \cdot \Delta t \tag{4-6}$$

where: λ is the washout rate [h^{-1}];
MH is the mixing height in neutral weather condition [m];
Δt is the total duration of rainfall [h] during the growing season;
the dry deposition component in Equation (4-5) is defined by $0.15C_a$,
where 0.15 is a best estimate constant (IAEA 2001b)

The resultant ^{3}H activity concentrations in plant material are assumed to represent the diet of herbivorous animals. Subsequently, the activity concentrations estimated for herbivores is used to estimate the diet of carnivores. The transfer of ^{3}H to animals has been estimated using the approach derived recently by Galeriu *et al.* (2001; 2002). The ^{3}H activity concentration is estimated as the sum of four transfer processes:

$$F_{HH} = I_{HTO} \cdot \frac{v_{bw}}{WF} \tag{4-7}$$

$$F_{HO} = I_{HTO} \cdot \frac{0.25 m_{ot}}{0.111 WF} \tag{4-8}$$

$$F_{OH} = I_{OBT} \cdot F_{Dom} F_{HH} \tag{4-9}$$

$$F_{OO} = I_{OBT} \cdot \frac{0.75 m_{ot}}{I_{OBH}} \tag{4-10}$$

where: F_{HH} is the transfer from dietary HTO to body HTO [d kg^{-1}];
F_{HO} is the transfer from dietary HTO to body OBT [d kg^{-1}];
F_{OO} is the transfer from dietary OBT from diet to body OBT [d kg^{-1}];
F_{OH} is the transfer from dietary to body HTO [d kg^{-1}];
I_{HTO} and I_{OBT} are the daily intakes of HTO and OBT respectively [Bq d^{-1}];
v_{bw} is the body water fraction;
m_{ot} is the bound hydrogen content [kg kg^{-1} FW];
F_{Dom} is the digestibility coefficient of organic matter in food;
WF is the total daily water flux [L d^{-1}]; and
I_{OBH} is the daily intake of bound hydrogen [kg d^{-1}]

The animal metabolism parameters required for Equations 4-7 to 4-10 can be derived from the literature. The body compositions of wild animals and diet digestibility's were taken from Robbins (1993) and Crocker *et al.* (2002). To estimate dry matter intake and water flux relationships based upon organism live-weight of Nagy (2001) and Robbins (1993) respectively were used; both authors present relationships for different groups of animals (e.g. carnivorous mammals) and relationships appropriate to the reference organism being modelled were used here.

4.2.1 Tritium Results

Predictions of ^{3}H concentrations in reference organism have been made for four different climatic regions of Europe, these being Arctic, Continental, Maritime and Mediterranean. The required climatic input parameters were derived from the University of East Anglia Climate Research Units' gridded (0.5° × 0.5° resolution) regional climatology for Europe data set[6]. Tables 4-1 to 4-3 present predicted HTO, OBT and total tritium activity concentrations in animals representative of FASSET reference organisms assuming a constant 1 Bq m^{-3}. Across climates the highest concentrations are predicted for the Arctic and the lowest for the Mediterranean. Considering various animals, there is a factor of two difference in total tritium concentrations but more than a factor of four for OBT, largely reflecting the influence of body composition. Predicted variation is, however, low compared to the observed variation for the other radionuclides considered here (see Tables 3-1 to 3-5). On the basis of estimates presented in Table 4-3, an assumed activity concentration in all reference organisms of 150 Bq ^{3}H kg^{-1} FW per Bq ^{3}H m^{-3} in air would be appropriate and generally conservative.

Table 4-1 Predicted HTO activity concentrations (Bq kg^{-1} FW) in species representative of FASSET reference organism groups.

Climate	Grass	Worm	Mole	Rabbit	Moose	Weasel	Red fox	*Lagopus* spp. egg
Mediterranean	54	58	42	51	52	56	52	49
Continental	73	74	56	68	69	75	70	66
Maritime	86	98	70	82	83	88	83	80
Arctic	94	88	77	85	87	96	93	83

Table 4-2 Predicted OBT activity concentrations (Bq kg^{-1} FW) in species representative of FASSET reference organism groups.

Climate	Grass	Worm	Mole	Rabbit	Moose	Weasel	Red fox	*Lagopus* spp. egg
Mediterranean	20	8.8	22	29	25	30	27	19
Continental	27	12	29	39	34	40	36	26
Maritime	31	14	35	46	40	47	43	30
Arctic	34	19	44	50	43	51	48	33

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[6] See also http://www.cru.uea.ac.uk/~mikeh/datasets/regional/europe.htm

Table 4-3. Predicted total ^{3}H activity concentrations (Bq kg^{-1} FW) in species representative of FASSET reference organism groups.

Climate	Grass	Worm	Mole	Rabbit	Moose	Weasel	Fox	*Lagopus* spp. egg
Mediterranean	74	67	64	80	76	85	79	68
Continental	10	86	85	110	100	120	110	92
Maritime	117	110	100	130	120	140	130	110
Arctic	128	110	120	140	130	150	140	120

4.3 Carbon-14

A simpler specific activity approach is used to model ^{14}C (Killough & Rohwer, 1978; Wirth, 1982). As we are assuming a constant concentration of ^{14}C in air of 1 Bq m-3, the specific activity in air, SA_{air} (Bq g^{-1} C), is:

$$SA_{air} = 1/0.18 \qquad (4\text{-}11)$$

where: 0.18 g m^{-3} is the current carbon content of air.

The specific activity in herbage, SA_{herb}, will equal that in air:

$$\frac{1}{0.18} = \frac{^{14}C_{herb}}{C_{herb}} \qquad (4\text{-}12)$$

where: $^{14}C_{herb}$ (Bq kg^{-1} DM) and C_{herb} (g kg^{-1} DM) are the ^{14}C activity and stable carbon concentrations in herbage, respectively.

Therefore, the ^{14}C activity concentration in herbage is:

$$^{14}C_{herb} = 5.56(C_{herb}) \qquad (4\text{-}13)$$

Similarly, the ^{14}C activity concentration in animals, $^{14}C_{anim}$ (Bq kg^{-1} FW), is:

$$^{14}C_{anim} = 5.56(C_{anim}) \qquad (4\text{-}14)$$

where: C_{anim} is the stable carbon concentration in animals (g kg^{-1} FW).

The stable carbon content of herbage and different animal species were taken from Robbins (1993) and Crocker *et al.* (2002).

4.3.1 C-14 Results

Predicted ^{14}C activity concentrations (assuming a constant 1 Bq ^{14}C m^{-3} air) are: grass – 890 Bq kg^{-1} (FW); mammal – 1340 Bq kg^{-1} (FW); worm 430 Bq kg^{-1} (FW); bird egg 890 Bq kg^{-1} (FW).

5 FASTer – Dynamic model for predicting radionuclide transfer in semi-natural ecosystems

5.1 Introduction

The model developed is primarily derived from those described by IAEA (1982) and Karlsson *et al*. (2001) (a model developed for application in dose assessments at the Forsmark site (Sweden) and which includes a pathway to enable the modelling of releases to ground water). The conceptual model for the transfer of radionuclides from aerial deposition through a simplified grass – herbivorous mammal – carnivorous mammal food chain is presented as an interaction matrix in Figure 5-1.

1.1 **Source**	1.2 *Ingestion*	1.3 *Soil ingestion*	1.4 Deposition Interception	1.5 *Deposition*	1.6	1.7
2.1	2.2 **Carnivore**	2.3	2.4	2.5	2.6	2.7 Excretion
3.1	3.2	3.3 **Herbivore**	3.4	3.5	3.6	3.7 Excretion
4.1	4.2	4.3	4.4 **Grass**	4.5 *Weathering*	4.6	4.7
5.1	5.2 *Resuspension* *Inhalation*	5.3 *Resuspension* *Inhalation*	5.4 *Root uptake*	5.5 **Top Soil**	5.6 Diffusion Advection	5.7 Run-off Erosion
6.1	6.2	6.3	6.4	6.5 *Diffusion* *Advection*	6.6 **Deep Soil**	6.7 *Diffusion* *Advection.*
7.1	7.2	7.3	7.4	7.5	7.6	7.7 **Loss**

Figure 5-1 The FASTer conceptual model represented as an interaction matrix. Processes not currently included in the model are shown in italics.

5.2 Model Description

The rate of change of the radionuclide inventory in the compartments (diagonal elements of the matrix) is described with a linear differential equation of the form:

$$dA_i / dt = \sum_j k_{ji} * A_j - \sum_j k_{ij} * A_i \tag{5-1}$$

where

k_{ji} is the transfer rate from compartment *"j"* to compartment *"i"*
kij is the transfer rate from compartment *"i"* to compartment *"ji"*

The transfer rate is a mathematical representation of the corresponding processes in the matrix (Figure 5-1).

5.2.1 Interception by grass of radionuclides deposited from air (from element 1.4)

The fraction (*r*) of deposited radionuclides that is intercepted by grass is described as a function of the biomass using the equation proposed by Chamberlain & Garland (1991)):

$$r = \exp(-\mu * B_{grass}) \tag{5-2}$$

where

μ is the interception coefficient [m^2 kg^{-1}] – a value of 2.5 which can be considered a conservative value for grass (IAEA 1982) has been assumed for all radionuclides
B_{grass} is the yearly average biomass of grass [kg m^{-2}]
The fraction deposited on topsoil is calculated as *1-r*.

5.2.2 Transfer of radionuclides from grass to soil (from element 4.5)

The transfer from grass to soil is described by the weathering rate constant, k_{weath}. A weathering half-time of 14 days has been assumed for all radionuclides (IAEA 1994).

In case of a chronic deposition the radionuclide content in grass, accumulated via direct deposition from the air, can be calculated as:

$$A_{grass}(air) = Q_c * \frac{r}{k_{weath}} * (1 - \exp(-k_{weath} * t)) \tag{5-3}$$

where

$A_{grass}(air)$ is the radionuclide inventory in grass coming from the air [Bq m^{-2}]
r is the fraction of deposited radionuclides that is intercepted by grass [relative units]
k_{weath} is the weathering constant [d^{-1}]
Q_c is the deposition rate [Bq m^{-2} d^{-1}]

In case of an acute deposition the radionuclide content in grass, accumulated via direct deposition from the air, can be calculated as:

$$A_{grass}(air) = Q_a * r * \exp(-k_{weath} * t) \tag{5-4}$$

where

Q_a is the initial deposition [Bq m^{-2}]

5.2.3 Transfer from soil to grass (from element 5.4)

We assume equilibrium between the upper 10 cm of the soil (root zone; top soil on Figure 5-1) and the grass. The radionuclide content in grass, accumulated by root uptake, is calculated as:

$$A_{grass}(soil) = CR * A_{topsoil} * \frac{B_{grass}}{d_{topsoil} * h_{topsoil}} \tag{5-5}$$

where

$A_{grass}(soil)$ is the radionuclide inventory in grass coming from the soil [Bq m^{-2}]
CR is the concentration ratio from soil to grass [relative units]
$A_{topsoil}$ is the radionuclide inventory in the topsoil [Bq m^{-2}]
$d_{topsoil}$ is the topsoil density [kg m^{-3}]
$h_{topsoil}$ is the assumed thickness of the topsoil [m]

The total inventory of radionuclides in grass (A_{grass}) is calculated as the sum of $A_{grass}(soil)$ and the radionuclide inventory coming from the air (i.e. $A_{grass}(air)$)

5.2.4 Uptake of radionuclides by mammals (from elements 2.7 and 3.7)

The uptake of radionuclides (into body tissues) by herbivorous mammals via ingestion of grass (UH_{grass}) per unit of body weight is calculated as:

$$UH_{grass} = \frac{DMI_H}{W_H} * \frac{A_{grass}}{B_{grass}} * f1_H \tag{5-6}$$

where

DMI_H is the daily dry matter intake by herbivorous mammals [kg d^{-1}]
$f1_H$ is the fractional gut uptake for herbivorous mammal [relative units]
W_H is the live-weight of the herbivorous mammal [kg]

The quantity of radionuclides taken up by the animals is not substracted from the grass compartment.The uptake of radionuclides by herbivorous mammals via ingestion of soil (UH_{soil}) per unit of body weight is calculated as:

$$UH_{soil} = fs * \frac{DMI_H}{W_H} * \frac{A_{topsoil}}{d_{topsoil} * h_{topsoil}} * f1s \tag{5-7}$$

where

$f1s$ is the fractional gut uptake of radionuclides ingested with soil [relative units]
fs is the fraction of the daily matter intake by herbivorous mammals that is soil [relative units]

The quantity of radionuclides taken up by the animals is not subtracted from the soil compartment.

The daily dry matter intake by herbivorous mammals is calculated using the allometric relationship:

$$DMI_H = a1h * W_H^{\ b1h} \tag{5-8}$$

where

$a1h$ is the multiplication constant in the allometric (weight dependent) relationship for dry matter intake for herbivorous mammals [kg d^{-1}]
$b1h$ is the exponent in the allometric relationship for dry matter intake for herbivorous mammals [relative units]

The uptake of radionuclides (into body tissues) by carnivorous mammals via ingestion of herbivorous mammals (*UC*) per unit of body weight is calculated as:

$$UC = \frac{FMI_C}{W_C} * C_H * f_{soft} * f1_C \tag{5-9}$$

where

FMI_C is the daily fresh matter intake by carnivorous mammals [kg d^{-1}]
W_C is the live-weight of the carnivorous mammal [kg]
C_H is the radionuclide concentration in the herbivorous mammal [Bq/kg]
$f1_C$ is the fractional gut uptake for carnivorous mammal [relative units]
f_{soft} is the relative activity concentration of soft tissues to whole body of herbivores [relative units]

As implied above by the inclusion of f_{soft} it is assumed that radionuclide incorporated into the bone of herbivorous animals is not available for transfer to carnivorous consumers. The quantity of radionuclides taken up by the animals is not subtracted from the compartment for herbivorous mammals.

The fresh matter intake by carnivorous mammal is calculated using the allometric relationship:

$$FMI_C = fwc * a1c * W_C^{\ b1c} \tag{5-10}$$

where

$a1c$ is the multiplication constant in the allometric relationship for dry matter intake for carnivorous mammals [kg d^{-1}]
$b1c$ is the exponent in the allometric relationship for dry matter intake for carnivorous mammals [relative units]
fwc is the conversion factor from dry to fresh weight for herbivorous tissue [relative units]

5.2.5 Release of radionuclides from the mammal's body (from elements 1.2 and 1.3)

For Cs, Cl, Sr, I, Tc, Pu, Am and Cm the release from the mammal's body (*R*) is assumed to be proportional to the radionuclide content in the whole body and is calculated with the following expressions:

$$R_H = \frac{\ln(2)}{Tb_H} * A_H, \quad R_C = \frac{\ln(2)}{Tb_C} * A_C \tag{5-11}$$

where

Tb_H and Tb_C are the biological half-life for herbivorous and carnivorous mammals respectively [d]
A_H and A_C are the radionuclide inventories in the body of the herbivorous and carnivorous mammals respectively [Bq]

The biological halftimes Tb_H and Tb_C are calculated using the allometric relationship:

$$Tb = a2 * W^{b2} \tag{5-12}$$

where

Tb is the biological half-life of the mammals [d]
$a2$ is the multiplication constant in the allometric relationship for biological half-life [d]
$b2$ is the exponent in the allometric relationship for biological half-life [r.u]
W is live-weight [kg]

For Ni, Nb, Ru and Np the biological half-life is calculated using a retention function (*RF*) as no published dependence on live-weight is available:

$$RF = \sum m_i * e^{-n_i * t} \tag{5-13}$$

where

m_i [relative units] and n_i [d^{-1}] are reported coefficients for the metabolic model.

5.2.6 Correction to account for animal life time

For radionuclides with a long biological half-life (in comparison with the life time of the animal) the activity concentration in herbivorous and carnivorous mammals will never reach equilibrium with dietary intake and can be calculated by the following expression:

$$A_{corr}(t) = A(t) - \frac{A(t - T_{life})}{(1 + \lambda_{eff})^{T_{life}}} \tag{5-14}$$

where

$C_{corr}(t)$ is the corrected activity concentration in the mammal at death [Bq kg^{-1}]
$C(t)$ is the activity concentration in the mammal calculated without consideration for the life time [Bq kg^{-1}]
λ_{eff} is the effective release rate of the radionuclide from the animal body by radioactive disintegration and via the metabolism [d^{-1}]
T_{life} is the expected life time of the mammal [d]

If unknown, the expected life time of the animal can be calculated from the allometric relationship (Calder 1984):

$$T_{life} = 368.6 * W^{0.35} \tag{5-15}$$

Note that corrected for life time activity concentrations in herbivorous mammals should be used in Equation (5-9) as the input into carnivorous mammals.

5.2.7 Vertical migration in the soil column (from elements 5.6 and 6.5)

The vertical migration in the soil column is described in simplified way as a net leaching from the root zone, with a rate constant, k_{leach}.

5.2.8 Flows from the system (from elements 5.7 and 6.7)

Flows of radionuclides from the system are not considered.

5.3 Model Parameters

Parameter values as used within the current work and their sources are presented in Tables 5-1 to 5-7.

Table 5-1. Radionuclide independent parameters.

Parameter	Units	Best estimate	Reference
B_{grass}	kg m^{-2}	0.15	Beresford 2002^{+}
$h_{topsoil}$	m	0.1	-
$d_{topsoil}$	kg m^{-3}	1400	IAEA 1994
a_{1c}	kg d^{-1}	0.0486	Nagy 2001
b_{1c}	r.u.*	0.834	Nagy 2001
a_{1h}	kg d^{-1}	0.0658	Nagy 2001
b_{1h}	r.u.	0.628	Nagy 2001
f_s	r.u.	0.05	-
f_{wc}	r.u.	5	-

*Relative units
$^{+}$Values for semi-natural pastures in the UK

Table 5-2. Soil-to-plant CR.

Nuclide	Best estimate	Min.	Max	Reference
Cs	0.2	0.02	2	Karlsson *et al*. 2001
Sr	1	0.4	3	Karlsson *et al*. 2001
Cl	30	10	100	Karlsson *et al*. 2001
I	0.6	0.06	6	Karlsson *et al*. 2001
Tc	8	0.8	80	Karlsson *et al*. 2001
Ni	0.2	0.02	2	Karlsson *et al*. 2001
Nb	0.005	0.0005	0.05	Karlsson *et al*. 2001

Pu	0.0004	0.00005	0.7	Karlsson *et al*. 2001
Am	0.001	0.0005	0.2	Karlsson *et al*. 2001
Cm	0.001	0.0001	0.004	Karlsson *et al*. 2001
Np	0.069	0.0014	0.5	IAEA 1994
Ru	0.02	0.01	0.1	Prosser 1994

Table 5-3 Radionuclide half time in the root zone (years) (IAEA 2001a).

Nuclide	**Best estimate**	**Min.**	**Max.**
Cs, I, Pu, Am, Nb, Ru, Cm, Ni, Np	20	10	40
Sr	10	5	20
Tc, Cl	5	1	10

Table 5-4. Parameters of the allometric relationship for the biological half-life (d).

Nuclide	a_2	b_2	**Reference**
Cs	13.22	0.237	Beresford *et al*. 2003
Sr	645	0.26	Higley *et al*. 2003
Cl	2.38	0.25	FASSET*
I	16.7	0.13	Higley *et al*. 2003
Tc	4.8	0.4	USDoE 2002
Pu	1140	0.731	FASSET[+]
Am	1140	0.731	FASSET[+]
Cm	1140	0.731	FASSET[+]

* Derived from data presented in Coughtrey *et al.* (1983) and Bishop *et al.* (1989)

[+] Derived from data presented in Coughtrey *et al*. (1983) and ICRP (1979).

Table 5-5. Parameters for the retention functions in mammals (d) (Coughtrey *et al*. 1985).

Nuclide	m_1	n_1	m_2	n_2	m_3	n_3	m_4	n_4
Ni	0.7	2.77	0.3	$5.8\text{x}10^{-4}$	-	-	-	-
Nb	0.5	0.166	0.5	$3.4\text{x}10^{-3}$	-	-	-	-
Np	0.1	0.139	0.1	$6.94\text{x}10^{-3}$	0.5	$4.64\text{x}10^{-3}$	-	-
Ru	0.15	2.31	0.35	0.087	0.3	0.02	0.2	$6.9\ \text{x}10^{-4}$

Table 5-6 Parameters for the assimilation of radionuclides in the gut of mammals.

Nuclide	$f1_H$	$f1_C$	$f1s$	Reference
Cs	1	1	0.1	Coughtrey *et al.* 1985; Beresford *et al.* 2000
Sr	0.2	0.2	0.2	Coughtrey *et al.* 1985; Beresford *et al.* 2000
Cl	1	1	1*	Bishop *et al.* 1989
I	1	1	1	Beresford *et al.* 2000
Tc	0.1	0.1	0.1*	Bishop *et al.* 1989
Ni	0.05	0.05	0.05*	Coughtrey *et al.* 1985
Nb	0.002	0.002	0.002*	Coughtrey *et al.* 1985
Pu	0.0005	0.0005	0.0001	Coughtrey *et al.* 1985; Beresford *et al.* 2000
Am	0.0005	0.0005	0.0001++	Coughtrey *et al.* 1985; Beresford *et al.* 2000
Cm	0.0003	0.0003	0.0001++	Coughtrey *et al.* 1985; Beresford *et al.* 2000
Np	0.001	0.001	0.001*	Coughtrey *et al.* 1985
Ru	0.05	0.05	0.001*	ICRP 1979

*Where no specific data for $f1_s$ it is assumed to be the same as $f1_H$
++$f1_s$ assumed to be the same as values presented for Pu by Beresford *et al.* (2000).

Table 5-7. Fraction of total activity in the soft tissues (relative units) (based on Coughtrey *et al.* 1985).

Nuclide	f_{soft}	Nuclide	f_{soft}
Cs	1	Nb	0.3
Sr	0.09	Pu	0.6
Cl	1	Am	0.6
I	1	Cm	0.6
Tc	1	Np	0.5
Ni	1	Ru	1

5.4 Model Predictions

Predictions have been made using the model described for two scenarios: (i) equilibrium at time after a single deposition event (Table 5-8); (ii) after 50 years of a constant chronic deposition (Table 5-9). Results are normalised to a soil activity concentration of 1 Bq kg^{-1} DM and a 1 Bq m^{-2} year^{-1} deposition respectively and both are presented for year 50 after the start of the scenario. Animals were assumed to have the live-weights of a rabbit (herbivorous mammal) and red fox (carnivorous mammal) (see Appendix 2, Sections 1.1.13 and 1.1.15). The 5 %, 50 % and 95 % percentiles presented on Table 5-9 are obtained from a probabilistic simulation with FASTer using probability distributions for the soil-to-grass CR, the biomass of grass, the weathering half-time and the leaching half-time. Best estimate values are used for all other parameters. The best estimate results shown in the table are the simulation results obtained using best estimate values for all model parameters.

Comparison of predicted values within Table 5-8 and observed CR values collated for mammal reference organism groups in Table 3-5 allows limited comment on the validity of model predictions. For Cs, predicted values are within the observed range, and the predicted increase in activity concentrations from prey-carnivore is similar to that expected from the literature (see review in Strand *et al.* 2001). The best estimate prediction for ^{90}Sr in herbivorous mammals is in good agreement with the observed mean. Predicted values for ^{226}Ra and uranium appear reasonable if somewhat high and low respectively compared with the limited available data. Model predictions for the transfer of actinides elements to herbivorous mammals appear low compared to the minimal data presented within Table 3-5. For comparison Copplestone *et al.* (2001) recommend CR values for herbivorous and burrowing mammals of $1x10^{-4}$ and $5x10^{-4}$ respectively for ^{239}Pu for use within environmental impact assessments. From the review of Hakonson *et al.* (1981) of studies conducted within the United States a range in CR values for Pu to *native animals* (predominantly rodents) of 10^{-5}-10^{-2} (FW) can be estimate. The 50th percentile value predicted here is comparable to the lower end of this range although our best estimate value is comparatively low. However, Hakonson and Nyhan (1980) report that the pelt and gastrointestinal tract (GIT) contained 50 % and 46 % respectively of the whole body Pu content of rodents trapped at a contaminated US site. Therefore, we could expect the model outputs to be approaching two orders magnitude lower than reported whole body activity concentrations which include pelt and GIT; the model prediction representing absorbed radionuclide activity concentrations.

The scenario modelled here is for herbivorous and carnivorous mammals with typical live-weights and average life-times of a rabbit and fox respectively (see Appendix 2, Sections 1.1.13 and 1.1.15). Whilst from some of the equations incorporated into the model variations in transfer would be expected as a consequence of live-weight it can be shown that this is unlikely to lead to soil – organism CR values which vary by more than 100 % the values presented in Tables 5-8 and 5-9. Therefore, variation as a consequence of live-weight is comparatively low given all the other uncertainties and the predictions presented within Table 5-8 and 5-9 can be assumed to be representative for the two mammalian reference organism groups.

5.4.1 Limitations of current model

We accept that the existing model has many limitations and some of these are discussed below. However, it represents a valuable first attempt to dynamically model transfer to natural biota and on the basis of available data with which to compare it the resulting predictions are reasonable.

The model obviously considers a very simplified foodchain which will not accurately reflect the complex dietary intakes of many species. Similarly, soil ingestion, an important route of intake for some radionuclides, would be expected to vary between species as a consequence of dietary and other habits, and not solely in proportion to dry matter intake as currently modelled; there are also few data on the availability of ingested soil associated radionuclides for transfer across the gastrointestinal tract. Observed variation in a number of other parameters (e.g. gastrointestinal absorption) could be included in the probability assessment with further literature review.

The observation of Hakonson and Nyhan (1980) that >95 % of whole body plutonium of rodents was in the pelt and GIT highlights a potential underestimation of the transfer of radionuclides (with low f_1 values) through food chains as radionuclides associated with these compartments would not be assumed to be consumed by carnivores. Toal *et al*. (2002b) concluded that the cadmium ‘dose’ of predators could be underestimated by up to a factor of 10 if the GIT contents of wood mice (*Apodemus sylvaticus*) were neglected.

Soil and other ecosystem characteristics of European semi-natural pastures/heathlands vary widely and could be expected to lead to considerable spatial variation in the transfer of radionuclides to reference organisms. Whilst in this respect the model presented here is no different than the majority of human foodchain models we can perhaps expect that semi-natural ecosystems to represent extremes of (for instance) soil parameters.

Many reference organisms are not represented within this initial model. For some radionuclides – reference organism combinations there is perhaps sufficient data to enable their incorporation in future evolutions of the model.

Table 5-8 Concentration ratios at equilibrium from soil to reference organisms of the semi-natural ecosystem predicted using FASTer model.

	Grass (Bq kg^{-1} DM per 1Bq kg^{-1} DW soil)				Rabbit (Bq kg^{-1} FW per 1Bq kg^{-1} DW soil)				Fox (Bq kg^{-1} FW per 1Bq kg^{-1} DW soil)			
	5%	*50%*	*95%*	*BE*[+]	*5%*	*50%*	*95%*	*BE*	*5%*	*50%*	*95%*	*BE*
Cl-36	1.4E+01	3.1E+01	4.7E+01	3.0E+01	3.0E+00	6.5E+00	9.9E+00	6.3E+00	3.1E+00	6.8E+00	1.0E+01	6.6E+00
Ni-59	4.1E-02	2.0E-01	4.8E-01	2.0E-01	4.8E-02	2.3E-01	5.6E-01	2.3E-01	3.2E-01	1.6E+00	3.8E+00	1.6E+00
Ni-63	4.1E-02	2.0E-01	4.8E-01	2.0E-01	4.8E-02	2.3E-01	5.6E-01	2.3E-01	3.2E-01	1.6E+00	3.8E+00	1.6E+00
Sr-90	5.5E-01	1.0E+00	1.5E+00	1.0E+00	2.1E+00	3.9E+00	5.7E+00	3.8E+00	3.8E+00	7.3E+00	1.1E+01	7.0E+00
Nb-95	1.0E-03	4.9E-03	1.2E-02	5.0E-03	5.4E-06	2.6E-05	6.2E-05	2.6E-05	3.3E-08	1.5E-07	3.6E-07	1.5E-07
Tc-99	1.6E+00	7.9E+00	1.9E+01	8.0E+00	7.3E-02	3.6E-01	8.7E-01	3.7E-01	2.1E-02	1.0E-01	2.4E-01	1.0E-01
Ru-103	1.3E-02	2.4E-02	3.4E-02	2.0E-02	2.0E-03	3.7E-03	5.2E-03	3.1E-03	1.2E-03	2.1E-03	3.0E-03	1.7E-03
Ru-106	1.3E-02	2.4E-02	3.4E-02	2.0E-02	1.5E-02	2.7E-02	3.8E-02	2.3E-02	9.2E-02	1.7E-01	2.4E-01	1.2E-01
I-129	1.2E-01	6.0E-01	1.4E+00	6.0E-01	1.7E-01	8.2E-01	2.0E+00	8.2E-01	1.0E+00	4.9E+00	1.2E+01	4.9E+00
I-131	1.2E-01	6.0E-01	1.4E+00	6.0E-01	5.2E-02	2.5E-01	6.1E-01	2.5E-01	8.6E-02	4.2E-01	1.0E+00	4.1E-01
Cs-135	4.1E-02	2.0E-01	4.8E-01	2.0E-01	4.7E-02	2.3E-01	5.5E-01	2.3E-01	2.7E-01	1.3E+00	3.2E+00	1.3E+00
Cs-137	4.1E-02	2.0E-01	4.8E-01	2.0E-01	4.7E-02	2.3E-01	5.5E-01	2.3E-01	2.7E-01	1.3E+00	3.2E+00	1.3E+00
Ra-226	2.7E-02	8.0E-02	1.5E-01	8.0E-02	8.1E-02	2.4E-01	4.4E-01	2.4E-01	1.2E-01	3.6E-01	6.7E-01	3.7E-01
Th-230	2.3E-03	1.1E-02	2.6E-02	1.1E-02	8.9E-06	4.4E-05	1.0E-04	4.4E-05	1.2E-07	5.9E-07	1.4E-06	5.9E-07
Th-232	2.3E-03	1.1E-02	2.6E-02	1.1E-02	8.9E-06	4.4E-05	1.0E-04	4.4E-05	1.2E-07	5.9E-07	1.4E-06	5.9E-07
U-234	4.8E-03	2.3E-02	5.5E-02	2.3E-02	1.1E-04	5.6E-04	1.3E-03	5.5E-04	1.4E-05	6.7E-05	1.6E-04	6.6E-05
U-235	4.8E-03	2.3E-02	5.5E-02	2.3E-02	1.1E-04	5.6E-04	1.3E-03	5.5E-04	1.4E-05	6.7E-05	1.6E-04	6.6E-05
U-238	4.8E-03	2.3E-02	5.5E-02	2.3E-02	1.1E-04	5.6E-04	1.3E-03	5.5E-04	1.4E-05	6.7E-05	1.6E-04	6.6E-05
Np-237	4.0E-03	4.1E-02	1.5E-01	7.0E-02	8.5E-05	8.8E-04	3.2E-03	1.5E-03	5.3E-06	5.5E-05	2.0E-04	9.3E-05
Pu-238	1.3E-04	1.2E-03	3.8E-03	4.0E-04	1.4E-06	1.3E-05	4.1E-05	4.2E-06	5.3E-08	4.7E-07	1.5E-06	1.6E-07
Pu-239	1.3E-04	1.2E-03	3.8E-03	4.0E-04	1.4E-06	1.3E-05	4.1E-05	4.2E-06	5.3E-08	4.7E-07	1.5E-06	1.6E-07
Pu-241	1.3E-04	1.2E-03	3.8E-03	4.0E-04	1.4E-06	1.3E-05	4.1E-05	4.2E-06	5.3E-08	4.7E-07	1.5E-06	1.6E-07
Am-241	7.9E-04	2.1E-03	3.6E-03	1.0E-03	8.4E-06	2.2E-05	3.9E-05	1.1E-05	3.2E-07	8.5E-07	1.5E-06	4.0E-07
Cm-242	1.9E-04	7.9E-04	1.7E-03	1.0E-03	5.1E-07	2.1E-06	4.6E-06	2.6E-06	3.3E-09	1.4E-08	3.0E-08	1.7E-08
Cm-244	1.9E-04	7.9E-04	1.7E-03	1.0E-03	1.2E-06	5.0E-06	1.1E-05	6.4E-06	2.8E-08	1.1E-07	2.5E-07	1.4E-07

[+] Best estimate

Table 5-9. Activity concentrations, normalised by the deposition rate, in the top 10 cm layer of the soil and in reference organisms of the semi-natural ecosystem assuming a constant annual deposition. Values presented are predictions after 50 years of a chronic deposition.

	Soil (Bq kg^{-1} DM per Bq m^{-2} year^{-1})				**Grass (Bq kg^{-1} DM per Bq m^{-2} year^{-1})**			
	5%	*50%*	*95%*	*BE*	*5%*	*50%*	*95%*	*BE*
Cl-36	3.1E-02	6.1E-02	9.2E-02	5.5E-02	9.1E-01	2.1E+00	4.1E+00	1.9E+00
Ni-59	1.4E-01	2.0E-01	2.4E-01	1.8E-01	1.1E-01	2.8E-01	1.0E+00	3.1E-01
Ni-63	1.3E-01	1.7E-01	2.1E-01	1.6E-01	1.0E-01	2.8E-01	1.0E+00	3.0E-01
Sr-90	6.2E-02	8.9E-02	1.1E-01	8.2E-02	1.6E-01	3.2E-01	1.1E+00	3.5E-01
Nb-95	7.2E-04	8.4E-04	9.4E-04	8.4E-04	5.4E-02	1.6E-01	6.5E-01	1.9E-01
Tc-99	2.7E-02	6.0E-02	9.3E-02	5.5E-02	2.5E-01	7.7E-01	1.9E+00	7.2E-01
Ru-103	8.4E-04	9.8E-04	1.1E-03	9.8E-04	5.6E-02	1.6E-01	6.8E-01	2.0E-01
Ru-106	1.0E-02	1.0E-02	1.1E-02	1.0E-02	7.1E-02	2.2E-01	9.5E-01	2.6E-01
I-129	1.4E-01	2.0E-01	2.4E-01	1.8E-01	1.5E-01	3.7E-01	1.2E+00	3.8E-01
I-131	9.4E-05	1.4E-04	1.8E-04	1.3E-04	2.8E-02	7.9E-02	3.0E-01	9.5E-02
Cs-135	1.4E-01	2.0E-01	2.4E-01	1.8E-01	1.1E-01	2.8E-01	1.1E+00	3.1E-01
Cs-137	1.0E-01	1.3E-01	1.6E-01	1.3E-01	1.0E-01	2.7E-01	1.0E+00	3.0E-01
Ra-226	7.6E-02	1.2E-01	1.7E-01	1.1E-01	8.3E-02	2.4E-01	1.0E+00	2.8E-01
Th^{+}	1.4E-01	2.0E-01	2.4E-01	1.8E-01	7.6E-02	2.3E-01	1.0E+00	2.7E-01
U^{++}	1.4E-01	2.0E-01	2.4E-01	1.8E-01	7.8E-02	2.4E-01	1.0E+00	2.7E-01
Np-237	1.4E-01	2.0E-01	2.4E-01	1.8E-01	8.4E-02	2.4E-01	1.0E+00	2.8E-01
Pu-238	1.2E-01	1.7E-01	2.1E-01	1.6E-01	7.4E-02	2.3E-01	1.0E+00	2.7E-01
Pu-239	1.4E-01	2.0E-01	2.4E-01	1.8E-01	7.4E-02	2.3E-01	1.0E+00	2.7E-01
Pu-241	7.7E-02	9.5E-02	1.1E-01	9.1E-02	7.4E-02	2.3E-01	1.0E+00	2.7E-01
Am-241	1.4E-01	1.9E-01	2.4E-01	1.8E-01	7.4E-02	2.3E-01	1.0E+00	2.7E-01
Cm-242	4.4E-03	4.6E-03	4.7E-03	4.6E-03	6.8E-02	2.1E-01	9.0E-01	2.5E-01
Cm-244	8.4E-02	1.1E-01	1.2E-01	1.0E-01	7.4E-02	2.3E-01	1.0E+00	2.7E-01
	Rabbit (Bq kg^{-1} FW per Bq m^{-2} year^{-1})				**Fox (Bq kg^{-1} FW per per Bq m^{-2} year^{-1})**			
Cl-36	1.9E-01	4.3E-01	8.6E-01	4.1E-01	2.0E-01	4.6E-01	9.0E-01	4.3E-01
Ni-59	1.3E-01	3.3E-01	1.2E+00	3.6E-01	8.4E-01	2.2E+00	8.2E+00	2.4E+00
Ni-63	1.2E-01	3.2E-01	1.2E+00	3.6E-01	8.2E-01	2.2E+00	8.1E+00	2.4E+00
Sr-90	6.0E-01	1.2E+00	4.2E+00	1.3E+00	1.1E+00	2.3E+00	7.8E+00	2.5E+00
Nb-95	2.8E-04	8.2E-04	3.4E-03	1.0E-03	1.7E-06	4.8E-06	2.0E-05	5.7E-06
Tc-99	1.2E-02	3.5E-02	8.5E-02	3.3E-02	3.2E-03	9.9E-03	2.4E-02	9.2E-03
Ru-103	8.5E-03	2.5E-02	1.0E-01	3.0E-02	4.9E-03	1.4E-02	5.9E-02	1.7E-02
Ru-106	7.8E-02	2.4E-01	1.1E+00	2.9E-01	5.0E-01	1.5E+00	6.7E+00	1.8E+00
I-129	2.0E-01	5.1E-01	1.6E+00	5.2E-01	1.2E+00	3.0E+00	9.5E+00	3.1E+00
I-131	1.2E-02	3.3E-02	1.3E-01	4.0E-02	1.9E-02	5.5E-02	2.1E-01	6.5E-02
Cs-135	1.2E-01	3.3E-01	1.2E+00	3.5E-01	7.2E-01	1.9E+00	7.0E+00	2.0E+00
Cs-137	1.2E-01	3.1E-01	1.2E+00	3.4E-01	6.7E-01	1.8E+00	6.8E+00	2.0E+00
Ra-226	2.5E-01	7.3E-01	3.1E+00	8.5E-01	3.8E-01	1.1E+00	4.6E+00	1.3E+00
Th^{+}	2.9E-04	9.0E-04	3.9E-03	1.1E-03	4.0E-06	1.2E-05	5.3E-05	1.4E-05
U^{++}	1.9E-03	5.7E-03	2.4E-02	6.6E-03	2.3E-04	6.8E-04	2.9E-03	7.9E-04
Np-237	1.8E-03	5.2E-03	2.1E-02	6.0E-03	1.1E-04	3.3E-04	1.4E-03	3.8E-04
Pu-238	7.9E-04	2.5E-03	1.1E-02	2.9E-03	3.0E-05	9.3E-05	4.0E-04	1.1E-04
Pu-239	7.9E-04	2.5E-03	1.1E-02	2.9E-03	3.0E-05	9.3E-05	4.0E-04	1.1E-04
Pu-241	7.9E-04	2.5E-03	1.1E-02	2.9E-03	3.0E-05	9.3E-05	4.0E-04	1.1E-04
Am-241	7.9E-04	2.5E-03	1.1E-02	2.9E-03	3.0E-05	9.3E-05	4.0E-04	1.1E-04
Cm-242	1.8E-04	5.5E-04	2.4E-03	6.6E-04	1.2E-06	3.7E-06	1.6E-05	4.4E-06
Cm-244	4.7E-04	1.5E-03	6.4E-03	1.7E-03	1.1E-05	3.3E-05	1.4E-04	3.9E-05

$^{+}$Predictions for both thorium isotopes (^{230}Th, ^{232}Th) are the same; $^{++}$Predictions for all uranium isotopes (^{234}U, ^{235}U, ^{238}U) are the same.

6 Review of models applicable to the prediction of radioactivity concentrations in aquatic environmental media

6.1 Introduction

The Framework for ASSessment of Environmental ImpacT (FASSET) programme is concerned with the determination of a methodology to predict the impact of radioactivity on the environment. As part of this work programme radioactivity concentrations in environmental media will be required. Westlakes Scientific Consulting has carried out a review of presently available aquatic dispersion models to assess the suitability of aquatic models for predicting environmental activity concentrations in regional seas, coastal areas, estuaries, rivers and lakes.

An initial review covering both commercially available and research models[7] identifies the main aquatic models potentially suitable for modelling radionuclide dispersion. From this group, commercially-available compartment models (based on simple linear transfer rates), as well as more complex dynamic equation-solving models, were chosen for further review. The current trends in model development are also discussed so as to give an overall review of the current trends in modelling.

The suitability of each model for application to FASSET has been determined based on its potential to predict the long and short-term effects of acute and chronic environmental exposure based on past, ongoing, future and accidental releases.

It has been concluded that compartment models based on first-order linear transfer rates provide a more suitable approach to modelling long term and long range transport of radionuclides. Numerical models, which provide a numerical solution of complex fluid dynamic equations, may be best applied to short term releases and local impacts. A brief description of the GLObal MARine Database (GLOMARD) has also been included so as to provide further information on environmental activity concentration monitoring data available.

By its very nature, this review is limited in that it is not possible to know all the possible models that are potentially available to generate concentrations of radionuclides in the aquatic environment. Westlakes' experience is strong in the area of marine modelling, particularly when applied to coastal regions of the Irish Sea. Perhaps in future this review can be turned into an indexed database of models with contributions from other FASSET participants, a potentially useful result setting options for future environmental assessments.

A number of other model reviews are also available in the literature. One of which is similar in its investigation of models suitably for prediction of environmental radioactivity and associated dose calculation is Hilton *et al.* (2002). By their nature, each model review is focused towards different aspects of model implementation.

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[7] Research models are classified as those available for use by non-commercial organisations or those from which only output data may be provided

6.2 General introduction to different modelling approaches

This section introduces some of the relevant issues and terminology used to understand the way models work. If the reader is not technically orientated, this section can be bypassed and such reader is advised to skip directly to Section 3.

6.2.1 Initial considerations

The general purpose of numerical modelling is to provide predictions of physical, chemical and biological processes, using a mathematical approach. One must choose a specific modelling process that will fulfil the objectives and be as close as possible to the processes required. It is not desirable to employ very complex models to get results that could be obtained by simple models using the right set of assumptions. Complex models can unnecessarily increase the coding and running time, require a large amount a data for calibration and introduce greater uncertainty. **In other words, academic overkill should be avoided, especially in a programme such as FASSET, which is supposed to provide a solid framework for practical environmental assessments.**

The models used in assessments should be adequately calibrated and validated. The response of the model to various boundary conditions, or its sensitivity to different parameters, empirical or not, can be assessed by using sensitivity tests. It is desirable to use models that have a solid pedigree, i.e. that they have already been tested with regards to stability and the limits of the model. The accuracy of modelled results is underpinned by fine tuning and data analysis. Again this comes to the point of avoiding academic overkill. Some models can be quite esoteric and sophisticated, to the point that they are dependent on a refined calibration procedure that is not immediate to understand.

6.2.2 Model types

Depending on the requirements and assumptions, models can have zero, one, two or three dimensions.

- Zero-dimensional models (0D) have very limited applications. The variables are averaged in a volume where mixing is considered to be instantaneous. The volume itself and the fluxes at the boundaries control them.
- One-dimensional models (1D) compute parameters in one direction, assuming that advection and mixing are not significant in the other directions. These models are mainly used in long and narrow channels, such as rivers, where a well-mixed flow across a section can be assumed.
- Two-dimensional models (2D) calculate the parameters in a plane, either horizontal or vertical. Horizontal 2D models are depth-averaged. They are used in channel flow type systems where the flow is depth variable.
- Three-dimensional models (3D) give a full and more realistic representation of flow but they are very complex. The calibration of such models requires a large amount of data, and typically more computing time than the other types of models.

Solution of the appropriate equations derives model variables over the area of interest. Different solution methods have been developed. The most common models in marine modelling are finite difference, finite element or box models. The box models are subdivided into large areas in which the parameters are integrated (see, for instance, Lepicard *et al.*, 1998). Each area behaves as a 0D model where transfer at the boundaries is calculated depending on values of the parameters in the next area. They may also need input, such as the hydrodynamic flow field, from data or other models.

Finite element models calculate the variables integrated over small cells, typically triangular in shape. Cell sizes are variable, so that the grid can fit complicated coastline boundaries more realistically. However the computation of the equations is complex and time consuming. The TELEMAC model, developed at the Laboratoire National d'Hydraulique (LNH) in Chatou (France), and used by HR Wallingford (UK) and Bristol University, is a good example of a finite element model.

Finite difference models have been the most popular models in marine modelling. Model equations are solved using Taylor series to get finite difference expressions from the partial derivatives. The grids are typically composed of rectangular cells. Unlike finite element modelling, the different parameters are not computed at the same grid points. Various schemes have been used to solve the equations: explicit, implicit, ADI, QUICK, *etc.*

6.2.3 Model choice and constraints

The choice of the model is dictated by the simplifications necessary to obtain the most realistic results commensurate with minimum cost. For example, the dispersion of a pollutant in a long and narrow river can be simulated accurately enough by using a 1D model, while the hydrodynamics or sediment dynamics in a large shallow area, such as a bay, can be well represented with a 2D horizontal model. Typical uses of the different model types are discussed below.

1D models are principally used in river flows or in narrow well–mixed estuaries. If the vertical transport, either by advection or dispersion, is assumed not to be negligible, then 2D vertical models are preferable. Some additional refinement can be introduced for complex estuaries (Hamilton, 1990). In the modelling of large areas, such as oceans, seas or wide estuaries, 2D horizontal models are preferred, based on the shallow water approximation. In the presence of significant vertical fluxes (melting or formation of an ice field, for example) or gradients (thermocline, river plume) over large areas, 3D models may be required.

Because of their simplicity, finite difference models are chosen for most studies. However, significant errors or imprecision can be introduced around complicated coastlines, such as harbours, where discontinuities prevent the use of curvilinear grids. In this case, finite element models give more accurate and reliable solutions.

The choice of the model time step depends on the characteristic timescale of the dominant processes involved. For example, in a model where tides are one of the dominant phenomena, the time step must not be larger than the tidal period of the main constituents. In the case of long range marine models, where residual transport is studied over long periods, the choice of time step is based on other factors such as the dispersion factors or a decay rate of radioactive contaminants. The grid size will determine the accuracy of the model, but the smaller the grid size is, the longer the computing time will be, and the more unstable the model is likely to be with explicit schemes.

The stability and the accuracy of the model determine the choice of the numerical procedure to be used. Sometimes the time and grid steps are such that the model threatens to be unstable at some time or location, and a more refined numerical scheme needs to be adopted. In long range modelling, residual fluxes are studied and the dispersion effects are one of the dominant processes involved. Therefore, the numerical schemes used for the advection-dispersion must not introduce too much numerical dispersion.

Model constraints come from the amount and the quality of the available data, used either as input parameters (bathymetry, fluxes and concentration of matter from a discharge source, residual flow fields) or from the calibration and validation of the model. The computing facilities can also be a limiting factor in the choice of a model.

Aquatic dispersion models are developed around several stages, hydrodynamics, advection-dispersion and sediment dynamics. Below is a description of the different types, based on lessons learnt from marine modelling.

6.2.4 Hydrodynamic models

Hydrodynamic processes determine the characteristics of the water flow: water levels, pressure, velocity, fluxes, salinity, density and temperature. The assumptions of conservation of mass and momentum give the driving equations for the hydrodynamics: the continuity equation and the well-known Navier-Stokes equations apply here.

The continuity equation is used to compute the vertical velocities or the water levels, from the horizontal velocity field, depending on its domain of integration (local cell or water column). The Navier-Stokes equations give a relationship between the velocity and the pressure. When the hydrostatic assumption is made, the pressure is proportional to the water level. These equations include many processes: the transfer of momentum by advection, the turbulent dispersion, the bed friction, the pressure gradients due to the surface elevations (barotropic mode) and to the density differences (baroclinic mode), the Coriolis force generated by the earth rotation on geophysical flows, etc. Physical processes that influence the hydrodynamics and are generated at the boundaries of the domain by external phenomena (tides, waves, wind, heating and cooling, discharge of a river) or occur within the domain (diffusion and dispersion, bed friction) are all modelled.

6.2.5 Advection-dispersion models

The advection-dispersion models simulate the transport of substances that are supposed to have no significant effect on the hydrodynamics. The concentration of the substance is computed by an advection-diffusion equation. In addition to the transport of the substance by advection and turbulent dispersion, the equation can include buoyancy, decay terms, or source-sink terms to represent the adsorption-desorption on cohesive sediments.

Various kinds of substances can be considered in advection-dispersion models: dissolved or particulate matter, conservative or decaying tracers interacting or not with other substances (sediment, salt, etc.), substances having different weight or density such as oil slicks. The salinity can be solved as a, conservative, “non-interactive” tracer. Therefore it can be used for the calibration of purely advective and diffusive effects.

Radioactive elements discharged into the aquatic environment are typically metals. Hence, processes involved in heavy metals modelling may also be used for radioactive discharge modelling. Most heavy metal models can give suspended, deposited and pore water heavy metal concentrations. The distributions of the radionuclide between different aquatic compartments is modelled using partition coefficients defined as the ratio of adsorption and desorption coefficients. Radioactive decay also plays a part. Biological uptake or release are usually modelled by means of concentration factors, and the assumption is made that organisms are in equilibrium with the surrounding water milieu, an assumption that is often not correct.

6.2.6 Sediment models

Modelling the dispersion of some contaminants, like radionuclides or heavy metals, may require the modelling of sediment transport. A broad range of type sediment of models has been developed during the past decades. The most suitable type of models would be box models or 2D horizontal models, including a wide range of processes (hydrodynamics, advection-dispersion process, sediment dynamics and biological activity). Different models are characterised by their dimensions, the area covered, and the processes involved. Sediment dynamics modelling will typically attend to the type of sediment: non-cohesive (sand, gravel) and cohesive (silt, clay). An advection-dispersion equation that includes vertical settling (Stokes' Law and empirical laws of settling velocity) and erosion terms represents the way the sediments are transported by the bed load (rolling on the bottom), saltation (intermittently jumping) or suspended load (in suspension in the water column). More complex formulations (Mehta *et al.* 1982; and Hayter, 1986) have been developed to take into account bed sediment consolidation, which can be affected by biological activity (bioturbation).

6.3 Initial model review

The objective of this document is to pinpoint what aquatic models may be more suitable for use in FASSET. This implies a selection of a several models after having reviewed a wider gamut of models that might be available. This section presents the initial overview, in the form of brief summaries condensing information as was available to us about the operation and performance of the models from publications, trade literature, internet searches and personal communications. Section 4 then goes on to discuss the most significant models in greater detail.

The review process has determined the models weaknesses and strengths by considering:

- Key processes included in the model
- Performance- accuracy, precision and uncertainty
- Calibration and validation history
- Quality Assurance (QA) history
- Data requirements
- User-friendliness
- Software demands
- Hardware demands
- Reputation of the model within the European research community

Numerous publications on the dispersion of radionuclides in the European continental shelf seas have also been found in our WSC marine database. This information includes research models developed by various European institutes. Based on reviews performed by Stripling

(1994) and Maul *et al.* (1997) and on information gathered through our own searches, the following models have been selected for review:

- *River and Lake Models*
    - MIKE11 model developed by the Danish Hydraulic Institute, Water and Environment
    - PRAIRIE, developed by AEA Technology
    - RIVTOX, developed as part of the EU RODOS programme
    - LAKECO, developed as part of the EU RODOS programme
    - IAEA, FASSET river model, developed by IRSN
    - CASTEAUR, developed by IRSN
    - DETRA, developed by VTT Nuclear, Finland.
    - Swedish dose assessment model, developed by S. Karlsson, U. Bergström and M. Meili (2001), Swedish Nuclear Fuel and Waste Management Company

- *Estuarine Models*
    - VERSE, developed by Westlakes Scientific Consulting Ltd
    - DIVAST, developed by Dr Roger Proctor, POL
    - ECoS, developed by Plymouth Marine Laboratory

- *Coastal Area Models*
    - MIKE21 model developed by the Danish Hydraulic Institute, Water and Environment
    - Delft 3D model (Netherlands)
    - BSH model, developed and used by the Bundesamt für Seeschiffart und Hydrographie in Hamburg (Germany)
    - GHER model, developed at the University of Liege (Belgium)
    - IFREMER model, developed and used by the Institut Francais de Recherche pour l’Exploitation de la Mer, in Brest (France)
    - TELEMAC, developed by Electricité de France and Laboratoire National d’Hydraulic (LNH) in Chatou (France), used by HR Wallingford.
    - COASTOX, developed as part of the EU RODOS programme
    - Coastal Area Ecosystem Model, developed by L. Kumblad, Department of Systems Ecology and U. Kautsky, Swedish Nuclear Fuel and Waste Management Company
    - NRPA box model (ARCTICMAR), developed by M. Iosjpe, Norwegian Radiation Protection Authority.
    - HAMSOM (Hamburg Shelf Ocean Model), developed from the HOPE (Hamburg Ocean Primitive Equation Model

- *Regional Sea Models*
    - MEAD, developed by Westlakes Scientific Consulting (UK), based on the Harwell CUMBRIA77 code.
    - POLCOMS model, developed by the Proudman Oceanographic Laboratory
    - POSEIDON, developed by the CEPN, in Paris (France).
    - PC CREAM, developed by the NRPB (UK)
    - SCREAM, developed by CEFAS
    - NAOSIM (North Atlantic-Arctic Ocean Sea Ice Model), developed by Alfred Wegener Institute for Polar and Marine Research

Brief reviews of these models are given below.

6.3.1 River and Lake Models

6.3.1.1 MIKE11

MIKE11 is a modular software package for the simulation of flows and transport in rivers, estuaries and other narrow channels. It is produced by DHI Water and Environment and is a widely used system in the UK and overseas. Models constructed using MIKE11 are one-dimensional, operating down the length of the river channel(s), with cross sections supplied at intervals along the river to give the channel dimensions and shape. The core of the system is the Hydrodynamic (HD) module that simulates the water flows giving predictions of water depth and discharge along the river(s). The Advection-Dispersion (AD) module is used in parallel with the HD module to simulate the transport and dispersion of contaminants. This module solves the one-dimensional advection-dispersion equation, assuming complete lateral dispersion across the river channel (DHI, 2000). These two MIKE11 modules have been applied by WSC to the rivers Ehen and Calder around Sellafield to simulate the transport of dissolved radioactive contaminants from accidental releases. Model-predicted environmental concentrations were then used to determine the dose to the local critical group (Smith and Vives Lynch, 2002).

This model is suitable for determination of environmental radioactivity concentrations associated with short-term and accidental releases.

6.3.1.2 PRAIRIE

PRAIRIE (Pollution Risk from Accidental Influxes into RIvers and Estuaries) is an UK river management software tool (Welsh, 1992). It uses a 1 dimensional time dependent, finite difference solution to the advection diffusion equation and can be applied in either deterministic or stochastic mode (Thiessen *et al.*, 1997). PRAIRIE is a risk assessment software tool for predicting the risks associated with accidental releases of hazardous materials into rivers or estuaries. The program was developed on behalf of the UK HSE, Department of Environment and National Rivers Authority. It can be used for either deterministic or probabilistic assessments. The model's aquatic dispersion calculations are based on methods advocated by the US EPA and account for volatilisation, photolysis, hydrolysis, oxidation and cationic exchange. The effects of weirs, and chemical partitioning between the dissolved and undissolved phase and interaction with suspended sediment, are also simulated. Databases include chemical specific data, toxicological data and river-specific hydrological data.

This model is suitable for determination of environmental concentrations associated with accidental releases.

6.3.1.3 RIVTOX

RIVTOX is a one-dimensional model for the simulation of the transport of radionuclides in a network of river channels The RIVTOX model, developed at IMMSP, Cybernetics Centre, Kiev, simulates the radionuclide transport in networks of river channels. The sources of radionuclide can be a direct release into a river or the runoff from a catchment. In the latter case, the output from RETRACE-2 (Popov, 1996) is used as the input of RIVTOX. The stream functions, the concentrations of suspended sediment and radionuclide are averaged over a cross-section of a river. A 'diffusion wave' model, derived from the one-dimensional Saint-Venant's equation, describes the water discharge. An advection-diffusion equation simulates the transport of the suspended sediments in the river channel. Its sink/source terms describe the rate of sedimentation and re-suspension as a function of the difference between the actual and equilibrium concentration of suspended matter with respect to the transport capacity of the flow. The latter is calculated on the basis of semi-empirical relations. The dynamics of the upper contaminated riverbed is driven by the equation for the erosion of the bottom layer. The radionuclide transport sub-model of RIVTOX describes the dynamics of the cross-sectionally averaged concentrations of activity in solution, in suspended sediments and in bottom depositions (Zheleznyak, 2000a).

This model is suitable for determination of environmental concentrations associated with accidental releases.

6.3.1.4 LAKECO

LAKECO is a model for studying the transfer of radionuclides in a lake ecosystem. The model has been developed to be one of the aquatic models within the emergency decision support system RODOS (Real-Time On-line Decision Support System for off-site emergency management in Europe). LAKECO is a dynamic model, able to predict the radionuclide levels in water, in sediments, and in fishery produce in various types of lake ecosystems. Sensitivity and uncertainty analyses demonstrate the dominant parameters. In order to develop a more flexible and applicable model, LAKECO has been modified and extended with process based sub-modules to assess these parameters, which increased the predictive power of the model. Subsequent validation tests with this new release showed a higher reliability of the model predictions. LAKECO will be integrated into RODOS and coupled to dose and countermeasure models to assess both the short and long-term radiological consequences due to the aquatic exposure routes in accident circumstances (Heling, 1997).

This model is suitable for determination of environmental concentrations associated with accidental releases.

6.3.1.5 FASSET river model derived from IAEA

The key processes included in the model are radioactive decay, dilution, and sorption/desorption. The model predicts a rough estimation of radionuclide concentration in water and sediment in rivers. These kinds of generic models result in conservative assessments of radionuclide transfers in the environment, in chronic situations. As a result of the simplifying assumptions used in its derivation, this generic methodology strictly applies only if the following conditions are satisfied:

- the surface water geometry (e.g. river cross-section) does not change greatly with distance;
- the flow characteristics (e.g. flow velocity, water depth) do not change significantly with distance or with time;
- Radionuclides in water and sediment, under the conditions of a routine release, can be considered to be in equilibrium.

This model has been derived from the IAEA safety reports series No. 19, titled “Generic models for use in assessing the impact of discharges of radioactive substances to the environ ment”. Its validation and quality-assurance histories are reported in this document.

Parameter values have been provided for the list of radionuclides considered in FASSET. Only two data are needed to make calculations with this simple model:

- Q_i, the average discharge rate for each radionuclide considered (Bq s^{-1}),
- q_r, the mean river flow rate (m^3 s^{-1}).

Calculations are very easy, because the user has only to multiply or to divide the numbers put in a table. There is no significant software or hardware demand, because this very simple model is not a calculation code.

This model is suitable for determination of environmental concentrations associated with short-term and accidental releases.

6.1.3.6 CASTEAUR

The key processes included in this one dimensional model are:

- Hydrography: oriented succession of reaches.
- Hydraulics: uniform, permanent and fluvial condition.
- Sedimentology: advection - diffusion - deposition.
- Ecology: simplified linear food-chain (3 levels).
- Radioecology: advection – diffusion; sorption/desorption; dynamic accumulation from water and food, dynamic depuration.
- Releases: punctual permanent release, punctual pulse release, punctual sequential release at variable rate, linear sequential release at variable rate, individually or in combination.
- Pollutants: radioactive or not, individually or in combination (data for Ag, Am, Co, Cs, Mn, Ru implemented in the prototype).

The outputs consist of pollutant concentrations in water (Bq m^{-3}), suspended matter (Bq kg^{-1} w.w.), deposited matter (Bq kg^{-1} w.w.), zooplankton (Bq kg^{-1} f.w.), macrobenthos (Bq kg^{-1} f.w.), planctonivorous and omnivorous fish (Bq kg^{-1} f.w.), distributed in time or/and in space. The spatial validity is based on the homogeneous mixing sector of rivers whilst the temporal validity is based on the hydrological season (steady-state uniform conditions for hydraulics and biology). An uncertainty and sensitivity study will be soon available (Boyer and Duchesne, 2003).

A first exercise was achieved (Beaugelin-Seiller *et al.* 2002). A second one is planned for 2003. The model is currently only available as a β-version. All the QA aspects will be treated for the definitive version. The data requirement for the model consist of the following:

- Geometry of the reaches: length (m), slope (m m^{-1}), bottom width (m) and angle of the riverbanks ϕ_r (rad).
- Hydraulics (Manning-Strickler relations): river discharge (m^3 s^{-1}), Strickler coefficient ($m^{1/3}$ s^{-1}), water depth (m) (knowing two of these three variables, the tool calculates the third one).

- Sediment modelling: entering suspended matter load (kg m^{-3}), erosion rate (kg m^{-2} s^{-1}) and critical erosion shear stress (N m^{-2})
- Food-chain representation: growth and feeding rates of omnivorous species, and its dietary distributed among the three following compartments, zooplankton, macrobenthos, planktonivorous fish.
- Radioecological transfers:
  - Equilibrium distribution coefficient for suspended matter, phytoplankton and deposited matter, desorption kinetics for the same components, for each pollutant (proposed by default);
  - Accumulation and depuration kinetics for all biotic components, for each pollutant (proposed by default)
- Releases: dissolved and particulate contents of each pollutant considered, rate of discharge, time of beginning and end of the release.

The model is user friendly with successive screens, each of them dedicated to a module (hydrographic network, matter, omnivorous fish, pollutants, transfer parameters, releases, calculations) and containing dialog boxes with fields to select or to fill. It is also possibility to return at the previous step on each screen. Help on line is planned for the definitive version and not activated for the beta version. The model requires *Excel 97* or superior, for Windows NT or 2000, running on a standard PC with processor Pentium II

This model will be used in the European network EVANET-HYDRA, for a comparison exercise planned to begin in March 2003. It is potentially suitable for short term releases, bases on results of the planned comparison exercise and upgrade from beta version.

6.3.1.7 DETRA (Doses via Environmental Transfer of Radionuclides)

The computer code DETRA is a generic tool for environmental transfer analysis of radioactive or stable substances. The DETRA code employs a dynamic compartment approach. It was developed and applied by VTT (Technical Research Centre in Finland). DETRA is relatively simple, but it was intended to account for the most important processes known to contribute to the concentrations in aquatic ecosystems.

The primary source term is the direct deposition on the lake surface. The activity source from the drainage area is called the secondary source term. The code is a general model that can be applied for analyses of various lakes. In each specific application relevant input data have to be used. The flow rates of solid material and water between compartments have to be specified. Additionally the local sorption circumstances have to be considered by selecting reasonable distribution coefficients K_d for elements. The dynamic fish model represents a general modelling approach for non-predator-intermediate-predator fish chain.

The accuracy of the model predictions depends on the specific application, but in the aquatic food chain the intended accuracy is roughly within a factor of ten, based on the estimation of uncertainty related to the conceptual models and the input parameters.

Sensitivity analyses have shown that the predictions using this model are sensitive to the parameter values for the water exchange rate, sedimentation, suspended sediment load, sorption distribution coefficient (K_d), solubility and consumption rates of plankton by fish.

When model predictions were compared with the empirical values (Cs), DETRA seemed to underestimate the activity content in water in the long-term.

Most of the results were satisfactory (Cs in fish and water). However, the comparison of predicted concentrations to the observed values indicates that especially the modelling of the drainage area including the effect of seasonally and sedimentation need improvement.

The model requires values for such parameters as surface area and mean depth of the lake, amount of suspended solids in water, sedimentation rate, re-suspension from sediment to water, water exchange rate, K_d.

DETRA has been modernised for PC environment with Windows interface. The Windows interface with available help texts makes the use of the code flexible for the user. Different segments of the input file are presented in their own window displays. The calculation results can be illustrated in several ways. The graphs of the concentrations of radionuclides in compartments are easily available for the user after simulation. The graphical displays of the compartment model structure and of the radionuclide chains decay schemes are also generated by the code.

VTT has successfully participated with this model to VAMP aquatic and multiple pathways modelling projects co-ordinated by the IAEA. In the wake of further development, this model is potentially suitable for predicting short term impacts.

6.3.1.8 Dose assessments model used in safety assessments for a reactor operational waste repository (SFR) in Sweden

This modelling suite is used for estimating of doses to critical group from multiple radionuclides calculated to be released from the SFR-repository. The models used are a modification of the models used for SR 97 (latest safety assessment for spent fuel in Sweden) and when possible they are adapted to site data. The following is a description of the lake model. For further description see Karlsson *et al.*, (2001).

Briefly, the model consists of one dynamic part where the radionuclides undergo turnover due to various processes. Water and suspended matter are the carriers of the radionuclides. The turnover of radionuclides is thereby described by a set of first order differential equations. These are solved numerically with ACTIVI, a subprogram from the BIOPATH code package. The obtained amounts in various compartments are then divided by masses or volumes for obtaining the concentrations to be further used for exposure calculations. The exposure from e.g. consumption of fish is obtained by the total radioactivity content in water and suspended matter divided by water volume and multiplied with element specific bio-accumulation factors.

All parameter values used except for dose coefficients are generated from pre-determined statistical distributions. Selection of the distributions reflects mostly the uncertainties in parameter values. This leads to a set of input data, which are used for calculating the results. This is performed by the PRISM-package (Gardner *et al.*, 1983). The statistical tools in the code helps to identify major contributions to the uncertainty in the results. The numerical tools and type models are combined in what we call our EBS-system. The system also includes a database of parameter values, both generic as well as element or nuclide-specific.

The key processes included in the model include water turnover, decay, adsorption to suspended matter, desorption from suspended matter. Sedimentation rates, particle deposition velocity, resuspension, uptake and excretion from biota. Care is also taken of the division of the sediments in transportation and erosion bottoms. For dose estimates, uptake of radionuclides in fish is considered through element-specific bioaccumulation factors and consumption rates. Dose conversion according to EURATOM (1996) are used. Due to the circumstances water may also be transferred to adjacent soils by irrigation. Hence, additional compartments for upper and deeper soil are included. This leads also to additional processes to be considered such as, such as interception, advection, erosion, bioturbation,, root-uptake and translocation.

The precision of results may vary partly due to which result is to be studied. The precision of the statistical properties such as standard deviations is also a function of how many sets of values and thereby runs are performed. The numerical tools for solving differential equations have been verified; the numerical accuracy could not be regarded as a main contributor to uncertainties in results. The solution methods applied in ACTIVITY have been subject to several verifications (e.g. Person and Nilsson, 1978). ACTIVITY and PRISM were parts of one total model intercomparison called PSAC (Klos *et al.*, 1993).

No calibration has been applied to the model. However, results for ^{14}C have been compared to the result from an ecological model for the flow of ^{14}C in the system. Earlier versions have also been participating in several model comparison test such as BIOMOVS 1 and II. The compartment structure using time units of weeks and months and also considered uptake in the aquatic food web dynamically by using compartments for plankton, herbivorous and predator fishes have been validated against observations from the Chernobyl fallout of ^{137}Cs. This was performed in BIOMOVS I (1990) as well as in the VAMP-program.

The general data requirements for the model include water turnover rates (per year), volumes, surfaces, average and maximum depths, concentration of suspended matter, fraction of accumulation bottoms, particle falling velocity, mass sediment growth and consumption values. Element-specific data includes the K_d-distribution factors for suspended matter, decay times, biological uptake factors, dose coefficients and external dose conversion factors.

The model is user-friendly to run and modify with regard to expressions, site-specific data, responses, etc. There is a graphical interface where compartments and their intersections are directly drawn on the screen. The user easily selects type and number of responses. Selected results from various steps in the calculations for a number of radionuclides are easily put in tables for reporting. Results may also be presented graphically especially concerning main contribution to the uncertainty in results.

The model runs on *Windows 95* and requires activity data from the BIOPATH code package and further input data from the Prism-system, running on a standard PC with 80486 or higher Intel-based processor. Computer hardware must also satisfy requirements enumerated in the "Windows 95 system requirements" and "Lahey Fortran 90 system requirements" sub-chapters.

This model is still under development and will require calibration and validation before suitability for FASSET purposes can be stated.

6.3.2 Estuarine Models

6.3.2.1 VERSE

The 2D mathematical model VERSE, developed by Westlakes Scientific Consulting Ltd, is capable of simulating the hydrodynamics, sediment dynamics, radionuclide and trace metal dispersion in partially mixed estuaries (Gleizon, 2002). The model has been specifically developed for the Ribble Estuary, but is designed so that it may be easily adapted for use in other estuarine systems. Coverage of the model extends from a point near to Lytham, to beyond the tidal limit. The model allows the user to investigate dissolved and suspended particulate radionuclide activity concentrations in the water column, as well as radionuclide activity concentration profiles in bed sediments. Additionally, the location, type and duration of contaminant input may be specified, allowing the choice between point and diffuse sources. This adaptability enables the model to be applied to a wide range of environmental and discharge scenarios (Watts and Gleizon, 2002).

This model is suitable for determination of environmental concentrations associated with short term and accidental releases.

6.3.2.2 DIVAST

DIVAST, developed by Prof. Roger Falconer, University of Cardiff, is a 2-dimensional flow and contaminant transport estuarine/coastal model. It is a numerical hydrodynamic, solute and sediment transport model for estuarine and coastal waters. It is typically used for Environmental Impact Assessment projects. Model capabilities include the ability to compute water surface elevations, 2D flow velocities, predict BOD, total coliforms or other water quality indicators from out-falls or river discharges and assessing sediment transport and erosion processes. The source code, experience and support are readily available to users. However, extensive training or modelling experience is required. The model output includes distribution of water quality parameters, e.g. salinity, T, faecal and total coliforms, disease burden risk levels, BOD, DO, N, P and heavy metals (http://www.cf.ac.uk/engin/-research/water/2.1/2.1.1.6/erdf2.1.1.6.html).

This model may require substantial investment of time and resources to code it for investigation of radioactive discharges.

6.3.2.3 ECoS

Developed at Plymouth Marine Laboratories*,* ECoS provides a modelling environment for ‘non-coders’. The modelling shell is suitable for modelling spatially and temporarily variable processes. However, a series of templates that can be used to construct an estuarine model are supplied with the software. Initially developed under a DETR contract to model the transport and degradation of TBT in the Tamar Estuary in the early 1980’s, subsequent model development has been funded by Shell, Environment Agency and most recently through the LOIS study. The model was initially designed to concentrate on biogeochemical reactions with limited representation of physical transport properties (initial Tamar model 1 D tidally averaged) the model has evolved and templates are supplied to model tidal scenarios at various levels of complexity.

Current templates supplied with the model are suitable for 1D simulations. Although some 2D-simulation work has been conducted the model at present is not suited to these tasks. Templates supplied with the software specific to setting up an estuarine model include:

- Defining estuarine cross-sectional area through aspect ratio or area-depth polynomials.
- Defining dynamics (velocity and elevation) through cubature (volume of a square section of channel) or hydrodynamic calculations.
- Defining tides at the mouth and river flows.
- Dispersion of solutes.
- Light attenuation through water column.
- Gas exchange across the water surface.
- Mechanistic or empirical methods for predicting sediment transport.
- Partitioning of metals or organic compounds and
- Oxygen, nutrients and algal blooms.

ECoS is a modelling environment, not a model. Templates guide the user through the set up and running of an estuarine model, but these can be adapted, deleted or entirely replaced. The model is used by the EA and has been applied to the Humber (Tappin *et al.*, 2000), Tamar, Tweed (Punt *et al.*, 2002) (Liu *et al.,* 1998), Gironde (Pham *et al.*, 1997) and Ria de Averio (Abreu *et al.*, 1998).

This model may require substantial investment of time and resources to code it for investigation of radioactive discharges.

6.3.3 Coastal area models

6.3.3.1 MIKE21

MIKE21 was developed by the Danish Hydraulic Institute Water and Environment and is a modular 2D modelling system for free surface flows (DHI, 2001). It is a widely used tool for hydraulic modelling in lakes, estuaries, coastal waters and seas.

There are two primary concerns about using a large-scale hydrodynamic model such as MIKE21 for long term and long range assessments. Firstly, the complexity of the model means that a large amount of data will be required to obtain an accurate model set up. Secondly, the detail in the model will require long run times to reach solutions. However it is an important model as it represents a major class of “state of the art” models.

For investigating sediments as a source for radioactivity in the environment it would be appropriate to use the MIKE21 Heavy Metal (ME) module. This module simulates the interaction of metals with sediments in the marine environment.

This model is suitable for predicting environmental concentrations due to short-term release and accidental discharges. Duration of exposure, months or years, would be dependent on the size of the model and the computing resources available.

6.3.3.2 DELFT Hydraulics Model

Initial interest in the Delft modelling system was through the Northwest European Shelf Pilot Model. This pilot model, freely available, covers the region of interest for this review and has an example of modelling Sellafield discharges over 3 years.

The pilot model is uncalibrated and is for demonstration purposes only. It cannot therefore be used for the long range modelling study in its current state The pilot model has been developed using the DELFT3D modelling system in its 2D mode. As it has proven to be capable of carrying out the type of assessment relevant to this review it has come under consideration here.

The Delft Hydraulics pilot model is a two dimensional depth-integrated model of the North sea (Postma *et al.,* 1987). It has been developed to study the long-term impact of pollution from river discharges on the North Sea waters. The model has been used mainly to assess the dispersion of contaminants such as nitrates and cadmium along the Dutch coast. It includes tidal and wind effects, and salinity horizontal gradients. Two nested grids have been used, the first one with a grid step resolution of 8km, the second with a resolution of 3.2 km. This is a flexible, integrated modelling environment, capable of simulating:

- Flows due to tide, wind, density gradients and waves;
- Propagation of directionally spread waves over uneven bathymetries;
- Advection and dispersion of effluents;
- Water quality phenomenon with up to 140 standard contaminants and over 50 processes;
- Initial and/or dynamic (time varying) 2D-morphological changes (including the effects of waves on sediment stirring and bed-load transport).

The model area covers the Northwest European shelf system. This is the same area as covered by the Northwest European Shelf Pilot Model.

DELFT3D is organised in a set of modules, all accessed via a Graphical User Interface (GUI). The hydrodynamic module, DELFT3D-FLOW is at the core of the modelling environment. All other modules can be coupled with the FLOW module.

The system can be switched between 3D and 2D modelling simply by changing the number of layers. This feature enables the user to set up and investigate the model behaviour in 2D mode before going into full 3D simulations.

This model is suitable for predicting environmental concentrations due to short-term release and accidental discharges. Duration of exposure, months or years, would be dependent on the size of the model and the computing resources available.

6.3.3.3 BSH Model

The BSH model has been developed under European funding (MAST-CT900052). Its initial purpose was to study the dispersion of ^{99}Tc from Cap de la Hague, in the English channel and the North Sea (Schönfeld, 1995). The BSH model is a three-dimensional baroclinic circulation model and a Lagrangian dispersion model. The variables are calculated on a fixed Cartesian grid, with a grid step of 1 km. Nested grid of resolution 1 nautical mile in coastal areas and 6 nautical miles on off shore areas have been used around the German and Danish coasts so as to improve the model resolution. The Baltic Sea has also been included with a grid resolution of 12 nautical miles. The model includes tides, wind friction, Coriolis acceleration and simulates heat fluxes. The model has been used to simulate the residual dispersion of ^{99}Tc during a 21-month period (from March 1991 to December 1992). The model's data have been stored with a temporal resolution of 15 min and can be accessed on line.

Complex research tool

6.3.3.4 GHER Model

The GHER model is a 3D model used to determine the residual dispersion of ^{137}Cs on the European continental shelf seas (Djenidi *et al.*, 1987). The model has a large coverage with a variable grid step. The model has been developed by an academic institution and is not yet commercially available.

Complex research tool

6.3.3.5 IFREMER Model

The IFREMER model is a 2D depth-integrated Lagrangrian model of the English Channel (Salomon *et al.,*1987). IFREMER has been used to determine the residual dispersion of ^{125}Sb from Cap de la Hague, in the Golfe Breton-Normand. It may be applied to larger areas, such as the English Channel and the North Sea. As far as we know, this model is not commercially available but it has been used by COGEMA at the nuclear reprocessing power plant at La Hague.

Suitability for FASSET may be for long term exposures, but the model is not generic and may only be applied to the English Channel area.

6.3.3.6 TELEMAC

TELEMAC is a modelling system for simulating physical processes associated with rivers, estuaries and coastal waters. Within a fully integrated package TELEMAC includes modules for the simulation of:

- Steady state flow
- Tidal, wind or wave-driven hydrodynamics
- Dispersion of pollutants including heat
- Transport, erosion and deposition of sand and mud
- Water quality
- Failure of dams or dykes
- Wave dynamics

Developed by Electricité de France and Laboratoire National d'Hydraulic (LNH) in Chatou (France), the TELEMAC system is now both used and marketed by HR Wallingford in close partnership with EDF.

TELEMAC uses the latest finite element techniques to solve the shallow water equations either vertically averaged in 2D or layered in 3D. It has a completely unstructured grid allowing simulation in fine detail of flow plus coverage of large areas. The Scottish Environment Protection Agency uses this system for water quality management in the Forth Estuary.

Using finite element techniques, TELEMAC solves the shallow water equations, either vertically averaged in two dimensions or layered in three dimensions. Using unstructured triangular grids, boundary conditions can be applied away from areas of interest, which in turn can be modeled in fine detail. TELEMAC includes horizontal turbulence options for the simulation of very detailed flow patterns, spherical co-ordinates for very large area models, simulation of wetting and drying within the model domain and solution for transcritical flow. TELEMAC-3D can also simulate three dimensional flow affected by stratification (thermal or saline), wind or wave breaking. Turbulence models available include k-epsilon and mixing length. The results from TELEMAC-2D and TELEMAC-3D are often used as input to other modules to study, for example, water quality and sediment transport.

This model is suitable for predicting environmental concentrations due to short-term release and accidental discharges. Duration of exposure, months or years, would be dependent on the size of the model and the computing resources available.

6.3.3.7 PLUMES

PLUME-RW is a well-established model developed for studies of pollutant dispersion in estuaries and coastal waters. Both dissolved pollutants, such as bacteria from sewage discharges, and suspended pollutants, such as sediment released during dredging operations, can be simulated. Based on a particle based random walk technique, and available in 2 and 3 dimensions, PLUME-RW uses flow data derived using TELEMAC to simulate transport by prevailing currents, turbulent dispersion, bacterial decay and deposition and re-suspension of particulates at the sea bed. Typically, PLUME-RW is used to optimise outfall location and waste treatment levels.

Suitable for both short-term and accidental release scenarios.

6.3.3.8 COASTOX

COASTOX is a two-dimensional model of lateral-longitudinal distribution of radionuclides in water bodies. The two-dimensional COASTOX model uses the depth averaged Navier Stokes equations to calculate the velocity field in rivers, lakes and reservoirs generated from the combined influence of discharge, wind and bottom friction. The steady state approximation without advection terms and the system of the unsteady shallow water equation are used. The same approach as in RIVTOX is applied to simulate the radionuclide exchange in the system: solution - suspended sediments - bottom deposits. The 2-D advection-diffusion equations and the equations of flow dynamics are solved numerically by using finite difference methods. Necessary input to COASTOX is the geometrical data of the river/lake bed with the sufficiently fine spatial resolution. COASTOX was widely used by IMMSP CC to simulate the radionuclide transport in the Kiev Reservoir and the Pripyat River floodplain close to the Chernobyl Nuclear Power Plant. The Measurements for a ^{90}Sr release from the floodplain after ice jams, January 1991 and February 1994, confirmed the results of predictions based on simulations with COASTOX. Within INCO-COPRERNICUS project, COASTOX was tested and verified on the basis of measured ^{137}Cs and ^{90}Sr distributions during the 1987 spring flood in the Kiev reservoir and ^{137}Cs distribution in the bottom sediments of the Kralova Reservoir, the Vakh River, Slovakia. These results demonstrate the importance of applying different adsorption and desorption rates to describe the transfer of ^{137}Cs between solute and bottom deposits (Zheleznyak *et al.*, 2000b).

This model is suitable for determination of environmental concentrations associated with accidental releases.

6.3.3.9 Coastal Area Ecosystem Model

This is a radionuclide transfer model that could be used for any ecosystem provided that suitable data is available. This is a brief description of a generic radionuclide transfer model for aquatic environments. The model is developed for a brackish water bay where a repository for low and intermediate level radioactive waste (SFR) is located. The model is site-specific but could be used for any aquatic ecosystem if environmental data for the sites of interest is available. Established for the hierarchical level of population, based on a mass-balance model approach, is mechanistic and dynamic (on an annual basis).

The radionuclide transfer between the model components is funded on an ecosystem process-based carbon and nutrient flux model (CNP-model), which identifies and quantifies the most important processes for fluxes of matter at the site. The CNP-model includes both abiotic (e.g. water and particulate material) and biotic (e.g. plankton, fish, benthos and seabirds) components and calculates the transfer of carbon and nutrients both between the model components and over the system boundaries. In the model, carbon and nutrients enter the food web through autotrophic organisms and is then channelled through the system (model) in agreement with the food web interactions of the ecosystem. The key processes involved in the CNP-model are photosynthesis, respiration, consumption, water flow, sedimentation and exchange of carbon at the air-sea interface. Temperature and light intensity are also included to compensate for seasonal variations in the metabolic rates and C:N:P ratios for the varying nutrient content in different type of organisms.

Radioactive carbon, nitrogen and phosphorous can easily and with high degree of certainty be modelled with the CNP-model as a fraction of the general flow of the stable isotopes. To enable food web modelling of radionuclides that distinguishes from the properties of carbon, nitrogen and phosphorous, some radionuclide specific processes are involved as a function of the metabolic rates for the organism groups included in the CNP-model. The radionuclide-specific processes considered are (i) radionuclide adsorption to organic surfaces (including organisms), (ii) radionuclide uptake by autotrophic organisms and (iii) radionuclide elimination by heterotrophic organisms. The connection between radionuclide specific mechanisms and the ecosystem dynamics constrains the radionuclide uptake and elimination to estimated proportions of the maximum ingestion and excretion of carbon and nutrients and the availability of radionuclides in media, such as in the water or in the prey items. The radionuclide entry to the food web is assumed to primarily be via autotrophic organisms and ingestion of radionuclides adsorbed to organic surfaces. This is modelled with transfer factors related to the rate of photosynthesis and surface normalised K_d values. The transfer of radionuclides to higher trophic levels is driven by consumption rates and radionuclide concentrations in prey and is regulated by radionuclide elimination rates.

Sensitivity analysis of the relative importance of the radionuclide specific mechanisms for the resulting concentrations in the ecosystem components indicate that uptake of radionuclides via consumption contributes more than adsorption to the organism surfaces. Further analysis of the radionuclide elimination process indicate that even if the elimination rate is extremely low, concentrations in the organisms are just slightly higher than if the elimination rate is the same as for carbon. This emphasises the importance of including data on consumption rates and food web relationships in transfer models.

The performance of the model is ecologically sound and relates the radionuclide dynamics to predominant ecosystem processes. Thus, as long as the ecosystem model is accurate, the radionuclide concentrations in the organisms estimated by the model are within the probable range.

The accuracy of the CNP-model was checked by a validation against eight carbon budgets for adjacent areas, which showed good compliance with the magnitudes of storage and flows for most organism groups. The CNP-model was calibrated for the few deviations found. A comparison of bio-concentration factors (BCFs) derived for organisms at higher trophic levels than primary producers from the model results with empirically obtained BCFs made out another accuracy check of the model. The comparison showed fairly good agreement for the about 40 radionuclides tested.

The model poses large data requirements, especially for environmental data. To run the CNP-model, data on biomass and metabolic rates for all organism groups are needed as well as estimations of amount of dissolved and particulate matter in the water, the water exchange rate and the hypsography of the area. To model the radionuclide dynamics, uptake rates for plants is needed as well as K_d-values. Data on elimination rates improves the simulations but this has shown to be of minor importance compared to other processes.

The model is conceptualised in the software *Matlab* with *Simulink*, which does not require more hardware capacity than a PC. The model is fairly complex and unfortunately not very user-friendly for users that not are familiar with the *Matlab* environment.

The models are to be published in scientific journals. A description of the CNP-model has been accepted for publication in Ecological Modelling, and a description of extension of the CNP-model to the generic radionuclide transfer model has been submitted to the Environmental Pollution journal. These models are until now not well-known to a wider research community except for those that have listened to oral presentations of the models.

This package is potentially suitable for both annual and sub-annual prediction of environmental radioactivity concentrations.

6.3.3.10 RCTICMAR 2

The NRPA box model (ARCTICMAR) is an improved version of the compartmental model developed by Nielsen *et al*. (1997). The model is based on the modified approach for box modelling (Iosjpe *et al*., 2002), which includes dispersion of radionuclides during time (non-instantaneous mixing in oceanic space).

The model includes the processes of advection of radioactivity between compartments, sedimentation, diffusivity of radioactivity through the pore water, resuspension, mixing due to bioturbation and a burial process of activity in deep sediment. Processes of radioactivity incorporation into sea ice through the freezing process, the transfer of radioactivity by ice and incorporation to the water column through melting process are also included into the model.

Radioactive decay is included in all compartments. The contamination of biota is further calculated from the radionuclide concentrations in filtered seawater in the different water regions. Doses to man are calculated on the basis of data for the catch of seafood and assumptions about human diet.

Comparison between predicted and experimental data for sea water, sediment and biota is within an order of magnitude and for many sea regions such comparison is within a range of experimental data distribution. Uncertainties of the dose assessment are dominated by definition of the concentration factors and sediment distribution coefficient.

Comparison between predicted and experimental data for levels of ^{99}Tc in northern European waters following discharges from Sellafield and La Hague show satisfactory agreement (Iosjpe *et al*., 2002).

The structure of water model boxes for surface, mid-depth and deep waters is developed with regards to improved description of Polar, Atlantic and Deep waters in the Arctic Ocean and the Northern Seas (Karcher & Harms, 2000) and site-specific information for description of the boxes. The volume of the water layers in each box has been calculated by using a detailed bathymetry in geographical information system (IBCAO, 2001; ETOPO5, 2002). Results of calculations are compared with 3-D hydrodynamic oceanographic model (Karcher *et al*., 2002).

All necessary information is included into the model. If a scenario of radionuclides discharges is prepared, initial information includes choosing radionuclide, an initial time step for calculations and time intervals for simulation and printing of final information.

The model is prepared in user-friendly "Time-Zero" modelling environment (Kirchner, 1989), but for preparation of the discharge scenarios and modification of initial information some FORTRAN knowledge is required. A TIME-ZERO integrated modelling environment and FORTRAN is required for this model, designed to run on a standard PC.

Compartmental/box modelling has been recommended by the European Commission for radiological assessment (EC, 1995) and selected for many investigations (e.g. EC, 1997; Nielsen *et al*., 1997; IASAP, 1999). The NRPA box model has been developed and applied in two, earlier EU projects including ARMARA (1995-1999) and ARCTICMAR (1998-2001) and is one of a suite of models currently being applied in the EC Project REMOTRANS (2000-2003).

6.3.3.11 HAMSOM (Hamburg Shelf Ocean Model)

HANSON is a hydrostatic, 3-D, level type model based on finite differences. The primitive equation model with a free surface utilises two time-levels, and is defined in Z co-ordinates on the *Arakawa* C-grid. Stability constraints for surface gravity waves and the heat conduction equation are avoided by the implementation of implicit schemes. With a user-defined weighting between future and present time levels a hierarchy of implicit schemes is provided to solve the free surface problem, and for the vertical transfer of momentum and water mass properties. In the time domain a scheme for the Coriolis rotation is incorporated which has second order accuracy. Time and space-dependent vertical exchange and diffusivity coefficients are determined from a simple zero-order turbulence closure scheme which has also been replaced by a higher order closure scheme (GOTM). The resolution of a water column may degenerate to just one grid cell. At the seabed a non-linear (implicit) friction law as well as the full kinematic boundary condition are applied. Seabed cells may deviate from an undisturbed cell height to allow for a better resolution of the topography. The HAMSOM coding excludes any time-splitting, i.e. free surface and internal baroclinic modes are always directly coupled. Simple upstream and more sophisticated advection schemes for both momentum and matter may be run according to directives from the user. Successful couplings with eco-system models (ECOHAM, ERSEM), an atmospheric model (REMO), and both Lagrangian and Eulerian models for sediment transport are reported in the literature. For polar applications HAMSOM was coupled with a viscous-plastic thermo-hydrodynamic ice model of *Hibler* type.

The accuracy and precision of HAMSOM has been tested and documented recently during the NOMADS-2 model intercomparisson project. Uncertainties of HAMSOM are associated mostly with the applied forcing and with the general features of the model type, e.g. the step wise resolution of the topography or the parametrisation of physical processes might lead to certain constrains.

The development of the HAMSOM coding goes back to the mid eighties where it emerged from a fruitful co-operation between *Backhaus* and *Maier-Reimer* who later called his model HOPE (Hamburg Ocean Primitive Equation Model). From the very beginning HAMSOM was designed with the intention to allow simulations of both oceanic and coastal and shelf sea dynamics. Since about 15 years in Hamburg, and overseas in more than 30 laboratories, HAMSOM is already being in use as a so-called community model. There has been a continuous improvement of the model code by the community. HAMSOM is maintained now by the Inst. for Oceanography, University Hamburg. There is a HAMSOM home page;

http://www.ifm.uni-hamburg.de/~wwwsh/res/HAMSOM/hamsom.html

presenting the current distribution, a user group, information on the freely available code etc.

Depending on the application, HAMSOM needs initial and forcing data such as open boundary sea levels, temp. + salinity, meteorological data (wind stress, air temperatures etc.).

HAMSOM is well documented. The home page gives all information needed. The user group helps with trouble shooting. There are no specific software demands. The code is written in Fortran 77, a vectorised version, partly written in Fortran 90, is now available. HAMSOM runs on PCs under Windows and Linux, on workstations and main frame systems such as Cray or NEC.

HAMSOM has been used in the last 15 years in a large number of national and international, EU-funded projects. It is currently used at the Inst. for Oceanography, Hamburg, in at least 5 EU-funded projects (METACOD, LIFECO, ASOF-N 'MOEN', ESTABLISH, NOMADS-2). There are numerous other applications in national projects and at other Institutes.

6.3.4 Regional Seas models

6.3.4.1 MEAD

MEAD is used for the prediction of long term radionuclide dispersion over the Irish Sea. This is achieved by modelling the variation in annually averaged radionuclide activity concentrations over periods up to 100 years. The MEAD model includes the interaction between bed and suspended sediment. The most recent version includes ionic exchange processes between the dissolved phase and the bed sediment. It is currently set up for the Irish Sea area.

Suitable for the prediction of annual doses from routine releases

6.3.4.2 POLCOMS Model

The POL model has been developed to look at the dispersion of ^{137}Cs in and out of the Irish Sea (Prandle, 1984). It uses a fixed grid with a grid step of 35 km but aims for a finer resolution. It covers the European continental shelf seas (Irish Sea, North Sea and English Channel). The model is depth-integrated and includes tides and winds, as well as the effects of the horizontal density gradients. The model calculates the residual advection-dispersion of ^{137}Cs for a 10-year period. A sensitivity study of the dispersion factors has also been made. The model gives an assessment of the influence of both contributions from Sellafield and Cap de la Hague nuclear reprocessing plants and on the release of material in continental shelf waters. The POL model is in the public domain and has already been used for *bona fide* research (Needle, 1993) but cannot be used for commercial purposes.

Complex research model still under development.

6.3.4.3 POSEIDON Model

POSEIDON is a box model of the European continental shelf seas. The model has been developed by the CEPN (France) and has been recently exhibited at the Third SRP meeting in Southport (UK) (Lepicard, 1999). The model does not solve the hydrodynamics of the region but determines the path and dispersion of radioactive contaminants.

Suitable for the prediction of annual doses from routine releases and short term exposure due to accidental releases.

6.3.4.4 PC CREAM

This is a suite of models and data for the radiological impact assessment of routine and continuous discharges. The model consists of six programmes one of which, DORIS, is a marine dispersion model for European waters capable of calculating activity concentrations in seawater and marine sediments. The marine model covers four aquatic areas, namely rivers, estuaries, local and regional waters.

The flow of water is approximated by exchange rates between compartments and it is assumed that instantaneous uniform mixing takes place in each compartment, with the transfer of contaminants being proportional to the inventory of material in the source compartment. The output of the model is the total activity concentration in each compartment, from which seawater concentrations can be calculated. Typical processes such as sediment interaction are also considered within the model.

The development of PC CREAM was carried out by NRPB under contract to the European Commission. It is based in part on the MIRMAID model but has a more complete description of water movements.

Suitable for the prediction of annual doses from routine releases and short term exposure due to accidental releases.

6.4.4.5 CSERAM

CSERAM is a model for prediction of marine radionuclide transport in both particulate and dissolved phases (Aldridge, 1998). The model attempts to go beyond the traditional box model approach in describing the underlying physical processes in a more realistic way. Application to real problems is at the preliminary stage but an illustration is given of the predicted and observed concentrations of a conservative radionuclide, ^{99}Tc. CSERAM includes a 2-dimensional hydrodynamic description of the tidal and wind-induced flows; a wind-wave model to provide the wave-induced bed stress that controls the behavior of the suspended and settled sediments; and, a physically-based transport model to simulate the movement of both the dissolved and particle-bound radionuclides.

Suitable for the prediction of annual doses from routine releases and short term exposure due to accidental releases.

6.3.4.6 NAOSIM (North Atlantic-Arctic Ocean Sea Ice Model)

NAOSIM (North Atlantic-Arctic Ocean Sea Ice Model) is a 3-dimensional coupled ice-ocean-model developed at the Alfred Wegener Institute for Polar and Marine Research. Its oceanic part derives from the Geophysical Fluid Dynamics Laboratory modular ocean model MOM-2. For the advection of tracers a FCT-scheme is employed, which is characterised by a low implicit diffusion and avoids false extremes ("overshooting") in advected quantities. Friction is implemented as a bi-harmonic diffusion of momentum. The model domain of the most frequent applications encloses the North Atlantic, the Nordic Seas and the Arctic Ocean or partial domains thereof. The model boundary may be opened according to the user specifications. The boundary conditions allow the outflow of tracers and the radiation of waves. At inflow points determined by the model, temperature and salinity are specified by the user. For the ocean the model solves the hydrodynamic equations on an Arakawa-B grid. It has an optional free or rigid-lid surface. The spherical horizontal grid is rotated to avoid numerical difficulties at the Pole.

The ocean model is coupled to a dynamic-thermodynamic sea ice model which employs a viscous-plastic rheology. Freezing and melting are calculated by solving the energy budget equation for a single ice layer with optional snow cover. The freezing point of sea water is salinity dependent. The sea ice and ocean models use the same time step and the same horizontal grid. The surface heat flux is calculated from standard bulk formulae using prescribed atmospheric data and sea surface temperature predicted by the ocean model.
Some applications of the model can be found on the web-page:

http://www.awi-bremerhaven.de/Modelling/ARCTIC/index.html.

NAOSIM participates with two applications in an international effort of inter-comparison and improvement of Arctic and Nordic Sea models. Details of the AOMIP (Arctic Ocean Model Inter-comparison Project) can be found on the web-page:

http://fish.cims.nyu.edu/project_aomip/overview.html.

The project results so far show NAOSIM to be one of the top state-of-the-art models for this area.

The model has been successfully used in the past to investigate the circulation of ice and ocean in the Arctic Ocean and the Nordic Seas (e.g. Koeberle and Gerdes, 2003; Karcher *et al.,* 2003; Kauker *et al*., 2003). The same model has also been used in a study to calculate the possible spreading of radionuclides from the sunken submarine 'Kursk' (Gerdes *et al*., 2001) and a study to investigate the dispersion of ^{99}Tc in the Nordic Seas (Karcher *et al*., 2003)

The model uses river runoff and atmospheric forcing data for wind stress or wind velocities, air temperatures, dewpoint temperatures or humidity, cloudiness and precipitation. In addition, forcing functions for tracers can be prescribed. At the open boundaries barotropic transports and hydrographic data are needed.

The modular concept of the model allows easy exchange of parameterisations or addition of new modules describing physical processes.

The model code is written in Fortran 77 and exists in non-paralleled and paralleled versions. The model code is appropriate for running on various platforms like SUN, Vector and Parallel mainframes. NAOSIM has been used as key model in various EU funded projects in the recent decade (VEINS, CONVECTION, ASOF-N). In addition NAOSIM has been utilised in numerous other national and international projects.

6.4 Model selection criteria

The Westlakes Research Institute's aquatic modelling capability focuses on coastal and regional sea models; based on the initial review reported in Section 6.3 and our experience, four marine models were singled out for further reviewing. These models are:

- PC CREAM
- POSEIDON
- DELFT 3D
- MIKE21
- MIKE11

The reasoning behind this choice is that these models satisfy the following conditions:

a) Models are commercially available with full support and quality assurance.
b) Some of them have already been applied to the Northwest European shelf sea area and the Mediterranean Sea.
c) Two of the models are compartmental whereas the DELFT and MIKE21 models are based on numerical simulation; hence the two types of model are well represented and contrasted.
d) Westlakes already owns copies of the PC CREAM, POSEIDON and MIKE21 models. An advantage of the compartmental models PC CREAM and POSEIDON is that they are specifically set up to determine doses to man due to external exposure and consumption pathways. Hence, they have potential for modification to determine dose to biota.

Reasons for not considering further other models were as follows:

- *River and Lake Models*
  - MIKE11: sediment modelling methods unknown.
  - PRAIRIE: not known if the model could be adapted to radionuclide discharges.
  - RIVTOX: it is unknown if this model might be made available for the FASSET programme.
  - LAKECO: it is unknown if this model might be made available for the FASSET programme.

- *Estuarine Models*
    - VERSE code is proprietary and protected by commercial confidentiality; hence it is not available to external groups.
    - DIVAST: not known if the model could be adapted to radionuclide discharges.
    - EcoS: not known if the model could be adapted to radionuclide discharges.

- *Coastal Area Models*
    - BSH: this is an academic model only applicable for the North Sea and Baltic Sea.
    - GHER: this is an academic model and no further information is available at this time regarding commercial availability.
    - IFREMER: while already used for La Hague predictions, the model does not appear to be commercially available.
    - COASTOX: it is unknown if this model might be made available for the FASSET programme.

- *Regional Sea Models*
    - MEAD: the model is currently only set-up for the Irish Sea and the code is unavailable commercially.
    - POLCOMS: still under development and only being used as an academic research tool.
    - SCREAM: academic code may not be commercially available

6.5 Models selected

Suitable models should be able to predict the dispersion of radionuclides over the European Shelf system and have sediment interaction as a constituent model process. Based on this criterion, the following models: PC-CREAM, POSEIDON, DELFT 3D and MIKE21 were selected as meriting further detailed review.
PC CREAM

6.5.1 PC-CREAM

..1.1 Background

PC CREAM is a suit of codes comprising models and data which can be used to perform the radiological impact assessments of routine and continuous discharges from virtually any type of installation including nuclear power plants and nuclear fuel cycle facilities. It was developed for the European Union but parts of the system have been used throughout the world.

The main features of the package include:

- A suite of environmental transfer models to estimate the transfer of radionuclides through the environment.
- Modelling of annual discharges to the atmospheric, marine or river environments.
- A comprehensive list of exposure pathways.
- Results can be expressed in terms of individual or collective doses, using effective dose as defined in ICRP-60 (ICRP, 1991).
- The facility to assess the radiological impact of discharges of radionuclides from virtually any site, for regulatory purposes, and/or health impact studies.

PC CREAM consists of six programmes: the assessment code, and five supporting programmes. The supporting programme for aquatic discharges is DORIS, a marine dispersion model for European waters capable of calculating activity concentrations in seawater, and marine sediments.

6.5.1.2 Key processes included in the model

The PC-CREAM model is a simple 2D compartmental model (depth averaged, first order differential). (European Commission, 1995). The compartments are relatively large (only 44 to cover the whole of the European Shelf). Unfortunately, because of the size of the whole PC CREAM model only subsets of the full set-up could be accommodated in PC CREAM. The model is divided into modules of 15 compartments each. Radionuclides are assumed to be released into a local compartment that is connected to the rest of the compartments. The processes to model water movements are advection and diffusion; the model can calculate the fraction of activity in solution and adsorbed by suspended sediments using K_d values. Sedimentation processes are also represented in the model. Two extra compartments representing seabed sediments are connected to each water compartment. The processes modelled are: settling of particles (modelled using K_ds, sedimentation rates and suspended sediment loads), diffusion and bioturbation. The latter two processes are treated as diffusive processes.

6.5.1.3 Compartment Modelling

Instantaneous mixing is assumed with transfer between compartments being proportional to the inventory of material in the source compartment. The equation that describes variation in activity (A_i) in the compartment i is:

$$\frac{dA_i}{dt} = \sum_{j=1}^{n} k_{ij} A_j - \sum_{j=1}^{n} k_{ij} A_i - k_i A_i + Q_i$$

where:

A_i is the activity present at time t in compartment i
k_{ji} and k_{ij} are the rates of transfer between compartments
k_i is the loss of material without transfer to another compartment
Q_i is a source of continuous input
n is the number of compartments in the system

6.5.1.4 Daughter Products

In some cases, daughter products are in secular equilibrium with the parent radionuclide and thus the parent determines their behaviour in the aquatic environment. However in non-secular equilibrium cases, PC CREAM models the daughter product chain by adding, for each daughter in the chain, a further set of compartments identical to those of the parent. Physical parameters are the same but radionuclide dependent parameters differ according to the radionuclide being modelled. Thus daughter products are modelled in a mirror system of the parent radionuclide.

6.5.1.5 Radioactive Decay

This is modelled by using an effective transfer rate from a compartment that takes into account loss of material from the compartment without transfer to another.

6.5.1.6 River Models

PC CREAM uses a semi-empirical river model since this is the simplest model that will provide the necessary degree of predictive accuracy and the best starting point for generic model development. The model used is an extension of the algorithm developed by Schaeffer (1975), and described in the 1979 CEC methodology report. Two points of difference between Schaeffer's river models and PC CREAM are:

- An effluent plume model in PC CREAM has replaced assumption of instantaneous dilution.
- In addition to bed sediment transport the latest version in PC CREAM incorporates a river-beach or river-bank compartment.

6.5.1.7 Estuary Models

Estuaries are complex aquatic environments where the interface between fresh and marine waters can lead to changes in radionuclide uptake by sediments. In such areas the discharge site is important in relation to the required model complexity. Since generic estuarine models are difficult to produce, PC CREAM has recognised two broad categories of estuary:

- Fjords
- Shallow Muddy Estuaries

Account is taken for water stratification and the associated sediment interactions.

6.5.1.8 Marine Modelling

In the case of marine models both local and regional compartments are used. The local compartments take into account local environmental conditions that may be important in determining exposures. Three types of local compartment are available:

- Estuarine
- Coastal
- Exposed Coastal

The model describes the significant water movements in European coastal seas. The data is similar to that used in Charles *et al.*, (1990), but PC CREAM has a more complete description of water movements.

Sediment interaction is taken into account, with desorption accounted for by use of a net absorption factor. The bed consists of three layers and the movement of sediment between these layers is also modelled.

The purpose of the estuarine and marine models is to calculate the radionuclide concentrations in the filtrate fraction of the water and on suspended sediment.

6.5.1.9 Performance, Validation and Limitations

The PC CREAM model is used for assessing the consequences of continuous discharges. The use of annually averaged data for water flow parameters is considered adequate when assessment on an annual basis or integrated over a longer time-scale is required. Thus, this model is not appropriate for short term releases or deriving information on concentrations over short times (*e.g.* less than one year).

The model has been validated by comparison of model predictions with environmental data not used in its construction. The simplified river model was validated against data obtained from a study of the River Molse Bette in Belgium. Predicted concentration rates of radionuclides in bed sediment and the external gamma dose rates were compared with measured data. Agreement between prediction and measurement was within a factor of five (Lawson *et al.*, 1991).

The marine model has also been validated against environmental measurements for a number of different cases as part of the MARINA project. Results were compared for releases of ^{137}Cs, ^{99}Tc and ^{239,240}Pu. For ^{137}Cs and ^{239,240}Pu dispersion in European shelf seas, the agreement between prediction and measurement was within a factor of two, however plutonium in *Fucus* seaweed was mainly overestimated. For ^{99}Tc, agreement was in general, acceptable close to the discharge point but predictions tended to underestimate at greater distances (Charles *et al.*, 1990).

Further validation was performed for ^{137}Cs and ^{239,240}Pu in the Irish Sea (Wilkins *et al.*, 1994) and comparisons were found to be in reasonable agreement. No systematic under or over-prediction by the model was found, which is considered adequate for the estimation of radiation doses.

The original model was developed as part of the EU MARINA project in the 1980's and was the combination of different models for specific areas. Extensions were added at a later stage (Irish Sea and the Spanish waters). As stated above, the model in PC CREAM is not exactly the model described in the EU report, but a smaller size version. The differences between the radionuclide concentrations predicted by the two models is within 10% in the compartments near the release point but could be as high as a factor of 3 to 5, depending on the compartment and radionuclide, in compartments distant from the release point. These differences are not significant if the model is used for its purpose, to calculate individual doses near the discharging site or collective doses. The differences can be considered more significant if the model is used to predict radionuclide concentrations at a greater distance from the site.

6.5.1.10 A History

The Environmental Assessments Department of NRPB that developed PC CREAM has a Quality Management System, which is certified under ISO-9001. The development of the model started before full certification. Therefore PC CREAM was only partially developed under the Departmental Quality System.

The latest version is PC CREAM 98, which was released in October 1998. This version contains revised data sets mainly used in the calculation of collective doses.

6.5.1.11 Data Requirements

The model can only calculate mean annual concentrations following a continuous release at an annual rate specified by the user. The user has to select the radionuclides from a list for which default data for parameters like K_d are provided. The user can change these parameter values. Other parameters required by the model for each compartment (volume, depth, suspended sediment load, sedimentation rate, sediment diffusion and bioturbation coefficients, sediment density) as well as the exchange rate due to advection and diffusion (summed together) are provided but can be changed by the user.

6.5.1.12 User Friendliness

PC CREAM is written in Visual Basic. The user can use all the Microsoft Windows features (text boxes, option buttons, and check boxes, menus, dialog boxes) to enter data.

6.5.1.13 Hardware and Software Demands

PC CREAM has been designed to run on a stand alone PC under Microsoft Windows version 3.1, 3.11 or Windows 95, with the following system specifications:

- 80486 DX2 processor or higher
- 4 Mbytes of RAM
- a 8.9 cm floppy drive
- a mouse

6.5.1.14 Pedigree of model with UK and EU groups

The development of PC CREAM was carried out by NRPB under contract to the European Commission, Directorate-General Environment, Nuclear Safety and Civil Protection, DGXI. A recent PC CREAM users group meeting took place at the NRPB, from the 3rd to the 4th of December 1998. At this meeting an overview of development and distribution of PC CREAM was presented. Observations on distribution were:

- Sold 91 copies to: government departments and agencies, regulatory organisations, research institutes and organisations, nuclear and non-nuclear industries, ‘small users’, consultancies, environmental groups.
- UK 29% of sales, rest of EU 40%, rest of Europe 9%, rest of world 22%.

6.5.1.15 Licence costs

PC CREAM single user licence copy is €713 including VAT. This also includes a copy of Radiation Protection 72, Report EUR 15760.
POSEIDON

6.5.2 POSEIDON

6.5.2.1 Background

POSEIDON is a PC based MS Windows box model used to assess the radiological consequences of a continuous or pulse discharge of radioactive materials in European sea waters. It can be run on medium and long-term scales and there is a large scale geographic area of dispersion.

6.5.2.2 Important processes included in model

POSEIDON is a box model which takes into account radionuclide leaching and sediment exchange processes, where leaching is considered to be the continuous surface erosion of the immersed material during a user defined time period. Sedimentation processes modelled consist of depletion of suspended material in equilibrium, diffusion of radioactivity between layers and bioturbation which is modelled as a diffusive process on the boundary layer. The sediment boundary layer is modelled as a three-layer system (upper, middle and deep sediment). Decay chain daughter products are considered by replication of the compartments system. The methodology used for these processes has been derived from EC projects such as MARINA (European Commission, 1990) and PC CREAM (European Commission, 1995).

Different release types are considered in the POSEIDON model, these are:

- Instantaneous (accidental release).
- Time Variable.
- Continuous leaching of an immersed solid material.

Dispersion is modelled by the exchange between compartments being modelled as an annual transferred volume of water. Biota concentrations are determined from seawater concentrations by use of CF values. This data combined with dose per unit intake values and consumption rates are used in dose calculations.

Both collective and individual doses can be calculated with doses being based on ingestion and external exposure. As well as using the critical groups supplied with the model it is also possible to set up user defined critical groups.

6.5.2.3 Performance, Validation and Limitations

POSEIDON V2.0 was used to compare ^{137}Cs predicted concentrations in the Irish Sea with measured concentrations based on discharges from the Sellafield and La Hague reprocessing plants. Calculated values covering the period 1967 to 1985 were found to be within the same range as the measured values.

6.5.2.4 QA History

The first version of this software was developed at CEPN for internal purposes in 1992-1994, in response to several requests from different French institutes asking for calculations of collective doses resulting from routine releases into the marine environment, as well as for ship transportation accidents in the Channel. Special features included:

- The possibility to consider multiple products of a given decay chain (especially for Pu decay chains);
- The possibility to consider routine releases and accidental releases, giving a time profile of releases as input to the programme.

The calculation code was designed to run on a PC. Validation of the results was performed in close interaction with the NRPB. Activity concentrations and collective doses were compared with the outputs from NRPB. A validation was also performed for the results on activity concentrations in water, making a comparison between measurement data in the Irish Sea and the North Sea and the predicted concentrations associated with the routine releases from La Hague and Sellafield reprocessing plants.

In 1997, a second version of the software was produced including new features:

- An individual dose module, to perform individual dose calculations for default or user-defined reference groups of population;
- The introduction of sedimentation modelling.

In 1998, the French Institute for Nuclear Protection and Safety (IPSN), which had partly supported the previous developments of POSEIDON, decided to choose this code for its own calculations of radiological impacts of routine/accidental releases into the marine environment. Since then, complementary developments have been performed to produce the latest third, version, including:

- The possibility to consider multiple radionuclides as input releases (a mixture of radionuclides);
- The possibility to enter a source term resulting from the leaching of a solid material immersed (for ship transportation accident purposes for example);
- Additional graphic displays (maps of contamination, etc.).

Since October 1998, an updated version of POSEIDON has been included in the European Real-time Online DecisiOn Support (RODOS) project.

6.5.2.5 Data Requirements

POSEIDON was designed to be adaptable to any compartment system. The description of the marine compartment system is computed into different text files and the user is free to create a new compartment system or to adapt the existing one to its own requirements (for example improving the spatial resolution in specific areas). The creation of a new compartment system is not a tool directly included in the software. This work must be performed independently of POSEIDON, assuming that the user has all the required information on all compartments (hydrological characteristics, water exchanges, sedimentation parameters, etc.).

Releases are entered as "profiles". This means that there is no restriction in the releases possibilities. For any release period, the user enters the duration and the release rate, without any limitation of the number of release periods.

The radionuclide database includes a list of about 90 radionuclides. Ingestion dose factors are those taken from ICRP 72 (ICRP, 1997).

"Observation times" - time periods for which the user asks for results on activity concentrations in the environment (water and sediments) and doses calculations (collective and individual) - are directly entered in years. No limitation is imposed on the number of observation times.

6.5.2.6 User Friendliness

The model is a MS Windows based system.

6.5.2.7 Hardware and Software Demands

POSEIDON version 3 is designed to run on a PC, MS Windows 95 environment. There is no special software requirement. The software interface was developed with Visual Basic. Calculation routines were computed in Fortran, in a special library, which is provided in the software package.

6.5.2.8 Pedigree of model with UK and EU groups

Used and developed by CEPN (Centre d'Étude pour la Protection Nucléaire). As mentioned above, POSEIDON was chosen by the French IPSN for its own calculations. POSEIDON was also selected to be included into the European RODOS project.

POSEIDON was recently used in France to calculate collective and individual doses resulting from the routine releases of La Hague, Marcoule and Romans nuclear plants (request from COGEMA). Calculations of accidental releases were performed for ship transportation of vitrified wastes to Japan (request from COGEMA). Calculations were also made for an accidental release of PuO_2 into the Channel (request from IPSN).

6.5.2.9 License costs

POSEIDON single user licence copy is about €400 excluding VAT.

6.5.3 MIKE21

6.5.3.1 Background

MIKE21 is a modular 2D modelling system for free surface flows. It is a widely used tool for hydraulic modelling in lakes, estuaries, coastal waters and seas. MIKE21 is currently used at Westlakes Scientific Consulting Ltd for modelling discharges to the Irish Sea for companies such as BNFL, North-west Water and the Environment Agency. These studies have only covered both short-term and short-range discharge studies.

6.5.3.2 Key Processes included in the model

The modular make up of the model allows different components to be purchased separately for different tasks.

6.5.3.3 The Hydro-Dynamic module

Central to the modelling system is the Hydro-Dynamic (HD) module, which calculates the basic hydrodynamics that are then used as a basis for computations in additional modules.

This module generates water levels and fluxes (velocities) across a rectangular grid covering the area of interest. The system solves the full time-dependent non-linear equations of continuity and conservation of momentum. The solution is obtained using a finite difference scheme.

6.5.3.4 The Advection-Dispersion module

The Advection-Dispersion (AD) module is necessary for modelling transport processes. It simulates the spreading of dissolved or suspended material in the aquatic environment. The concentration of the substance is calculated at every point of a rectangular grid covering the area of interest. The AD module takes hydraulic output from the HD module as the basis of its calculations. Additional information is needed regarding the discharge source and boundary conditions such as decay rates and background concentrations. First order linear decay is important for radionuclides over long time-scales.

6.5.3.5 The Heavy Metals Module

There are many additional modules that can be added onto a MIKE21 system for specific purposes. These include modules for mud and sand transport, water quality, eutrophication and waves. Of particular interest to this study is the heavy metal (ME) module that will enhance the modelling of the absorption and desorption of the metals from sediments. Specific process represented are:

- Adsorption/desorption of metals to suspended matter.
- Sedimentation of sorbed metals to the bed.
- Resuspension of settled metal.
- Exchange of metal between bed sediment and interstitial water.
- Diffusive exchange between metal dissolved in water column and the interstitial water of the bed.

The module is designed as a general module but the various metals behave differently and the module therefore needs a number of metal specific parameters to achieve optimum performance for a particular metal.

6.5.3.6 Performance, Validation and limitations

The strength of this model is that the hydrodynamics are represented explicitly, thus making it an accurate and realistic representation of these processes, provided that a full calibration and validation against measured data is performed. The heavy metals module will also allow processes relating to sediment interactions to be represented realistically.

The model is potentially very realistic, but there are still some inherent weaknesses in applying it to such a large scale project that are important to consider. Although setting up the relevant bathymetry and initial boundary conditions is feasible, the amount and quality of calibration data required for such a large dynamic area may be difficult. Running the model is computationally expensive. For example, the computational time required by the hydrodynamic simulation depends on the region covered by the model, number of time steps, specified simulation features and the general computer speed. Thus, the duration of a simulation over a large area and a long time period could make the use of this model impractical.

6.5.3.7 QA history

Professor Mike Abbot created the MIKE21 model suite for the Danish Hydraulic Institute. Further developments have been made by DHI. The latest version of the MIKE21 software is MIKE 1999. This is a fully integrated MS Windows programme allowing the user access to data display and modification through a typical MS Windows system.

6.5.3.8 Data requirements

Large quantities of data are required to use this model and hence a PC capable of storing these data is necessary. The inputs to the hydrodynamic module are important for correct calibration and validation of the model. Typical boundary conditions of sea surface elevations are obtainable from the POL tidal prediction model. These data can also be obtained for points within the model area for calibration and validation.

6.5.3.9 User friendliness

The latest version of MIKE21 is an MS Windows based package, which makes use of the model very easy.

6.5.3.10 Hardware and Software demands

Since MIKE21 is a PC based package, it would be advisable to run the model on the most powerful PC available because the duration of the simulations is dependent on the computational speed of the machine.

6.5.3.11 Pedigree of model with UK and EU groups

MIKE21 is a widely used engineering tool in the UK, Europe and the world. Some international projects undertaken using this modelling suite are:

Bahrain	GPIC Marine Works, Hydraulic Model Study, Bahrain.
Bangladesh	Flood Management Model, Bangladesh (1994).
Brazil	The Iguaçu Upper River Basin.
Denmark	Environmental Impact Assessment of a Fixed Link between Denmark and Sweden (1994 - 1997).
	Environmental Management Support System, Denmark (1995-1996).
	Modelling of the Eutrophication Effects of Sewage Discharges to Øresund, The Sound between Denmark and Sweden (1995-1996).
Europe	Dynamics of Connecting Seas (DYNOCS).
France	Loire River and Estuary, France.
Indonesia	Eutrophication Modelling of a Tidal Influenced Mangrove Area in Bali
Norway	Environmental Conditions (1993-97).
	Environmental Design Criteria (1991-92).

6.5.3.12 License costs

The purchase price of the modules is:

MIKE21-Pre/Post Processor	€5,150
MIKE21-Hydrodynamics	€21,600
MIKE21-Advection Dispersion	€11,100
MIKE21-Heavy Metals	€4,120

6.5.4 DELFT MODEL

6.5.4.1 Background

The *DELFT3D* system is a flexible modular modelling system capable of simulating:

- Flows due to tide, wind, density gradients and wave induced currents
- Advection dispersion of effluents
- Water quality

It can also switch between the 2D vertically averaged mode and 3D mode by simply changing the number of layers. The modules of interest for this project are FLOW, WAQ and SED. Further information on each module and typical applications are given below.

6.5.4.2 FLOW module

The hydrodynamic model is a multi-dimensional hydrodynamic simulation programme that calculates non-steady and transport phenomena resulting from tidal and meteorological forcing. The module is based on the full Navier-Stokes equations with the shallow water approximation applied. The equations are solved with a highly accurate unconditionally stable solution procedure. The supported features are:

- Three co-ordinate systems, i.e. rectilinear, curvilinear and spherical in the horizontal directions and a sigma co-ordinate transformation in the vertical;
- Simulation of drying and flooding of intertidal flats (moving boundaries);
- Coriolis force and (optionally) tide generating forces;
- Density gradients due to a non-uniform temperature and salinity concentration distribution;
- Inclusion of density (pressure) gradients terms in the momentum equation (density driven flows);
- Turbulence model to account for the vertical turbulent viscosity and diffusivity based on the eddy viscosity concept;
- Shear stresses exerted on the bottom, based on a quadratic Chézy or Manning's formula;
- Simulation of the thermal discharge, effluent discharge and the intake of cooling water at any location and any depth in the computational field (advection-diffusion module).
- On-line analysis of model parameters in terms of Fourier amplitudes and phases enabling the generation of co-tidal maps;
- On-line visualisation of model parameters enabling the production of animations (in preparation).

DELFT3D-FLOW can be applied to the following application areas:

- Salt intrusion in estuaries;
- Fresh water river discharges in bays;
- Thermal stratification in lakes and seas;
- Cooling water intakes and waste water outlets;
- Transport of dissolved material and pollutants;
- Storm surges;
- River flows;
- Wave driven flows.

The results of the hydrodynamic module are used in all other modules of DELFT3D. The results are dynamically exchanged between the modules through the use of a communication file. Basic (conservative) water quality parameters, such as concentrations of dissolved material and pollutants, can be included in the computations. For more dedicated water quality simulations, the hydrodynamic module is coupled with the far-field water quality module (DELFT3D-WAQ) and the near-field particle tracking module (DELFT3D-PART). A coupling with the sediment transport module (DELFT3D-SED) is available to simulate cohesive and non-cohesive sediment transport processes, e.g. in the case of erosion and sedimentation studies. For wave-current interaction a dynamic coupling is provided with the wave module (DELFT3D-WAVE) and for morphodynamic simulations the hydrodynamic module is integrated with the wave module and a sedimentation and erosion module into a morphodynamic module (DELFT3D-MOR).

6.5.4.3 WAQ module

The transport of substances in surface and ground water is commonly represented by the so-called advection-diffusion equation. The water quality module, DELFT3D-WAQ, is based on this equation and it offers different computational methods to solve it numerically (in one, two or three dimensions) on an arbitrary irregular shaped grid, on a grid of rectangles, triangles or curvilinear computational elements. In order to model waste loads and water quality processes the advection-diffusion equation is extended with an extensive water quality library of source/sink terms. The model is capable of describing any combination of constituents and is not limited with respect to the number and complexity of the water quality processes.

The water quality processes may be described by arbitrary linear or non-linear functions of the selected state variables and model parameters. For many water quality problems, these process formulations have been standardised in the form of a library, which smoothly interfaces with the water quality module. The library contains over 50 water quality process routines covering 140 standard substances. A graphical user interface within the WAQ module enables the user to select substances and associated water quality processes.

In many cases the water quality processes in the model are determined by meteorological conditions, by other (modelled or non-modelled) constituents or by other (modelled or non-modelled) processes. Examples are wind, water temperature, acidity (pH), primary production and the benthic release of nutrients. These entities are referred to as "forcing functions". Water quality process formulations are often of an empirical or semi-empirical nature and contain "model parameters" that are subject to tuning or calibration. Because of this, DELFT3D-WAQ allows complete freedom in selecting the set of water quality processes and the relevant forcing functions and model parameters may vary between individual applications. It therefore provides flexible input facilities for constants, spatially varying parameters, functions of time and functions of space and time.

Water quality processes are incorporated in the advection diffusion equation by adding an additional source in the mass balance. Examples of water quality processes are:

- Exchange of substances with the atmosphere (oxygen, volatile organic substances, temperature);
- Adsorption and desorption of toxicants and ortho-phosporous;
- Deposition of particles and adsorbed substances to the bed;
- Re-suspension of particles and adsorbed substances from the bed;
- The mortality of bacteria;
- Biochemical reactions like the decay of BOD and nitrification;
- Growth of algae (primary production);
- Predation (e.g. zooplankton on phytoplankton).

DELFT3D-WAQ can be applied to the following application areas:

- Bacterial decay processes;
- Chemical processes;
- Nutrient cycling and eutrophication processes;
- Sedimentation and resuspension of particulates;
- Interaction between water and bottom (including diffusive and benthic mixing);
- Evaporation, re-aeration and other surface processes;
- Transport and chemical processes regarding heavy metals and organic micropollutants;
- Recirculation of cooling water.

6.5.4.4 SED module

The sediment transport module, DELFT3D-SED, can be applied to model the transport of cohesive and non-cohesive sediments, i.e. to study the spreading of dredged materials, to study sedimentation/erosion patterns, to carry out water quality and ecology studies where sediment is the dominant factor.

It is in fact a sub-module of the water quality module. For a detailed description of the general aspects refer to the description of the water quality module. DELFT3D-SED can be applied to the following areas:

- Effects of dredging on the environment;
- Sedimentation and resuspension of sediment in general;
- Sand transport.

6.5.4.5 Performance, Validation and limitations

The DELFT3D model has been used to produce the "Northwest European Shelf Pilot Model" and "Screamotox". Both these models are publicly available, either through the internet or directly from Delft Hydraulics. It is known that the "Northwest European Shelf Pilot Model" has not been directly validated against measured data however it does show what is achievable using the Delft modelling system.

6.5.4.6 QA history

No information was available on the quality assurance of the model.

6.5.4.7 Data requirements

It is reasonable to assume that a substantial amount of input data will be required.

6.5.4.8 User friendliness

All features are embedded in a state of the art graphical user interface based on the MS Windows (Wintel-Platforms) standards. An application can be completely defined through this menu driven user friendly graphical interface. The system is also embedded in a project and scenario management tool which allows the definition, selection and archiving of simulations by reference to project or scenario names.

6.5.4.9 Hardware and Software demands

DELFT3D and its accompanying programmes ARE supported on UNIX workstations (HP, SUN and SG) and PC's (Windows 95 and Windows NT). The minimal and preferred configurations are as follows:

Configuration Item	**Minimal**	**Preferred**
Processor Speed	166 MHz	333 MHz or more
Internal Memory	64 Mb	128 Mb
Swap space	1.5 * internal memory	2.5 * internal memory
Hard disk	2 Gb	10 Gb
Monitor	43 cm colour	48 cm colour
CD-ROM	Standard	Standard
Printer	PCL, HPCL	Postscript

6.5.4.10 Pedigree of model with UK and EU groups

The "Northwest European Shelf Pilot Model" is a DELFT3D application. This model has been one subject of modelling comparison studies at three workshops organised by the Assessment and Modelling Group (ASMO) of OSPARCOM.

6.5.4.11 License costs

The approximate cost in for the FLOW, WAQ and SED modules, FLOW model add-ons and Pre and Post processing module is €95,000. However it is also possible to obtain dedicated versions of the model. The dedicated version is a restricted use of the system to a single or restricted number of predefined models with fixed grid dimensions and/or geometry. It may be used in models designed for a specific project. A discount of 75% on the list price is granted for a standard dedicated version licence for the required programme package. The price for a specific dedicated version depended on the specification of the project and the involvement of Delft Hydraulics in the project and is decided on a project to project basis.

6.6 The Global Marine Database

The IAEA-MEL (Internaional Atomic Energy Agency's Marine Environment Laboratory) has been acting as a central facility for the collection, synthesis and interpretation of data on marine radioactivity in the world ocean. Ultimately, all available data on radioactivity in seawater, sediments and biota will be stored in the IAEA-MEL Global Marine Radioactivity Database (GLOMARD) which is under development. The database is designed to serve as follows:

- To provide immediate and up-to-date information on radionuclide levels in the seas and oceans.
- To provide a snapshot of activities at any time in any location.
- To investigate changes with time in radionuclide levels.
- To identify gaps in available information.

To meet these objectives and ensure maximum utility of the information contained in the database, the data format has been rigorously prescribed. The degree of detail is extensive (general sample information including type, method of collection and location as well as physical and chemical treatment), to allow the data to be validated and its quality assured. In addition, the database has links to IAEA-MEL's in-house analytical quality control database allowing immediate checks on laboratory practice. Information is stored in such a way as to facilitate data interrogation and analysis. The data base provides critical evidence in the evaluation of the environmental radionuclide levels of the region and in assessment of the radiation doses to marine biota, and local, regional and global human populations. It enables users to perform:

- Evaluation of nuclide ratios - the identification of the different contributions to radioactivity in a given region is critical, given the multiple nature of source-terms, e.g. global fallout, Chernobyl fallout, releases from nuclear reprocessing plants and leakage from dumped radioactive wastes.
- Investigation of time trends - given the temporally varying nature of the known sources to the region, the database helps to estimate their contribution to the environmental concentrations and thus increase the sensitivity with which any small residual change or trend may be detected.
- Inventory calculations - the ability to carry out budget calculation may again permit detection of any imbalances.
- Dose estimation - combining the environmental levels with a pathway model allows dose estimation (to first order) of the dose to marine biota and local and regional populations.
- Model validation - to provide reliable predictions of the impact of real or theoretical discharges, it is necessary to use validated models and this requires access to the existing

appropriate experimental data (either in the form of time series of observation or a snap-shot of activities).

More than 33 MB of data from 19 seas have already been entered, with initial emphasis on IAEA-MEL's own measurements during over 30 years of operation. IAEA-MEL are further actively searching for data and would be glad to have further contributions from all marine laboratories (http://www.iaea.org/monaco/glomard.html).

GLOMARD is still in a development stage and it is planned to have some data available for the public via the internet by the end of 2002. The area of greatest coverage for data is the NW Pacific and NE Atlantic. Data has also been provided on CD-ROM for the Marina II project (P. Povinec, personal communication, 2002).

An Asia/Pacific marine radioactivity database (ASPAMARD) has also been compiled. The database uses to the fullest extent practical, the reporting format designed by IAEA/MEL for its Global Marine Radioactivity Database (GLOMARD). The database contains data particularly on Cs-137 and Pu-239+240 in seawater, sediment and biota. These two radionuclides are deemed important because they are significant indicators of radioactivity pollution of the marine environment for purposes of dose assessment and are good tracers of water movement. In addition, available data on other radionuclides in seawater, sediment and biota were also included in the compilation. This database will be useful for assessing the short-term and long-term impact of man-made sources of marine radioactivity for the region and will be available as one of the database with the IAEA/RCA programmes (http://www.rca.iaea.org/regional/html/projects/marine.htm).

ASPAMARD has been developed under the Regional Co-operative Agreement (RCA) for Research, Development and Training Related to Nuclear Science and Technology, 1997.

6.7 Discussion and recommendations

There are essentially two issues to be considered in order to select a model for prediction of environmental radioactivity concentrations. Firstly, whether a box model or a numerical model would be more suitable, then secondly, choose a suitable model within this specified model type. The suitability of the model will depend on the size of area and time interval (*e.g.* daily or annual output) required. The detailed review of the features of all the models considered should be helpful in order to provide an initial structured comparison.

Box models are specifically designed for the task of long-range and duration radionuclide transport, and for dose assessment. On the other hand, numerical models have a greater accuracy. The fundamental question is whether the potential benefits of better process representation inherent to numerical models overcome effort and cost, keeping in mind the FASSET tenet of avoiding academic overkill.

The box models, PC CREAM and POSEIDON, are both simple, quick, easy to use and ready-for-purpose models of radionuclide transport. The concerns are over the simplicity of the representation of transport processes. The transport dynamics between the boxes has been developed carefully and should ensure a reasonable representation of the processes. The nuclide processes include a broad range of interactions with sediment and biological activity.

In order to determine whether a numerical model will better this, their problems and potential benefits need reviewing. The problems of applying a numerical model to a long-range and long-term study are severe and have been elucidated in the text previously. Generally, these involve the setting-up of the model over such a large area including boundary and initial conditions, calibration and validation, computational time and, last but perhaps not least, the expense of buying such a system. Another problem is that these models are not necessarily designed for radiological assessments. In order to carry out a radiological assessment a post-processor would have to be developed to evaluate biota concentrations and calculate doses. Many box models have an in-built post-processing for dose to humans and coding changes will be required to interface models with routines capable of assessing exposure to biota.

The main benefit of using a numerical model is the improved accuracy of prediction. Whether this potential benefit would in fact be realised is not so clear. In order to obtain the benefits the model must be set up very well and applied very carefully. Specifically to this study, points of concern arise, firstly, that these models would perhaps be applied beyond their ideal scope both in terms of range and duration and, secondly, the vast amounts of data that would be needed to enable a detailed model calibration and validation. Both of these suggest that in this case the potential accuracy benefits would be reduced. This in turn leads to a potential problem of false confidence in the results. This can simply arise where a complex model is applied in less than ideal conditions and the predictions suffer. Because the model is state of the art, the predictions are thought to be better than they perhaps are. The simpler box model however, reduces the expectation on the results, particularly by the end user. This would appropriately reflect the uncertainty inherent in long-range modelling, a point that might be overlooked with numerical models.

In the light of these points it would appear that box models provide a more suitable approach to modelling long-term and long-range transport of radionuclides, whilst numerical models may be best applied to short-term releases and local impacts.

Although model predictions are a useful tool in the determination of potential doses to biota and critical groups the availability of measured data should not be over looked. Databases such as GLOMARD may prove to be of benefit provided monitoring data is supplied and that it is updated on a regular basis.

By its very nature, this review is limited in that it is not possible to know all the possible models that are potentially available to generate concentrations of radionuclides in the aquatic environment. Westlakes' experience is strong in the area of marine modelling, particularly when applied to coastal regions of the Irish Sea. Perhaps in future this review can be turned into an indexed database of models with contributions from other FASSET participants, a potentially useful resource setting options for future environmental assessments.

7 Transfer to marine biota under equilibrium conditions

7.1 Introduction

Concentrations factors[8] have been widely used in modelling the transfer of radionuclides from the water column to biota and numerous reviews and summaries of the available literature have been made (Harrison, 1986; Gomez *et al.* 1991). Probably the most widely–used concentration factor values, in the fulfilment of human dose assessments, are those reported by the International Atomic Energy Agency in Techdoc. 247 (IAEA, 1985) and the updated version of this document (IAEA, in press). Selected data from these publications corresponding to the radionuclides chosen for further analysis within the FASSET project (with the exception of ^{40}K), are shown in Table 7-1.

Table 7-1 Concentration factors for generic marine organisms (IAEA in press)

Element	**Phytoplankton**	**Macroalgae**	**Zooplankton**	**Mollusca***	**Crustaceans**	**Fish**
Cs	2 x 10^1	5x10^1	**4x10^1**	**6x10^1**	**5x10^1**	1x10^2
Tc	**4 x 10^0**	**3 x 10^4**	1 x 10^2	**5 x 10^2**	1 x 10^3	**8 x 10^1**
Sr	**1 x 10^0**	**1 x 10^1**	**2 x 10^0**	**1 x 10^1**	**5 x 10^0**	**3 x 10^0**
U	2 x 10^1	1 x 10^2	**3 x 10^1**	3 x 10^1	1 x 10^1	1 x 10^0
Th	**4 x 10^5**	2 x 10^2	1 x 10^4	1 x 10^3	1 x 10^3	6 x 10^2
Pu	**2 x 10^5**	**4 x 10^3**	**4 x 10^3**	3 x 10^3	**2 x 10^2**	**1 x 10^2**
Am	2 x 10^5	8 x 10^3	**4 x 10^3**	**1 x 10^3**	**4 x 10^2**	**1 x 10^2**
Cm	**2 x 10^5**	**5 x 10^3**	***4 x 10^3***	**1 x 10^3**	***4 x 10^2***	**1 x 10^2**
Np	1 x 10^2	5 x 10^1	***4 x 10^2***	4 x 10^2	***1 x 10^2***	**1 x 10^0**
Ra	2 x 10^3	1 x 10^2	1 x 10^2	**1x10^2**	1 x 10^2	**1 x 10^2**
Pb	**1 x 10^5**	1 x 10^3	1 x 10^3	**5 x 10^4**	**9 x 10^4**	2 x 10^2
Po	**7 x 10^4**	1 x 10^3	3 x 10^4	**2 x 10^4**	**2 x 10^4**	2 x 10^3
C	9 x 10^3	1 x 10^4	2 x 10^4	2 x 10^4	2 x 10^4	2 x 10^4
H	1 x 10^0	1 x 10^0	1 x 10^0	1 x 10^0	1 x 10^0	1 x 10^0
Nb	1 x 10^3	3 x 10^3	*2 x 10^4*	1 x 10^3	2 x 10^2	3 x 10^1
Ni	3 x 10^3	2 x 10^3	1 x 10^3	2 x 10^3	1 x 10^3	1 x 10^3
Ru	2 x 10^5	2 x 10^3	3 x 10^4	**5 x 10^2**	1 x 10^2	2 x 10^0
I	**8 x 10^2**	**1 x 10^4**	3 x 10^3	1 x 10^1	**3 x 10^0**	**9 x 10^0**
Cl	1 x 10^0	5 x 10^{-2}	1 x 10^0	5 x 10^{-2}	**6 x 10^{-2}**	**6 x 10^{-2}**

Values in bold indicate those that have been updated from IAEA (1985).
Italicized = best estimates
*excluding cephalopods

The CF approach has the advantage of being simple and provides the assessor with a large and easily-accessible data-base. It is therefore a useful starting point for our assessment of transfer and uptake of radionuclides within any marine environmental impact assessment.

The updated version of IAEA Techdoc 247 (IAEA, in press) also provides information on uptake to marine mammals for selected radionuclides/elements as shown in Table 7-2.

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[8] The concentration factor (CF) is usually defined as the ratio of the concentration of the radionuclide in the organism or tissue to that in the ambient seawater.

Table 7-2 Data on marine mammals from IAEA (in press).

Element	Pinniped[9] muscle	Pinniped liver	Polar Bear muscle	Polar Bear liver	Cetacean[10] muscle	Cetacean Liver
Cs	4 x 10^{2}	3 x 10^{2}	1 x 10^{2}	n.a.	3 x 10^{2}	n.a.
Ni	n.a.	n.a.	n.a.	n.a.	< 2 x10^{3}	n.a.
Pb	3 x 10^{3}	1 x 10^{5}	n.a.	n.a.	4 x 10^{4}	6 x 10^{4}
Pu	n.a.	8 x 10^{0}	7 x 10^{1}	n.a.	n.a.	3 x 10^{0}

n.a. – not available

7.2 Applicability of CF data to transfer-uptake assessments for non-human biota

Although the generic organism groups considered in IAEA (1985) and the updated version of this document are similar, and in some cases identical, to those selected as reference organisms within FASSET (Table 7-3), the applicability of these data to the present work is partly limited.

Table 7-3 : (candidate) reference organisms selected in FASSET (see Strand *et al.*, 2001)

Bacteria	**Crustacean**	**Mammal**
Worm	**Bivalve Mollusc**	**Wading bird**
Vascular plant	**Benthic fish**	**Phytoplankton**
Macroalgae	**Pelagic Fish**	**Zooplankton**

In view of the fact that the intended use of CF data would be in human dose assessments, the approach adopted in IAEA (1985) and the updated version of this document involved the collation of data for organism forming parts of food-chains leading to man, i.e. edible plants and animals. Furthermore, to the extent possible with the available data, the information was reported for the edible body parts of these edible organisms. Clearly, a question of data compatibility exists here. Within marine environmental impact assessments, non-edible organisms forming parts of food-chains that have no connection with man should be given equal consideration to those dealt with in human dose-assessment. It is also clearly of importance to consider not only those parts of an organism eaten by man but also those body parts that might be of interest from a dosimetric or dose-effects perspective for the organism *per se*. Such organs/body parts might include, where relevant, the hepatic system (where high accumulation of heavy metal contaminants can occur) and gonads (important from the perspective of fertility).

In view of these limitations, a data collation exercise was conducted at the NRPA in order to derive information that would be of use in an environmental impact assessment. The data collated in the following section of the report are intended to provide a substantial supplement to the more generic values provided in IAEA (in press).

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[9] Pinniped - any of various families of aquatic, fin-footed mammals consisting of the families: Odobenidae (walrus) and Phocidae and Otariidae (seal).

[10] Cetacaean - any member of the entirely aquatic group of mammals commonly known as whales, dolphins, and porpoises.

7.3 Concentration factors

The concentration factor method assumes that the organism is in biochemical equilibrium with its surroundings. The time required for equilibrium to be attained depends on the half-life of the radionuclide and the biological half-life of the element in the organism (Till and Meyer, 1983). The physicochemical form of the element and its route of entry into the organism are among factors that affect CF value. Radionuclides may exist in different physicochemical forms with a distribution that varies according to the radionuclide and the features of the ecosystem under consideration. Environmental factors, including temperature, light (in the case of algae), salinity and pH affect the growth and metabolism of organisms, and consequently the uptake of radionuclides (Meinhold and Hamilton, 1991).

7.3.1 Scope of CF data collation exercise

Owing to time constraints certain limiting search criteria were applied.

(i) Publications older than 1984 were not included in this report based on the assumption that these data were considered in a comprehensive manner in IAEA (1985). With extended time and resources it would be fruitful to revisit the data compiled in IAEA (1985), and other historical data-sets, and reconfigure them in a form more suitable for biota impact assessment.
(ii) The focus of this study has been on concentration factor data pertaining to European marine ecosystems[11] in line with the objectives of the FASSET project.
(iii) Potassium-40 has not been considered, owing to the fact that a scenario whereby this radionuclide would be released at rates high enough to significantly augment the activity concentrations above those observed under natural conditions could not be envisaged.
(iv) Benthic bacteria were not considered in terms of CFs. External exposure from contaminated sediments will completely dominate the dose rate to these organisms.
(v) Only radionuclide data have been collated. A means of augmenting the data set would involve the collation of CFs derived from the simultaneous measurement of stable isotopes in biota and seawater.

7.3.2 Organization of the collected data

Data compiled from the literature were organized into subsets of interest using Excel spreadsheets. These compiled data were first divided geographically into European and non-European subsets and then each divided further methodologically into: field data and data from laboratory experiments. Data subsets were then analysed statistically to produce ranges, medians and means.

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[11] For definition of European marine system boundaries, FASSET Deliverable 1 (Strand *et al.*, 2001; Appendix 2) can be consulted.

Table 7-4 shows categories of marine organisms (along with their subdivisions) for which data have been found. Subdivisions of the original reference organism categories (Table 7-3) were created to accommodate reported differences in uptake between orders or families of organism under each category. In the case of crustaceans, reported differences in the uptake of Tc-99 prompted the inclusion of subdivisions: lobster and crab. Differences in uptake of naturally-occurring Po-210 prompted the creation of the subdivision: prawn and isopod.

Table 7-4 Organism categories considered in the present study.

Categories	Subdivisions
Microalgae	None
Zooplankton	None
Macroalgae	Chlorophyta (Green) Phaeophyta (Brown) Rhodophyta (Red) Unspecified
Molluscs	Bivalve Gastropods Cephalopod
Crustaceans	Prawn Shrimp Crab Lobster Isopod
Fish	None
Mammals	None
Worms	None
Seabirds	None

In the original reference organism list, a differentiation was made between benthic and pelagic fish owing to the fact that animals included under these 2 categories might be expected to be exposed to quite dissimilar external dose-rates. In contrast to this, there was no strong evidence to suggest that pelagic and benthic fish would exhibit significantly different CFs and therefore the combination of these 2 groups into the single category “fish” (within a single Excel spreadsheet) was considered to be sensible. It should be noted that the nature of the data-base allows these 2 categories to be differentiated if deemed necessary. The following outline presents the general compilation scheme used in organizing the collected data.

7.3.3 Categorisation of reported studies (Excel spreadsheets):

An attempt has been made to document parameters which are generally addressed in most of the publications. The following factors were registered, if available, for both field and experimental studies:

- Organism category
- Tissue or organ analysed
- Habitat
- Sampling location
- Number of samples
- Time of measurement
- Location of measurement
- Type of water: filtered or unfiltered

The inclusion of the last entry is due to the fact that the type of water, filtered or unfiltered, used in studies can be a major source of variance, particularly for coastal waters. In addition to these categories, CFs reported from laboratory experiments also include:

- Time of exposure in days
- Exposure concentration
- Biological half-life, if available
- Temperature condition

The last column in the database is entitled comments and contains noted special conditions such as:
- Data reported as mean or as range rather than as individual observations
- Data estimated rather than directly measured
- Any special remarks, insights, conclusions, warnings, etc. cited
- Equilibrium condition, whenever information was available, particularly for laboratory data

The tissue/organ categories used in cataloguing the data were:
1. Crustaceans and bivalve: Whole, flesh, shell, hepatopancreas, gill, and other
2. Gastropods: whole, flesh, shell, digestive gland (viscera)
3. Cephalopods: whole, muscle, hepatopancreas
4. Fish: whole, muscle/flesh, bone, liver, gonads, and other.

The excel spreadsheets are not included as part of this report. These data have been used in the production of summary tables as described below.

7.4 General remarks

The amount of information/details reported in each study differs according to the objective(s) of that study. Consequently, in order to reduce this diversity and, at the same time, produce a reasonable database, the following conventions were adopted:

- For data reported as a range, the mean of the reported upper and lower bounds was given. In these cases the number of observations was chosen to be two (the lowest possible number).
- For data reported as less or larger than, the recorded limit was chosen.
- Seawater was used by default whenever the type of water was not mentioned.

- The number of observations was set equal to one whenever information on sample numbers was not provided.
- Whole body was used by default whenever the tissue or organ was not clearly specified.
- Concentration expressed as dry weights, e.g. Masson *et al.* (1995), were converted to wet weight concentration by multiplying the given value by 0.2 as recommended by IAEA (1985).

7.5 Table of summaries

In order to analyse the compiled data, CF values have been pooled together despite their inevitable diversity. In an attempt to reflect (different aspects of the collected data) this variety three different averages have been calculated: Median, arithmetic and weighted mean. These along with other parameters are presented in Tables 7-6 to 7-19. The following section presents explanatory information associated with the calculated averages as well as other table parameters.

N
Total number of data entries, normally based on means from individual articles/publications, that have been used to calculate means and medians. This value dose not reflect the total number of observations from each of the considered studies but instead the number of reports/articles/publications.

n
This parameter represents the total number of observations (many articles derived CFs for a given organism based on multiple measurements). Unfortunately, this value was among those least reported. Its deduction was not straightforward and often use was made of one or more of the conventions listed above. This value has been used to calculate the weighted means, M_{weight}[*].

Weighted mean
Using N to calculate medians and means can mask the scale of different investigations. In order to overcome this shortcoming the weighted mean was calculated, whenever possible.

Areas
As indicated earlier, the objective of this review was to collect concentration factor data for European marine. In the column entitled 'areas' code numbers have been used to represent the areas where the corresponding investigation has been undertaken. Although the Baltic Sea has been categorized as brackish waters by FASSET, some data pertaining to this area has been included in the present study. Numbers stated correspond to the areas:

1. Black Sea
2. Mediterranean Sea (including Adriatic Sea)
3. North-east Atlantic

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[*] $M_{weight} = \sum ((n_i * (CF)_i)/n)$; where n_i = number of observations in article "i"; CF_i = representative (mean)CF derived from article "i"; and $n = \sum n_i$ i.e. the total number of observations summed over all articles.

4. English Channel
5. Irish Sea
6. North Sea
7. Norwegian Sea
8. Barents Sea
9. Greenland Sea
10. Kattegat and Skagerrak
11. Baltic Sea

7.5.1 Paucity of data

Table 7-5 is an attempt to illustrate the scarcity of data. The grey boxes represent the presence of one or more data entries and the blank boxes the absence of data for the given radionuclide-biota combination. Whilst the availability of data indicates that some form of CF can be derived for use in an environmental impact assessment, the limitations on the application of the associated CF are not specified. Such limitations may result from the extrapolation of laboratory derived data to field conditions or on a lack of information for specific body parts or organs that may be of interest/concern form a dosimetric perspective. Few data were found for radionuclides such as I, Ru, Ra, Np, Cm and U no data were found for Th, C, H, Nb, Ni and Cl. No data for vascular plants were found. Marine birds, polycheate worms and mammals are particulary poorly characterised using CF datasets.

Table 7-5: Availability of data in the present study and updated TRS247 (IAEA, in press)

Elem.	**Mac. Al**	**Mollusc**	**Crusta.**	**Fish**[a]	**Zoopl.**	**Phytopl.**	**Worm**	**Mamm.**	**Bird**
Cs	▨	▨	▨	▨	▨	▨		▨	▨
Tc	▨	▨	▨	▨	▨	▨			
Sr	▨	▨	▨	▨	▨	▨		▨	
U	▨	▨	▨	▨	▨	▨			
Th	▨	▨	▨	▨	▨	▨			
Pu	▨	▨	▨	▨	▨	▨		▨	
Am	▨	▨	▨	▨	▨	▨			
Cm	▨	▨	▨	▨	▨	▨			
Np	▨	▨	▨	▨	▨	▨	▨		
Ra	▨	▨	▨	▨	▨	▨			
Pb	▨	▨	▨	▨	▨	▨		▨	
Po	▨	▨	▨	▨	▨	▨	▨		
C	▨	▨	▨	▨	▨	▨			
H	▨	▨	▨	▨	▨	▨			
Nb	▨	▨	▨	▨	▨	▨			
Ni	▨	▨	▨	▨	▨	▨		▨	
Ru	▨	▨	▨	▨	▨	▨			
I	▨	▨	▨	▨	▨	▨			
Cl	▨	▨	▨	▨	▨	▨			

[a] Includes both benthic and pelagic fish as reported in the main body of text.

Due to the interplay of many factors, both biological and environmental, variance of CFs is great within and between studies. Sazykina (1998) found that the values of Cs-137 CF for fish (Barents Sea, cod) were not constant, but gradually changed from 28±5 in 1979 up to 182±48 in 1992. This study clearly illustrates what is usually a major source of variation in many studies; lack of equilibrium within the period of observations. Consequently, caution must be used when an organism contamination is predicted by the radionuclide ‘concentration factor’ approach.

7.6 Points for discussion

A comparison between field data and those derived from laboratory studies shows that laboratory measurements tend to yield lower CF values than field measurements.

From data presented in Table 7-6, seabirds appear to accumulate comparatively high concentrations of Cs. This has been explained by the fact that the birds come into contact with seawater primarily for catching prey and cannot depurate themselves of accumulated contaminants through desorption in the way that other marine organisms can (Fisher *et al*., 1999).

Brown seaweeds have historically been used as a biological indicator of Tc-99 (Aarkrog *et al*. 1986 & 1987). Hurtgen *et al.* (1988) showed that, generally, apical fragments of *Fucus spiralis* possess a lower content of Tc-99 than the middle and basal ones. This indicated that the older parts of this brown alga accumulate more Tc than younger ones. Crustaceans display a wide range of CF values of Tc-99 with crab at the lower end and lobster at the higher end of this range. The highest CF value belongs to the lobster’s green gland, which has exhibited a CF as high as 65000 in the Irish Sea (Busby *et al*., 1997).

As it is clear from Table 3.8, the distribution of Po-210 is highly non-uniform with highest accumulations occurring in hepatic and digestive organs. Mussels display a higher affinity for Po-210 relative to winkles or prawns. This can be explained by considering their respective ecological niches and feeding habits (McDonald *et al*. 1993).

Table 7-6: Concentration factors for **Cs-137** in European marine organisms obtained from field studies.

Organism/Taxonomy/Tissue		N	Range	Median	Mean	n	$M_{weig.}$	Areas	References	Comments
Crustaceans	Whole	1	52 -			5		8	4	
Fish	Whole	15	15 - 189	51.5	72	72	65	1,2,5	11,28,32	
	Flesh	14	28 - 182	100	108.1	196	83.7	1,6,8	1,3,4	
Mammals	Seal, muscle	1	13 - 70		42	2		8	4	
	Harbour porpoise	4	100 - 600	350	350	25	366	3,5,6	11	
Seabird	Muscle	1	414 -			7		8	4	
	Liver	1	530 -			5		8	4	
Molluscs	Soft tissue	2	16 - 24		20	3	21.3	1,5	15,32	
	Muscle	2	6 - 15		10.5			5	15	
Gastropod	Whole	8	18 - 134	37.5	51.6			5,6	26	
	Shell	2	4 - 6		5			5	15	
	Digestive glands	2	20 - 55		37.5			5	15	
			-							
	Soft tissue	5	5 35	20	18.4	8	21.2	1,5	15,32	
	Muscle	2	2 - 4		3			5	15	
Bivalve	Whole	11	11 - 83	43	40	23	52	5,6,8	4,26	
	Shell	2	1 - 3		2			5	15	
	Gill	2	1 - 4		2.5			5	15	
Macroalgae	Brown	2	26 - 124		75.2	13	116.4	2,8	4,37	
	Red	3	35.7 - 80	51.5	55.7	4	54.6	1,2	32,37	
	Green	4	14.3 - 39.5	31.7	29.3	7	31.5	1,2	32,37	
	Unspecified	12	17 - 56	34.5	35.3			4,5,6	26	
	Brown	1	31.5 -					2	37	Cs-134
	Red	2	46 - 127		86			2	37	Cs-134
	Green	1	15.2 -					2	37	Cs-134

Table 7-7: Concentration factors for **Tc-99** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue		N	**Range**			**Median**	**Mean**	**n**	$M_{weig.}$	**Areas**	**References**	**Comments**
				-								
Fish	Edible parts	1	12					49		5	25	
	Whole	1	486	-						7	8	
Molluscs	Soft tissue	7	150	-	970	390	454	27	606	5	10,25	
	Brown	5	12200	-	36000	30000	26240	6	25500	4,6,7,10	8,31,33	
Macroalgae												
	Apical fragment	1	4000	-	20000		12000			6	33	Brown algae
	Middle fragment	1	10000	-	51000		30500			6	33	Brown algae
	Basal fragment	1	9000	-	46000		27500			6	33	Brown algae
Crustaceans												
	Whole	1	386	-						7	8	
Shrimp/prawn												
	Edible parts	1	2800	-						5	25	
					150	98	91.5					
	Muscle	4	20	-						5	10	
Crab	Hepatopancreas	1	160	-						5	10	
	Gills	1	70	-						5	10	
					8000	1085	2482					
	Muscle	8	40	-	7700		5000			5,7	8,10,25	
Lobster	Hepatopancreas	2	2300	-	1600		1400			5	10	
	Gills	2	1200	-	1600		1350			5	10	
	Shell	2	1100	-						5	10	
	Green glands	1	65000	-						5	10	

Table 7-8: Concentration factors for **Po-210** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue		**N**	**Range**	**Median**	**Mean**	**n**	**M$_{weig.}$**	**Areas**	**References**	**Comments**
Phytoplankton	Whole	4	2800 - 7400	4700	4900			11	24	
Zooplankton	Macrozoop. Whole	3	12400 - 25400	17000	18300			11	24	>303µm
	Mesozoop. Whole	17	8400 - 140800	35000	42000			11	24	>202µm
	Natural composition	23	600+ - 140800	30000	36500			3,11	23,24	+Jellyfish
Macroalgae	Seaweed	13	70 - 2585	890	1000			4,5,6	26	
Worm	Whole	3	9000 - 23000	17000	16330			11	22	
Fish++	Liver	1	300000 -					3	23	++Epipelagic
	Gonad	1	60000 -					3	23	
	Bone	1	30000 -					3	23	
	Muscle	1	6000 -					3	23	
Crustaceans Prawn	Cardiac fore-gut	1	26700 -					5	15	
	Hepatopancreas	1	22100 -					5	15	
	Gill	1	3800 -					5	15	
	Shell	2	1700 - 5000		3350			5	15	
	Abdom. Muscle	1	300 -					5	15	
Isopod	Whole$^{\lambda}$	3	24000 - 38000	28000	30000	20	28300	11	22	$^{\lambda}$Deca- and Amphipod
	Hepatopancreas	1	377000 -					11	22	
	Alimentary tract	1	24000 -					11	22	
	Gill	1	19000 -					11	22	
	Shell	1	14000 -					11	22	
	Muscle	1	8000 -					11	22	
Molluscs Gastropod	Whole	9	2410 - 31590	13780	13723			4,5,6	26	
	Digestive glands	1	29000 -					5	15	
	Pallial complex	1	9700 -					5	15	
	Total soft tissues	1	5500 -					5	15	
	Muscle	1	1700 -					5	15	

Table 7-8 (cont.): Concentration factors for **Po-210** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue			N	**Range**			**Median**	**Mean**	**n**	**M$_{weig.}$**	**Areas**	**References**	**Comments**
Molluscs													
	Bivalve	Whole	18	2200	-	74260	17500	25370			5,6,11	22,26	
		Soft tissues	34	3600	-	174000	46426	52380	235	58450	5,9,11	15,21,22,27	
		Hepatopancreas	1	66000	-						11	22	
		Viscera	1	58400	-						5	15	
		Alimentary tract	1	35000							11	22	
		Gills	2	13100	-	27000		20050	5	24220	5,11	15,22	
		Shell	8	3000	-	36000	9500	13000	11	10272	11	22	

Table 7-9: Concentration factors for **Pu-239+240** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue		**N**	**Range**	**Median**	**Mean**	**n**	**M$_{weig.}$**	**Areas**	**References**	**Comments**
Fish	Muscle	1	- 500					8	4	
Macroalgae	Brown	19	633 - 16000	3467	4637	34	5221	2,4,8	4,35,36,37	
	Red and Green	3	1037 - 3760	1320	2040	85	1931	2	36,37	
	Unspecified	10	1310 - 4950	2450	2693			5,6	26	
Molluscs										
Gastropod	Whole	25	400 - 4600	1500	1865			4,5,6	26,35	
	Pallial complex	1	15800 -					5	15	
	Digestive glands	2	7100 - 9700		8400			5	15	
	Total soft tissues	1	5700 -					5	15	
	Shell	2	1500 - 1800		1650			5	15	
	Muscle	2	600 - 1100		850			5	15	
Bivalve	Whole	9	380 - 2240	1480	1233			5,6	15,26	
	Byssal threads	1	29500 -					5	15	
	Viscera	1	4800 -					5	15	
	Total soft tissue	1	1400 -					5	15	
	Gill	1	1000 -					5	15	
	Muscle	1	800 -					5	15	

Table 7-10 : Concentration factors for **Sr-90** from field studies in European marine organisms

Organism/Taxonomy/Tissue		N	**Range**			**Median**	**Mean**	**n**	**$M_{weig.}$**	**Areas**	**References**	**Comments**
Macroalgae	Brown	1	182	-						8	4	
Mammals												
Seal	Muscle	1	0.4	-	1.2		0.8			8	4	
	Liver	1	0.2	-	3		1.6			8	4	
Fish	Whole	4	36	-	49	44	43			2	28	
	Muscle	1	4	-						8	4	
Crustaceans												
Shrimp	Shell	1	26	-						8	34	
	Muscle	1	11	-	19		15			8	34	

Table 7-11 : Concentration factors for **Pb-210** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue		N	**Range**			**Median**	**Mean**	**n**	**$M_{weig.}$**	**Areas**	**References**	**Comments**
Macroalgae	Seaweed	13	10	-	440	170	183			4,5,6	26	
Molluscs												
Gastropod	Whole	10	60	-	6930	715	1180			5,6	26	
Bivalve	Whole	10	30	-	7360	575	1508			5,6	26	

Table 7-12: Concentration factors for **U-238** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue		N	Range	Median	Mean	n	$M_{weig.}$	Areas	References	Comments
Macroalgae	Seaweed	13	21 - 130	46	50			4,5,6	26	
Molluscs										
Gastropod	Whole	9	12 - 82	23	28.4			5,6	26	
Bivalve	Whole	10	4 - 83	13	20.2			5,6	26	

Table 7-13: Concentration factors for **Am-241** in European marine organisms obtained from field studies

Organism/Taxonomy/Tissue		N	Range	Median	Mean	n	$M_{weig.}$	Areas	References	Comments
Macroalgae										
	Red	2	1020 - 3760		2400			2	36,37	
	Green	1	836 -					2	36,37	
	Brown	1	440 -					2	36,37	
Molluscs										
Gastropod	Whole	1	600 - 900		700*			5	40,41,42	
Bivalve	Whole	1	200 - 1300	700				5	40,41,42	

Table 7-14 : Concentration factors for **Cm-242, Ru-106** and **I-131** in European marine organisms obtained from field studies.

Organism/Taxonomy/Tissue	**Radionuc.**	**N**	**Range**	**Median**	**Mean**	**n**	**$M_{weig.}$**	**Areas**	**References**	**Comments**
Macroalgae	Cm-242									
Red		2	5950 - 51700		28800			2	36	
Green		1	1520 -					2	36	
Brown		1	1320 -					2	36	
Macroalgae	Ru-106									
Red		2	1343 - 3854		2600			2	37	
Brown		1	428 -					2	37	
Green		1	219 -					2	37	
Macroalgae	I-131									
Red		2	12690 - 85000		48800			2	37	
Green		1	921 -					2	37	
Brown		1	418 -					2	37	
Mussel	I-131				96			2	44	*Mytelis edulis*

Table 7-15: Concentration factors for **Am-241** in European marine organisms obtained from experimental studies

Organism/Taxonomy/Tissue			**N**	**Range**	**Mean**	**t**[a]	**C**[b]	**T**[c]	**Areas**	**References**	**Comments**
Macroalgae		Brown	1	329 -		12	6840	2	6	13	
			1	437 -		12	6840	12	6	13	
Molluscs											
	Bivalve	Whole	1	380 -		30	?	2	8	14	60% in soft tissues
		Whole	2	150 - 200	175	50	1000	13±2	1	17	Cs-134, Equilibrium
Crustaceans											
	Crab	Whole	1	145 -		8	1169	13	2	20	
		Gills	1	960 -		8	1169	13	2	20	
		Exoskeleton	1	240 -		8	1169	13	2	20	
		Hepatopancreas	1	5 -		8	1169	13	2	20	
		Muscle	2	5 - 25	15	8	1169	13	2	20	
Phytoplankton					2.9×10^5					43	*Thalassiosira pseudonana*

a Time of exposure in days
b Exposure concentration in Bq/l
c Temperature under which the experiment has been undertaken

Table 7-16: Concentration factors for **Po-210** in European marine organisms obtained from experimental studies

Organism/Taxonomy/Tissue			N	**Range**		**Mean**	**t**^{a}	**C**^{b}	**T**^{c}	**Areas**	**References**	**Comments**
Crustaceans												
	Shrimp	Whole	1	139	-		21	260	14	2	19	Uptake from seawater
		Muscle	1	24	-		21	260	14	2	19	
		Hepatopancreas	1	316	-		21	260	14	2	19	
		Exoskeleton	1	272	-		21	260	14	2	19	
	Shrimp	Whole	1	168	-		14	260	14	2	19	Uptake from seawater and food
		Muscle	1	54	-		14	260	14	2	19	
		Hepatopancreas	1	958	-		14	260	14	2	19	
		Exoskeleton	1	303	-		14	260	14	2	19	

Table 7-17: Concentration factors for **Pb-210** in European marine organisms obtained from experimental studies

Organism/Taxonomy/Tissue			N	**Range**		**Mean**	**t**^{a}	**C**^{b}	**T**^{c}	**Areas**	**References**	**Comments**
Crustaceans												
	Shrimp	Whole	1	682	-		21	260	14	2	19	Uptake from seawater
		Muscle	1	47	-		21	260	14	2	19	
		Hepatopancreas	1	297	-		21	260	14	2	19	
		Exoskeleton	1	1738	-		21	260	14	2	19	
	Shrimp	Whole	1	663	-		14	260	14	2	19	Uptake from seawater and food
		Muscle	1	32	-		14	260	14	2	19	
		Hepatopancreas	1	928	-		14	260	14	2	19	
		Exoskeleton	1	1813	-		14	260	14	2	19	

a Time of exposure in days
b Exposure concentration in Bq/l
c Temperature under which the experiment has been undertaken

Table 7-18: Concentration factors for **Cs-137** and **Cs-134** in European marine organisms obtained from experimental studies

Organism/Taxonomy/Tissue		**N**	**Range**	**Mean**	**t**^{a}	**C**^{b}	**T**^{c}	**Areas**	**References**	**Comments**
Macroalgae	Brown	1	3.3 -		12	8950	2	6	13	Cs-134
	Brown	1	4.6 -		12	8950	12	6	13	Cs-134
	Green	1	2 -		18	1000	16	2?	18	
Fish	Whole	1	3 -		18	1000	16	2?	18	
Crustaceans										
Isopod	Whole	1	22 -		18	1000	16	2?	18	
Molluscs										
Bivalve	Whole	1	14 -		30	?	2	8	14	
	Whole	2	2.57 - 2.8	2.7	50	1000	13±2	1	17	Cs-134, Equilibrium
Phytoplankton		5	≈0 - 40		10	$3x10^5$	12	?	39	Growing cells,
		5	≈0 - 100		6	$3x10^5$	12		39	Non-growing cells

a Time of exposure in days
b Exposure concentration in Bq/l
c Temperature under which the experiment has been undertaken

Table 7-19: Concentration factors for different radionuclides in European marine organisms obtained from experimental studies

Organism/Taxonomy/Tissue		**Radionucl.**	**N**	**Range**	**Mean**	**t**[a]	**C**[b]	**T**[c]	**Areas**	**References**	**Comments**
Macroalgae	Brown	Ru-106	1	88.5 -		12	7110	2	6	13	
Macroalgea	Green	Tc-99	1	350 -		10	37000	18-21	6	9	
	Red	Tc-99	1	0.1 - 5	2.5	10	37000	18-21	6	9	
	Brown	Tc-99	2	9000 - 12000	10500	10	37000	18-21	6	9	
Worm	Whole	Np-239	1	1.5 -		13	2146000	14±1	6	38	Equilibrium
Molluscs											
Bivalve	Soft tissues	Np-239	1	14 -		13	2146000	14±1	6	38	
	Shell	Np-239	1	47 -		13	2146000	14±1	6	38	
Bivalve	Byssus	I-125	1		1040					45	
	Foot		1		2.4					45	Experimental organism was *Perna viridis* – not native to European waters
	Soft tissue		1		6.4					45	
	Shell		1		2.1					45	
Bivalve	Whole body	I-131	1		3.9					46	
	Shell		1		4.7					46	
	Soft parts		1		4.9					46	Experimental organism not native to European waters
Crustaceans											
Crab	Muscle	Pu-237	2	5 - 8	6.5	8	1169	13	2	20	
	Hepatopancreas	Pu-237	1	2 -		8	1169	13	2	20	
	Exoskeleton	Pu-237	1	70 -		8	1169	13	2	20	
	Gills	Pu-237	1	340 -		8	1169	13	2	20	
Megabalanus tintinnambulum		I-131	1		4.3					46	Experimental organism not native to European waters

a Time of exposure in days
b Exposure concentration in Bq/l
c Temperature under which the experiment has been undertaken

References cited in the table of summaries

1 Steele, A. K. (1990)

3 Sazykina, T. G. (1998)

4 Fisher, N. S., Fowler, S. W. , Boisson, F. , Carroll, J. , Rissanen, K. , Salbu, B. , Sazykina T. G. and Sjoeblom K. L. (1999)

8 Brown, J. E. Kolstad, A. K. Brungot, A. L. , Lind, B. , Rudjord, A. L., Strand P. and Føyn, L. (1999)

9 Bonotto, S. , Robbrecht, V. , Nuyts, G. , Cogneau M. , and Van der Ben D. (1988)

10 Busby, R. , McCartney, M. and McDonald, P. (1997)

11 Berrow, S. D. , Long, S. C., McGarry, A. T., Pollard, D., Rogan, E. , Lockyer, C. (1998)

13 Boisson, F., Hutchins, D. , Fowler, S. W. , Fisher, N. S. and Teyssie, J. L. (1997)

14 IAEA (1999)

15 McDonald, P. , Baxter, M. S. and Fowler, S. W. (1993)

17 Gungor, N., Tugrul, B.. Topcuoglu, S. , Gungor, E. (2001)

18 Topcuoglu, S. (2001)

19 Fernando P. Carvalho, Fowler, S. W. (1993)

20 Guary, J. C., Fowler, S. W. (1990)

21 Stepnowski, P., Skwarzec B. (2000)

22 Skwarzec, B., Falkowski L. (1988)

23 Carvalho F. P. (1988)

24 Skwarzec, B., Bojanowski R. (1988)

25 Smith, V., Fegan, M., Pollard, D., Long, S., Hayden, E., Ryan T. P. (2001)

26 McDonald, P., Cook, G. T. , Baxter M. S. (1992)

27 Ryan, T. P., Dowdall, A. M. , McGarry, A. T. , Pollard, D., Cunningham, J. D. (1999)

28 Franic, Z., Lokobauer, N (1993)

31 Masson, M., Van Weers, A. W., Groothuis, R. E. J., Dahlgaard, H., Ibbett R. D. & Leeonard, K.S. (1995)

32 Osvath, L., Bologa, A., Dovlete C. (1990)

33 Hurtgen, C., Koch, G., Van der Ben, D., Bonotto S. (1988)

34 J. Brown, M. Iosjpe (2001)

35 Germain, P. Leclerc, G., le Cavelier, D., Solier, L., Baron Y. (2000)

36 Holm, E., Ballestra, S., Lopez, J. J., Bulos, A., Barci-Funel, G., Ardisson G. (1991)

37 Holm, E., Ballestra, S., Lopez, J. J., Bulos, A., Barci-Funel, G., Ardisson G. (1994)

38 Germain, P., Gandon, R., Masson, M., Gueguenìat P. (1987)

39 Heldal; H.E., Stupakoff, I., & Fisher, N.S.(2001).

40 Mitchell, P.I., Vives Batlle, J., Ryan, T.P., McEnri, C., Long, S., O'Colmain, M., Cunningham, J.D., Caulfield, J.J., Larmour, R.A. and Ledgerwood, F.K. (1991).

41 Mitchell, P.I., Vives Batlle, J., Ryan, T.P., McEnri, C., Long, S., O'Colmain, M., Cunningham, J., Caulfield, J.J., Larmour, R.A. and Ledgerwood, F.K. (1992).

42 Vives i Batlle (1993).

43 Rheinfelder, J.R. and Fisher, N.S. (1991).

44 Whitehead NE, Ballestra S, Holm E, Huynh-Ngoc L. (1998).

45 Shunhua C, Qiong S and Xiaokui Z (1997).

46 Mayr L, Moraes R, Lopes MA, Vicente C, Mauro JN (1988).

8 Modelling approach to derive CFs for sea mammals and birds

8.1 Introduction

A simple modelling approach has been adopted for the purpose of deriving suitable sea mammal and seabird CF values for inclusion in the look-up tables presented in Appendix 1 of the report (Appendix 1, Section 1.5). Models have been applied for all 20 "FASSET" radionuclides with the exception of K, C and H. In some cases, e.g. Cs, Pu and Pb in sea mammals, appropriate empirical data were available. In these cases the predictions made from biokinetic models could be presented for comparison purposes in order to evaluate, at a cursory level, whether the model were providing realistic CF values.

The models for both sea mammals and seabirds are semi-empirical in the sense that they use empirical CF data from the look-up tables as input to the models. In the case of seal, it is assumed that the animal lives entirely from a diet of fish exhibiting the appropriate radionuclide specific CF as specified in the look-up tables. In the case of seabirds, it is assumed that the diet consists entirely of molluscs, again with a concomitant whole body activity concentration derived from radionuclide specific data from the look-up tables.

The model, based on earlier work (Thomann, 1981; Landrum *et al.*, 1992; Fisher, 2002) considers uptake via food only (transfer from the water column to these vertebrates is considered to be negligible), while the excretion/elimination rate is considered to be independent of the uptake route, and the assimilation efficiency is considered to be independent of food type. We also make the simplifying assumption that the growth rate for all organisms is zero and predictions are made for adults. This latter assumption may be a particularly poor one (Thomann, 1981), but the complexity of the weight dynamics for the organisms in question is outside the scope of this work.The following equation defines the model :

$$\frac{dC_o}{dt} = (AE_o \cdot IR_o \cdot C_p) - C_o \cdot k_o \qquad \text{(8-1)}$$

where : AE_o = assimilation efficiency (dimensionless) for organism (seal or bird); IR_o = ingestion rate per unit mass of organism (kg d^{-1} per kg, f.w.); C_p = activity concentration in prey species (Bq kg^{-1}, f.w.); C_o = activity concentration in organism (Bq kg^{-1}, f.w.); k_o = loss rate from organism (d^{-1})

The loss rate is primarily a function of the excretion rate from the organism but for short lived radionuclides, it would include a component attributable to radioactive decay.

In the case of sea mammal, assuming a unit concentration in seawater, i.e. $C_w = 1$ Bq l^{-1} :

$$C_p = CF_{f,r} \qquad \text{(8-2)}$$

where : $CF_{f,r}$ = the radionuclide specified concentration factor (l kg^{-1}) for fish for radionuclide r

and for seabird (again and assuming a unit concentration in seawater, i.e. $C_w = 1$ Bq l^{-1}) :

$$C_p = CF_{m,r} \qquad (8\text{-}3)$$

where : $CF_{m,r}$ = the radionuclide specified concentration factor (l kg^{-1}) for mollusc for radionuclide r.

It can be shown that at equilibrium:

$$C_o = (AE_o \cdot IR_o \cdot C_p)/k_o \qquad (8\text{-}4)$$

where all parameters have been defined above.

8.2 Parametrisation of model

8.2.1 Assimilation efficiencies for sea mammals and birds

Assimilation efficiencies are presented in Table 8-1; We assume that the assimilation efficiencies for man can serve as an initial estimate for sea mammals and birds, in line with the approach adopted by the USDoE (USDoE, 2002).

Table 8-1: Mammal and bird Assimilation efficiencies for selected radionuclides

Radionuclide	**Organism**	**AE**	**Reference**
Cs	Mammal, bird	*(0.5)* - 1	*Thomann (1981)*; USDoE (2002);
Tc	Mammal, bird	0.8	USDoE (2002); ICRP 30 (Part 2)
Sr	Mammal, bird	0.3	USDoE (2002); ICRP 30 (Part 1)
U	Mammal, bird	0.05	USDoE (2002); ICRP 30 (Part 1)
Th	Mammal, bird	2x10^{-4}	USDoE (2002); ICRP 30 (Part 1)
Pu	Mammal, bird	*(0.1)*-1x10^{-3}	*Thomann (1981)* ; USDoE (2002);
Am	Mammal, bird	1x10^{-3}	ICRP 30 (Part 4), USDoE (2002);
Cm	Mammal, bird	1x10^{-3}	ICRP 30 (Part 4)
Np	Mammal, bird	1x10^{-3}	ICRP 30 (Part 4)
Ra	Mammal, bird	0.2	USDoE (2002); ICRP 30 (Part 1)
Pb	Mammal, bird	0.2	ICRP 30 (Part 2)
Po	Mammal, bird	0.1	ICRP 30 (Part 1)
C	Mammal, bird	*(1)*	ICRP 30 (Part 3)
H	Mammal, bird	*(1)*	USDoE (2002); ICRP 30 (Part 1)
Nb	Mammal, bird	0.01	ICRP 30 (Part 1)
Ni	Mammal, bird	0.05	ICRP 30 (Part 3)
Ru	Mammal, bird	0.05	ICRP 30 (Part 2)
I	Mammal, bird	1	USDoE (2002); ICRP 30 (Part 1)
Cl	Mammal, bird	1	ICRP 30 (Part 2)

Values and references not used are italicized in parentheses.

8.2.2 Ingestion rates for sea mammals and birds

8.2.2.1 Sea mammals

Innes *et al.* (1987) have provided the following allometric relationship for adult seals:

$$I_{seal} = 0.079M^{0.71} \qquad (8\text{-}5)$$

Where : I_{seal} = rate of biomass ingestion (kg/d f.w.); M = Body mass (kg, f.w.)

Within FASSET we have selected a seal with associated mass of 180 kg on the basis of life history information (Appendix 2, Section 1.3.9) . Applying this value we can derive a (mass-normalised ingestion rate, *IR*, of 0.175 (kg d^{-1} per kg, f.w.).

8.2.2.2 Seabirds

Allometric functions describing food ingestion rates (expressed as dry weight) have been presented by Nagy (1987). The following relationship has been reported for the generic group "birds":

$$I_{bird} = 0.0582\ M^{0.651} \qquad (8\text{-}6)$$

Where I_{bird} = Ingestion rate (kg $_{(d.w.)}$ per day); M = body mass (kg)

The percent water content of bivalves, in addition to other aquatic invertebrates, is taken to be 82 % (Sample *et al.,* 1997). For the sake of consistency with the dosimetric models, a duck mass of 0.6 kg has been selected in the derivation of ingestion rates.

Using Equation (8-6), and making the assumption that the bird is feeding entirely on bivalve molluscs), we derive a fresh weight ingestion rate of 0.23 kg (f.w.) per duck per day. Normalising this value to body mass we attain a value of 0.383 (kg d^{-1} per kg, f.w.)

8.2.3 Loss rate for sea mammals and birds

In theory the loss rate should include components of both biological elimination/excretion and radioactive decay. In view of the long physical half-lives of most of the radionuclides of interest, coupled to the fact that CF data are often based on stable element data (i.e. for the sake of compatibility), radioactive decay has not been considered in most cases. The exception to this approach can be noted for Cm and Ru where commonly occuring isotopes of these radionuclides, i.e. ^{242}Cm and ^{106}Ru were deemed to have sufficiently short half-lives (compared to biological half-lives) to render a consideration of physical decay appropriate.

2 approaches can be adopted in the derivation of elimination rates:

8.2.3.1 Allometric approach

In this approach we invoke the mass of sea mammals and birds (as specified above) and empirically derived allometric relationships to derive information on biological half-lives.

For example, an allometric relationship may be used to estimate the ^{137}Cs k_e for seal. The following equation has been applied by the USDoE (USDoE, 2002) based on earlier considerations (Whicker & Shultz, 1982).

$$\lambda_i = \frac{\ln 2}{18.36 M^{0.24}} \tag{8-6}$$

where λ_i = biological decay constant (d^{-1}), M = mass of animal (kg, f.w.).

This expression yields a value of **0.01085** d^{-1} for seal with a reference mass of 180 kg. Assuming that excretion is the only process by which the organism loses contamination, the biological decay constant can be set equal to the excretion rate. This value can be used directly in Equation (8-4) above to derive the equilibrium CF for Cs..

8.2.3.2 Man-analogue approach

We can assume that the retention in sea mammals and, more tentatively, seabirds, is similar to that for man or other specified mammals. For example, ICRP 30, Part 4 (ICRP, 1988), provide the following equation defining the retention of Pu in man:

$$R\ (t) = 0.45\, e^{-0.693\, t / 50} + 0.45 e^{-0.693\, t / 20} + 0.00035 e^{-0.693\, t / \infty} \tag{8-7}$$

(Remaining approx 0.1 goes directly to excretion.)

where R = retention; t = time (years) and where the 1st terms related to a fraction of 0.45 entering a skeleton compartment, i.e. a half life of 100yrs, the 2nd term relates to a biological compartment representing the liver with half-life 40 yrs and the 3rd relates to a biological compartment representing the gonads with infinite biological half-life.

The excretion term, $C_o k_o$, in Equation (8-7), therefore, now takes the form of a multi-compartmental model as shown in the following Figure 8-1 for the specific example of Pu.

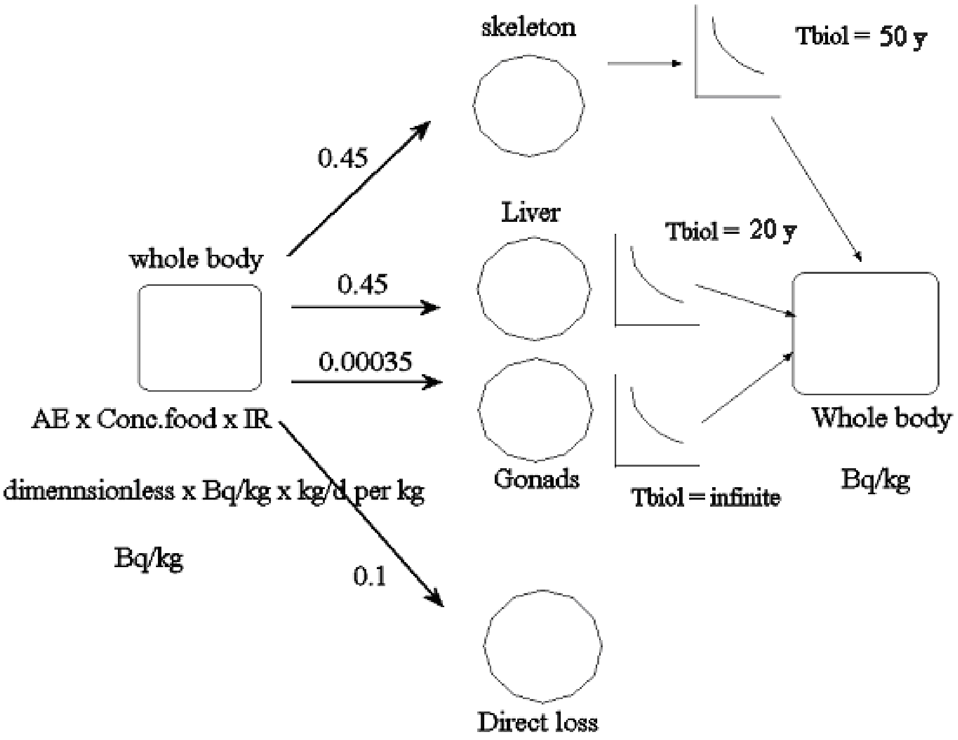


Figure 8-1 Excretion model for Pu based on biokinetic model for man (ICRP30, Part 4).

The “whole-body” activity concentration (Bq/kg f.w.) at any time instant is the sum of the activity values in the various biological compartments.

Retention models have been reported in ICRP30 (Parts 1-4). The following models have been applied for predicting elimination rates for the radionuclides in question (Table 8-2) :

Table 8-2. (1) Retention and (2) excretion models for radionuclides. References for each of the models presented in brackets.

Radionuclide	Retention and/or elimination Equation
Cs	1. $R(t) = 0.1e^{-0.693t/2} + 0.9e^{-0.693t/110}$ (ICRP30, Part 1) 2. $\lambda_i = \frac{\ln 2}{18.36M^{0.24}}$ (Whicker & Schultz, 1982, USDoE, 2002)
Tc	1. $R(t) = 0.76e^{-0.693t/1.6} + 0.19e^{-0.693t/3.7} + 0.043e^{-0.693t/22}$; (ICRP30, Part 2) 2. $\lambda_i = \frac{\ln 2}{4.8M^{0.4}}$ (Whicker & Schultz, 1982, USDoE, 2002)
Sr	$\lambda_i = \frac{\ln 2}{645W^{0.26}}$ (Whicker & Schultz, 1982, USDoE, 2002)

U	1a. $R_{bone}(t)=0.2e^{-0.693t/20}+0.023e^{-0.693t/5000}$ 1b. $R_{kidney}(t)=0.12e^{-0.693t/6}+0.00052e^{-0.693t/1500}$ 1c. $R_{other}(t)=0.12e^{-0.693t/6}+0.00052e^{-0.693t/1500}$ 0.464 eliminated directly. (ICRP 30, Part 1) 2. $\lambda_i=\frac{\ln 2}{5.545M^{0.28}}$ (USDoE, 2002)
Th	1. $R\ (t)=0.1e^{-0.693t/0.5}+0.7e^{-0.693t/8000}+0.04e^{-0.693t/700}+0.16e^{-0.693t/700}$ 0.1 eliminated directly (ICRP 30, Part 1); Biological half-life in transfer compartment = 0.5days 2. $\lambda_i=\frac{\ln 2}{888M^{0.81}}$ (USDoE, 2002)
Pu	1. $R\ (t)=0.45e^{-0.693t/18250}+0.45e^{-0.693t/7300}+0.00035e^{-0.693t/\infty}$ Remaining approx 0.1 goes directly to excretion. (ICRP 30, Part 4) 2. $\lambda_i=\frac{\ln 2}{1140M^{0.731}}$; (FASSET)
Am	1. $R\ (t)=0.45e^{-0.693t/18250}+0.45e^{-0.693t/7300}+0.00035e^{-0.693t/\infty}$ Remaining approx 0.1 goes directly to excretion. (ICRP 30, Part 4) 2. $\lambda_i=\frac{\ln 2}{1140M^{0.731}}$; (FASSET)
Cm	$\lambda_i=\frac{\ln 2}{1140M^{0.731}}$; (FASSET)
Np	$R\ (t)=0.75e^{-0.693t/18250}+0.15e^{-0.693t/7300}+0.00035e^{-0.693t/\infty}$ (ICRP 30, Part 4)
Ra	$R\ (t)=0.69e^{-0.693t/0.683}+0.24e^{-0.693t/8.15}+0.07e^{-0.693t/338}$ (see note 1)
Pb	$R\ (t)=0.7e^{-0.693t/12}+0.17e^{-0.693t/180}+0.13e^{-0.693t/5000}$ (Lloyd *et al.*, 1975 as cited in ICRP-30, Part 2) (see note 2)
Po	$R\ (t)=e^{-0.693t/50}$ (ICRP 30, Part 1); $\lambda_i = 0.01386$
C	n/a (see note 3)
H	n/a (see note 3)
Nb	$R\ (t)=0.5e^{-0.693t/6}+0.5e^{-0.693t/200}$ (ICRP-30, Part 1)
Ni	$R\ (t)=0.02e^{-0.693t/0.2}+0.3e^{-0.693t/1200}$ Remaining 0.68 goes directly to excretion (ICRP-30, Part 3)
Ru	$R\ (t)=0.15e^{-0.693t/0.3}+0.35e^{-0.693t/8}+0.3e^{-0.693t/35}+0.2e^{-0.693t/1000}$ (ICRP-30, Part 2)
I	$\lambda_i=\frac{\ln 2}{16.7M^{0.13}}$ (Whicker & Schultz, 1982, USDoE, 2002)
Cl	$\lambda_i=\frac{\ln 2}{2.38M^{0.25}}$ (FASSET)

Where R(t) = retention of radionuclide; λ_i = excretion rate (d^{-1}) M = f.w. of organism (seal or seabird) in kg

Note 1 : The retention function for Ra requires special mention because this was not directly taken from published information. The retention function was derived by taking values predicted by the Norris power function (see ICRP Publication 20; ICRP, 1973) and then fitting a 3 component, 6 parameter, exponential function to the line using the multiple regression tool provided in Sigmaplot©. The 3 component exponential fit provides an output, for approximately the first 1000 days of retention, that is similar to the Norris model but fails to adequately predict whole body retentions at protracted time intervals. The 3 components are arbitrarily derived and do not relate to any specific organ or body part.

Note 2. This retention equation was derived for whole body retention of Pb in dogs. The 3rd component was apparently not well defined by these experiments.

Note 3: Special carbon and tritium modeling was not conducted for the purpose of deriving values for the look-up tables.

8.3 Summary tables of parameters used for biokinetic modelling

The parameters used in the biokinetic models are summarised for seal and seabirds in Tables 8-3 and 8-4 respectively.

Table 8-3 Parameters used in biokinetic models for seal

Nuclide	**AE**	**IR**	**C_{prey}**	**λ_{biol}[a] (d^{-1})**	**λ_{phys} (d^{-1})**
Cs	1	0.0175	90	0.01085	-
Tc	0.8	0.0175	45	0.018	-
Sr	0.3	0.0175	17	$2.78x10^{-4}$	-
U	0.05	0.0175	1	0.0292	-
Th	$2 x 10^{-4}$	0.0175	600	$1.16 x 10^{-5}$	-
Pu	$1 x 10^{-3}$	0.0175	100	$1.365 x 10^{-5}$	-
Am	$1 x 10^{-3}$	0.0175	100	$1.365 x 10^{-5}$	-
Cm (Cm-242)	$1 x 10^{-3}$	0.0175	100	$1.365 x 10^{-5}$	$4.25 x 10^{-3}$
Np	$1 x 10^{-3}$	0.0175	1	See note 4	-
Ra	0.2	0.0175	100	See note 4	-
Pb	0.2	0.0175	200	See note 4	-
Po (Po-210)	0.1	0.0175	6000	See note 4 + 5	-
Nb	0.01	0.0175	30	See note 4	-
Ni	0.05	0.0175	1000	See note 4	-
Ru	0.05	0.0175	2	See note 4	$1.88x10^{-3}$
I	1	0.0175	9	0.021	-
Cl	1	0.0175	0.06	0.0795	-

[a] Used for the simple single component exponential loss model only.

Note 4 : No simple allometrically derived (single component) loss rate available. Loss rates had to be modeled using a multi-compartmental (or in the exceptional case of Po a single compartmental) model only.

Note 5 : Po-210, the most likely environmental form of Po, has a half-life of 138 days and therefore, on cursory inspection, qualifies as a special case where decay should be accounted for in the derivation of CFs using biokinetic models. However, in view of the fact that the fraction of unsupported Po-210, in prey or target organism body, is not quantifiable for a generic scenario (presumably in most real cases at least some Po-210 would be supported by parent nuclides from ^{238}U decay chain), the application of a loss by decay was deemed inappropriate.

Table 8-4 : Parameters used in biokinetic models for Seabird

Nuclide	**AE**	**IR**	**C_{prey}**	**$\lambda_{biol}{}^{a}$ (d^{-1})**	**λ_{phys} (d^{-1})**
Cs	1	0.386	60	0.043	-
Tc	0.8	0.386	500	0.177	-
Sr	0.3	0.386	10	1.2×10^{-3}	-
U	0.05	0.386	20	0.144	-
Th	2×10^{-4}	0.386	1000	1.2×10^{-3}	-
Pu	1×10^{-3}	0.386	1230	8.83×10^{-4}	-
Am	1×10^{-3}	0.386	700	8.83×10^{-4}	-
Cm (Cm-242)	1×10^{-3}	0.386	1000	8.83×10^{-4}	4.25×10^{-3}
Np	1×10^{-3}	0.386	200	See note 4	-
Ra	0.2	0.386	100	See note 4	-
Pb	0.2	0.386	1500	See note 4	-
Po	0.1	0.386	14000	See note 4 + 5	-
Nb	0.01	0.386	1000	See note 4	-
Ni	0.05	0.386	2000	See note 4	-
Ru (Ru-106)	0.05	0.386	500	See note 4	1.88×10^{-3}
I	1	0.386	100	0.044	-
Cl	1	0.386	0.05	0.331	-

8.4 Seal CFs derived from biokinetic-allometric modeling

Transfer factors for seal, derived from model simulations, are presented in Table 8-5. In some cases equilibrium has only been obtained after the passage of considerable time – in this case the time to reach equilibrium and, where appropriate, the concentration ratio following a 10 year contact period are expressed.

Table 8-5 : Seal CFs

Radionuclide	**CF**
Cs	225[a]; 145[b]
Tc	3[a]; 35[b]
Sr	320[b]
U	0.15[a] (>50 y to equilibrium); 0.065[a] (10 y) 0.03[b]
Th	17[a] (> 50 y to equilibrium) ; 5[a] (10y) 180[b] (> 1000 y to equilibrium) ; 7.5[b] (10 y)
Pu	30[a] (>250 y to equilibrium) ; 5[a] (10 y) 130[b] (1000 y to equilibrium); 6[b] (10 y)
Am	30[a] (>250 y to equilibrium) ; 5[a] (10 y) 130[b] (1000 y to equilibrium); 6[b] (10 y)
Cm[c] (Cm-242, $T_{1/2}$ = 163 d)	0.4[b]
Np	0.4[a] (>250 y to equilibrium); 0.05[a] (10y)
Ra	25[a]
Pb	25[a]; *10 000[a](note 6)*
Po	760[a]
C	n/a
H	n/a
Nb	0.1[a]
Ni	450[a] (>25 y to equilibrium); 400[a] (10 y)
Ru[c] (Ru-106, $T_{1/2}$ = 368 d)	0.2[a]
I	8[b]
Cl	0.01[b]

[a] derived using a multi-compartmental (single for Po) model for elimination (Table 8-2), normally based on models for man (ICRP-30, Parts 1-4); [b] Derived using an allometric relationship to derive a single component elimination rate (See Table 8-3); [c] Decay considered (radioisotope and half-life specified in brackets)

Note 6 : In view of the great difference associated with crustacean and fish Pb CFs, i.e. 90 000 and 200 respectively, it was of interest to run the model using crustaceans as the only dietary component. It is known that many seal species in fact ingest significant quantities of crustaceans (see for example Dommasnes *et al*., 2001). In this case a Seal Pb CF of circa 10 000 is obtained. This value is somewhat closer to the empirically derived value used in the look-up tables and exhibits the fact that limited information concerning the diet of marine mammals may lead to high uncertainty in the prediction of uptake using biokinetic models.

8.5 Seabird CF derived from biokinetic-allometric modelling

Transfer factors for seabird, derived from model simulations, are presented in Tables 8-6. In accordance with the approach adopted for seal, concentration ratios have been provided for a 10 year contact period, for cases where the time to equilibrium is significantly longer than the life expectancy of the reference seabird (See Appendix 2, Section 1.3.8).

Table 8-6 Seabird CFs

Radionuclide	**CF**
Cs	540[b]
Tc	870[b]
Sr	940[b] (> 10 y to equilibrium)
U	3[b]
Th	65[b] (> 10 y to equilibrium)
Pu	540[b] (> 10 y to equilibrium)
Am	310[b] (> 10 y to equilibrium)
Cm[c] (Cm-242, $T_{1/2}$ = 163 d)	75[b]
Np	1650[a] (>250 y to equilibrium); 230[a] (10 y)
Ra	520[a]
Pb	3900[a]
Po	39 000[a]
C	n/a
H	n/a
Nb	100[a]
Ni	20 000[a] (>25 y to equilibrium); 17 500[a] (10 y)
Ru[c] (Ru-106, $T_{1/2}$ = 368 d)	920[a]
I	880[b]
Cl	0.06[b]

[a] derived using a multi-compartmental (single for Po) model for elimination (Table 8-2), normally based on models for man (ICRP-30)
[b] Derived using an allometric relationship to derive a single component elimination rate (See Table 8-4)
[c] Decay considered (radioisotope and half-life specified in brackets)

8.6 Comparison with empirical data – Cs, Pu and Pb for sea mammals and birds

In some cases empirical data for CFs in sea mammals were available to allow a comparison with the predictions made by the biokinetic models. Fisher *et al*. (1999) reported a CF value of 40 (range 13-70) for ^{137}Cs in seal muscle whereas the IAEA (IAEA, in press) provide a recommended value of 400. This compares with the 2 values derived from biokinetic modeling of 145 and 225. Fisher *et al*. (1999) also provide a CF for seabirds, 400, which agrees favourably with model predictions, 540. For the case of radiocaesium, therefore, the CFs derived from modeling work appear to be sensible. In the case of Pu, a recommended CF value of 8 has been present by the IAEA (IAEA, in press) for pinniped liver. The available modeling approaches have provided an equilibrium CF of 30 (based on human retention equations) and 130 (based on allometrically-derived elimination rates) but also provide information suggesting that the time to reach such an equilibrium would be protracted, i.e. significantly longer than the life expectancy of the reference organism ingesting this radionuclide. Providing a concentration ratio for contact periods that are of the same order as the life expectancy of our reference marine organisms, in this case arbitrarily taken to be 10 years although the life expectancy of Eider duck and Harp seal can be somewhat longer (see Sections 1.3.8 & 1.3.9), is arguably more appropriate in an environmental context. If one accepts the 10 year concentration ratio as a more relevant transfer factor than the (equilibrium) concentration factor, the biokinetic model output is similar to the value reported from empirical observation. Finally, empirical data are available for pinniped muscle. The recommended value provided by IAEA is 3000 (IAEA, in press). This value is clearly quite different to the value derived from the biokinetic model. A brief examination of the available transfer information leads us to the conclusion that the reason for this discrepancy may lie in the activity concentration (as defined by lower trophic level CFs) for the prey of the seal. Crustaceans can form a substantial proportion of the diet of some seal species. Assuming that the seal is ingesting crustaceans only, thereby applying the appropriate Pb CF to the system of equations leads to a much higher seal CF, i.e. 10 000, as discussed above. This exercise, moreover, demonstrates the sensitivity of the models to the selection of parameter values and provides insight into possible uncertainties and limitations associated with the approach. In order to make confident predictions concerning activity concentrations in marine mammals, and sea birds, quite detailed information concerning diet may be required but may be unattainable.

In conclusion, it can be noted that the simple biokinetic models applied in this study provide sensible estimates for CF in the case of Cs for seabirds and Cs, Pu and Pb for sea mammals. Extending this conclusion to other radionuclide-organism combinations, however, is not possible without verification using empirical data sets. In many cases, the values used to parameterize the model are far from ideal, notably in the case of seabirds, where elimination data for humans needed to be used, in many cases, for lack of more appropriate information. Parametrisation of the models is no simple task and may be a source leading to great uncertainty in model predictions.

9 River modelling

9.1 Generic models

Within FASSET, modelling is recommended to be as simple as possible and needing a very small number of parameters, in order to give generic assessments.
The Safety Reports Series No.19 (IAEA, 2001) provides these kinds of generic models resulting in conservative assessments of radionuclide transfers in the environment, in chronic situations. As a result of the simplifying assumptions implicit in its derivation, this generic methodology strictly applies only if the following conditions are satisfied:

- the surface water geometry (e.g; river cross-section) does not change greatly with distance ;
- the flow characteristics (e.g. flow velocity, water depth) do not change significantly with distance or with time ;
- radionuclides in water and sediment, under the conditions of a routine, long term release, can be considered to be in equilibrium.

Concerning the calculation of radionuclide concentrations in rivers, the following equation can be used :

$$C_{w,tot} = \frac{Q_i}{q_r} \exp\left(-\frac{\lambda_i x}{U} \right), \text{ with}$$

$C_{w,tot}$	radionuclide total (including suspended sediment) concentration in water ($Bq.m^{-3}$),
Q_i	average discharge rate for radionuclide i ($Bq.s^{-1}$),
q_r	mean river flow rate ($m^3.s^{-1}$),
λ_i	radioactive decay constant (s^{-1}),
x	distance between the discharge point and the receptor (m),
U	net freshwater velocity ($m.s^{-1}$).

Q_i is the input data. It has been decided in FASSET to provide the results for 1 $Bq.s^{-1}$.

Tables given in the safety reports series No. 19 (IAEA, 2001) allow to calculate the values for any of three parameters out of four (q_r, U, B, D) since one of them is known.
B is the river width (m)
D is the river depth (m)

We shall see in the following that the distance between the discharge point and the receptor has not a very great influence on the result.

However, in this model, a difference is made between the riverside where the discharge occurs and where the calculation is made. A correction factor is applied in one of the two cases. If the calculation is made very close to the discharge point, where the complete mixing is not yet obtained, the present model neglects dilution effects and assumes that exposure occurs at the point of discharge. This approach is independent of the type of water body and the concentration in water is calculated as follows:

$$C_{w,tot} = \frac{Q_i}{F}, \text{ with}$$

F is the flow rate of the liquid effluent ($m^3.s^{-1}$).

9.2 Model testing

The objective was to evaluate the influence of the different parameters in the model to see if some simplifications could be operated. For this reason, some runs with the model were executed to test four parameters :

- river width,
- assessment on the same or the opposite riverside where the discharge has occurred,
- distance between the point of calculation and the point of release,
- radioactive decay.

9.2.1 River width

More than 30 values (taken in IAEA (2001), from 5.75 to 2000 m) for the average river width have been tested (cf. Figure 9-1). This set of tested values are that given by IAEA.

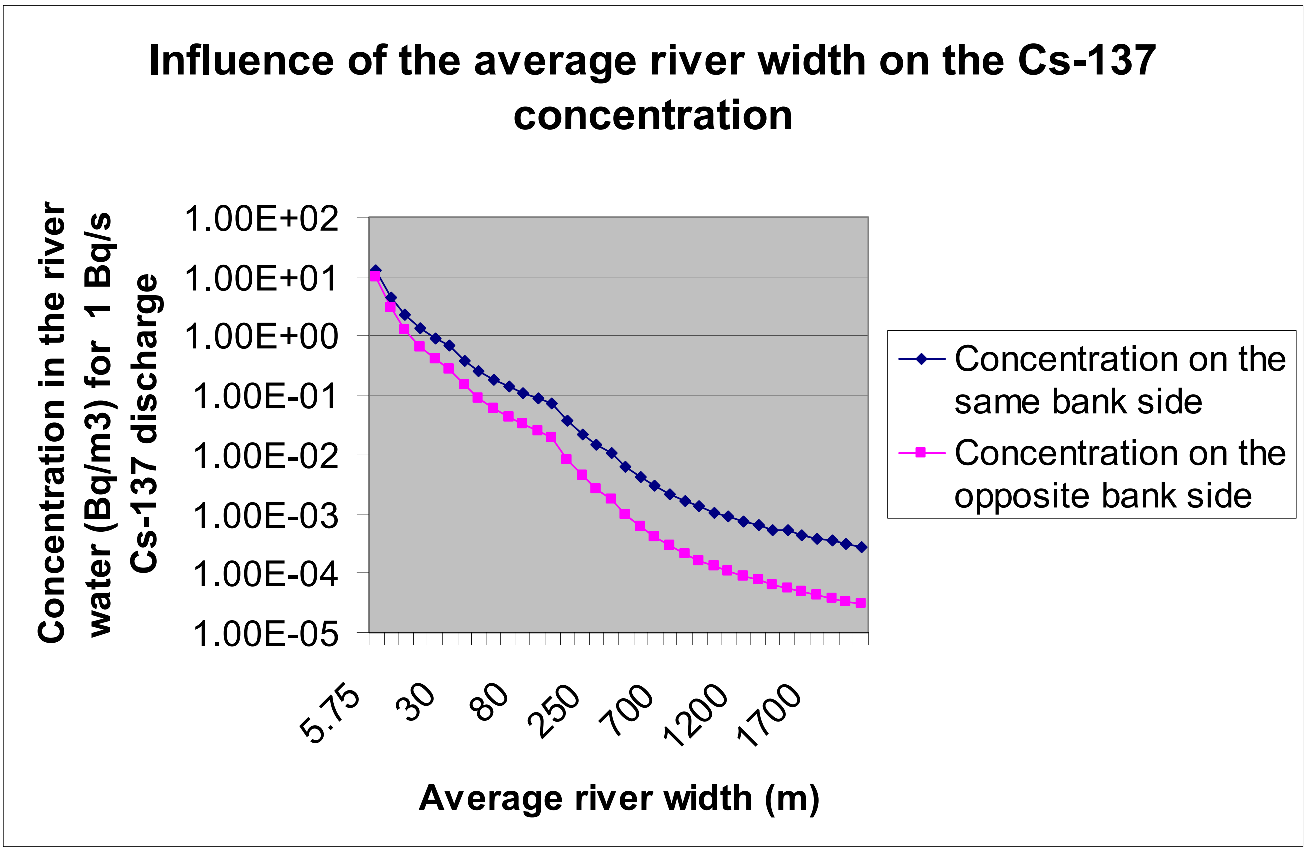


Figure 9-1 : Influence of the average river width on the radionuclide concentration in water.

Concerning the radionuclide concentration in water, there is a factor of 10^6 between the smallest (5.75 m) and the largest (2000 m) rivers modelled, which exactly corresponds to the ratio of the river flow rates (simple dilution hypothesis).

The river width or the river flow rate seems to be a crucial input data that determines totally the value of the assessment. Consequently, it seems useful to provide assessment data for a limited number of generic rivers that will be characterized by some extreme and average river widths.

9.2.2 Same or opposite bank side?

Depending on the river bank for which the calculation is made, the model is corrected via a coefficient that represents the effect of water mixing. In figure 1, it can be seen that this correction is negligible for the low values of river width (5 - 20 m). Beyond this range, there is up to a factor of ten for the largest rivers tested (2 km).

At a distance of $\frac{3 \times B^2}{D}$ (B, river width and D, river depth), the concentrations in water on the same or the opposite bank of the river are equal. Below that distance, the difference is about a factor of ten, whatever the river width (see figures 9-1 and 9-2).

It is sure that the difference that is made between the same or the opposite bank side where the calculation and the discharge are made could be of concern for an application to a real site. However, for generic assessment, it seems that this consideration is of minor importance, in comparison to the river width or the river flow rate, for example. For this reason, this peculiarity will not be taken into account for the purpose of FASSET.

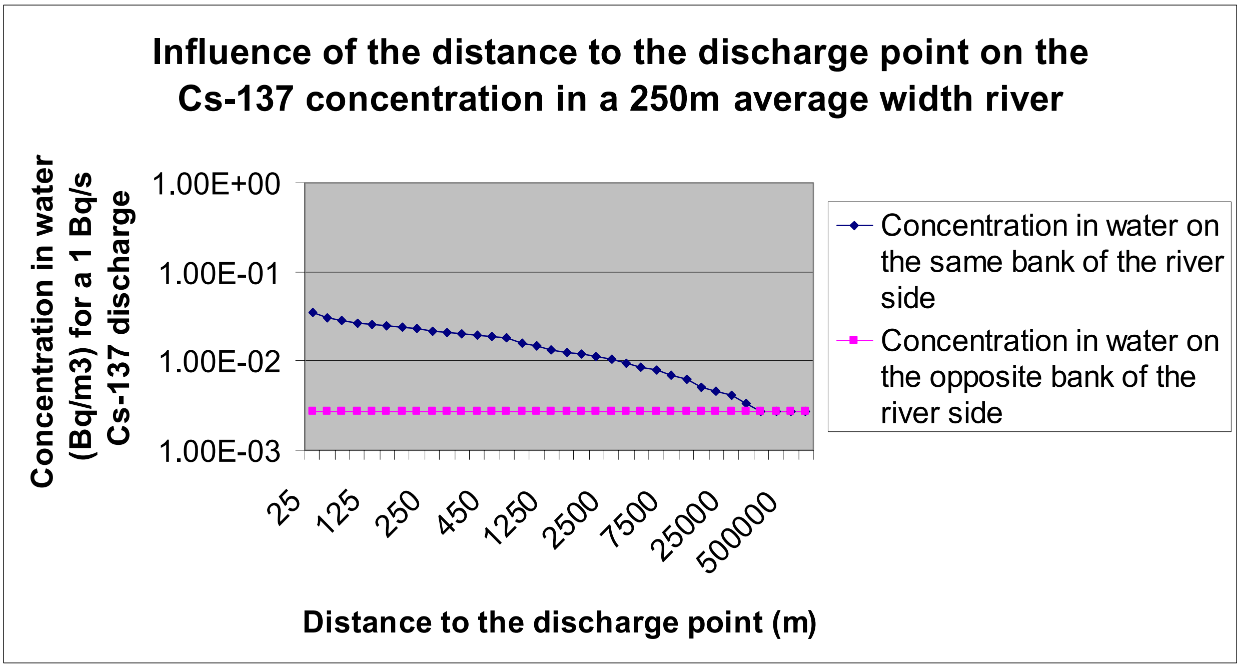


Figure 9-2 : Influence of the distance between discharge and calculation points.

9.2.3 Distance between calculation and discharge points

The effect of the distance between discharge and calculation points has been tested. It can be seen, on Figure 9-2, that there is a maximal factor of ten between the radionuclide concentration calculated close or far from the discharge point.

9.2.4 Radioactive decay

For this matter, two extreme radionuclides in FASSET have been tested : Th-231 (25 hours) and Th-232 (1.4x10^{10} years). The Figure 9-3 shows that the effect of the radioactive decay is negligible compared to the river width and to the difference made between opposite or same bank side. For the very low radioactive period (especially, Th-231), the concentration in water decreases sharply at a distance of 1000 times the river width from the discharge point. In a screening perspective, this parameter could be neglected, taking into account that the least exposing scenarios are not especially interesting for FASSET.

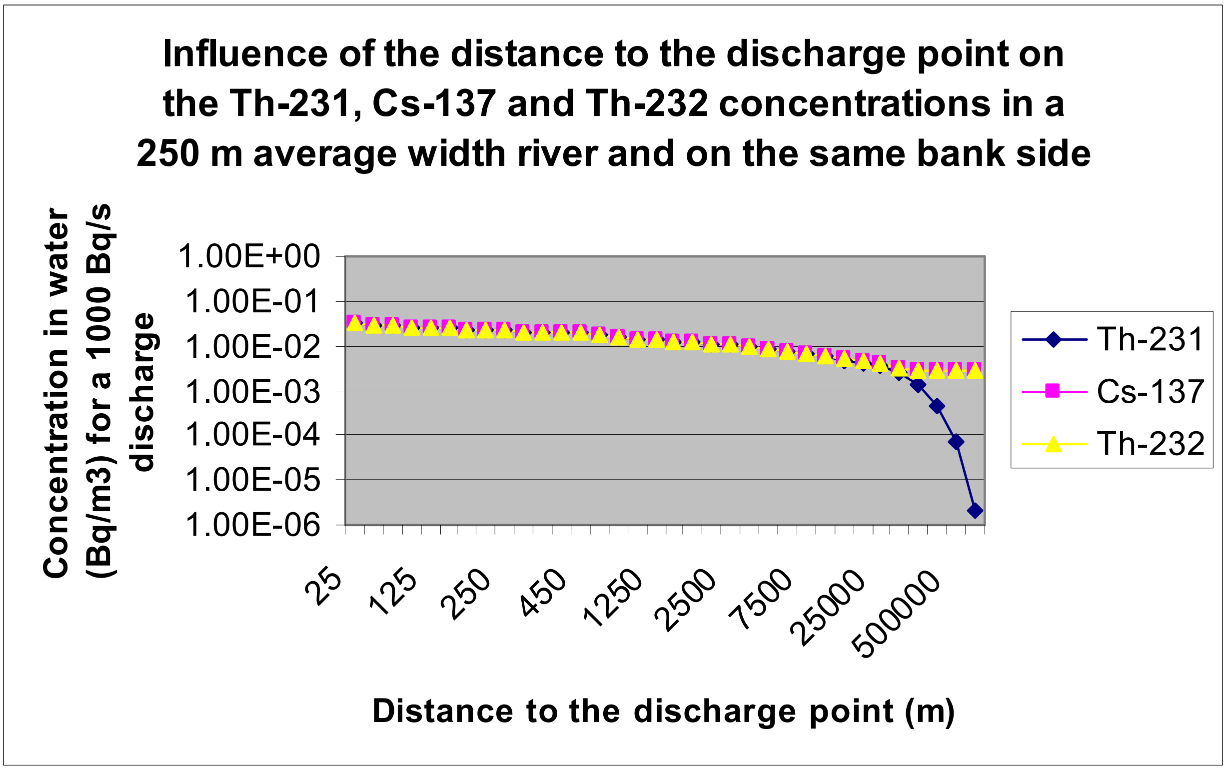


Figure 9-3 : Influence of the radioactive decay on the radionuclide concentrations in water.

9.3 Results

9.3.1 Radionuclide concentration in water

Examining the generic equation for radionuclide concentration in river water, it is possible to mathematically confirm the different simplifications proposed before.

$$C_{w,tot} = \frac{Q_i}{q_r} \exp\left(-\frac{\lambda_i x}{U} \right)$$

For the radionuclides considered in FASSET, λ_i belongs to the interval $10^{-6} - 10^{-18}$ s^{-1}.

To assess the minimum value of the term $\exp\left(-\frac{\lambda_i}{U} x \right)$, we can use the extreme realistic scenario of the Volga river (the largest European river). The different parameters are as follows :

- mean annual flow rate = 8400 $m^3.s^{-1}$,
- 30 year low annual river width = 398 m,
- 30 year low annual river depth = 5.84 m,
- 30 year low annual river flow rate = 3000 $m^3.s^{-1}$.

If the calculation is made at the complete mixing point, $x = 3B^2/D = 81372$ m
The water velocity, U, is equal to $q_r/BD = 1.29$ $m.s^{-1}$.

Consequently, the term $\left(-\frac{\lambda_i}{U} x \right)$ will belong to the interval (-0.06 ; $-6.3x10^{-14}$). The exponential of this value will then belong to the interval (0.94 – 1). The equation could then be simplified as follows :

$C_{w,tot} = \frac{Q_i}{q_r}$, simple dilution calculation used in a lot of routine models

Q_i is the average discharge rate for radionuclide i ($Bq.s^{-1}$),

q_r is the 30 year low annual river flow rate ($m^3.s^{-1}$). This parameter is considered by IAEA (2001) as 1/3 of the mean annual river flow rate.

9.3.2 Radionuclide concentration in sediment

IAEA (2001) recommends that calculations for screening purposes of doses from drinking water, fish and shellfish be based on the total concentration in water (including suspended sediments). In this case, sediment effects should be taken into account only for the purpose of calculating doses arising from exposure to sediment. The radionuclide concentration $C_{W,\, tot}$ is estimated as described above, Section 9.3.1., and already corresponds to the total concentration in water (including suspended sediments).

9.3.2.1 Radionuclide concentration in suspended sediment

The radionuclide concentration, $C_{s,w}$ (Bq.kg^{-1}), adsorbed by suspended sediment can be obtained by :

$$C_{s,w} = \frac{0{,}001 K_d C_{w,tot}}{1 + 0{,}001 S_s K_d} \qquad K_d \text{ in l.kg}^{-1} \text{ and } S_s \text{ in kg.m}^{-3}.$$

K_d values can be obtained in IAEA document (IAEA, 2001), in Coughtrey *et al.* (1985) and in Huang & Mouchel (1995) and Shafer *et al.* (1997). But for K and Po, K_d values on freshwater sediments have not been found. Default values for soils have been taken from Baes III *et al.* (1984). A default value of 5.10^{-2} kg.m^{-3} for the suspended sediment concentration, S_s is given in IAEA (2001).

9.3.2.2 Radionuclide concentration in bottom sediment

Bottom sediment can contain radionuclides owing to deposition of suspended sediment, on which radionuclides are adsorbed, and to direct adsorption on to bottom sediment of dissolved radionuclides from overlying water. The apparent K_d value for bottom sediment is generally assumed to be one tenth of the K_d value associated with suspended sediment (IAEA, 2001). This assumption is still likely to overestimate the K_d values of the bottom sediment. Radioactive decay during accumulation of the radionuclide on the river bottom is taken into account in calculating the radionuclide concentration in the bottom sediment, $C_{s,b}$ (Bq.kg^{-1}), as follows :

$$C_{s,b} = 0{,}1 C_{s,w} \times \frac{1 - e^{-\lambda_i T_e}}{\lambda_i T_e}$$

where T_e is the effective accumulation time (s). To give a conservative estimate of $C_{s,b}$ a default value of 3,15.10^{7} s (1 year) is recommended for the effective accumulation period T_e.

9.3.3 Radionuclide concentration table

The Table 9-1 presents the different radionuclide concentrations in water, suspended sediment and bottom sediment calculated with the different equations, hypothesis and simplifications described previously.

Calculations are made with Q_i = 1 Bq.s^{-1} and q_r = 1 m^3.s^{-1}. If a value is needed for Q_i = X Bq.s^{-1} and q_r = Y m^3.s^{-1}, the value of the table 1 must be multiplied by $\frac{X}{Y}$.

Some input data are needed and are given hereafter :

- suspended sediment concentration, $S_s = 5.10^{-2}$ kg.m^{-3},
- effective accumulation time, Te = 3,15 10^7 s,
- radioactive decay rate, λ_i is given in Table 9-1.

Table 9-1 Radionuclide concentrations in water and sediment for unit Q_i and q_r.

	Qi = 1 Bq/s		qr = 1 m3/s		
radionuclide	λ(s)	Kd (l/kg)	Cw (Bq/m3)	Csw (Bq/kg)	Csb (Bq/kg)
3H	1.78E-09	0	1	0.0E+00	0.0E+00
36Cl	7.27E-14	1		1.0E-03	1.0E-04
14C	3.83E-12	5		5.0E-03	5.0E-04
99Tc	1.03E-13	5			
40K	1.72E-17	5.5			
131I	9.98E-07	10		1.0E-02	3.2E-05
129I	1.40E-15	10			1.0E-03
237Np	1.03E-14	10			
234U	8.94E-14	50		5.0E-02	5.0E-03
235U	3.12E-17	50			
238U	4.92E-18	50			
106Ru	2.18E-08	500		5.0E-01	3.6E-02
210Po	5.80E-08	500			2.3E-02
226Ra	1.37E-11	500			5.0E-02
89Sr	1.59E-07	1000		1.0E+00	2.0E-02
134Cs	1.07E-08	1000			8.5E-02
90Sr	7.54E-10	1000			9.9E-02
137Cs	7.33E-10	1000			
135Cs	9.56E-15	1000			
242Cm	4.92E-08	5000		4.9E+00	2.5E-01
244Cm	1.21E-09	5000			4.8E-01
243Cm	7.71E-10	5000			
241Am	5.08E-11	5000			
231Th	7.55E-06	10000		9.5E+00	4.0E-03
227Th	4.27E-07	10000			7.1E-02
234Th	3.33E-07	10000			9.1E-02
228Th	1.15E-08	10000			8.0E-01
63Ni	2.20E-10	10000			9.5E-01
59Ni	2.93E-13	10000			
230Th	2.91E-13	10000			
232Th	1.56E-18	10000			
94Nb	1.08E-12	100000		6.7E+01	6.6E+00
238Pu	2.50E-10	100000			
239Pu	9.11E-13	100000			
240Pu	3.35E-12	100000			
241Pu	1.53E-09	100000			
210Pb	9.85E-10	300000		1.2E+02	1.2E+01

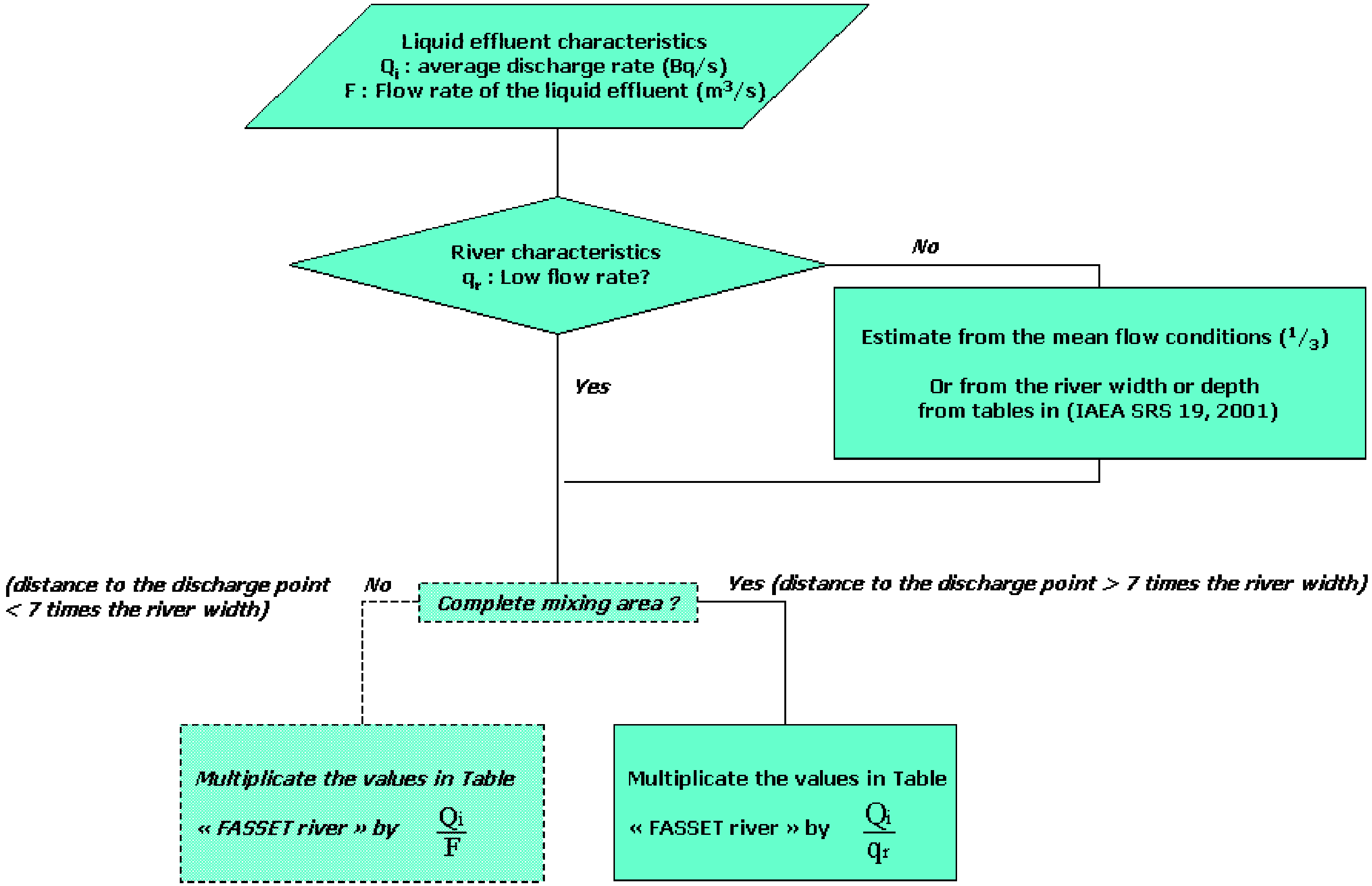
The total concentration in water, C_w, that includes suspended sediments, has to be used for the calculation of :

- radionuclide concentration in freshwater reference organisms using concentration factors
- doses arising from exposure to water for reference organisms

The radionuclide concentration in bottom sediment (C_{sb}) has to be used for the calculation of doses arising from exposure to sediment for reference organisms.

9.3.4 How to use the FASSET river table?

The following flow chart illustrates the different steps to reach and use the river table.



10 References

10.1 References for Section 2 (Forest ecosystems)

Avila, R. (1998). Radiocaesium Transfer to Roe Deer and Moose – Modelling and Experimental Studies. Doctor´s dissertation. Agraria 136, Uppsala.

Cederlund, G., Ljungqvist, H., Markgren, G. And Stålfelt, F. (1980). Foods of moose and roe deer at Grimsö in central Sweden. Results of rumen content analyses. *Swedish Wildlife Res*., **11**,171-247.

Garten, C. T., Gardner, R. H. and Dahlman, R. (1978). A compartment model of plutonium dynamics in a deciduous forest ecosystem. *Health Physics*, **34**, 611-639.

Garten, C.T., Jr. (1987) Technetium-99 cycling in deciduous forests: Review and ecosystem model development. *Environment International*, **13**, 311-321.

10.2 References for Sections 3 to 5 (semi-natural ecosystems)

Albers, B.P., Steindl, B.P., Schimmack, W. & Bunzl, K. (2000). Soil-to-plant and plant-to-cow's milk transfer of radiocaesium in alpine pastures: significance of seasonal variability. *Chemosphere*, **41**, 717-723.

Arctic Monitoring and Assessment Programme (AMAP) (1998). Assessment Report: *Arctic Pollution Issues*. Arctic Monitoring and Assessment Programme, Oslo.

Anderson, I., Lonsjo, H. & Rosen, K. (1992). Transfer of radiocaesium from mountain pasture land to vegetation and to grazing lambs in northern Sweden. Torshavn, Faroarna: Sjatte Nordiska Radioekologiseminariet.

Avila, R., Johanson, K.J. & Bergstrom, R. (1999). Model of the seasonal variations of fungi ingestion and Cs-137 activity concentrations in roe deer. *Journal of Environmental Radioactivity*, **46**, 99-112.

Bakunov, N.A., Panasenkova, O.I. & Drichko, V.F. (1998). ^{90}Sr, ^{137}Cs and Natural Radionuclides in the Ecosystem of a Deep Lake. *Russian Journal of Ecology*, **30**, 361-363.

Balonov M.I. (Ed.) (1999). Long-term consequences of potential radioactive contamination in the Northern areas. Interim report from the field work performed in 1998-1999 in cooperation with the Norwegian Radiation Protection Authority. St Petersburg: Research and Technical Centre “Protection”.

Balonov M.I. (Ed.) (2000). Long-term consequences of potential radioactive contamination in the Arkhangelsk region. Interim report from the field work performed in 1999-2000 in frame of AVAIL project. St Petersburg: Research and Technical Centre "Protection".

Belot Y., Roy M., Metivier H. (1996). Le Tritium: de l'environment a l'homme. Les Edition de Physique.

Beresford, N.A., Mayes, R.W., Crout, N.M.J., Dawson, J.M., Maceachern, P.J., Dodd, B.A., Soar, J.B., Barnett, C.L. & Lamb, C.S. (1997). Transfer of heavy metals from feeding stuffs and environmental sources into meat, milk and other foods of animal origin. Part 1. Arsenic, Cadmium and mercury. Final report. Ministry of Agriculture, Fisheries and Food. Grange-Over-Sands: Centre for Ecology and Hydrology.

Beresford, N.A., Mayes, R.W., Cooke, A.I., Barnett, C.L., Howard, B.J., Lamb, C.S & Naylor, G P.L. (2000). The importance of source dependent bioavailability in determining the transfer of ingested radionuclides to ruminant derived food products. *Environmental Science & Technology*, **34**, 4455-4462.

Beresford, N.A. (2002). Radiocaesium variability in upland sheep flocks. PhD. Thesis. University of Nottingham.

Beresford, N.A., Wright, S.M., Brown, J.E. & Sazykina, T. (Eds.) (2003). Transfer and Uptake Models for Reference Arctic Organisms. Deliverable for EPIC. EC Inco-Copernicus project ICA2-CT-2000-10032. Grange-over-Sands: Centre for Ecology and Hydrology.

Bishop, G.P, Beetham, C.J. & Cuff, Y.S. (1989). Review of literature for chlorine, technetium, iodine and neptunium. Epsom: Associated Nuclear Services.

Bunzl, K. & Kracke, W. (1984). Distribution of ^{210}Pb, ^{210}Po, stable lead and fallout ^{137}Cs in soil, plants and moorland sheep of a heath. *The Science of the Total Environment*, **39**, 143-159.

Bunzl, K. & Kracke, W. (1986). Accumulation of fallout ^{137}Cs in some plants and berries of the family Ericaceae. *Health Physics*, **50**, 540-542.

Bunzl, K. & Kracke, W. (1989). Seasonal variation of soil-to-plant transfer of K and fallout Cs-134, Cs-137 in peatland vegetation. *Health Physics*, **57**, 593-600.

Bunzl, K., Albers, B. P., Schimmack, W., Belli, M., Ciuffo, L. & Menegon, S. (2000). Examination of a relationship between Cs-137 concentrations in soils and plants from alpine pastures. *Journal of Environmental Radioactivity*, **48**, 145-158.

Calder, W.A. (1984). Size, function and life history. Cambridge, Harvard University.

Chamberlain, A.C. & Garland, J.A. (1991). Interception of radioactive fallout by vegetation. AERE R 13826. Harwell: UKAEA.

Copplestone, D., Johnson, M.S., Jones, S.R., Toal, M.E. & Jackson, D. (1999). Radionuclide behaviour and transport in a coniferous woodland ecosystem: vegetation, invertebrates and wood mice, Apodemus sylvaticus. *The Science of the Total Environment*, **239**, 96-109.

Copplestone, D., Bielby, S., Jones, S.R., Patton, D., Daniel, P. & Gize, I. (2001). Impact Assessment of ionising radiation on wildlife. R&D Publication 128. ISBN 1 85705590. Bristol: Environment Agency.

Coughtrey, P.J. & Thorne, M.C. (1983). Radionuclide distribution and transport in terrestrial and aquatic Ecosystems – A critical review of data. Volume 1. Rotterdam: A.A. Balkema.

Coughtrey, P.J., Jackson, D. & Thorne, M.C. (1985). Radionuclide distribution and transport in terrestrial and aquatic Ecosystems – A compendium of data. Volume 6. Rotterdam: A.A. Balkema.

Cristaldi, M., Ieradi,L. A., Mascanzoni, D. & Matei, T. (1991). Environmental impact of the Chernobyl accident: mutagenesis in bank voles from Sweden. *International Journal of Radiation Biology*, **59**(1), 31-40.

Crocker, D., Hart, A., Gurney, J. & McCoy, C. (2002). Methods For Estimating Daily Food Intake Of Wild Birds And Mammals. PROJECT PN0908. Central Science Laboratory. London: Department for Environment, Food & Rural Affairs.

Diercxsens, P., Deweck, D., Borsinger, N., Rosset, B. & Tarradellas, J. (1985). Earthworm contamination by PCB's and heavy metals. *Chemosphere*, **14**, 511-522.

Gaare, E. & Staaland, H. (1994). Pathways of fallout radiocaesium via reindeer to man. In: Nordic Radioecology - The transfer of radionuclides through Nordic ecosystems to man. 303-334. Dahlgaard, H. (Ed.) New York: Elsevier.

Galeriu, D. (1994). Transfer parameters for routine release of HTO, incorporation of OBT. Atomic Energy of Canada report, AECL 11052, COG -94-76. Chalk River, Province of Ontario, Canada: Atomic Energy of Canada.

Galeriu, D., Crout, N.M.J., Melintescu, A., Beresford, N.A., Peterson, S.R. & van Hees, M. (2001). A metabolic derivation of tritium transfer coefficients in animal products. *Radiation & Environmental Biophysics*, **40**, 325-334.

Galeriu, D., Melintescu, A., Beresford, N.A., & Crout, N.M.J. (2002). The derivation of tritium transfer parameters for farm animals on the basis of a metabolic understanding. *Radioprotection - Colloques*, **37**, 361-366.

Galeriu, D. & Belot, Y. (2002). A Standard Guide For Dose Assessment Of Routine Releases Of Tritium For Any Tritium Facility. Report WP3 IDRANAP 31-02/2002 available from: http://www.nipne.ro./idranap

Gaschak, S., Chizhevsky, I., Arkhipov, N.A., Beresford, N.A. & Barnett, C.L. (2003). The transfer of Cs-137 and Sr-90 to wild animals within the Chernobyl exclusion zone.

International Conference on the Protection of the Environment from the Effects of Ionizing Radiation. 6–10 October 2003, Stockholm, Sweden. IAEA.

Gastberger, M., Steinhausler, F., Gerzabek, M.H., Lettner, H. & Hubmer, A. (2000). Soil-to-plant transfer of fallout caesium and strontium in Austrian lowland and Alpine pastures. *Journal of Environmental Radioactivity*, **49**, 217-233.

Hakonson, T.E. & Nyhan, J.W. (1980). Ecological relationships of plutonium in southwest ecosystems. 403-419. In: Transuaranic elements in the environment. Hanson W.C. (Ed.) Washington: US Department of Energy.

Hakonson, T.E., Watters, R.L. & Hanson, W.C. (1981). The transport of plutonium in terrestrial ecosystems. *Health Physics*, **40**, 63-69.

Hendriks, A.J., W.-C Ma., Brouns, J.J., de Ruiter-Dijkman, E.M. & Gast, R. (1995). Modelling and monitoring organochlorine and heavy metal accumulation in soils, earthworms and shrews in Rhine Delta floodplains. *Archives of Environmental Contamination and Toxicology*, **29**, 115-127.

Higley, K.A., Domotor, S.L. & Antonio, E.J. (2003). A kinetic-allometric approach to predicting tissue radionuclide concentrations for biota, *Journal of Environmental Radioactivity*, **66**, 61-74.

Holtzman, R.B. (1966). Natural levels of ^{210}Pb, ^{210}Po and ^{226}Ra in humans and biota of the Arctic. *Nature*, **210**, 1094-1097.

Howard, B.J., Beresford, N.A. & Hove, K. (1991). Transfer of radiocaesium to ruminants in unimproved natural and semi-natural ecosystems and appropriate countermeasures. *Health Physics*, **61**, 715-725.

Howard, B.J., Wright, S.M., Barnett, C.L., Salbu, B., Ragnhild, L., Lind, O.C., Hove, K. & Skuterud, L. (2002). Long-term consequences of potential radioactive contamination in the Northern areas: Northern Norway. Final Report. CEH Project No T07050t9. Grange-over-Sands: Centre for Ecology and Hydrology.

International Atomic Energy Agency (IAEA) (1982). Generic models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases. Exposure of Critical Groups. Safety Series No. 57. Vienna: International Atomic Energy Agency.

International Atomic Energy Agency (IAEA) (1994). Handbook of transfer parameter values for the prediction of radionuclide transfer in temperate environments. IAEA TECDOC Series No. 364. Vienna.

International Atomic Energy Agency (IAEA) (2001a). Generic models for use in assessing the impact of discharges of radioactive substances. Safety Reports Series No. 19. Vienna: International Atomic Energy Agency.

International Atomic Energy Agency (IAEA) (2001b). Biomass programme, version beta 2. Vienna: International Atomic Energy Agency. (CD).

International Commission on Radiological Protection (ICRP) (1979). Limits for intakes of radionuclides by workers. ICRP publication 30. Part 1. Volume 2. Oxford: Pergamon Press.

Ireland, M.P. (1979). Metal accumulation by the earthworms Lumbricus Rubellus, Dendrobaena Veneta and Eiseniella Tetraedra living in heavy metal polluted sites. *Environmental Pollution*, **13**, 201-206.

Janssen, M.P. Glastra, M., P. & Lembrechts, J.f.m.m. (1996a). Uptake of Cs-134 from a sandy soil by two earthworm species: The effects of temperature. *Archives of Environmental Contamination and Toxicology*, **31**, 184-191.

Janssen, M.P. Glastra, M.P. & Lembrechts, Jfmm. (1996b). Uptake of caesium-134 by the earthworm species Eisenia foetida and Lumbricus rubellus. *Environmental Toxicology and Chemistry*, **15**, 873-877.

Johanson, K.J. (1994). Radiocaesium in game animals in the Nordic countries. In: Nordic Radioecology: the transfer of radionuclides through Nordic ecosystems to man. Dahlgaard, H. (Ed.). Studies in Environmental Science 62. New York: Elsevier.

Johanson, K.J. & Bergstrom, R. (1989). Radiocaesium from Chernobyl in Swedish moose. *Environmental Pollution*, **61**, 249-260.

Johanson, K. J. & Bergstrom, R. (1994). Radiocesium transfer to man from moose and roe deer in Sweden. *Science of the Total Environment*, **157**, 309-316.

Johanson, K.J., Bergstrom, R., Eriksson, O. & Erixon, A. (1994). Activity concentrations of Cs-137 in moose and their forage plants in mid-Sweden. *Journal of Environmental Radioactivity*, **22**, 251-267.

Karlsson, S., Bergström, U. & Meilli, M. (2001) Models for dose assessment. Models adapted to the SFR-area, Sweden. Svensk Kärnbräslehantering AB: Stockmolm.

Kauranen, P. & Miettinen, J.K. (1969). Po-210 and Pb-210 in the Arctic food chain and the natural radiation exposure of Lapps. *Health Physics*, **16**, 287-295.

Keith-Roach, M.J. & Livens, F.R. (Eds.). (2002). Radioactivity in the environment Volume 2: Interactions of microorganisms with radionuclides. 179-203. Oxford: Elsevier Science Ltd.

Killough, G.G. & P.S. Rohwer. (1978). A New Look at the Dosimetry of 14C Released to the Atmosphere as Carbon Dioxide. *Health Physics*, **34**, 141-159.

Litver, B.Ya., Nizhnikov, A.I., Ramzaev, P.V., Teplykh, L.A. & Troitskaya, M.N. (1976). Pb-210, Po-210, Ra-226, Th-228 in the biosphere of Far North of USSR. Moscow: Atomizdat. (In Russian).

Livens, F.R., Horrill, A.D. & Singleton, D.L. (1991). Distribution of radiocaesium in the soil-plant systems of upland areas of Europe. *Health Physics*, **60**, 539-545.

Lubashevsky, N., Balonov, M., Basalaeva, L., Bruk, G., Ivanova, N., Shvidko, N. & Shutov, V. (1993). Radioactive contamination of the Yamal Peninsula and assessment of radiation protection of its population. *Ecology*, **4**, 39-45. (In Russian).

Mahon, D.C. & Mathews, R.W. (1983). Uptake of Naturally-Occurring Radioisotopes by Vegetation in a Region of High Radioactivity. *Canadian Journal of Soil Science*, **63**, 281-290.

Maslov, V.I., Maslova, K.I. & Verkhovskaya, N. (1966). Characteristics of the radiological groups of mammals and birds of biogeocoenoses with high natural radiation. In: *Radiological Concentration Processes*. 561-571. Proceedings of an International Symposium Stockholm 25th -29th April, 1966. Pergamon Press: Oxford.

Matishov, G.G., Matishov, D.G., Szczypa, J. & Rissanen, K. (1994). Radionuclides in the Ecosystem of the Barents and Kara Seas Region. Apatity: Kola Science Center. (In Russian).

Miretsky G., Alekseev, P.V., Ramzaev, O.A., Teodorovich, I.E. & Shuvalov, I.E. (1993). New radioecological data for the Russian Federation (from Alaska to Norway). 269-272. In: Environmental Radioactivity in the Arctic and Antarctic. Strand, P. & Holm.E. Østerås: Norwegian Radiation Protection Authority.

Morgan, J.E. & Morgan, A.J. (1990). The distribution of cadmium, copper, lead, zinc and calcium in the tissues of the earthworm Lumbricus rubellus sampled from one uncontaminated and 4 Polluted Soils. *Oecologia*, **84**, 559-566.

Morgan, J.E. (1991). The metabolism of toxic metals by domestic animals – lead, cadmium, mercury and arsenic. ANS Report No. 2437-R1. Epsom: ANS consultants Ltd.

Morris, B. & Morgan, A.J. (1986). Calcium-lead interactions in earthworms- observations on Lumbricus terrestris sampled from a calcareous abandoned lead mine site. *Bulletin of Environmental Contamination and Toxicology*, **37**, 226-233.

Nagy, K.A. (2001). Food requirements of wild animals: predictive equations for free-living mammals, reptiles, and birds. *Nutrition Abstracts and Reviews, Series B*, **71**, 21-31.

Nelin, P. (1995). Radiocesium uptake in moose in relation to home-range and habitat composition. *Journal of Environmental Radioactivity*, **26**, 189-203.

Nelson, W., Beyer, R.L., Chaney, L. & Mulhern, B.M. (1982). Heavy metal concentrations in earthworms from soil amended with sewage sludge. *Journal of Environmental Quality*, **11**, 381-385.

Pálsson, S.E., Eglisson, K., P¢risson, S., Magn£sson, S.M., Olafsd¢ttir, E.D. & Indridason, K. (1994). Transfer of radiocaesium from soil and plants to reindeer in Iceland. *Journal of Environmental Radioactivity*, **24**, 107-125.

Peterson, S. R. &. Davis, P. A. (2002). Tritium Doses from Chronic Atmospheric Releases: A New Approach Proposed for Regulatory Compliance. *Health Physics*, **82**, 213-225.

Pietrzak-Flis, Z., Radwan, I., Rosiak, L. & Wirth, E. (1996). Migration of 137Cs in soils and its transfer to mushrooms and vascular plants in mixed forest. *Science of the Total Environment*, **186**, 243-250.

Pietz, R.I., Peterson, J.R., Prater, J.E. & Zenz, D.R. (1984). Metal concentrations in earthworms from sewage sludge amended soils at a strip mine reclamation site. *Journal of Environmental Quality*, **13**, 651-654.

Pokarzhevskii, A.D. & Krivolutzkii, D.A. (1997). Background concentrations of Ra-226 in terrestrial animals. *Biogeochemistry*, **39**, 1-13.

Pröhl, G. (Ed.). (2003). Dosimetric models and data for assessing radiation exposures to biota. Deliverable for FASSET. EU Contract No. FIGE-CT-2000-00102. Stockholm: Swedish Radiation Protection Institute.

Prosser, S.L. (1994). Review of data on the transfer of ruthenium-106 in the environment. NRPB-M513. Didcot: National Radiological Protection Board.

Ramzaev, P.V. (1967). Hygienic investigation of radiation situation in the Far North of the USSR attributed to global fallout. Dissertation of Doctor of Medical Science. Leningrad. (In Russian).

Rantavaara, A.H., (1990). Transfer of radiocaesium through natural ecosystems to foodstuffs of terrestrial origin in Finland. 202-209. In: Transfer of radionuclides in natural and semi-natural environments. Desmet, G., Nassimbeni P. & Belli M. (Eds.). New York: Elsevier.

Read, H.J. & Martin, M.H. (1993). The effect of heavy metals on populations of small mammals from woodlands in Avon (England); with particular emphasis on metal concentrations in Sorex Araneus L. and Sorex Minitus L. *Chemosphere*, **27**, 2197-2211.

Regional Centre for Sanitary Inspection (RCSI). (1974-1998). Annual Reports of the Regional Centre for Sanitary Inspection. Murmansk: Regional Centre for Sanitary Inspection. (In Russian).

Robbins, C. T. (1993). Wildlife feeding and nutrition. London: Academic Press.

Spencer, F.S. & Vereecken-Sheehan, L. (1994). HTO relationships between air, soil and vegetation and seasonal dependence of HTO deposition to soil. Ontario Hydro Report No. A-NBP-94-36-K. Toronto: Ontario Hydro Technology.

Spurgeon, D.J. (1996). Risk assessment of the threat of secondary poisoning by metals to predators of earthworms in the vicinity of a primary smelting works. Science of the *Total Environment*, **187**, 167-183.

Strand, P., Beresford, N.A. & Avila, R. Jones, S.R. & Larsson, C.R. (Eds.) (2001). Identification of candidate reference organisms from a radiation exposure pathways perspective. A Deliverable for FASSET. EU Contract No. FIGE-CT-2000-00102. Stockholm: Swedish Radiation Protection Institute.

Toal, M.E., Copplestone, D., Johnson, M.S., Jackson, D & Jones, S.R. (2002a). Quantifying ^{137}Cs aggregated transfer coefficients in a semi natural woodland ecosystem adjacent to a nuclear reprocessing facility. *Journal of Environmental Radioactivity*, **63**, 85-103.

Toal, M.E., Walker, L.A. & Shore R.F. (2002b). Modeling cadmium dynamics in the guts and tissues of small mammals: dose implications for predators. *Environmental Toxicology and Chemistry*, **21**, 2493-2499.

Thomas, P.A., Sheard, J.W. & Swanson, S. (1994). Transfer of Po-210 and Pb-210 through the lichen-caribou-wolf food-chain of Northern Canada. *Health Physics*, **66**, 666-677.

Troitskaya, M.N. (1981). Hygienic assessment of increased levels of exposure of population of the Far North. Dissertation of Doctor of medical sciences. Leningrad.

United States Environmental Protection Agency (USEPA) (1989). User's Guide for AIRDOS-PC, Version 3.0. EPA 520/6-89-035. Washington: Environmental Protection Agency.

United States Department of Energy (USDoE) (2002). A graded approach for evaluating radiation doses to aquatic and terrestrial biota. Technical Standard DoE-STD-1153-2002. Washington: United States Department of Energy.

Verhovskaya, I.N. (Ed.). (1972). Radioecological investigations in natural biogeocenoses. Moskow. (In Russian).

Wei chun, Ma. (1982). The influence of soil properties and worm related factors on the concentration of heavy metals in earthworms. *Pedobiologica*, **109**, 1-9.

Wei-chun, Ma. (1987). Heavy metal accumulation in the mole (Talpa europea) and earthworms as an indicator of metal bioavailability in terrestrial environments. *Bulletin of Environmental Contamination and Toxicology*, **39**, 933-938.

Wirth, E., (1982). The Applicability of the C-14-Labeled Specific Activity Model. *Health Physics*, **43**, 919-922.

10.3 References for Section 6 (Review of physical transport models)

Abreu SN, Pereira ME, Duarte AC (1998). The Use of a Mathematical Model to Evaluate Mercury Accumulation in Sediments and Recovery Time in a Coastal Lagoon (Ria de Aveiro, Portugal. *Water Science and Technology*, **37**(6-7), 33-38.

Aldridge JN (1998). CSERAM: A Model for Prediction of Marine radionuclide Transport in both Particulate and Dissolved Phases. *Radiat. Prot. Dosim.*, **75**(1-4), 99-103.

Beaugelin-Seiller *et al.* (2002). *Health Physics*, **83** (4), 539-542.

BIOMOVS (1990). Dynamics within Lake Ecosystems. Ed: Björn Sundblad. Swedish Radiation Protection Institute, Stockholm (BIOMOVS Technical Report No. 12).

Boyer P. and Duchesne S. (2003). CASTEAUR: Objectives, Steps and Prototype Presentation. In: Evaluation and Network of EC-Decision Support Systems in the field of Hydrological Dispersion Models and of Aquatic radioecological Research (Evanet - Hydra): First Meeting, Rome, 20 - 22 February 2002; Environ. Modelling & Software (In Press).

Charles D, Jones M, Copper JR (1990). Radiological impact on EC member states of routine discharges into Northern European Waters report of working groups IV of CEC Project MARINA, NRPB-M172. National Radiological Protection Board, Chilton.

Danish Hydaulic Institute (DHI) (2000). MIKE11, a modelling system for rivers and channels. Users Guide.

Danish Hydaulic Institute (DHI) (2001). MIKE21, a modelling system for estuaries, coastal waters and seas.

Djenidi S, Nihoul JCJ, Garnier A (1987). Modèle mathématique du transport des radionucléides sur le plateau continental nord-européen, in Radionuclides: A tool for oceanography, 373-383.

ETOPO-5, (2002). “ETOPO-5” Five Minute Gridded World Elevation. U.S. National Geophysical Data Center (NGDC). Boulder, Colorado, USA. http://www.grid.unep.ch/datasets/earth.html.

European Commission (EC) (1990). The radiation exposure of the population of the European Community from radioactivity in North European marine waters. RP 47, EUR 12483.

European Commission (EC) (1995). Methodology for assessing the radiological consequences of routine releases of radionuclides to the environment. RP 72, European Commission, Luxembourg, EUR 15760 EN.

European Commission (EC) (1997). Evolution of the radiological situation around the nuclear reactors with spent fuel which have been scuttled in the Kara Sea, European Commission, Luxembourg, EUR 17634 EN.

Europeiska Unionen (1996). Rådets direktiv 96/29/Euratom av den 13 maj 1996. Published in Europeiska gemenskapernas officiella tidning L 159, 29 juni 1996 (in Swedish).

Gardner R.H., Röjder B. and Bergström U. (1983). PRISM: A systematic method for determining the effect of parameter uncertainties on model prediction. Studsvik Energiteknik AB, Sweden (STUDSVIK/NW-83/555).

Gerdes R., M. Karcher F. Kauker, and C. Koeberle (2001): Prediction for the spreading of radioactive substances from the sunken submarine "Kursk" in the Barents Sea, *EOS transactions, AGU*, **82** (No. 23), 253-257.

Gleizon P., (2002). Ribble Estuary model VERSE programmers guide. Westlakes Scientific Consulting Report 000191/08.

Hamilton P., (1990). Modelling salinity and circulation for the Columbia river estuary, *Progress in Oceanography*, **25**, 113-156.

Hayter E.J. (1986). Estuarial sediment bed model, Estuarine cohesive sediment dynamics, Mehta A.J. ed.,326-359.

Heling R. (1997). LAKECO: Modelling the Transfer of Radionuclides in a Lake Ecosystem. *Radiat. Prot. Dosim.*, **73** (1-4), 191-194

Hilton J., Small S., Hornby D., Scarlett P., Harvey M., Simmonds J., Bexon A., and Jones A. (2002). Modelling the Combined Impact of Radionuclide Discharges Reaching Rivers. Environment Agency R&D Technical Report P3-068.

IASAP (1999). Radioactivity in the Arctic Seas, Report for the International Arctic Seas Assessment Project (IASAP). International Atomic Energy Agency, Vienna, IAEA-TECDOC-1075.

IBCAO (2001). International Bathymetric Chart of the Arctic Ocean (IBCAO). Jakobsson, M., N.Z. Cherkis, J. Woodward, R. Macnab, and B. Coakley. New grid of Arctic bathymetry aids scientists and mapmakers; Eos, Transactions, *American Geophysical Union*, **81** (No.9), p.89, 93, 96. http://www.ngdc.noaa.gov/mgg/bathymetry/arctic/arctic.html.

International Commission on Radiological Protection (ICRP) (1991). Recommendations of the International Commission on Radiological Protection. Publication 60. Annals of the ICRP 21. Oxford Pergamon Press (Oxford).

International Commission on Radiological Protection (ICRP) (1997). Age dependent doses to members of the public from the intake radionuclides. Part 5 Compilation of ingestion and inhalation dose efficients. ICRP Publication 72, Pergomon Press.

Iosjpe M., Brown J. & Strand P., (2002). Modified approach for box modelling of radiological consequences from releases into marine environment. *Journal of Environmental Radioactivity*, **60**, 91-103.

Karcher M.J., R. Gerdes, F. Kauker, C.Koeberle (2003): Arctic warming - Evolution and Spreading of the 1990s warm event in the Nordic Seas and the Arctic Ocean, *J. Geophys. Res.*, (in press).

Karcher M.J., S. Gerland, I. H. Harms, M. Iosjpe, H. E. Heldal and P. J. Kershaw (2003). The dispersion of ^{99}Tc in the Nordic Seas and the Arctic Ocean in the 1990s according to model results and observations, submitted to JER.

Karcher M.J. & Harms I.H., (2000). Estimation of water and ice fluxes in the Arctic for an improved box structure of the NRPA box model. In: Iosjpe M. (Ed.): Transport and fate of contaminants in the Northern Seas, NRPA, (2000).

Karcher M.J., Gerland S., Harms I. H., Iosjpe M., Heldaland H. E. and Kershaw P. J. (2002). The dispersion of 99Tc in the Nordic Seas and the Arctic Ocean in the 1990s according to model results and observations. In: Proceedigs from the international conference on radioactivity in the environment, 1-5 September 2002 in Monaco (Eds. P. Børretzen, T. Jølle & P. Strand), 511-515.

Karlsson S., Bergström, U. And Meili M. (2001). Models for dose assessments. Models adopted to the SFR-area, Sweden Swedish Nuclear Fuel and Waste Management Co, SKB TR-01-04 Stockholm.

Kauker, F., R. Gerdes, M. J. Karcher, C. Koeberle and J. Lieser (2003): Variability of Northern hemishere sea ice: A combined analysis of model results and observations for the period 1978-2001, *J. Geophys. Res.*, (in press).

Kirchner T.B. (1989). TIME-ZERO, The integrated modelling environment, Ecological Modelling, 47:33-52.

Klos R.A., Sinclair J.E., Torres C., Bergström U., Galson D.A. (1993). An international Code Intercomparison Exercise on a Hypothetical Safety Assessment Case Study for Radioactive Waste Disposal Systems. PSACOIN level 1 B. Intercomparison, Probabilistic System Assessment Group. Nuclear Energy Agency. OECD.

Koeberle C., and R. Gerdes (2003). Causes of Arctic sea ice volume variability, submitted to J. Clim.

Lawson G., Attwood C.A., Simmonds J.R., Fayer C.A. (1991). Assessment of the radiological impact of radiological liquid waste discharges to the Molse Nete River. National Radiological Protection Board, Chilton.

Lepicard S. (1999). Poseidon: a software for assessing radiological impacts of radionuclides in european seawater environment, Southport 99, proceedings of the 6th SRP international symposium, 343-346.

Lepicard S., Raffestin D., Rancillac F. (1998). POSEIDON: A dispersion computer code for assessing radiological impacts in a European sea water environment. *Radiation Protection Dosimetry*, **75** (Nos. 1-4), 79-83.

Liu Y.P., Millward G.E., Harris J.R.W. (1998). Modelling the distribution of dissolved Zn and Ni in the Tamar Estuary using hydrodynamics coupled with chemical kinetics. *Estuarine, Coastal and Shelf Science*, **47**, 535-546.

Mehta A.J., Parchure T.M., Dixit J.G., Ariathurai R. (1982). Resuspension potential of deposited cohesive sediment beds, Estuarine Comparison, Kennedy V. ed., Academic Press, New York, 591-609.

Maul P.R., Watkins B.M., Clark K.J., Waters D.J. (1997). Aquatic Dispersion Models for radiological assessment. QuantiSci Report IE6008A-1 Version 1.0.

Needle C.L. (1993). Application of a general purpose hydrodynamics model to the Northwest European shelf seas. MSc report on industrial project, University of Aberdeen.

Nielsen S.P., Iosjpe M. and Strand P. (1997). Collective doses to man from dumping of radioactive waste in the Arctic Seas. *The Science of the Total Environment*, **202**, 135-146.

NRPB - CEA (1979). Methodology for evaluating the radiological consequences of radioactive effluents released in normal operations. Commission of the European Communities, Doc. No.V/3865/1/79.

Persson B. and Nilsson M. (1978). Review of the report “Final steps of the nuclear fuel cycle” part IV – Safety assessment especially concerning radioecological transport. Statens strålskyddsinstitut, Stockholm, PM.

Pham K.H., Martin J.M., Garnier J.M., LI Z., Boutier B., Chiffoleau J.F. (1997). On the possibility of using the commercially available ECoS model to simulat Cd distribution in the Gironde estuary (France). *Marine Chemistry*, **58**, 163-172.

Popov A.G., Heling R. (1996). Aquatic Transfer Models for RODOS RETRACE and LAKECO. Proceedings of the First International Conference, "The radiological consequences of the Chernobyl accident" p.1189-1192, ISSN 1018-5593.

Postma L., de Kok J.M., Markus A.A., van Pagee J.A. (1987). Long term and seasonal water quality modelling of the North Sea and its coastal waters, International Council for the Exploration of the sea, ICES, Hydrographic committee.

Povinec P. (2002). Personal communication.

Prandle D. (1984). A modelling study of the mixing of ^{137}Cs in the seas of the European continental shelf. *Phil. Trans. R. Soc. Lond.*, **A310**, 407-436.

Punt A.P., Millward G.E., Harris J.R.W. (2002). Modelling solute transport in the Tweed Estuary, UK using ECoS. *Science of the Total Environment*, (In press).

Salomon J.C., Guegueniat P., Orbi A., Baron Y. (1987). A Lagrangian model for long term tidally induced transport and mixing. Verification by artificial radionuclide concentration, in Radionuclides: A tool for oceanography, 384-394.

Schaeffer R. (1975). Consequences du deplacement des sediments sur la dispersion des radionuclides. In Proceedings of the Conference on impacts of Nuclear Releases into the Aquatic Environment, Otaniemi 1975. Vienna, IAEA, p263 (1976)IAEA-SM 198/4.

Schönfeld W. (1995). Numerical simulation of the dispersion of artificial radionuclides in the English channel and the North Sea. *Journal of Marine Systems*, **6**, 529-544.

Smith C., Vives Lynch S.M. (2002). Assessment of dose per unit discharge factors for accidental non-pipeline liquid discharges from BNFL Sellafield. Westlakes Scientific Consulting Report 010054/01.

Stripling S (1994). A review of transport models in the marine environment. PEP Research and Consultancy Limited.

Tappin A.P., Harris J.R.W., Uncles R.J. (2002). The fluxes and transformations of suspended particles, particulate organic carbon, particulate nitrogen, nitrate, nitrite and ammonium in the Humber estuarine system (UK) from 1994 to 1996: results from an integrated monitoring and modelling study. *Science of the Total Environment*, (In Press).

Thiessen K.M., Hoffman F.O., Rantavaara A., Hossain S. (1997). Environmental models undergo international test: The science and art of exposure assessment modelling were tested using real-world data from the Chernobyl accident. *Environmental Science and Technology*, **31**(8), 358A-363A.

Watts S.J., Gleizon P. (2002). Ribble Estuary model VERSE users guide. Westlakes Scientific Consulting Report 000191/06.

Welsh S. (1992). Assessment and management of risks to the environment. Major hazards onshore and offshore (Symposium Series No.130). Institute of Chemical Engineering.

Wilkins B.T., Simmonds J.R., Cooper J.R. (1994). Assessment of the present and future implications of radioactive comtamination of the Irish Sea coastal region of Cumbria. NRPB –R267.

Zheleznyak M., Donchytz G., Hygynyak V., Marinetz A., Lyashenko G., Tkalich P. (2000a). RIVTOX - one dimensional model for the simulation of the transport of radionuclides in a network of river channels. RODOS-WG4-TN (97)05.

Zheleznyak M., Shepeleva T., Mezhueva I. (2000b). COASTOX - two-dimensional model of lateral-longitudinal distribution of radionuclides in water bodies. RODOS-WG4-TN (97)07.

10.4 References for Section 7 (CF review for marine envrinonment)

Aarkrog , A. , H. Dahlgaard, L. Hallstadius, E. Holm, S. Mattsson and J. Rioseco, (1986). Time trend of ^{99}Tc in seaweed from Greenland waters. In: Technetium in the Environment, Desmet. G. and C. Myttenaere, Eds., Elsevier, 69-78.

Aarkrog, A., Boelskifte, S., Dahlgaard, H., Duniec, S., Hallstadius, L., Holm, E., and Smith, J. N. (1987). Technetium-99 and cesium-134 as long distance tracers in Arctic wateres. *Esturine, Coastal and Shelf Science*, **24**, 637-647.

Berrow, S. D., Long, S. C., McGarry, A. T., Pollard, D., Rogan, E. and Lockyer, C. (1998). Radionuclides (Cs-137 and K-40) in harbour porpoises Phocoena phocoena from British and Irish Coastal waters. *Marine Pollution Bulletin*, **36** (No. 8), 569-576.

Boisson, F., Hutchins, D. A., Fowler, S. W., Fisher, N. S. and Teyssie, J.-L. (1997). Influence of temperature on the accumulation and retention of 11 radionuclides by the marine alga Fucus vesiculosus (L.). *Marine Pollution Bulletin*, **35** (Nos. 7-12), 313-321.

Bonotton, S., Robbrecht, V., Nuyts, G., Cogneau, M. and Van Der Ben, D. (1988). Uptake of Technetium by marine algae autoradiographic localization. *Marine Pollution Bulletin*, **19** (No. 2), 61-65.

Brown, J. E., Kolstad, A. K., Brungot, A. L., Lind, B., Rudjord, A. L., Strand, P., Føyn, L. (1999). Levels of Tc-99 in seawater and biota samples from Norwegian Coastal waters and adjacent seas. *Marine Pollution Bulletin*, **38** (No. 7), 560-571.

Brown, J. and Iosjpe M. (2001). Development of Assessment tools. In : Radioactivity in the marine environment 1999. Strålevern Rapport (NRPA), 2001:9. Norwegian radiation Protection Authroity, Østerås, Norway, pp.39.

Busby, R., McCartney, M. and McDonald, P. (1997). Technetium-99 concentration factors in Cumbrian seafood. Radioprotection – Colloques, 32, C2.

Carvalho, F. P. (1988). Po-210 in marine organisms: A wide range of natural radiation dose domains. *Radiation Protection Dosimetry*, **24** (No 1/4), 113-117.

Fernando, P. C. and S. W. Fowler (1993). An experimental study on the bioaccumulation and turnover of polonium-210 and lead-210 in marine shrimp. *Mar. Ecol. Prog. Ser.*, 102, 125-133.

Fisher, N. S., Fowler, S. W., Boisson, F., Carroll, J., Rissanen, K., Salbu, B., Sazykina, T. G. and Sjoeblom, K. L. (1999). Radionuclide bioconcentration factors and sediment partition coefficients in Arctic Seas Subject to contamination from dumped nuclear wastes. *Environ. Sci. Technol.*, **33**, 1979-1982.

Franić, Z. and Lokobauer, N. (1993). Sr-90 and Cs-137 in pilchards from the Adriatic Sea. *Arh hig rada toksikol*, **44**, 293-301.

Germain, P., Gandon, R., Masson, M. and Guèguèniat, P. (1987). Experimental studies of the transfer of neptunium from sea water to sediments and organisms (Annelids and molluscs). *J. Environ. Radioactivity*, **5**, 37-55.

Germain, P., Leclerc, G., Le Cavelier, S., Solier, L. and Baron, Y. (2000). Évolution spatio-temporelle des concentrations, des rapports isotopiques et des facteurs de concentration du plutonium dans une espèce d'algue et deux espèces de mollusques en Manche. *Radioprotection*, **35** (No. 2), 175-200.

Gomez, L. S., Marietta, M. G. and Jackson, D. W. (1991). Compilation of selected marine radioecological data for the formerly utilized sites remedial action program: Summaries of available radioecological concentration factors and biological half-lives. Sandia National Laboratories Report SAND89-1585 RS-8232-2/71343. Albuquerque, New Mexico, Sandia National Laboratories.

Guary, J. C., Fowler, S. W. (1990). Experimental study of the transfer of transuranium nuclides in marine decapod crustaceans. *Marine Ecology Progress Series*, **60**, 253-270.

Güngör, N., Tuğrul, B., Topcuoğlu, S., Güngör, E. (2001). Experimental studies on the biokinetics of Cs-134 and Am-241 in mussels (Mytilus galloprovincialis). *Environmental International*, **27**, 259-264.

Harrison, F. L. (1986). Laboratory- determined concentration factors and elimination rates of some anthropogenic radiocuclides in marine vertebrates and invertebrates. Environmental Protection Agency Report EPA 520/1-85-015. Environmental Protection Agency, Washington DC, USA.

Heldal, H.E., Stupakoff, I., & Fisher, N.S. (2001). Bioaccumulation of 137Cs and 57 Co by five marine phytoplankton species. *J. of Environmental Radioactivity*, **57**, 231-236.

Holm, E., Ballestra, S., Lopez, J. J., Bulos, A., Barci-Funel G. and Ardisson, G. (1991). Transuranium elements in macroalgae at Monaco following the Chernobyl accident. *Eur. J. Solid State Inorg. Chem. T.*, **28**, 375-378.

Holm, E., Ballestra, S., Lopez, J. J., Bulos, A., Whitehead, N. E., Barci-Funel G. and Ardisson, G. (1994). Radionuclides in macro algae at Monaco following the Chernobyl accident. *Journal of Radioanalytical and Nuclear Chemistry, Articles*, **177** (No. 1), 51-72.

Hurtgen, C., Koch, G., van der Ben, D. and Bonotto, S. (1988). The determination of technetium-99 in the brown marine alga Fucus spiralis collected along the Belgian Coast. *The Science of Total Environment*, **70**, 131-149.

International Atomic Energy Agency (IAEA) (1985). Sediment Kds and concentration factors for radionuclides in the marine environment. IAEA, Technical Report Series No. 247, International Atomic Energy Agency, Vienna.

International Atomic Energy Agency (IAEA) (1999). Radioactivity in the Arctic Seas, report for the International Arctic Seas Assessment Project (IASAP). IAEA, VIENNA, 1999. IAEA-TECDOC-1075.

International Atomic Energy Agency (IAEA) (in press). Sediment K_ds and Concentration Factorsfor Radionuclides in the Marine Environment, Technical Reports Series No. 247 Revised, International Atomic Energy Agency, Vienna.

Masson, M., van Weers, A. W., Groothuis, R. E. J., Dahlgaard, H., Ibbett, R. D. and Leonard, K. S. (1995). Time series for sea water and seaweed of Tc-99 and Sb-125 originating from releases at La Hague. *Journal of Marine Systems*, **6**, 397-413.

Mayr L., Moraes R., Lopes M.A., Vicente C., Mauro J.N. (1988). Transit and absorption of nuclear industry derivatives by marine biota (Transito e absorçao de radionuclideos na biota marinha derivados de industria nuclear). In: Proc. 2. General Congress of Nuclear Energy, Rio de Janeiro (Brazil), 24-29 Apr. 1998. Vol. 3: 281-293.

McDonald, P., Cook, G. T., Baxter, M. S. (1992). Natural and anthropogenic radioactivity in coastal regions of the UK. *Radiation Protection Dosimetry*, **45** (No. 1/4), 707-710.

McDonald, P., Baxter, M. S. and Fowler, S. W (1993). Distribution of radionuclides in mussels, winkles and prawns. Part 1. Study of organisms under environmental conditions using conventional radio-analytical techniques. *J. Environ. Radioactivity*, **18**, 181-202.

Meinhold A. F. and Hamilton L. D. (1991). Radium concentration factors and their use in health and environmental risk assessment. BNL - 45035. Brookhaven National Laboratory, New York.

Mitchell, P.I., Vives Batlle, J., Ryan, T.P., McEnri, C., Long, S., O'Colmain, M., Cunningham, J.D., Caulfield, J.J., Larmour, R.A. and Ledgerwood, F.K. (1991), ‘Plutonium, americium and radiocaesium in sea water, sediments and coastal soils in Carlingford Lough’, In: Radionuclides in the Study of Marine Processes, P.J. Kershaw and D.S. Woodhead (Eds.), Elsevier Applied Science, London, pp. 265-275.

Mitchell, P.I., Vives Batlle, J., Ryan, T.P., McEnri, C., Long, S., O‘Colmain, M., Cunningham, J., Caulfield, J.J., Larmour, R.A. and Ledgerwood, F.K. (1992), ‘Artificial Radioactivity in Carlingford Lough’, Report published jointly by the Radiological Protection Institute of Ireland and the Dept. of the Environment for Northern Ireland, Dublin, 37 pp.

Osvath, I., Bologa, A. and Dovlete, C. (1990). Environmental Cs-137 concentration factors for Black Sea biota. Preliminary data. *Rapp. Comm. Int. Mer Medit*, **32**, 320.

Rheinfelder, J.R. and Fisher, N.S. (1991), 'The assimilation of elements ingested by marine copepods', *Science*, **251**, 794-797.

Ryan, T. P., Dowdall, A. M., McGarry, A. T., Pollard, D. and Cunningham, J. D. (1999). Po-210 in Mytilus edulis in the Irish marine environment. *Journal of Environmental Radioactivity*, **43**, 325-342.

Sazykina, T. G. (1998). Long-distance radionuclide transfer in the Arctic Seas related to fish migrations. *Radiation Protection Dosimetry*, **75** (Nos. 1-4), 219-222.

Shunhua C, Qiong S and Xiaokui Z (1997). Effect of body size on accumulation and distribution of 125I in the green mussel (Perna viridis). Report CNIC-01210, China Nuclear Information Centre, Beijing, BJ (China), Oct 1997, 8 pp.

Skwarzec, B. and Bojanowski, R. (1988). Po-210 content in sea water and its accumulation in southern Baltic plankton. *Marine Biology*, **97**, 301- 307.

Skwarzec, B. and Falkowski, L. (1988). Accumulation of Po-210 in Baltic invertebrates. *Journal of Environmental Radioactivity*, **8**, 99-109.

Smith, V., Fegan, M., Pollard, D., Long, S., Hayden, E., Ryan, T. P. (2001). Technetium-99 in the Irish marine environment. *Journal of Environmental Radioactivity*, **56**, 269-284.

Steele, A. K. (1990). Derived concentration factors for Cs-137 in edible species of North Sea fish. *Marine Pollution*, **21** (No. 12), 591-594.

Stepnowski, P. and Skwarzec, B. (2000). A comparison of Po-210 accumulation in molluscs from the southern Baltic, the coast of Spitsbergen and Sasek Wielki lake in Poland. *Journal of Environmental Radioactivity*, **49**, 201-208.

Strand, P., Beresford, N., Avila, R., Jones, S.R. Larsson, C-M. (2001). Idenitification of candidate reference organisms form a radiation exposure pathways perspective. A Deliverable Report for EC EURATOM Project (FIGE-CT-2000-00102). Norwegain Radiation Proectionn Authority, Østerås, Norway.

Till, J. E. and Mayer, H. R. (1983). "Radiological Assessment: A Textbook on Environmental Risk Analysis", NUREG/CR-3332, ORNL-5968, U.S. Nuclear Regulatory Commission, Washington, D. C.

Topcuoğlu, S. (2001). Bioaccumulation of cesium-137 by biota in different aquatic environments. *Chemosphere*, **44**, 691-695.

Vives i Batlle (1993). 'Speciation and Bioavailability of Plutonium and Americium in the Irish Sea and other Marine Ecosystems', PhD Thesis, National University of Ireland, 347 pp.

Whitehead NE, Ballestra S, Holm E, Huynh-Ngoc L. (1998). Chernobyl radionuclides in shellfish. *J. Environ. Radioact.*, **7**(2), 107-121.

10.5 References for Section 8 (biokinetic allometric in the marine environment)

Dommasnes, A, Christensen, V.,Ellertsen, B., Kvamme, C., Melle, W., Nøttestad, L., Pedersen, T., Tjelmeland, & S., Zeller, D. (2001). An ECOPATH model for the Norwegian Sea and Barents Sea. In Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. Fisheries Centre Research Reports 2001 Volume 9 Number 4. S. Guénette, V. Christensen & D. Pauly (eds.), pp. 213-240.

Fisher, N.S. (2002). Advantages and problems in the application of radiotracers for determining bioaccumulation of contaminants in aquatic organisms. In Proceedings from International Conference on radioactivity in the Environment, Monaco, 1-5 September, 2002, P. Børretzen, T. Jølle & P. Strand (eds.), pp. 573-576.

Innes, S., Lavigne, D.M., Earle, W.M., *et al.* (1987). Feeding rates of seals and whales. *J. Anim. Ecol.*, **56**, 115-130.

International Atomic Energy Agency (IAEA) (in press). Sediment K_ds and Concentration Factorsfor Radionuclides in the Marine Environment, Technical Reports Series No. 247 Revised, International Atomic Energy Agency, Vienna.

Internationl Commision on Radiological Protection (ICRP) (1973). ICRP Publication 20, ICRP Task Group on Alkaline Earth metabolism in Adult Man. Pergamon Press, Oxford, pp.92.

Internationl Commision on Radiological Protection (ICRP) (1979). ICRP Publication 30; Part 1. Limits of intakes of radionuclides by workers. Pergammon Press, Oxford, pp. 116.

Internationl Commision on Radiological Protection (ICRP) (1980). ICRP Publication 30; Part 2. Limits of intakes of radionuclides by workers. Pergammon Press, Oxford, pp. 71.

Internationl Commision on Radiological Protection (ICRP) (1981). ICRP Publication 30; Part 3 (including addendum to Parts 1 & 2). Limits of intakes of radionuclides by workers. Pergammon Press, Oxford, pp. 124.

Internationl Commision on Radiological Protection (ICRP) (1988). ICRP Publication 30; Part 4. Limits of intakes of radionuclides by workers. Pergammon Press, Oxford, pp. 163.

Landrum, P.F, Lee, H. & Lydy M.J. (1992). Annual Review: Toxicokinetics in aquatic systems: model comparison and use in hazard assessment. *Environ. Toxicol. Chem.*, **11**, 1709-1725.

Lloyd, R.D. Mays, C.W., Atherton, D.R., & Bruenger, F.W. (1975). ^{210}Pb studies in beagles. *Health Physics*, **28**, 575-583.

Nagy, K.A. (1987). Field metabolic rate and food requirement scaling in mammals and birds. *Ecol. Monogr.*, **57**, 111-128.

Sample, B.E., Aplin, M.S., Efroymson, R.A., Suter II, G.W., & Welsh, C.J.E. (1997). Methods and tools for estimation of the exposure of terrestrial wildlife to contaminants. ORNL/TM-13391. Oak Ridge National Laboratory, Oak Ridge, Tenesee, U.S.A., pp.113.

Thomann, R.V. (1981). Equilibrium model of fate of micorcontaminants in divserse aquatic food-chains. *Can. J. Fish. Aquat. Sci.*, **38**, 280-296.

USDoE (2002). A graded approach for evaluating radiation doses to aquatic and terrestrial biota. U.S. Department of Energy. Technical Standard DOE-STD-1153-2002. Washington D.C., U.S.A..

Whicker, F.W. & Schultz, V. (1982). Radiecology: Nuclear Energy and the environment. Vol. 2. CRC press, Inc. Boca Raton, Florida.

10.6 References for Section 9 (River modelling)

Baes III C.F., Sharp R.D., Sjoreen A.L., Shor R.W. (1984). A review and analysis of parameters for assessing transport of environmentally released radionuclides through agriculture, ORNL-5786.

Coughtrey P.J., Jackson D., Thorne M.C. (1985). Radionuclide distribution and transport in terrestrial and aquatic ecosystems. A.A Balkema eds.

Huang W.W. & Mouchel J.M. (1995). Piren Seine report 1994 III, IBM, Montrouge, Noisy-le-Grand, France.

International Atomic Energy Agency (IAEA) (2001). Generic models for use in assessing the impact of discharges of radioactive substances to the environment. Safety reports series No. 19, IAEA Vienna.

Shafer M.M., Overdier J.T., Hurley J.P., Armstrong D., Webb D. (1997). *Chemical Geology*, **136**, 71-97, Elsevier eds.