Currently available internal and external dose models

A deliverable report for EPIC
- Environmental Protection from Ionizing Contaminants in the Arctic

Project ICA2-CT-2000-10032

Edited by
V.Yu. Golikov & J. Brown

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Statens strålevern

REPORT ON CURRENTLY AVAILABLE INTERNAL AND EXTERNAL DOSE MODELS

A deliverable report for EPIC (<u>E</u>nvironmental <u>P</u>rotection from <u>I</u>onizing <u>C</u>ontaminants)

Project ICA2-CT-2000-10032

Edited by V.Yu. Golikov[1] & J. Brown[3]

Contributors:

A. Vlasov[1], I. Kryshev[2], T. Sazykina[2], Per Strand[3],

[1]Institute of Radiation Hygiene, 8 Mira Str., St. Petersburg, 197101, Russia.
[2]Scientific & Production Association "TYPHOON", 82 Lenin Av., Obninsk, Kaluga Region, 249038, Russia.
[3]Stratens Strålevern, Grini N?ringspark 13, P.O. Box 55, N1332, Østerås, Norway.

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Environmental Protection from Ionising Contaminants (EPIC)

To date, the protection of the environment from radiation is based on the premise that if Man is protected from harm, then all other components of the ecosystem will not be at risk. However, this has been increasingly questioned on the basis that it is not always true, it is inconsistent with environmental protection standards for other hazardous materials and conflicts with the recommendations of some international advisory bodies. The aim of the EPIC project is to develop a methodology for the protection of natural populations of organisms in Arctic ecosystems from radiation. This will be achieved by derivation of dose limits for different biota. The project therefore aims to (i) collate information relating to the environmental transfer and fate of selected radionuclides through aquatic and terrestrial ecosystems in the Arctic; (ii) identify reference Arctic biota that can be used to evaluate potential dose rates to biota in different terrestrial, freshwater and marine environments; (iii) model the uptake of a suite of radionuclides to reference Arctic biota; (iv) development of a reference set of dose models for reference Arctic biota; (v) compilation of data on dose-effects relationships and assessments of potential radiological consequences for reference Arctic biota; (vi) and integration of assessments of the environmental impact from radioactive contamination with those for other contaminants.

The EPIC project is funded under the EC Inco-Copernicus research programme and is co-ordinated by the Norwegian Radiation Protection Authority; project partners:

- Centre for Ecology & Hydrology, CEH-Merlewood, Grange-over-Sands, UK.
- Institute of Radiation Hygiene, St Petersburg, Russia.
- Scientific Production Association TYPHOON, Obninsk, Russia.

For further information on the EPIC project contact Dr. Per Strand (per.strand@nrpa.no).

Executive Summary

This report reviews some of the most commonly used models in the derivation of doses to biota. Dosimetry models have been developed to the greatest extent for aquatic organisms whereas application in terrestrial environments has been limited. Robust Dose calculations require input data relating to the organism's dimensions; concentrations and distributions of contamination in the biota's habitat; distribution of internal contamination; and the location of the organism in the surrounding media. In many cases, organisms are represented as geometrical figures such as spheres or ellipsoids. Radionuclides are normally assumed to be uniformly distributed throughout the organism, thus the resulting internal dose is calculated as an average value for the whole organism. For calculation of external doses, a static system may be assumed where a fixed target-source configuration is modelled. In more realistic formulations, the fractional occupancy of organisms per specified habitat may also be considered.

The damage induced by radiation is dependent upon the radiation type (α, β, or γ) and therefore radiation weighting factors may need to be invoked. The final choice of radiation weighting factor for alpha particles will depend on the selection of reference organism, end-point and dose (or dose-rate) range. Absorbed dose should be split into low LET and high LET define components in order to facilitate the incorporation of a radiation weighting factor once consensus has been achieved.

The accuracy of the dose rates estimations are generally limited by uncertainties in the parameters used in radionuclide transfer models e.g. concentration and distribution coefficients of different radionuclides in the media surrounding the organisms. Earlier studies have used reasonable assumptions to derive dose conversion factors for a number of radionuclides. Examples are given in this report where dose rates from both natural and artificial radionuclides have been estimated in selected Arctic marine environments. It is recognised that the inherent simplifications connected with the assumptions used in dose calculations leads to overestimated dose rates in many cases; gross over conservatism may not be desirable if results are to be used to regulate the nuclear industry.

More research is required to develop universal computer based models, which will allow both the modelling of radionuclide distribution in a specified environment and the subsequent estimation of doses for organisms with any shapes and sizes to be made. The algorithms of calculations should include both up-to-date data bases of constants for dose calculations and more realistic phantoms of plants and animals with, for example, separate internal organs.

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1. Introduction

Biota within the European Arctic are exposed to ionising radiation as a consequence of the routine operation of nuclear facilities, high levels of natural radioactivity, sites of peaceful nuclear explosions and fallout from accidental releases/above grounds weapons testing. In addition, the European Arctic contains a number of potential sources of radiological contamination including, nuclear power and reprocessing plants and civil and military nuclear powered vessels. Assessment of the exposure (or potential exposure) of biota following the release of radioactivity to the environment poses serious difficulties and thus simplifications are required. A simplified methodology could involve the selection of reference organism types that can be shown to be representative of large components of common ecosystems. In this context, the term *reference organism* has recently been defined as "a series of entities that provides a basis for the estimation of the radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects" [Larsson *et al.* 2002]. The reference organism approach provides a means of reducing the assessment to manageable proportions and may allow logical links/associations between sets of data attributed to different organism types to be established. In this way some insight into the potential environmental impacts of ionising radiation may be derived for components of the environment for which data are poor or absent. A criterion for the selection of reference organisms for the European Arctic, the area of concern within this work, has been detailed in a previous deliverable and suitable organism identified [Beresford *et al.* 2001]. Our second task within the EPIC project involves the development of a set of dose models for selected *reference* flora and fauna. The aim of this report is to review the current methodologies for the estimation of doses to biota.

2. Pathways of exposure for plants and animals to ionizing radiation

All organisms are exposed to radioactivity of natural and anthropogenic origin. Anthropogenic radionuclides can enter the environment through atmospheric wet and dry deposition and/or discharges to the water and give rise to radiation doses to biota through a variety of exposure pathways. Wildlife can be exposed to ionizing radiation (α, β and γ) though a number of different routes including:

- External exposure from contaminated air, water, soil or vegetation (flora and fauna);
- Internal exposure due to root uptake and foliar absorption (flora);
- Internal exposure due to inhalation of resuspended material and gaseous radionuclides (fauna);
- Internal exposure due to ingestion of plant material, animal material, soil and water (fauna).

There are many interactions between biota and their surroundings, which may influence the uptake and transfer of radionuclides. Radionuclides may be transferred through the food chain from the soil or sediment compartments through different trophic levels, e.g. plant uptake, into herbivores, omnivores, carnivores and higher predators.

Representations of some of the pathways of transfer and external exposure are shown in Figures 1 and 2, following atmospheric and aquatic discharges, respectively.

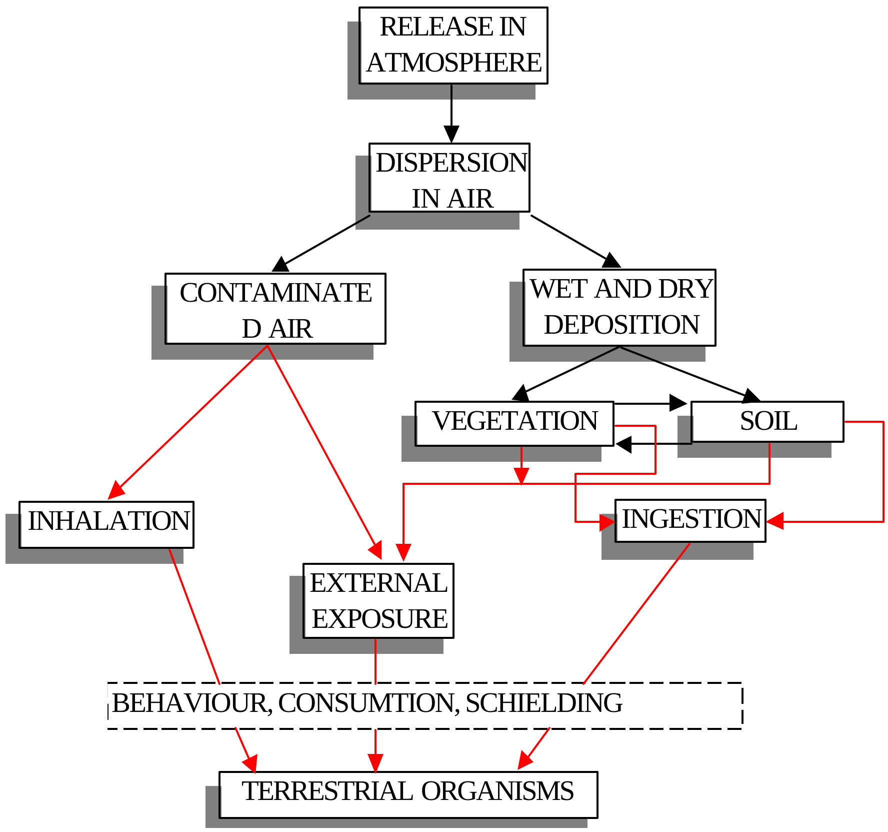

Figure 1. Schematic representation of exposure pathways of terrestrial organisms following the release of radionuclides to the atmosphere (black lines correspond to the transfer process, red lines correspond to the exposure process)

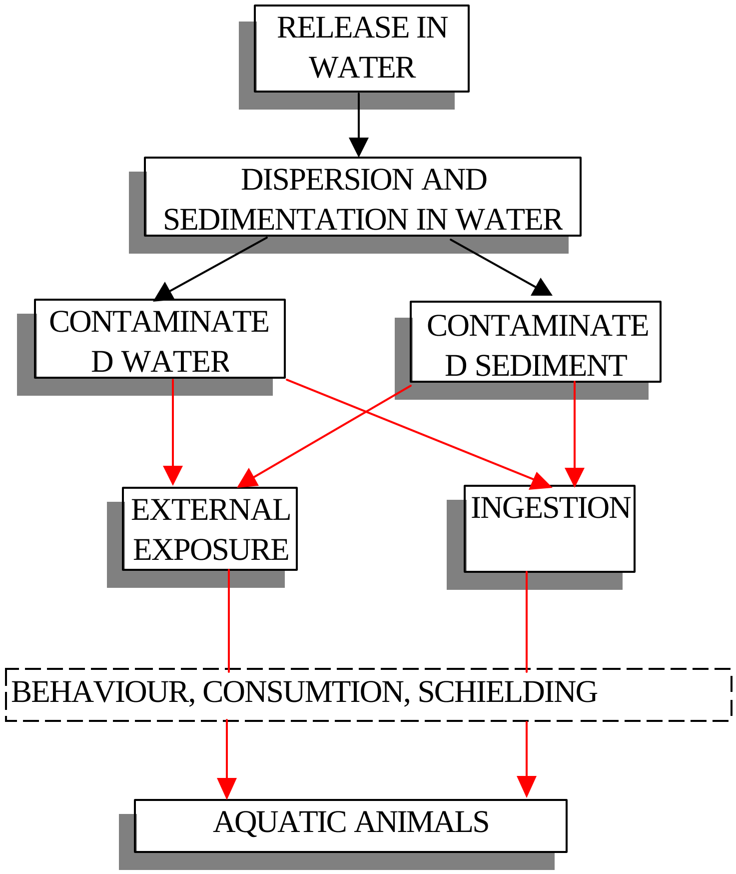

Figure 2. Schematic representation of exposure pathways of aquatic organisms following the release of radionuclides into a water body (black lines correspond to the transfer process, red lines correspond to the exposure process)

Dosimetric models can be developed to take into account the radiation type; the specific geometry of the target (e.g. the whole body, the gonads, the developing embryo or the plant meristem), and the source of exposure (e.g. radionuclides accumulated in body tissues, absorbed onto the body surface or distributed in the underlying soil). Clearly, there are limitations imposed by the paucity of data that are often available as input for the models, e.g. the spatial and temporal distributions of the radionuclides both within the organism and in the external environment. Additional limitations are imposed by the complexity of the systems we wish to model. Examples include the behaviour of mobile organisms modifying the exposure from external sources and, particularly, in the case of numerous aquatic organisms and many insects, from the occupation of different environmental niches at different stages of the life cycle. It is therefore a sensible strategy to develop models that are simplified and generalized but to attempt to avoid *losing the realism essential for the valid estimation of dose.*

3. Dosimetric phantoms for different representatives of biota

3.1. Selected reference organisms

In Beresford *et al.* [2001] we presented a practical approach for the identification of *reference Arctic organisms* (similar to the use of *reference man* in human radiation protection). The organisms selected are given in Table 1.

Table 1. Selected reference organisms for terrestrial and aquatic ecosystems [Beresford et al. 2001]

Terrestrial	**Aquatic**
Lichens & bryophytes	Benthic bacteria
Gymnosperms Monocotyledons	Macroalgae (marine) Aquatic plants (freshwater)
Dicotyledons	Phytoplankton
Soil micro-organisms	Zooplankton
Soil invertebrates	Molluscs
Herbivorous mammals	Polychaetes (marine)
Carnivorous mammals	Pelagic fish (planktotrophic)
Bird eggs	Benthic fish
	Pelagic fish (carnivorous)
	Carnivorous mammals
	Benthos eating birds
	Fish eggs

3.2. Derivation of phantoms

The current "state of art" in wildlife dosimetry involves a high degree of simplification, in particular concerning phantoms. Some approaches [e.g. Amiro 1997] assume organisms are simultaneously infinitely large (when calculating internal doses) and infinitely small (when calculating external doses). This approach is therefore highly conservative.

Representation of organisms using simple geometrical forms allows their volume and surface area to be readily estimated. Therefore, more realistic approaches, represent organisms as simple geometric forms (i.e. sphere, ellipsoid, cylinder) of appropriate dimensions [e.g. Woodhead 2000; Copplestone *et al.* 2001]. These dimensions define the approximate shape of an average animal or plant (of representative reference organisms) which is assumed to have a uniform density. For example, in Copplestone *et al.* [2001] ellipsoids and spheres of different sizes were used as phantoms for all organisms, from a sphere with diameter of $5\cdot10^{-5}$ cm representing a benthic bacteria up to an ellipsoid with dimensions $450 \times 87 \times 48$ cm representing a whale. At the present time, attempts have not been made to include separate organs inside the phantoms or to use more realistic shapes to represent different biota.

Any 3D solid object can be approximated via a set of points on some 3D lattice inside a defined boundary, normally a rectangular parallelepiped (description as volume), or as the interior of some surface. Formally, a description with such a surface is adequate, but a 3D lattice allows specific calculations to be performed simply and rapidly and also allows data about some characteristics of the 3D fields inside or around the object to be kept, if it is necessary for this particular kind of modelling.

For description of these surfaces a modern, universal approach is used with so called triangular meshes, i.e. any solid is approximated as a polyhedron with triangular faces. Such an approach allows us to use the same algorithm of calculations both for analytical objects (like sphere, ellipsoids, cylinders, etc.) and any 3D object. A typical representation of this sort is shown in Figure 3.

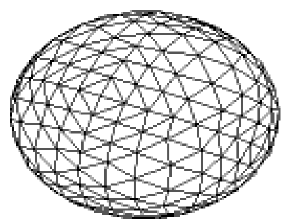

***Figure 3. Representation of a 3D ellipsoid using an array of triangular faces*.**

Because of this universal way of representing 3D forms, calculations for a biological object described with an arbitrary level of detail are possible, if there are input data about its shape. If such data are not available, it is possible to use an approximation with ellipsoids or cylinders as used in previous versions of biota phantoms. A more rigorous description may be useful for biota with complex (i) shapes, (ii) scenarios of contamination, and (iii) internal and external exposure models (an example may be the situation where it is necessary to take into account the shape and relative positions of different organs).

In Figure 4 the variants of future possible development of dosimetric phantoms for biota in this direction are presented.

a) b)

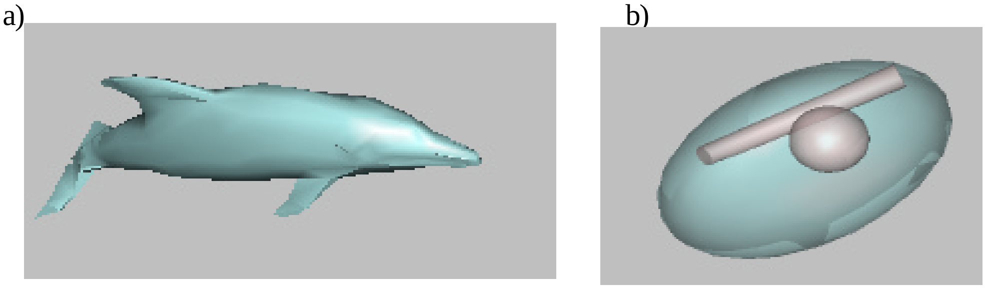

Figure 4. More realistic phantoms for representatives of biota: a) realistic shape of body of aquatic organism (dolphin); b) ellipsoid with internal organs (e.g. gastrointestinal tract and foetus).

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4. Dosimetric methods for dose calculations

4.1 General scheme of the dose calculations

Radiation dose may arise either from radionuclides present in the soil, sediment, water or air surrounding the organism (external dose) or from radionuclides taken up internally by the organism (internal dose).

The general strategy of estimating exposure doses consist of two main tasks:

- development of radionuclides transfer models to estimate the radionuclide content in the environment surrounding the organism and after that in organisms at any time after a radioactive release;
- development of dosimetric models for evaluation internal and external exposure of reference organisms, that is time-independent dose coefficient for given type of exposure.

For the assessment of internal dose it is necessary to estimate the fraction of energy which is absorbed within the organism. For the assessment of external dose, formulae are used for dose in an infinite or semi infinite absorbing medium.

If one assumes an infinite or semi-infinite volume with a uniform concentration *C(t)* of a radionuclide at time t, then the absorbed dose to biota, D_b, can be expressed as:

$$D_b = d_b \cdot \int C(t)dt, \qquad (1)$$

where d_b denotes the time-independent dose coefficient for given type of exposure, $Gy \cdot s^{-1}$ per $Bq \cdot kg^{-1}$.

The coefficient d_b represents the dose per unit time-integrated exposure, expressed in terms of the time-integrated concentration of the radionuclide. An alternative interpretation considers d_b to represent the instantaneous dose rate per unit activity concentration of the radionuclide in the environment.

The following standard models of estimating external exposure can be identified:

- exposure in infinite (semi-infinite) medium (air, water, soil) with uniform distribution of β, γ-emitting radionuclides;
- exposure from contaminated ground surface (infinite in a horizontal direction) with vertical distribution of radionuclides.

In the case where the individual (biota) is mobile the characteristics of the radiation field will clearly change as areas with differing contamination levels are traversed and/or occupied. This fact is taken into account with the help of the modifying factors (*location factors*), defined as the ratio of dose rate under specified conditions of an irradiating field (for example in/on the sediment) to the dose rate under standard conditions (in infinite medium). Thus the integrated exposure for an individual biota can be divided into partial exposures, where the basic assumptions of the basic, static models apply (e.g. uniform distributions of β and γ-emitters). Each of these individual models includes dose coefficients connecting concentration of radionuclides in environmental compartments with the characteristics of the external radiation field. This set of partial exposures situations used in combination with a set of *occupancy factors*, defined as the part of time spent under each of partial exposure, defines the common model of external dose formation for the organism.

Equations for estimating external and internal doses to biota under different pathways of exposure are described in the next section.

4.1.1 External exposure from an infinite source (air, water, and soil).

The value of absorbed dose for i-th group of organisms and k-th radionuclide $D^k_{i,c}$ is define by equation:

$$D^k_{i,c} = d^k_{i,c} \cdot C_{k,air} \cdot \sum_j p_{ij} \cdot f_j \qquad (2)$$

where:

$d^k_{i,c}$ is the dose coefficient for i-th group of organisms and k-th radionuclide ($Gy{\cdot}a^{-1}$ per $Bq{\cdot}m^{-3}$);
$C_{k,air}$ is integrated concentration of k-th radionuclide in air ($Bq{\cdot}a{\cdot}m^{-3}$);
f_j is the location factor in j-th location in the environment (dimensionless);
p_{ij} is the occupancy factor for i-th group of organisms and j-th location in the environment (dimensionless).

4.1.2 External exposure from contamination surfaces

The value of absorbed dose for i-th group of organisms and k-th radionuclide $D_{i,s}{}^k$ is define by the equation:

$$D^k_{i,s} = d^k_{i,s} \cdot A^k_s \cdot \sum_j p_{ij} \cdot f_j \qquad (3)$$

where:

$d^k_{i,c}$ is dose coefficient for i-th group of organisms and k-th radionuclide ($Gy{\cdot}a^{-1}$ per $Bq{\cdot}m^{-2}$);
$A_s{}^k$ is integrated surface activity of k-th radionuclide in soil ($Bq{\cdot}a{\cdot}m^{-2}$);
f_j and p_{ij} are defined above.

4.1.3 Internal exposure of animals (inhalation)

The value of absorbed dose for i-th group of organisms and k-th radionuclide $D^k_{i,inh}$ is define by equation:

$$D^k_{i,inh} = C^k_{air} \cdot B_i \cdot d^k_{i,inh} \cdot \sum_j p_{ij} \cdot f_j \qquad (4)$$

where:

$d^k_{i,inh}$ is the dose coefficient for i-th group of organisms and k-th radionuclide, $(Gy \cdot Bq^{-1})$;

C_{air}^k is integrated concentration of k-th radionuclide in air, $(Bq \cdot a \cdot ?^{-3})$;

B_i is inhalation rate for i-th group of organisms, $(m^3 \cdot a^{-1})$;

f_j is location factor in j-th location in the environment, dimensionless;

p_{ij} is occupancy factor for i-th group of organisms and j-th location in the environment.

4.1.4 Internal exposure of animals (ingestion)

The value of absorbed dose for i-th organism and k-th radionuclide $D^k_{i,ing}$ is define by equation:

$$D^k_{i,ing} = \sum_p C^k_p \cdot H_p \cdot d^k_{i,ing} \qquad (5)$$

where:

C^k_p is the integrated concentration of k-th radionuclide in p-th product, $(Bq \cdot a \cdot kg^{-1})$;

H_p is the feed rate of p-th product, $(kg \cdot a^{-1})$;

$d^k_{i,ing}$ is the dose coefficient for i-th organism and k-th radionuclide, $(Gy \cdot Bq^{-1})$.

The values of radionuclide concentrations in environmental objects included in equations (2 – 5) can be assumed to be time-dependent or to be at equilibrium. Transfer models can range from simple concentration ratios (e.g. Bq kg^{-1} in animal tissue:Bq kg^{-1} in diet) to more complex dynamic compartment models (although few of these exist for non-human foodchains). The above equations assume a uniform distribution of radionuclide contamination within an organism and within any one given media of its environment.

4.2 Methods of calculation of dose coefficients for aquatic ecosystems

For the aquatic environment, where authorized liquid discharges are made to the water column and inputs from accidental releases are likely to be to the water column or the water surface, the input concentration is taken to be in the water $(Bq \cdot m^{-3})$. Partitioning to sediment and uptake into aquatic organisms is normally determined by application of the relevant distribution coefficients and concentration factors, respectively.

The most realistic approach for calculation doses to aquatic biota, so far developed, is that presented by Woodhead and others (Woodhead, 1979; Woodhead, 2000; NCRP, 1991). In this approach organisms are represented by ellipsoids or spheres of appropriate dimensions, and the proportion of radiation absorbed within the volume of the organism is estimated using formulae which describe the distribution of radiation dose around point sources within the organism. It is necessary to integrate the resulting radiation doses over all hypothetical 'point sources' and 'point receptors' within the organism. In general this requires the use of numerical methods and suitable computer software. The empirical formulas for dose distribution function of α- and β- radiation around point isotropic source, taken from Woodhead [2000] are given below (see also IAEA 1979 for further details).

For α- radiation, the empirical point source dose distribution function, $D_\alpha(r)$, has the form:

$$D_\alpha(r) = \frac{4.59 \cdot 10^{-2}}{\rho \cdot r^2} \cdot (A + B \cdot r^2),\ \mu\mathrm{Gy} \cdot \mathrm{h}^{-1} \cdot Bq^{-1} \tag{6}$$

where:

ρ is the density of the medium (assumed to be soft tissue, freshwater or seawater with a density of 1000 kg·m^{-3});
r is the distance between the point source and the target point (μm) and is limited to less than the range of an α-particle at the emission energy ($E_{\alpha em}$);
A is the stopping power of the medium at the emission energy of the α-particle, MeV·μm^{-1}; and empirical parameter B is:

$$B = \frac{3 \cdot [E_{\alpha em} - A \cdot R(E_{\alpha em})]}{R(E_{\alpha em})^3},\ \mathrm{MeV} \cdot \mu\mathrm{m}^{-3} \tag{7}$$

The point source dose distribution function for β - particles, originally developed empirically by Loevinger *et al.* [1956], has been slightly modified [IAEA, 1979] to give a better fit to the scaled point source absorbed dose distribution for a wide range of radionuclides given by Berger [1971]. The modified point source dose distribution function for β - particles in water or soft tissue, $D_\beta(r)$, is:

$$D_\beta(r) = \frac{k}{(\rho\nu r)^2} \cdot \left\{ a \cdot \left[1 - \frac{\rho\nu r}{c} \cdot \exp(1 - \frac{\rho\nu r}{c}) \right] + \rho\nu r \cdot \exp(1 - \rho\nu r) \right\},\ \mu\mathrm{Gy} \cdot \mathrm{h}^{-1} \cdot Bq^{-1} \tag{8}$$

where:

$$\left[1 - \frac{\rho\nu r}{c} \exp(1 - \frac{\rho\nu r}{c}) \right] \equiv 0 \ \text{ for all } r > \frac{c}{\rho\nu};$$

$$k = \frac{4.59 \cdot 10^{-2} \cdot \rho^2 \cdot \nu^3 \cdot \bar{E}_\beta \cdot n_\beta}{ac(3 - e) + e} \quad \mu Gy \cdot h^{-1} \cdot Bq^{-1};$$

ρ and r are defined above;
ν is the apparent absorption coefficient in cm^2·g^{-1} and has the following dependence on the maximum β^+ - or β^- -particle emission energy $E_{\beta max}$:

$$\nu = 15.1 \cdot E_{\beta\max}^{-1.74}\ \ \mathrm{cm}^2 \cdot g^{-1}, \text{for } 0.0186\ \mathrm{MeV} \le E_{\beta\max} \le 0.92\ \mathrm{MeV}, \text{and}$$
$$\nu = 17.9 \cdot E_{\beta\max}^{-1.24}\ \ \mathrm{cm}^2 \cdot g^{-1}, \text{for } 0.92\ \mathrm{MeV} \le E_{\beta\max} \le 3\ \mathrm{MeV};$$

n_β is the fractional number of $\beta^{+/-}$ -particles of mean energy $\bar{E}_{\beta^{+/-}}$ emitted per disintegration;

a is dimensionless parameter given by:

$a = 1 + 3.43 \cdot \exp(-1.41 \cdot E_{b-\max})$, for $0.0186\ \mathrm{MeV} \le E_{b\max} \le 3\ \mathrm{MeV}$, for b^{-} – particles,

and

$a = 1.12$ for b^{+} -particles of all energies;

c is dimensionless parameter given by:

$c = 1 + 0.059 \cdot E_{b^{-}\max}^{-0.616}$, for $0.0186\ \mathrm{MeV} \le E_{b^{-}\max} \le 3\ \mathrm{MeV}$ for b^{-} -particles and

$c = 1.45 + 0.507 \cdot (E_{b^{+}\max} + 0.4)^{-3.65}$ for $0.324\ \mathrm{MeV} \le E_{b^{-}\max} \le 1.88\ \mathrm{MeV}$ for b^{+} – *particles*

The situation for γ-radiation is more complex due to the existence of several different processes of energy absorption and the fact that scattered radiation represents a significant proportion of the radiation field incident on the target tissue. For the internal contamination of small aquatic organisms (dimensions ~ 1cm) with γ-emitting radionuclides, it is usual to ignore absorption and scattering and employ the simple inverse square law to describe the radiation field from the point source, thus the dose distribution function, $D_\gamma(r)$, is:

$$D_g(r) = 4.59 \cdot 10^{-2} \cdot \sum_{E_g} \frac{m \cdot E_g \cdot n_g}{r \cdot r^2}\ m \cdot \mathrm{Gy} \cdot \mathrm{h}^{-1} \cdot Bq^{-1} \qquad (9)$$

where:

μ/ρ is the true mass absorption coefficient, at energy E_γ of the material (unit density tissue) being exposed;

n_γ is the fractional number of γ-rays of energy E_γ emitted per disintegration; and,

r is the target distance from the point source (cm).

This expression for $D_\gamma(r)$ relates to the 'kerma approximation' defined for the calculation of energy absorption from the γ-radiation field and does not take into account the energy deposition in tissue along the tracks of secondary electrons.

For internal contamination of larger aquatic organisms with γ-emitting radionuclides, and for photon exposure from their surrounding environment (water, sediment), the effects of absorption and scattering have to be taken into account. The results are expressed in terms of the energy-dependent absorbed fraction, $\Phi_\gamma(E)$:

$$\Phi_g(E) = \frac{\text{photon energy absorbed by the target}}{\text{photon energy emitted by the source}}$$

The mean dose rate to the target tissue volume is then:

$$D_g(r) = 5.76 \cdot 10^{-1} \cdot \sum_{E_g} \frac{E_g \cdot n_g \cdot \Phi_g(E)}{m}\ m \cdot \mathrm{Gy} \cdot \mathrm{h}^{-1} \cdot Bq^{-1} \qquad (10)$$

where:

m is the mass of the target organ;

n_γ is defined above.

The values of $\Phi_\gamma(E)$ have been computed for point and distributed sources with varying geometries, with and without the inclusion of a back-scattered contribution from the external environment [Brownell *et. al.* 1968; Ellett and Humes 1971].

On the basis of the data from Ellett and Humes (1971), the dependence of $\Phi_\gamma(E)$ from photon energy for two geometrical figures - sphere and ellipsoid (ratio of axes 1:3:8) of various masses (1 to 500 g) was estimated. In Figure 5 the dependence of values of absorbed fraction, calculated for central point sources of both geometrical figures, from photon energy are presented. As follows from these data the mass and shape of a geometrical figure (at identical mass) appreciably influences the values of absorbed fraction. The value of the absorbed fraction, $\Phi_\gamma(E)$, for a sphere can be 1½ times that of an ellipsoid (with dimensions as defined above) of identical mass for photon energies less than 0.1 MeV.

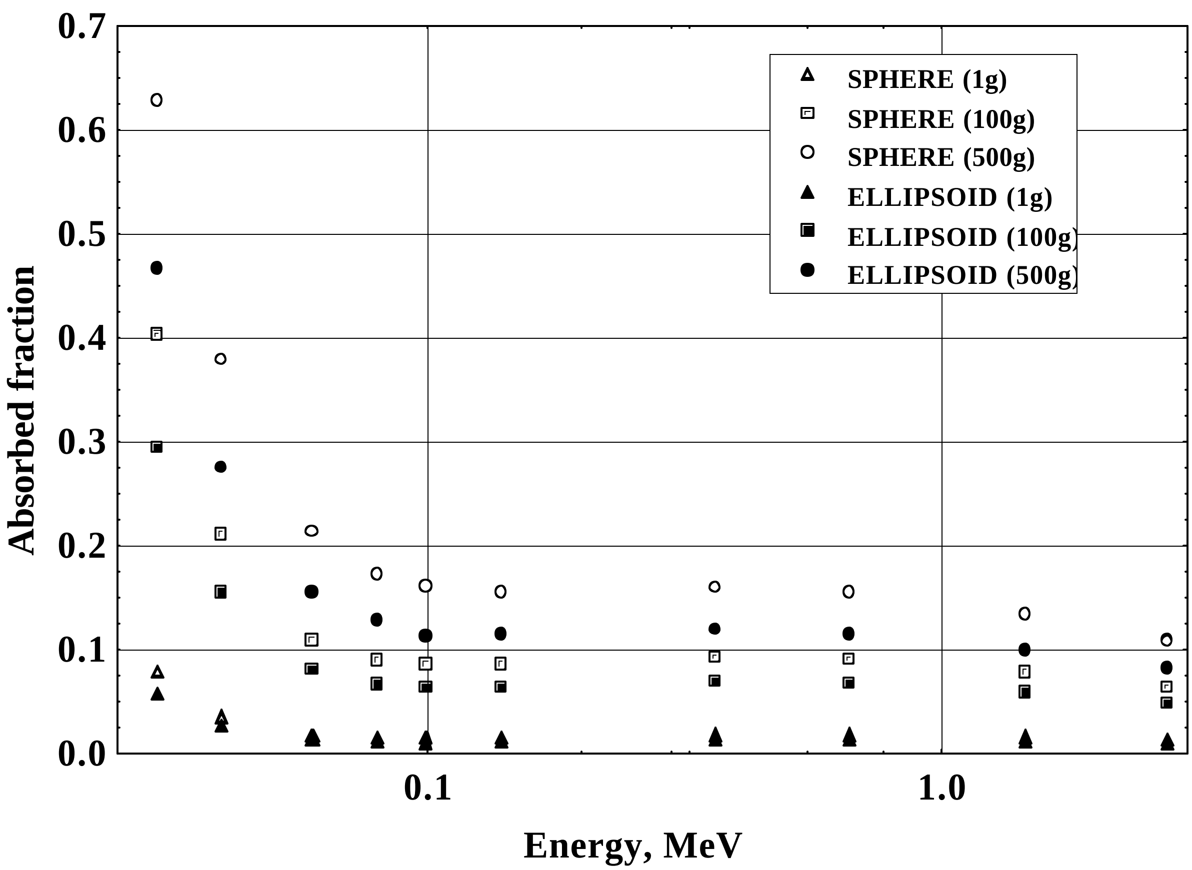

Figure 5. Photon absorbed fraction for central point sources in sphere and ellipsoid (ratio of axes 1:3:8) of various masses (from 1 to 500 g)

In Figure 6, the dependence of absorbed fraction, calculated for uniformly distributed sources in spheres and ellipsoids (axes ratio 1:3:8), of various masses (1 to 100 g), on photon energy are presented. As for the estimation of absorbed fraction the mass and shape of a geometrical figure (at identical mass) appreciably influences the values of absorbed fraction. It is further evident that the values of absorbed fraction, calculated for uniformly distributed sources are lower than corresponding values calculated for central point sources.

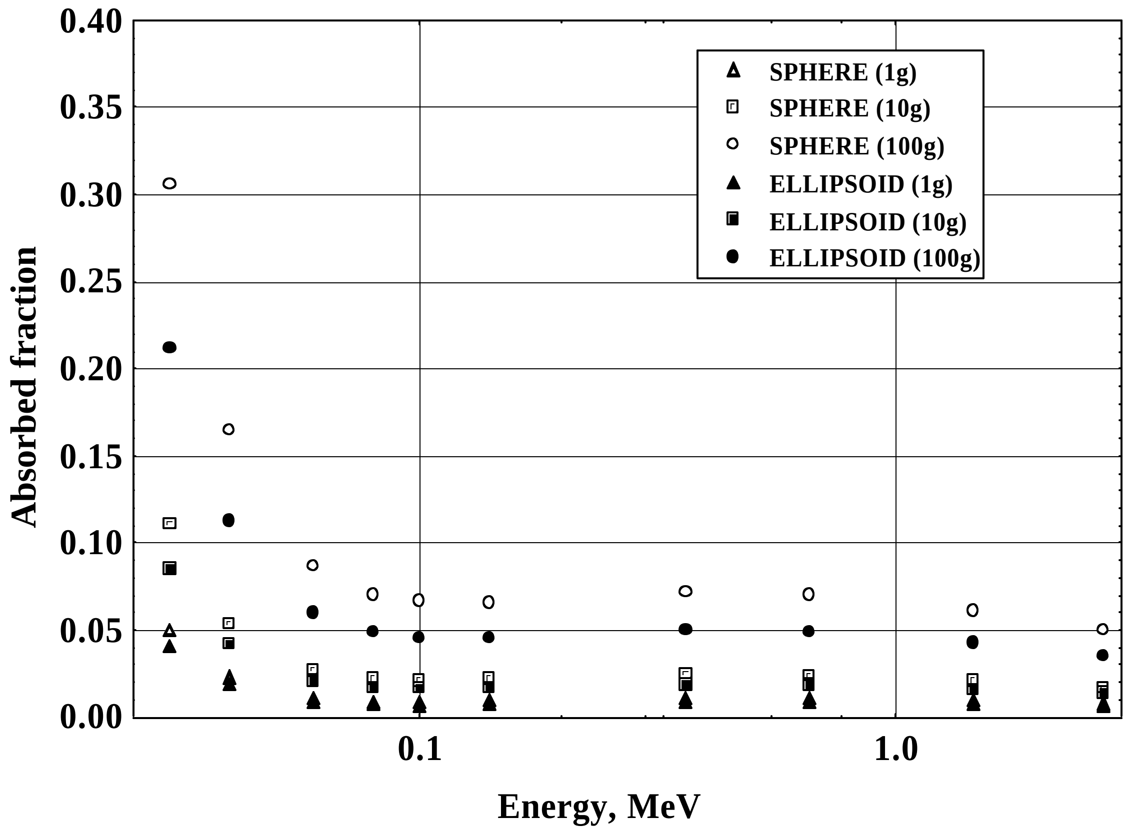

Figure 6. Photon absorbed fraction for uniformly distributed sources in sphere and ellipsoid (axes ratio 1:3:8) of various masses (1 to 100 gram)

4.2.1. Internal exposure

Phytoplankton. These organisms have been represented by a sphere of unit density tissue 50 μm in diameter (range is few microns – 200 μm) by IAEA [1976]. The radioactivity accumulated by these organisms has been assumed to be uniformly distributed throughout the volume. As a consequence of the limited size of the organism in relation to the ranges of the radiation being considered, a significant proportion of the total energy emitted by the incorporated radionuclides are dissipated in the surrounding water.

For alpha particles, if the average range is assumed to be 60 μm (i.e. the range of a 6 MeV α-particle in water) at constant linear energy transfer, the distribution of available path lengths within the sphere is such that approximately 30% of the emitted α-particle energy is absorbed within the volume [IAEA, 1976]. That is,

$$D_{ph}(\alpha) \sim 0.3\, D_{\alpha}(\infty), \tag{11}$$

where:

$$D_a(\infty) = 5.76 \cdot 10^{-1} \cdot \sum_{E_a} \bar{E}_a \cdot n_a\,, \quad m\text{Gy} \cdot \text{h}^{-1} \cdot (Bq \cdot g^{-1})^{-1} \tag{12}$$

n_α is the fractional number of α -particles of mean energy $\bar{E}_a$ in MeV emitted per disintegration;

This is equilibrium absorbed dose rate in a uniformly contaminated medium of effectively infinite extent.

For beta particles, with ranges in unit density tissue up to 2 cm, it is clear that, except at very low energies, a major fraction of the total energy emitted is deposited outside the organism (from more than 90% for beta particles with an energy of 2 MeV up to 20% for beta particles with an energy of 0.2 MeV).

The absorbed fraction for gamma radiation is so low that it can be assumed that gamma ray emission from radionuclides within the organism makes a negligible contribution to the overall dose rate received. However, in the exceptional case of very high phytoplankton densities (> 1 $\mu g \cdot g^{-1}$ water) and high concentration factors (> $2 \cdot 10^4$), the radioactivity associated with the plankton becomes a significant fraction of the total activity in the water volume and individual organisms then receive an absorbed dose rate from both the β- and γ-radiation emanating from the activity within/on neighbouring organisms.

Zooplankton. These organisms have been represented by a cylinder 0.5 cm long with a 0.2 cm diameter [IAEA, 1976] or an ellipsoid with axes 0.62 × 0.31 × 0.16 cm both with unit density tissue [IAEA, 1988, NCRP, 1991]. The radioactivity accumulated by these organisms has also been assumed to be uniformly distributed throughout the volume.

In the case of alpha radiation, the volume is sufficiently large, relative to the particle range, for it to be valid to assume that the mean absorbed dose rate closely approaches $D_\alpha(\infty)$.

For beta particles, the dimensions of the volume are the same order or less than the beta particle ranges and thus a variable proportion of the emitted energy escapes from the cylinder. The beta radiation dose rate, $D_\beta(P)$ at the centre of a series of cylinders is given in [IAEA, 1976] in terms of $D_\beta(\infty)$ as function of the maximum beta ray energy. In the case of an ellipsoid the values of absorbed fractions for beta particles can be estimated by help of the data given in Copplestone *et al.* [2001].

The calculation of the absorbed dose rate within a volume containing a distributed gamma ray sources also requires consideration of absorbed fractions. However, because of the small sizes zooplankton their values for gamma ray sources distributed within an organism as a rule consider equal to unit [IAEA, 1976].

Mollusc, crustacean and fish. The models adopted to represent each of these groups of animals are of such dimensions that it can be assumed that the dose rates from alpha and beta radiation closely approach $D_\alpha(\infty)$ and $D_\beta(\infty)$, respectively. The dimensions of the phantoms appropriate to the calculation absorbed fraction for gamma rays are as follows [IAEA, 1976; Copplestone *et al*., 2001]:

Mollusc: cylinder 1cm long and 4 cm diameter or ellipsoid ($2.5 \times 1.2 \times 0.62$ cm).
Crustacean: cylinder 15cm long and 6 cm diameter or ellipsoid ($3.1 \times 1.6 \times 0.78$ cm).
Fish: cylinder 50cm long and 10 cm diameter or ellipsoid ($45 \times 8.7 \times 4.9$ cm).

The tissue density has been assumed to be unity and the activity has been assumed to be uniformly distributed throughout the volume. For the fish the absorbed fraction corresponds to approximately 10% of the total gamma energy emitted in the range of 0.1 – 1 MeV, reaching 50 % at energy of 0.03 MeV.

4.2.2. External exposure

Phytoplankton. Since the dose rate from alpha radiation emitted within the organism is approximately $0.3 \cdot D_\alpha(\infty)$, it follows that the dose rates from alpha radiation incident from outside is approximately $0.7 \cdot D_\alpha(\infty)$, where $D_\alpha(\infty)$ is calculated from the concentration of alpha activity in water.

Similarly, for β-radiation from contaminated water the mean dose rate is given by:

$$\bar{D}_b = D_b(\infty) - \bar{D}_{sph}(r = 25\text{mm}), \tag{13}$$

where: $D_\beta(\infty)$ and $\bar{D}_b(r = 25mm)$ are calculated using the concentration of activity in water.

For γ- radiation the dose rates to the organisms is effectively equal to $D_\gamma(\infty)$ in water.

Zooplankton. It has been assumed [IAEA, 1976] that the dose rate from external alpha particles is negligible. For external beta particles the dose rate is calculated as for phytoplankton. The dose rate from external gamma radiation has been taken to be $D_\gamma(\infty)$ in water.

Molluscs, crustaceans and fish. It has been assumed that the dose rate from external alpha and beta particles are negligible and again the gamma-ray dose rate has been taken to be $D_\gamma(\infty)$ in water (same overestimation for low energy gamma radiation).

External dose rate from radionuclides in the sediment. For the radionuclides uniformly dstributed in sediment the dose rate at the sediment-water boundary has been taken to be $0.5 \cdot D_\infty(\beta)$ and $0.5 \cdot D_\gamma(\infty)$ for beta and gamma radiation, respectively from both water and sediment. Within the sediment the beta radiation dose rate increases to $D_{\infty}(\beta)$ at approximately 1 cm depth. Site-specific data about finite thickness of the source and variation of activity with depth in sediment should be considered where possible. This will determine the value of dose rate at the sediment-water boundary (similarly as for various distributions of radionuclides in soil). The dose rate from external alpha particles are negligible.

4.3. Methods of dose coefficients calculation for terrestrial ecosystems

Terrestrial ecosystems are more complex, in terms of dosimetry of external sources, due to both the much extended ranges of β- and γ- radiation in air and the presence of the substantial density variations between air, soil and plant and animal tissues. In the aquatic environment, it is reasonable to assume equivalence (at the level of accuracy required for environmental impact assessment) between the surrounding water and soft tissue in terms of radiation absorption and scattering properties.

The problems of estimating the absorbed dose to terrestrial plants and animals from external sources of γ-radiation have been discussed in UNSCEAR [1996]. It was concluded that the simple derivation of the absorbed dose rate from an estimate of air kerma would not be possible because it would depend on the assumption of photon field uniformity, secondary electron equilibrium and no photon scattering; these assumptions are unlikely to be valid in a contaminated environment with inhomogeneous distributions of both radionuclides and material densities.

With a number of simplifications Jacobi and Paretzke [1986] considered methods of calculating doses to plants, specifically leaves of trees and pine needles. The dose rate to the plant material, D_p, has external and internal components. If the radiation exposure results from radionuclide in air, the external dose rate, $D_{p,ext}$, may be expressed in terms of the radionuclide concentration in air:

$$D_{p,ext} = \frac{g \cdot w \cdot E \cdot C_a}{\rho_a} \qquad (14)$$

where:

g is a geometric factor which equals one for gamma rays and higher energy beta particles but drops to zero for low energy beta particles unable to penetrate the leaf cuticle to the growing cells (at a depth of around 0.1 mm) (dimensionless);
w is disintegration fraction (dimensionless);
E is the energy per disintegration (J);
C_a is the concentration of the radionuclide in air, $Bq \cdot m^{-3}$;
ρ_a is the density of air (may be taken to be 1.3 $kg \cdot m^{-3}$).

The cumulative doses to plant may be estimated from assumed mean lifetime for leaves of 0.5 years and needles of 7 years. The calculation of external dose takes no account of the possibility of self-shielding and will therefore overestimate external dose. The exposure from contaminated soil (and the rest of the plant/surrounding plants) is not considered and therefore will thus tend to underestimate the dose rate.

For estimating the internal dose, $D_{p,int}$, to leaves and needles Jacobi and Paretzke [1986] used the equation:

$$D_{p,\text{int}} = \frac{\Phi \cdot w \cdot E \cdot C_p}{\rho_p} \qquad (15)$$

where:

Φ is absorbed fraction of the energy released through disintegration;
E is the energy per disintegration (J);
w is disintegration fraction (dimensionless);
C_p is the concentration of the radionuclide in plant material $Bq \cdot m^{-3}$;
ρ_p is the density of plant material (taken to be 800 $kg \cdot m^{-3}$.

Because of the small dimensions of leaves and needles, the absorbed fraction Φ is small for gamma radiation and higher energy beta particles and increases to unity for lower energy beta particles and for alpha particles.

A similar degree of simplification was adopted by IAEA [1992] to estimate the absorbed dose rate to plants and animals from internal and external sources. For internal sources, the D(∞) value for the radionuclide was reduced by an absorption fraction relevant to the radiation type and energy, i.e. unity for α-particles; unity for β-particles except in the case of ^{32}P for which a value of 0.5 was adopted; and, 0.1 for γ-rays. The dose rate to plant tissues from external sources of γ-rays deposited on the ground was estimated to be 3.3 times that estimated for humans, owing to geometry and occupancy differences [UNSCEAR, 1982]. For external sources of β-radiation, it was concluded that, even for high energy emitters such as ^{32}P and ^{90}Y, the exposure would be less than 10% of that from the contamination on, and in, the plant. This contribution was, therefore, ignored.

A similar approach was followed for an animal. For internal sources, the absorption fraction for the reproductive tissues for α-, β- and γ-radiation were taken to be unity, unity and 0.3, respectively. The dose to animal tissues from external sources of γ-radiation was assumed to be the same as that for plants.

Some researchers [Amiro and Zach 1993; Amiro 1997] have estimated the dose conversion factors (DCF) for a number of generic terrestrial organisms – a plant, a mammal and a bird (in addition to pelagic and benthic freshwater fish) – for internal and external sources of radiation. The underlying dosimetry models were generalised and were made deliberately conservative to ensure that any consequent action provided the environment with the "benefit of the doubt". For the radionuclides taken up into, and assumed to be uniformly distributed within, the organisms, it was assumed that all the emitted energy was absorbed within the tissue (i.e., the absorbed dose rate was equivalent to $D_{abg}(\infty)$ evaluated at the radionuclide concentration in tissue). For the external exposure, organisms (including plants) are assumed to be infinitely small, i.e., the absorbed dose is again equivalent to $D_{abg}(\infty)$. Therefore, the dose conversion factors represent organisms of all sizes. External dose calculations are based on abiotic environmental concentrations (activity concentrations in air, water and soil) and take no account of biota size. Such an approach is clearly conservative.

Spirin [1992] developed a model to estimate radiation doses to agricultural crops from radionuclides in soil. The air plus the plant material is represented as a homogeneous medium of uniform density (intermediate between the densities of air and of plant material) that attenuates the radiation field. This is developed by integrating a point source isotropic dose function over the (plane) source in the soil and provides a dose distribution through the depth of the plant layer. Redistribution of the radionuclides from the surface soil into the 20 cm deep layer reduces the estimated beta dose by a factor of 30 and the gamma-dose by a factor of 3. The concept of a critical tissue for a plant is suggested (the apical meristem - the cells responsible for growth at the root or shoot tip).

In Golikov *et al.* [1999] air kerma rates 1 m above the ground created by gamma radiation from contaminated plane surfaces (for approximately 60 radionuclides) located at different depths in soil and at different heights in forest ecosystems are estimated. The forest ecosystem is represented by a three-layer composition. The two upper layers have the same elemental composition but different physical densities and represent the crowns and trunks of trees respectively; the third layer represents the soil. The calculations were performed for 18 source energies from 20 keV to 3 MeV and for the following coordinates of the source:

- depth in soil: 0, 0.3, 1, 3, 10, 30, 50, 70 cm (corresponding to cover from soil slabs with mass per unit area in the range of 0 to 70 g·cm^{-2});
- height in the forest layer: 0.05, 0.5, 1, 3, 10, 30, 50 m (corresponding to cover from forest slabs with mass per unit area in the range of 0 to 250 kg·m^{-2}).

On the basis of the results of calculations for plane isotropic sources formulas for calculating air kerma rate from the activity distribution in forest compartments are obtained.

The exponential distribution of contamination in soil is given by:

$$A_m^s(x_a) = A_a^s \cdot b \cdot \exp(-b \cdot x_a)$$

where $A_m^s(x_a)$ is the specific activity of soil (Bq·g^{-1}) at the depth $\boldsymbol{x_a}$ (g·cm^{-2}), A_a^s is the activity per unit area (Bq·cm^{-2}) and b is the depth distribution parameter, which is the reciprocal of the relaxation length (cm^2·g^{-1}). The kerma rate (nGy·h^{-1}), $K_a^s(b, r_1)$, at the point 1 m above the soil is then given by:

$$K_a^s(b, r_1) = 3.6 \cdot A_a^s \cdot \left\{ \frac{a_1 \cdot exp(-a_2 \cdot r_1)}{1 + \dfrac{a_3 \cdot exp(-a_4 \cdot r_1)}{b}} + \frac{a_5}{1 + \dfrac{a_6}{b}} \right\} \qquad (16)$$

where r_1 (kg·m^{-3}) is forest biomass density.

For a homogeneous distribution_of radionuclides in a given soil layer with densities between x_{a1} and x_{a2} (g·cm^{-2}, $x_{a1} < x_{a2}$) and a specific activity A_m^s (Bq·g^{-1}), is the kerma rate (nGy·h^{-1}), $K_a^s(x_{a2}, x_{a1}, r_1)$, at a point 1 m about the soil is:

$$K_a^s(x_{a2}, x_{a1}, r_1) = 3.6 \cdot A_m^s \cdot \left\{ \begin{array}{l} \frac{a_1}{a_3} \cdot exp((a_4 - a_2) \cdot r_1) \cdot exp(-a_3 \cdot exp(-a_4 \cdot r_1) \cdot x_{a1}) \cdot \\ \cdot \left[1 - exp(-a_3 \cdot exp(-a_4 \cdot r_1) \cdot (x_{a2} - x_{a1}))\right] \\ + \frac{a_5}{a_6} \cdot exp(-a_6 \cdot x_{a1}) \cdot \left[1 - exp(-a_6 \cdot (x_{a2} - x_{a1}))\right] \end{array} \right\} \quad (17)$$

Examples of the parameters values a_1 - a_6 for some radionuclides are presented in the Table 2.

Table 2. Numerical values of parameters in the formulas (17, 18) [Golikov et.al., 1999]

Radionuclide	a_1	a_2	a_3	a_4	a_5	a_6
Ba-140	1.19	0.091	0.555	0.045	0.78	0.0790
La-140	13.3	0.088	0.400	0.061	7.65	0.0569
Cs-134	9.46	0.093	0.470	0.055	6.31	0.0699
Cs-137→Ba-137m	3.39	0.094	0.492	0.051	2.38	0.0723
I-131	2.29	0.091	0.540	0.048	1.72	0.0845
Nb-95	4.59	0.094	0.474	0.053	3.13	0.0690
Zr-95	4.43	0.094	0.476	0.050	3.05	0.0697
Ru-106→Rh-106	1.21	0.090	0.472	0.055	0.81	0.0730

The variation in the estimated kerma rate at a point 1 m over the soil for the pair of radionuclides $^{137}Cs \rightarrow ^{137m}Ba$ with different exponential depth distribution of activity in soil and thickness of forest biomass are presented in Figure 7. The values modelled for the parameter of β, (i.e. exponential depth distribution in soil) correspond to the following empirical data:

- dry fallout (straight away after deposition, $\beta = 5$ cm^2·g^{-1});
- wet fallout (straight away after deposition, $\beta = 1.4$ cm^2·g^{-1});
- pine forest in the Bryansk region 7 years after deposition ($\beta = 0.9$ cm^2·g^{-1}).

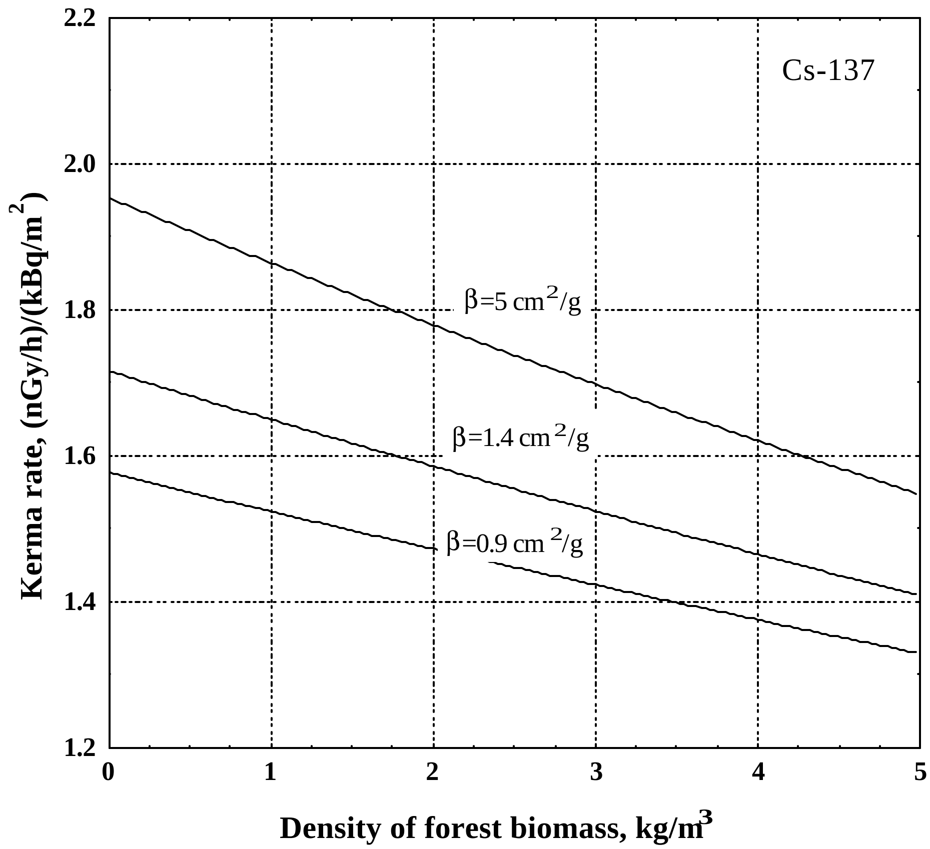

Figure 7. Estimated variation in air kerma rate at a point 1 m above the soil in a forest with different densities of biomass and exponential depth distributions of activity in soil.

From the data presented in the Figure 7, it is evident, that an increase in forest biomass density over the range modelled accounts for a 20-30 % reduction in air kerma rate. The depth distribution of activity in soil within the range modelled results in an approximately similar variation of air kerma rate.

These results could be used in the development of an algorithm for external dose calculations to plants and animals from contaminated soil, air and plants that takes into account shielding effects, different distributions of radionuclides in soil, air and plants and the behaviour of animals.

The PATHWAY model [Whicker F.W., Kirchner T.B., 1987] originally developed to estimate transfer through the human foodchain and doses to man has been used to estimate radionuclide concentrations in surface-living animals from aerial deposition [IAEA, 1992]. The domestic sheep was used as a model terrestrial animal. The dosimetry model assumed total absorption of alpha and beta radiation and 30% absorption of the gamma energy in the gonads.

Kozmin *et al*. [1992] developed models to estimate the radiation exposure of farm animals following accidental releases of radionuclides to the atmosphere. The radiation exposure pathways considered were external exposure from the plume, inhalation from the contamination ground-level air, contaminated pasture (external and internal sources) and contaminated soil (external and internal sources). In practice, it is internal exposure following ingestion that is of particular significance. Compartment models were developed to predict the radionuclide distribution within the animals and the consequent radiation exposure. In the model for external exposure the behaviour of animals was taken into account (for example the change of geometry of exposure arising from the observation that 30-60 % of the time, the animals lay on the ground, was modelled. The absorption of gamma radiation in the body of an animal is also taken into account. The model used geometrical figures (sphere, ellipsoid) to represent animal bodies. The average value of dose rates in a animal body for ?-radiation with energy 0.66 MeV arising from irradiation by contaminated soil (surface source) varied from 0.30 units for the cows up to 0.54 units for sheep.

5. Relative Biological Effectiveness (RBE) for biota.

Radiation protection standards for biota have generally been expressed in terms of absorbed dose [NRCC 1983; NCRP 1991; IAEA 1992; USDoE 1996]. These dose limits are based on studies of effects on biota resulting from exposure to photons [NCRP 1991; IAEA 1992]. In calculating doses resulting from exposure to α-emitting radionuclides deposited in tissues of animals or plants, consideration must be given to whether the absorbed dose due to α-particles should be modified by a radiation weighting factor. This factor would account for the fact that α-particles are more effective than photons (or electrons) in producing biological damage. Furthermore, it cannot be assumed that Relative Biological Effectiveness (RBE) values applied to α, β, or γ-radiation, and hence radiation weighting factors in human dosimetry are applicable to biota due to the vast differences in physiology between humans and many of the biota types under consideration, e.g. plants, invertebrates etc.. It has been argued that a methodology incorporating radiation weighting factors is required to allow the derivation of a “dose equivalent for biota” from absorbed doses (which would reflect the methodology in place for human radiological protection), in order to facilitate robust assessments of the risk posed to biota from ionising radiation [Pentreath 1999].

Some investigators have modified the calculated absorbed dose due to α-particles by a factor of 20 [IAEA 1992; Blaylock *et al.* 1993] based upon recommendations by the ICRP [1991] and the NCRP [1993] for human radiation protection. Other investigators have not modified the calculated absorbed dose due to α-particles [Baker and Soldat 1992; Amiro 1997], based in part on the view that the radiation weighting factor of 20 used in protection of humans may not appropriate for non-human biota [Amiro 1997]. The NCRP [1991] noted that values of the RBE for α-particles, which are used to develop an assumed radiation weighting factor, should be similar in tissues of humans and other organisms, but radiation weighting factors for α-particles were not used by NCRP, because of the degree of conservatism already built into the models used to calculate doses to biota (e.g. infinitely small and infinitely large geometries for external and internal radiation respectively).

Reviews of information on radiation effects in aquatic and terrestrial biota [NCRP 1991; IAEA 1992; UNSCEAR 1996] indicate that the critical biological endpoints of concern in the protection of populations of species involve impairment of reproductive capability. Indeed, recommended dose limits for biota are based on observed effects on the reproductive capability of organisms at different levels of acute and chronic exposure [NCRP 1991; IAEA 1992]. Biological endpoints, such as early mortality, are also of concern in protecting populations of species. However, other effects on populations are observed only at doses substantially higher than doses at which reproductive capability is impaired [NCRP 1991; IAEA 1992; UNSCEAR 1996].

Most importantly, the effects of ionizing radiation on the reproductive capability of organisms are considered to be a deterministic rather than a stochastic effect, as noted by UNSCEAR [1996]. Based on the observations that the biological effects of concern in protecting populations of species are deterministic, the radiation weighting factor of 20 for α-particles used in protection of humans [ICRP 1991; NCRP 1993] may not be appropriate, because this value was intended to represent RBEs for stochastic effects, primarily the induction of cancers [NCRP 1990; ICRP 1991].

?ased on the above, Kocher and Trabalka [2000] suggest using radiation weighting factors in the range 5 to 10 in the protection of biota. UNSCEAR [1996] suggests that the radiation weighting factors for α-particles of 5 is appropriate for non-human biota. This is on the basis that deterministic effects will be of greater significance than they are for human protection and that a lower factor than used for humans should therefore apply. Pentreath [1996] advances a similar argument in respect of aquatic organisms, although no specific value is recommended. Macdonald *et al.* [1998] also suggest that a radiation weighting factor of 20 for a-radiation may not be appropriate for shorter-lived non-human species and suggest a value of 10 based upon studies by Goodhead *et al.* [1993] and Barendsen [1994]. Both Woodhead [1984] and Blaylock *et al.* [1993] have suggested a weighting factor of 20 for aquatic organisms, on the grounds that this value incorporates the spectrum of effects, including stochastic effects. This value may, of course, be conservative in respect of deterministic effects.

Copplestone *et al.* [2001] recommended a weighting factor for α-particles of 20 based on the judgement that:

- the value for human protection is derived, partly, on data from other mammals, which are the most radiosensitive species, and that
- there is insufficient evidence from other non-human biota to influence this conclusion;
- the value of 20 is likely to be conservative in respect of deterministic effects.

A radiation weighting factor of 3 for mono-energetic electrons, or β-particles of average energy less than 10 keV was recommended by Copplestone *et al.* [2001].

The fact that the choice of RBE is a contentious issue has been highlighted most recently by Tracy & Thomas [2002]. These authors stressed the point that the choice of radiation weighting factor cannot be tied to a unique value of RBE since this quantity varies with species, end-point and dose range. Although examples exist in the literature where RBE's in excess of 350 have been calculated, these derivations have often been associated with a number of problems including poor statistics, high uncertainties or questionable dosimetry. Tracy & Thomas [2002] conclude that a radiation weighting factor of 10 would adequately protect the survival of most species in most cases. However, before blindly following this single-number guidance we should bear in mind that for the purposes of EPIC we may not be purely concerned with protecting the survival of a species. In many cases the protection of biota at an individual level may be appropriate. It should also be noted that our knowledge is far from complete in this respect and that further experimental data are required for large numbers of biota types and end-points of concern.

6. Results of current dose rate calculations to biota from natural and artificial radionuclides

6.1 General range of environmental absorbed dose rate

It is possible to indicate general limits of the radiation exposures for a number of situations [Woodhead 2000; UNSCEAR 1996]. For the natural background, the absorbed dose rates are normally up to ~1 $\mu Gy{\cdot}h^{-1}$ but, exceptionally, may be up to $2{\cdot}10^2$ $\mu Gy{\cdot}h^{-1}$. In all situations, α-particles appear to contribute a substantial proportion of the total dose rate (^{222}Rn + short lived daughters, and ^{210}Po). In environments receiving radioactive wastes, the absorbed dose rate from the contamination are generally < 10^2 $\mu Gy{\cdot}h^{-1}$, but may, exceptionally, rise to ~10^3 $\mu Gy{\cdot}h^{-1}$. The highest environmental doses rate have followed accidental releases of radionuclides (Kyshtym accident in 1957, accidental under ground explosions in Yakutia in 1974 and 1978, Chernobyl accident in 1986). In the vicinity of Kyshtym and Chernobyl, the initial absorbed dose rates were > 10^4 $\mu Gy{\cdot}h^{-1}$ (and locally up to more than 10^5 $\mu Gy{\cdot}h^{-1}$); these have declined to current values of < $1.5{\cdot}10^2$ $\mu Gy{\cdot}h^{-1}$ and 10^2 $\mu Gy{\cdot}h^{-1}$ respectively [Tikhomirov F.A., and Romanov G.N., 1993; Kryshev I.I., Alexakhin R.M., Makhon'ko K.P. et.al., 1992]. The significance of these ranges of dose rate is that they indicate the domain of the dose rate/response relationship over which information for the biological endpoints of interest is required.

In IAEA [1992] the radiation doses and/or dose rates to plants and animals which result when releases of radionuclides are controlled on the basis of the standards for the protection of humans are estimated. The conclusion was that it is highly probable that limitation of the exposure of the most exposed humans (the critical group), living on and receiving full sustenance from the local area, to 1 $mSv{\cdot}a^{-1}$ will lead to dose rates to plants and animals in the same area of less than 350 $mGy{\cdot}a^{-1}$. Despite of closeness of the received maximal value 350 $mGy{\cdot}a^{-1}$ to the value of 365 $mGy{\cdot}a^{-1}$ (1 $mGy{\cdot}d^{-1}$) recommended as a 'limit' for biota by IAEA [1992] the above made conclusion with a high degree is fair, as a methods of dose estimation used was were conservative

6.2 Examples of dose assessment to marine biota in the Arctic

Several sources of radionuclides have made their contribution to the contamination of the Arctic Seas. Among them are atmospheric fallout, the Chernobyl accident, waters of Siberian rivers contaminated by Russian reprocessing facilities, and sea currents carrying the wastes from the West European nuclear reprocessing plants. For over three decades, Russia practiced the dumping of solid radioactive wastes in the Kara Sea near the Novaya Zemlya Archipelago. Since 1993, large international efforts have been directed at the evaluation of long-term radiological consequences of radioactive wastes dumping in this shallow Arctic sea (AMAP 1998; IAEA 1998; Strand & Holm 1993; Strand et al. 1994; Strand & Cooke 1995; Strand 1997; Strand & Jolle 1999). Up to now the level of radioactive contamination in Arctic seas is relatively low compared to seas of Western Europe. However, accidental or continuous releases of radionuclides from the sources of potential radiation hazard may lead to radioactive contamination of some parts of the Arctic seas.

The modelling of doses to human and biota will constitute an important tool in the assessment of the impact of radionuclides released into the environment. Input data on the concentrations of radionuclides in biota and abiotic marine environment (water, sediments) are required for the calculation of doses. This will be derived either directly from analyses of monitoring data on the marine ecosystems contamination, or by modelling the transport and fate of radionuclides in marine ecosystems under different release scenarios.

The estimated doses for the biota of the Barents and Kara Seas obtained on the basis of actual radioactivity levels in components of the marine ecosystem are presented in Table 2 [Kryshev & Sazykina 1995]; additional doses from artificial radionuclides are estimated to be considerably lower than those from natural background radiation.

Table 2. The estimated internal exposures for Arctic marine biota nGy·day^{-1}

Marine biota	Barents Sea*	Kara Sea*	Natural background
Crustacean	2	3	3500
Molluscs	3	5	2700
Fish	20	30	800

* Dose rates derived from artificial radionuclides only.

Doses to marine organisms inhabiting the radioactive waste dumping sites in the bays of Novaya Zemlya were calculated using two sets of data: i) *current* (1992-94) data on the radiological situation in the Abrosimov Fjord and Tsivolki Fjord [Strand *et al.* 1997], and ii) predicted levels of future environmental contamination, associated with potential radionuclide releases from containers with radioactive materials [Sazykina *et al.* 1998]. The concentrations of radionuclides incorporated within the bodies of marine organisms were derived by applying the appropriate bioconcentration factors to the water concentration.

Model scenarios for the potential radionuclide releases patterns from the dumped containers with radioactive materials were developed within the framework of the International Arctic Seas Assessment Project (IASAP) between 1993 and 1996 under the auspices of the International Atomic Energy Agency [Sjoeblom & Linsley 1995; Lynn *et al.* 1995]. These IASAP source term scenarios provided predictions of annual release rates for a wide set of radionuclides from each radioactive waste dumping site over the period of 1000 years in the future. To evaluate the highest potential doses to marine biota, we used the release rates predicted for the 'plausible worst-case scenario', based on the assumption of an accidental disruption in 2050 AD of dumped fuel containers in the Tsivolki Fjord, with a 'spike' release of 110 TBq followed by much smaller releases in the subsequent years.

Estimated current dose levels to marine biota near radioactive waste disposal sites

The recent expedition to the Tsivolki Fjord showed a similar radiological situation as that of the open Kara Sea [Strand *et al.* 1997]. Consequently, the estimated internal doses to the local biota are the same as for the Kara Sea biota. External exposure from sediments in the Tsivolki Fjord was estimated to be about 0.11 µGy day^{-1}. During the survey of the Abrosimov Fjord [Strand *et al.* 1997], small localised areas of elevated contamination were found in the vicinity of some dumped radioactive wastes containers (but not near to those containing spent fuel). The highest levels of man-made radionuclides in bottom sediments within these localised areas were as follows: ^{137}Cs - about 2000 Bq kg^{-1} wet weight (w.w.); ^{60}Co - up to 15-21 Bq kg^{-1} w.w. A general increase in the ^{137}Cs levels in bottom sediments of the Abrosimov Fjord was observed (13±8 Bq kg^{-1} w.w., which is approximately two times higher than the level observed in sediments from the open Kara Sea). Small concentrations of ^{60}Co (2-5 Bq kg^{-1} w.w.) were also detected in the top layer (0-2 cm) of bottom sediments. The concentration of ^{137}Cs in water of the Abrosimov Fjord (about 3 Bq m-^{3}) did not differ from that in water of the open Kara Sea.

The calculated dose rates to molluscs living on highly contaminated areas of bottom sediments within the Abrosimov Fjord may be as high as 8.5 µGy·day^{-1}, whereas the average external exposure to molluscs from bottom sediments elsewhere in the Abrosimov Fjord is about 0.11 µGy·day^{-1}.

Potential doses to marine biota in the Tsivolki Fjord - worst-case release scenario

The calculated concentrations of the most important radionuclides in water and bottom sediments of the Tsivolki Fjord during the period of highest releases (2050-2055 AD) are given in Table 3 [Sazykina *et al.* 1998].

Table 3. Predicted levels of radionuclide concentrations in the Tsivolki Fjord in the case of realisation of the IASAP 'plausible worst case scenario' [Sazykina et al. 1998]

Radionuclide/ Year	Water, Bq·m^{-3}		Sediments (0-5 cm), Bq·kg^{-1}, d.w.	
	2050 AD	2055 AD	2050 AD	2055 AD
^{137}Cs	0.42E+05	0.32E+04	0.17E+03	0.64E+03
^{90}Sr	0.37E+05	0.28E+04	0.49E+02	0.19E+03
^{63}Ni	0.11E+05	0.15E+05	0.12E+05	0.21E+05
^{239}Pu	0.14E+04	0.37E+02	0.11E+03	0.36E+03
^{240}Pu	0.62E+03	0.16E+02	0.49E+02	0.16E+03
^{241}Am	0.28E+03	0.26E+01	0.49E+02	0.12E+03

The calculated dose rates to different types of marine biota are presented in Table 4. Because of the presence of large amounts of actinides, their contribution to potential dose rates dominates (Table 5). The contribution internal exposure to the total dose rate for all types of organisms is greater than 90 %.

Table 4. Predicted dose rates to marine biota in the Tsivolki Fjord in the case of realisation of the IASAP 'plausible worst case scenario'

Organism/ Year	Total dose rate, mSv h^{-1}	
	2050 AD	2055 AD
Molluscs	0.40	0.01
Small fish	0.0065	0.0004
Big fish	0.0067	0.0004
Sea mammal	0.0068	0.0004

Table 5. The percentage contribution of different radionuclides to the expected doses to the local marine biota during the year of highest releases (2050 AD, Tsivolki Fjord)

Organism	Actinides	Fission products	Activation products
Molluscs	99.88	0.07	0.05
Small fish	87.7	10.8	1.5
Big fish	86	12.6	1.4
Sea mammal	83.6	15	1.4

According to NCRP [1991], summarising the effects of radiation on aquatic organisms, dose rates no greater than 0.4 mGy h^{-1} should ensure the survival of populations, although some damage to individuals may occur. By analysing the predicted dose rates to marine biota in the Tsivolki Fjord (see Table 3), one can conclude that this dose level will be exceeded for some marine species, in particular for molluscs and some other benthic organisms. Therefore damage to local populations may occur under this scenario. It should be noted that the calculations were based on the assumption that released radionuclides were uniformly distributed in water and the seabed. However, a more realistic scenario would include the specification of local contamination "hotspots" in the vicinity of ruined containers.

7. Conclusion

A selection of the dosimetry models that have previously been employed to assess the radiation exposure of plants and animals in contaminated environments has been described. Dosimetry models have been developed to the greatest extent for aquatic organisms. There has been a more limited application of dosimetry models in the terrestrial environment.

Dose calculations recommended for impact assessment require information or estimates to be made of: the organism's dimensions; concentrations and distributions of contamination in the biota's habitat; distribution of internal contamination; and the location of the organism in the surrounding media. Usually dosimetry models represent organisms as geometrical figures such as spheres or ellipsoids. Radionuclides are normally assumed to be uniformly distributed throughout the organism, thus the resulting internal dose is calculated as an average value for the whole organism. For calculation of external doses, the fractional occupancy of organisms is considered, whether underground, on the soil/sediment surface, or fully immersed in infinite air or water.

As the damage induced by radiation is dependent upon the radiation type (α, β, or γ). It is proposed to use the following provisional radiation weighting factors:

- from 5 up to 20 for α-particles;
- 3 for low energy β-radiation (< 10 keV);
- 1 for β- radiation greater than 10 keV and γ-radiation.

The final choice of radiation weighting factor for alpha particles will depend on the selection of reference organism, end-point and dose (or dose-rate) range. Calculations of absorbed dose should be split into low LET and high LET define components in order to facilitate the incorporation of a radiation weighting factor once consensus has been achieved.

The accuracy of the dose rates estimations are generally limited by uncertainties in the parameters used in radionuclide transfer models [UNSCEAR, 1996], e.g. concentration and distribution coefficients of different radionuclides in the media surrounding the organisms. Based on reasonable assumptions, at the present time dose factors have been derived for a number of radionuclides. These have been applied to estimate dose rates both from natural and artificial radionuclides in various marine environments. It is recognised that the inherent simplifications often used in these assumptions leads to overestimated dose rates in many cases; gross over conservatism may not be desirable if results are to be used to regulate the nuclear industry.

More research is required to develop more universal computer based models, which will allow both the modelling of radionuclide distribution in a specified environment and the subsequent estimation of doses for organisms with any shapes and sizes to be made. The algorithms of calculations should include both up to date data bases of constants for dose calculations and more realistic phantoms of plants and animals with, for example, separate internal organs.

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