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Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment

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EXECUTIVE SUMMARY

The purpose of this report is to present a methodology for evaluating the potential for aquatic biota to incur effects from exposure to chronic low-level radiation in the environment. Aquatic organisms inhabiting an environment contaminated with radioactivity receive external radiation from radionuclides in water, sediment, and from other biota such as vegetation. Aquatic organisms receive internal radiation from radionuclides ingested via food and water and, in some cases, from radionuclides absorbed through the skin and respiratory organs. Dose rate equations, which have been developed previously, are presented for estimating the radiation dose rate to representative aquatic organisms from alpha, beta, and gamma irradiation from external and internal sources. Tables containing parameter values for calculating radiation doses from selected alpha, beta, and gamma emitters are presented in the appendix to facilitate dose rate calculations.

The risk of detrimental effects to aquatic biota from radiation exposure is evaluated by comparing the calculated radiation dose rate to biota to the U.S. Department of Energy's (DOE's) recommended dose rate limit of 0.4 mGy h^{-1} (1 rad d^{-1}). A dose rate no greater than 0.4 mGy h^{-1} to the most sensitive organisms should ensure the protection of populations of aquatic organisms. DOE's recommended dose rate is based on a number of published reviews on the effects of radiation on aquatic organisms that are summarized in the National Council on Radiation Protection and Measurements Report No. 109 (NCRP 1991). The literature identifies the developing eggs and young of some species of teleost fish as the most radiosensitive organisms. DOE recommends that if the results of radiological models or dosimetric measurements indicate that a radiation dose rate of 0.1 mGy h^{-1} will be exceeded, then a more detailed evaluation of the potential ecological consequences of radiation exposure to endemic populations should be conducted.

Dose rates have been calculated for biota in aquatic ecosystems associated with three national laboratories and one uranium mining and milling facility (NCRP 1991). At all sites, the dose rates were two orders of magnitude less than the value recommended by DOE for the protection of populations of aquatic biota. Therefore, it is highly unlikely that aquatic organisms will encounter dose rates in aquatic ecosystems that will be detrimental at the population level other than in man-made bodies of water associated with waste management activities or from accidental releases of radionuclides.

1. INTRODUCTION

Sources of radioactivity in the aquatic environment include naturally occurring radionuclides, fallout from the atmospheric, runoff from watersheds that have received atmospheric deposition, and radioactive effluents from medical, industrial, and nuclear facilities released either accidentally or routinely. Depending upon the element and the chemical form, radionuclides may accumulate in bottom sediment or remain in the water column in the dissolved state. From either location, they can subsequently accumulate in biota and be transferred through the aquatic food chain. Contamination of the environment by radionuclides inevitably results in an increase in the radiation exposure of natural populations of organisms that occupy the contaminated area. Aquatic organisms receive external radiation exposure from radionuclides in water, sediment, and from other biota such as vegetation. They also receive internal radiation exposure from radionuclides ingested via food and water and from radionuclides absorbed through the skin and respiratory organs.

Generally, the discharge of radioactive waste into the environment is such that it results in only long-term, low-dose-rate exposure of organisms. In most cases, acute mortality can be discounted. The very small increase in morbidity and mortality that is contributed by an increased exposure to chronic irradiation is unlikely to be detectable because of the natural fluctuations in the sizes of populations of organisms in the aquatic environment. The purpose of this report is to present a methodology for evaluating the ecological risk to aquatic organisms that are exposed to anthropogenic radionuclides released into the environment.

2. APPROACH

Ecological risk to aquatic organisms exposed to radiation from anthropogenic radionuclides in the environment will be assessed by 1) calculating the dose to the organism and 2) comparing that dose to levels of radiation below which no detectable effects have been observed. Special consideration will be given to effects on reproductive parameters such as fecundity and embryo viability which would be the most likely to be adversely affected by exposure to radiation.

Although most radiation effect studies have evaluated effects at the organism level, assessments of ecological risk are usually concerned with the viability and success of populations. Unlike the case for humans in which malignancies and genetic abnormalities can be a personal catastrophe, there usually is not a similar concern about the survival of individual organisms in nature. An exception exists for threatened or endangered species or species with low fecundity (typically uncommon in freshwater ecosystems), where the survival of an individual could influence the success of the population. In most cases, the potential for over-reproduction of aquatic organisms is large and most individuals either become part of the natural food chain to be consumed by other organisms or starve. Therefore, for aquatic organisms there is little concern about small increases in the frequency of malignancies or genetic abnormalities because the weakest individuals are usually eliminated first in the natural selection process.

3. EFFECTS OF RADIATION ON AQUATIC ORGANISMS

A large body of literature exists on the effects of radiation on aquatic organisms and has been reviewed extensively by a number of authors (IAEA 1976; Blaylock and Trabalka 1978; NRCC 1983; Egami and Ijiri 1979; Woodhead 1984; Anderson and Harrison 1986; NCRP 1991). The general consensus of the reviewers is that the most sensitive aquatic organisms known are teleost fish, particularly the developing eggs and young of some species. Additionally, the reviewers point out that most radiation effects studies have been conducted using acute exposures of radiation and less than 10% of the studies involved chronic or continuous irradiation. Because most environmental exposures are long-term, low-dose-rate exposures, data from chronic irradiation effect studies on the life cycle of organisms are the most useful in assessing the ecological risk to biota.

One approach that is used in assessing the risk of adverse ecological effects is to select indicator species of organisms for study. Indicator species are usually biologically significant organisms and are representative of the particular environment under investigation. An assessment of the environment will usually allow the identification of a few critical species of organisms for which dose estimates should be made. These species should provide adequate data for an assessment of effects from the radiation exposure to the community.

4. RADIATION DOSE TO BIOTA

Three approaches have been employed for calculating radiation doses to aquatic biota. Results of using these three approaches were evaluated by Woodhead (NCRP 1991). CRITR, a set of models and associated computer codes, was developed by Soldat et al. (1974) and recently revised by Baker and Soldat (1992) for application to discharges of effluents into surface waters. A simplified means was provided for calculating the concentrations of radionuclides in water, sediment, and two groups of organisms using a restricted number of parameters relating to the discharge source and the receiving water body.

A second approach involved two models, EXREM III and BIORAD (Trubey and Kaye 1973), which were developed from the starting point of a unit concentration of a radionuclide in water from which the concentration in an organism is determined by the application of a concentration factor. No means are given for estimating the concentration of a radionuclide in sediment or determining the exposure from contaminated sediment which may be significant.

A third approach, "Point Source Dose Distribution" (IAEA 1976, 1979), is advantageous in that it can be applied to any combination of radiation sources and target geometries. For any extended (nonpoint) source of ionizing radiation, the dose rate at a specified point can be obtained by the integration of an appropriate point source dose function over the source geometry. Although it is possible to derive theoretical expressions from first principles, these calculations are frequently complex due to the multiplicity of absorption and scattering phenomena which must be considered. For ease of computation, simple empirical expressions have been described for calculating doses to aquatic biota (IAEA 1976, 1979).

Several factors makes estimating the radiation dose to an organism difficult. Different radionuclides are differentially distributed among the organs and tissues of an organism, affecting the radiation dose that sensitive organs and tissues receive. In addition, the relative significance of internal and external sources of radiation to an organism can be markedly altered by the size and behavior of the organism.

Radiation exposure models have been developed that incorporate parameters accounting for differences in the size and shape of an organism. The "Point Source Dose Distribution" methodology provides a means for calculating the radiation dose to different size categories of aquatic organisms using simplified equations. Measurements used to represent different size categories for a select group of aquatic organisms are given in Table 1.

Table 1. Dimensions of organisms representing different size categories used in the Point Source Dose Distribution methodology for estimating radiation doses

Organism	Mass (kg)	Length of the major axes of the ellipsoid (cm)
Small insects and larvae	1.6×10^{-5}	0.62 x 0.31 x.0.16
Large insects and molluscs	1.0×10^{-3}	2.5 x 1.2 x 0.62
Small fish	2.0×10^{-3}	3.1 x 1.6 x 0.78
Large fish	1.0	45 x 8.7 x 4.9

4.1 γ-radiation

For large organisms with dimensions greater than a few cm, energy absorption and scattering become significant; therefore, a factor must be applied to account for these processes. Monte Carlo calculations have been made to include absorption and scattering for a number of geometries, and these calculations can be adapted for aquatic organisms (Brownell et al. 1968, Ellett and Humes 1971). The results are given in terms of the absorbed fraction which is defined as:

$$\Phi = \frac{\text{photon energy absorbed by target}}{\text{photon energy emitted by source}}$$

Absorbed fractions (Φ) which have been derived for the biota listed in Table 1 as a function of γ-ray energies (ICRP 1991) are given in Figures A.1 through A.3.

The γ-radiation dose rate from internal contamination is expressed as:

$$D_\gamma = 5.76 \times 10^{-4}\, E_\gamma\, n_\gamma\, \Phi\, C_o \qquad \mu Gy\ h^{-1} \qquad (1)$$

where

- E_γ is the photon energy emitted during transition from a higher to a lower energy state (MeV)
- n_γ is the proportion of disintegrations producing a γ-ray
- Φ is the absorbed fraction from Figures A.1 through A.3 of energy E_γ (MeV) (dimensionless)
- C_o is the concentration of the radionuclide in the organism (Bq kg^{-1} wet weight)

If a γ-emitter produces photons of different energy levels, the doses from all major γ-emissions should be included in the dose rate calculation.

It follows that the γ-radiation dose rate to the organism from radionuclides in water away from the sediment is

$$D_\gamma = 5.76 \times 10^{-4}\, E_\gamma\, n_\gamma (1-\Phi) C_w \qquad \mu Gy\ h^{-1} \qquad (2)$$

where

- C_w is the concentration of the radionuclide in water (Bq L^{-1})

The γ-radiation dose rate to organisms at the sediment-water interface from a uniformly contaminated sediment is

$$D_\gamma = 2.88 \times 10^{-4}\, E_\gamma\, n_\gamma (1-\Phi)\, C_s\, R \qquad \mu Gy\ h^{-1} \qquad (3)$$

where

- C_s is the concentration of the radionuclide in sediment (Bq kg^{-1} wet weight). A generic value of 0.75 can be used for converting sediment from dry weight to wet weight.
- R is the fraction of time that the organism spends at the sediment-water interface.

Because of deposition and resuspension of sediment, decay of the radioisotope, and the variability in the rate at which a radionuclide may be released into a aquatic system, sediment rarely presents a uniform, semi-infinite source of γ-radiation. Therefore, in most cases, equation (3) will over estimate the dose to biota at the sediment-surface water interface. In those cases where detailed information is not available, 0.5 times the D_γ in equation (3) can be used to account for the unequal distribution of radionuclides in the sediment (IAEA 1976, Woodhead 1984).

Table A.1 contains the average energy per transformation for a selected group of gamma emitters. These values were taken from ICRP Report 38 (1983) and can be used in place of E_γ and n_γ in the preceding equations to calculate the total γ-radiation dose rate in one step. Examples illustrating the calculation of γ-radiation dose rates are given in Appendix B.

4.2 β-radiation

The point source β-dose function (NCRP 1991, Woodhead 1979) was integrated over the geometries given in Table 1, assuming a uniform distribution of the radionuclide in the organism, to obtain the dose rate at the center of the organism as a fraction of the total β-dose rate. The results are shown in Fig. A.4 as a function of maximum β-particle energy for the three small geometries. For large fish and turtles, the internal β-dose rate is independent of the β-particle energy; therefore,

$$D_\beta = 5.76 \times 10^{-4}\, \overline{E}_\beta\, n_\beta\, C_o \qquad \mu Gy\ h^{-1} \qquad (4)$$

The internal β-radiation dose rate for the three small geometries is given by the following equation

$$D_\beta = 5.76 \times 10^{-4}\, \overline{E}_\beta\, n_\beta\, \Phi\, C_o \qquad \mu Gy\ h^{-1} \qquad (5)$$

where

- $\overline{E}_\beta$ is the average energy of the β-particle (MeV)
- n_β is the proportion of transitions producing a β-particle of energy $\overline{E}_\beta$ (MeV) (dimensionless)
- Φ is the absorbed fraction from Fig. A.4
- C_o is the concentration of the radionuclide in the organism (Bq kg^{-1} wet weight)

It is assumed that β-radiation from water contributes a negligible amount to the internal dose rate of large fish and turtles. The external β-dose rate from water for the smaller organisms described in Table 1 is

$$D_\beta = 5.76 \times 10^{-4}\, \overline{E}_\beta\, n_\beta (1-\Phi)\, C_w \qquad \mu Gy\ h^{-1}. \qquad (6)$$

where

- C_w is the concentration of the radionuclide in water (Bq L^{-1})

The external β-dose rate from sediment for organisms represented by the three small geometries that are in contact with the sediment surface is

$$D_\beta = 2.88 \times 10^{-4}\, \overline{E}_\beta\, n_\beta\, (1-\Phi)\, C_s R \qquad \mu Gy\ h^{-1}. \qquad (7)$$

where

- C_s is the concentration of the radionuclide in sediment (Bq kg^{-1} wet weight). A generic value of 0.75 can be used for converting sediment from dry weight to wet weight.
- R is the fraction of time that the organism spends at the sediment-water interface.

Some aquatic organisms may be surrounded by sediment during certain life stages and, in such cases, 5.76×10^{-4} instead of 2.88×10^{-4} would be the appropriate unit conversion factor.

Beta emitters that decay by alternative transitions produce an energy spectrum for each mode of transition. The dose rates from the major spectra must be included when calculating the total β-dose rate to an organism. Table A.1 contains a list of the maximum and average energies of selected β-emitters based on β-particles, conversion electrons, and Auger radiations. These values were obtained from ICRP Report 38 (1983). Examples demonstrating the use of the data in Table A.1 to calculate β-dose rates are given in Appendix B.

4.3 α-radiation

For organisms of the sizes represented in Table 1, the internal dose rate from α-radiation closely approaches the dose rate from an infinite source because essentially all the energy from α-particles is absorbed within the organism. The internal dose rate from α-radiation is calculated as follows:

$$D_\alpha = 5.76 \times 10^{-4}\, E_\alpha n_\alpha C_o \qquad \mu Gy\ h^{-1} \qquad (8)$$

where

E_α is the energy of the α-particle (MeV)

n_α is the proportion of transitions producing an α-particle of energy E_α(MeV) (dimensionless)

C_o is the concentration of the radionuclide in the organism (Bq kg^{-1} wet weight)

If α-particles of more than one energy level are produced during the decay of a radioisotope, the dose rate from all transitions are summed to obtain the total a-dose rate. It is assumed that external α-radiation from water and sediment is insignificant for organisms of the sizes shown in Table 1.

Table A.2 gives the average α-energies for selected α-emitters including those in naturally occurring α-decay chains. The average energy of β- and γ-emissions produced by the α-decay are also given. Examples illustrating the calculation of dose rates for a-emitters are presented in Appendix B.

The dose rates in this report are expressed in units of absorbed dose (μGy); however, different types of radiations differ in their relative biological effectiveness per unit of absorbed dose. A quality factor, Q, is normally used to account for the difference in biological effectiveness of the different radiations (NCRP 1987). Quality factors have been derived from data on humans and are intended to be used only for low doses, not high doses that might result from a nuclear accident. A quality factor of 1 is used for x-, γ-, and β-radiation and 20 for α-radiation. Therefore, to equate the relative biological effectiveness of the dose rate from α-radiation in μGy to the rate from γ- and β-radiations, the α-dose rate should be multiplied by 20. In effect, the resulting dose rate would be equivalent to microsieverts (μSv), the dose equivalent unit used for humans.

5. DOSE RATE CALCULATIONS FOR FISH EGGS

The calculation of a radiation dose to fish eggs/embryos exposed to radionuclides in the environment is a complex procedure that requires answers to a number of questions. These questions include: Is the radionuclide inside the egg or is it adsorbed to the outer shell or chorion? If the radionuclide is inside the egg, is it uniformly distributed? What is the diameter of the egg? Where is the developing embryo located? Do the eggs float, sink to the bottom, form clusters, adhere to vegetation or other objects, etc.? How long is the development period and does the radionuclide concentration change with time? If answers to these questions are available, it is possible to use mathematical models for different geometries and physical conditions to calculate the radiation dose rate to fish eggs/embryos (Adams 1968, Woodhead 1970, Ellett and Humes 1971, IAEA 1979). However, for most purposes a conservative estimate of the radiation dose rate is sufficient. The following discussion presents a simplified approach for estimating the dose rate to fish eggs/embryos from radionuclides in the environment.

γ-radiation to Fish Eggs

Most fish eggs are only a few millimeters in diameter; therefore, the radiation dose rate from internal γ-emitters would be insignificant (Ellett and Humes 1971, IAEA 1976). The external dose rate to an egg from γ-emitters in the surrounding water would be the average dose rate in an effectively infinite source (i.e., the dimensions of the source are much greater than the attenuation length of the radiation). The unit density of the fish eggs and the source (water) are assumed to be the same. The equation for the γ-dose rate from an infinite source is

$$D_\gamma(\infty) = 5.76 \times 10^{-4}\, E_\gamma\, n_\gamma\, C_w \qquad \mu\text{Gy h}^{-1} \quad (9)$$

where

E_γ is the photon energy emitted during transition from a higher to a lower energy state (MeV)

n_γ is the proportion of disintegrations producing a γ-ray of energy E_γ (MeV) (dimensionless)

C_w is the concentration of the radionuclide in water (Bq L^{-1})

Because the activity of most radionuclides in water is much lower than in biological tissue and because eggs of most species of freshwater fish hatch in a few weeks or less, it is unlikely that the radiation dose from γ-emitters in the environment would have a deleterious effect on fish eggs/embryos.

Fish eggs may receive external γ-radiation from other sources such as sediment and vegetation and a number of geometric factors would affect the dose rate. For most radionuclides, the activity in the sediment is much higher than in the water, so that the dose rate from the sediment will be higher than from the water. However, the dose rate to fish eggs would depend upon the photon energy and their distance from the sediment surface. Assuming that the sediment is a uniformly contaminated slab source of infinite area and the eggs are lying on the sediment surface, the following equation can be used to estimate the γ-dose rate to the eggs.

$$D_\gamma(\infty) = 2.88 \times 10^{-4} E_\gamma \; n_\gamma \; C_s \qquad \mu\text{Gy h}^{-1} \qquad (10)$$

where

- E_γ is the photon energy emitted during transition from a higher to a lower energy state (MeV)
- n_γ is the proportion of disintegrations producing a γ-ray of energy E_γ (MeV) (dimensionless)
- C_s is the concentration of the radionuclide in sediment(Bq kg^{-1} wet weight). A generic value of 0.75 can be used for converting sediment from dry weight to wet weight.

β-radiation to Fish Eggs

Equations for calculating the dose rate to fish eggs from internal β-emitters are complex and beyond the scope of this report. By assuming that all the energy from internal β-emitters is absorbed within the egg, the following equation can be used to estimate the dose.

$$D_\beta = 5.76 \times 10^{-4} \bar{E}_\beta \, n_\beta \, C_o \qquad \mu\text{Gy h}^{-1} \qquad (11)$$

where

- $\bar{E}_\beta$ is the average energy of the β-particle (MeV)
- n_β is the proportion of transitions producing a β-particle of energy $\bar{E}_\beta$ (MeV) (dimensionless)
- C_o is the concentration of the radionuclide in the organism (Bq kg^{-1} wet weight)

Results of equation (11) are approximately true for low-energy β-radiation; however, as the β-particle energy increases, the extent of over estimation increases. If the estimated dose rate indicates that harmful effects might occur, then a more accurate dose rate should be determined. Equations for calculating dose rates to fish eggs are available in the literature (IAEA 1979, Adams 1968, and Woodhead 1970).

If the range of the β-radiation in the surrounding water exceeds the radius of the eggs, then the dose rate to the eggs from the water is

$$D_\beta = 5.76 \times 10^{-4} \bar{E}_\beta \, n_\beta \, C_w \qquad \mu\text{Gy h}^{-1} \qquad (12)$$

where

- $\bar{E}_\beta$ is the average energy of the β-particle (MeV)
- n_β is the proportion of transitions producing a β-particle of energy $\bar{E}_\beta$ (MeV) (dimensionless)
- C_w is the concentration of the radionuclide in water (Bq L^{-1})

Equation (12) can be used to estimate the β-dose rate from water in instances where the range of the β-particle is less than the radius of the egg but the dose rate will be over estimated. As mentioned above, if the estimated dose rate indicates harmful effects might occur, then a more accurate estimate of the dose rate should be obtained.

Fish eggs can also receive β-radiation from contact with surfaces such as sediment or vegetation. The dose rate will depend upon the thickness and density of the material as well as the energy of the β-radiation. The following equation can be used to estimate to dose to eggs that are in contact with sediment although in most situations it will over estimate the dose rate.

$$D_\beta = 2.88 \times 10^{-4}\, E_\beta n_\beta C_s R \quad \mu Gy\ h^{-1}. \qquad (13)$$

where

C_s is the concentration of the radionuclide in sediment (Bq kg^{-1} wet weight). A generic value of 0.75 can be used for converting sediment from dry weight to wet weight

R is the fraction of time that the organism spends at the sediment-water interface.

α-radiation to Fish Eggs

Assuming that all the radiation from internal α-emitters remains within the egg and that all external α-radiation is stopped by the chorion, a reasonable estimate of the dose rate from α-emitters is given by

$$D_\alpha = 5.76 \times 10^{-4}\, E_\alpha n_\alpha C_o \qquad \mu Gy\ h^{-1} \qquad (14)$$

where

E_α is the energy of the α-particle (MeV)

n_α is the proportion of transitions producing an α-particle of energy E_α (MeV) (dimensionless)

C_o is the concentration of the radionuclide in the organism (Bq kg^{-1} wet weight)

6. DOSE CALCULATIONS AND EFFECTS

The previously listed equations can be used to calculate a dose rate to aquatic biota for most situations. Bioaccumulation factors for freshwater fish for selected radioisotopes are included in Tables A.1 and A.2. These factors can be used to estimate the concentration of a radioisotope in freshwater fish from the concentration in the surrounding water. Information on the decay schemes of additional radioisotopes can be obtained from ICRP 38 (ICRP 1983), Kocher (1981), and the Health Physics and Radiological Health Handbook (Shleien et al. 1984). Equations for calculating dose rates for other biota, such as fish eggs, phytoplankton, and zooplankton, can be found in IAEA Technical Reports Series No. 172 (1976) and Series No. 190 (1979). After determining the dose rate to an organism from each individual radioisotope in the environment, the total dose rate to the organism is determined by summing the dose rates (in dose equivalents) from all radioisotopes. The total dose rate can then be compared to literature values for radiation effects on the same or closely related organisms. The most appropriate values for comparison are those from chronic exposure studies conducted over the life cycle of an organism; however, it is often necessary to extrapolate the results of acute exposures to chronic exposures.

A number of reviews on the effects of radiation on aquatic organisms have been published over the last three decades (Polikarpov 1966, Templeton et al. 1971, Chipman 1972, IAEA 1976, Blaylock and Trabalka 1978, IAEA 1979, Egami 1980, NRCC 1983, Woodhead 1984, Anderson and Harrison 1986, NCRP 1991, and IAEA 1992). These reviews considered data from field and laboratory studies from both marine and freshwater environments. More data have been collected on marine than on freshwater species; however, where reasonable comparisons can be made, there is no evidence that significance differences in radiosensitivity exists between marine and freshwater organisms. NCRP Report No. 109 (NCRP 1991) contains summary tables of the effects of chronic irradiation on fish and invertebrates. Tables A.3 through A.6 are modifications of the NCRP tables. For information on specific organisms not contained in these tables, individual reviews can be consulted, for example, Woodhead (1984).

Methods for dose calculations for phytoplankton and zooplankton are not included in this document because these organisms are relatively resistant to irradiation exposure (Table A.5) (Marshall 1962, 1966). From reviews of the literature (IAEA 1976; Blaylock and Trabalka 1978; Woodhead 1984), detrimental effects on organisms of higher trophic levels should be detected before populations of phytoplankton and zooplankton are affected by exposure to radiation. Therefore, dose calculations for organisms of higher trophic levels are emphasized in this report. The methodology for calculating dose rates for phytoplankton and zooplankton is available in the IAEA Technical Report 172 (1976).

The U.S. Department of Energy's (DOE) guideline for radiation dose rates from environmental sources, which recommends limiting the radiation dose to aquatic biota to 0.4 mGy h^{-1} (1 rad day^{-1}), is based on results of previously cited reviews summarized in NCRP Report No. 109 (NCRP 1991). The conclusion from these reviews is that at 0.4 mGy h^{-1}, there is no evidence that deleterious effects have been expressed at the population level for aquatic biota. Tables A.3 through A.5 contain summaries from the literature reviewed in NCRP Report No. 109 on reproductive effects in fish exposed to chronic irradiation. In these chronic irradiation studies, effects were not detected unless the dose rates were much greater than 0.4 mGy h^{-1}. However, populations may be at risk from other factors, such as over exploitation or other environmental stresses, which might in combination with radiation have an undesirable impact. Therefore, it is desirable to conduct a comprehensive ecological evaluation of the radiation exposure regime in combination with other environmental factors in order to assess the potential for radiation contributing to effects at the population level. It is recommended (NCRP 1991) that where the results of radiological models or dosimetric measurements indicate a dose rate of 0.1 mGy h^{-1} or more to aquatic biota, a more detailed evaluation of the ecological consequences to the endemic biota should be conducted.

According to the radiation effects literature, the most radiosensitive aquatic organisms are the developing eggs and young of some species of teleost fish. With few exceptions, the developmental period for freshwater fish eggs is relatively short but it can range from 3 days for the common carp *Cyprinous carpio* to more than 70 days for some salmonidae species. For this reason, the accumulated radiation dose to fish eggs from chronic environmental radiation should be relatively small. It is highly unlikely that dose rates in natural aquatic ecosystems that receive routine releases of radioactive effluents would produce effects on developing eggs and young of fish that would influence the success of the population. Exceptions to this premise could occur as a result of accidental releases of unacceptable levels of radioactive effluents or in man-made waste disposal ponds where high concentrations of radionuclides may be present.

In NCRP Report No. 109 (NCRP 1991), dose rates to aquatic organisms were calculated for three DOE-operated sites and one site in Canada: Gable Mountain Pond, Hanford Plant, Washington; White Oak Lake, Oak Ridge National Laboratory, Tennessee; Savannah River Plant, South Carolina; and Beaverlodge Uranium Mining Area, Saskatchewan, Canada. The estimated whole-body doses received by aquatic organisms at these sites were more than two orders of magnitude below the proposed standard of 0.4 mGy h^{-1}. However, a few dose rates approached 0.1 mGy h^{-1}, which might in combination with environmental stresses have an undesirable impact. The highest dose rates occurred in man-made ponds associated with waste management activities and these ponds have no direct connection with natural bodies of water. Remedial actions have been implemented at these sites. Therefore, it is highly unlikely that environmental situations will be encountered where the risk from radiation exposure from releases of radioactive waste to the environment would produce detrimental effects on aquatic organisms at the population level.

The methodology for calculating conservative (upper-limit) radiation dose rates provided in this document can be used to estimate dose rates to biota inhabiting aquatic environments contaminated with radionuclides. If the dose rate to aquatic organisms is less than the DOE's recommended level of 0.4 mGy h^{-1} (1 rad day^{-1}), there should be no detrimental effects from radiation exposure at the population level, i.e., there should be no quantifiable risk to the biota. If estimated dose rates exceed 0.1 mGy h^{-1}, then studies should be implemented to determine whether effects can be detected at the individual and/or population level for biota inhabiting the environment.

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Appendix A

FIGURES AND TABLES

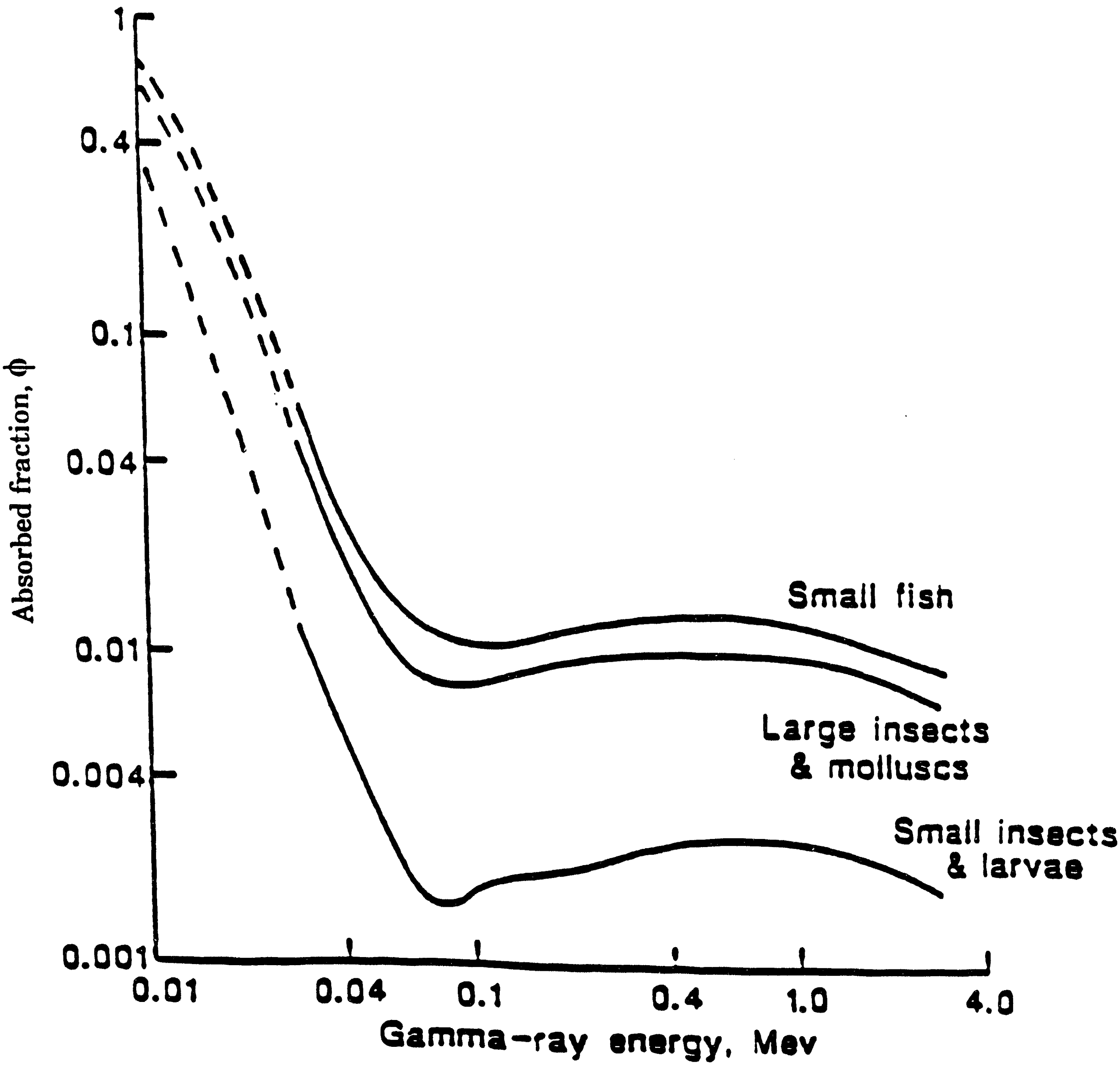

Fig. A.1 Derived absorbed fractions as a function of γ-ray energy (small fish, large insects and molluscs, and small insects and larvae).

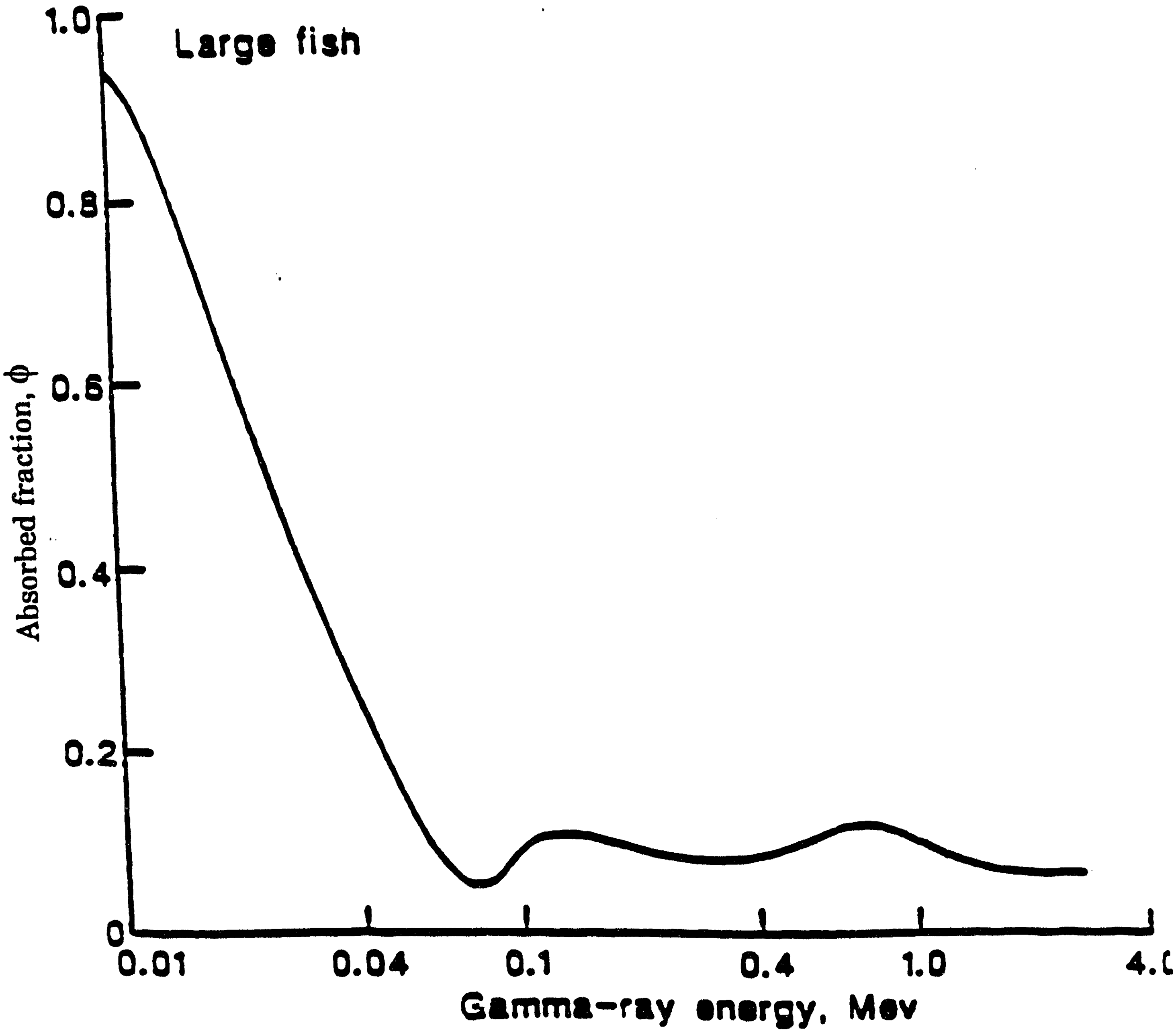

Fig. A.2 Derived absorbed fractions as a function of γ-ray energy (large fish).

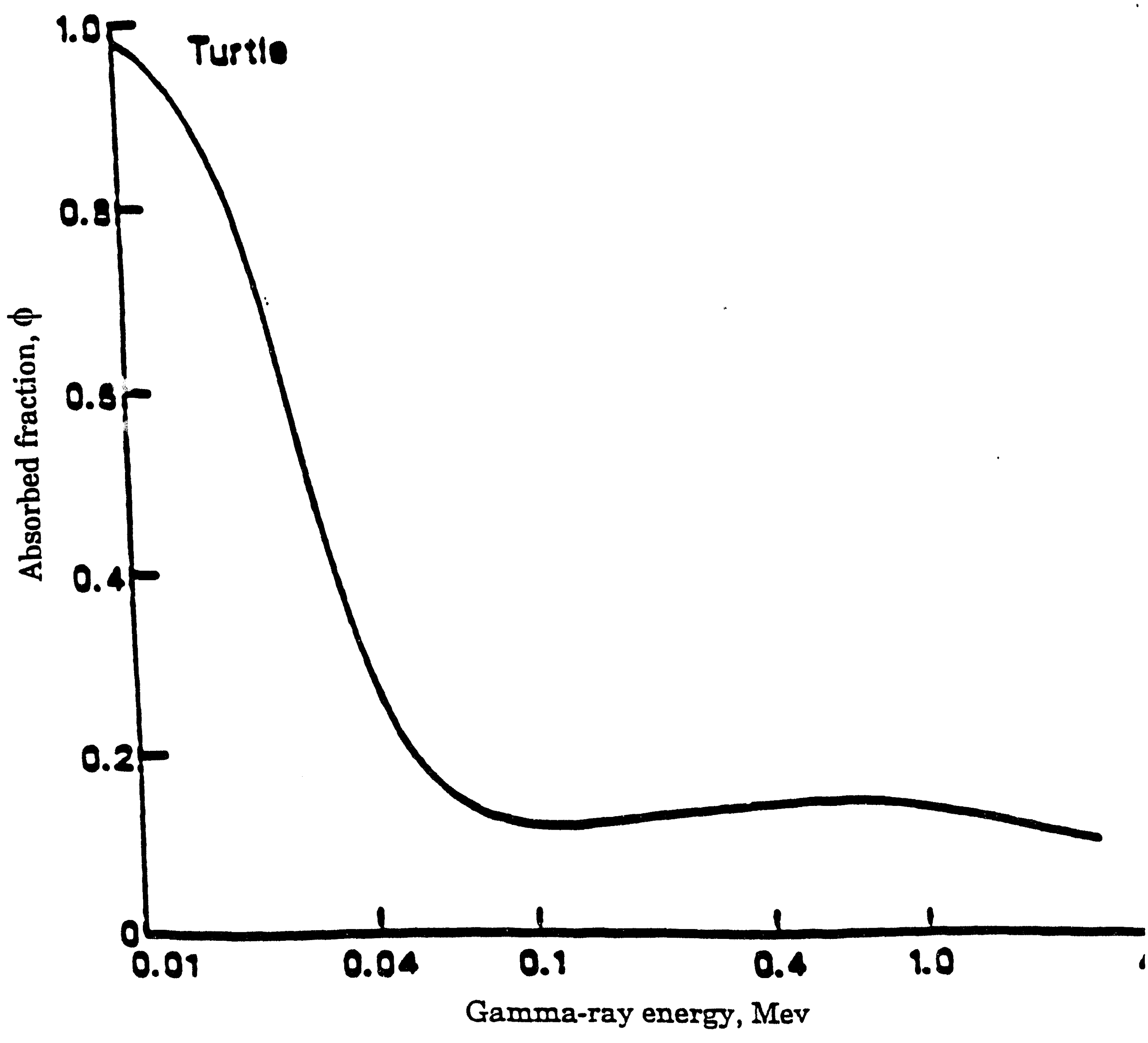

Fig. A.3 Derived absorbed fractions as a function of γ-ray energy (turtle).

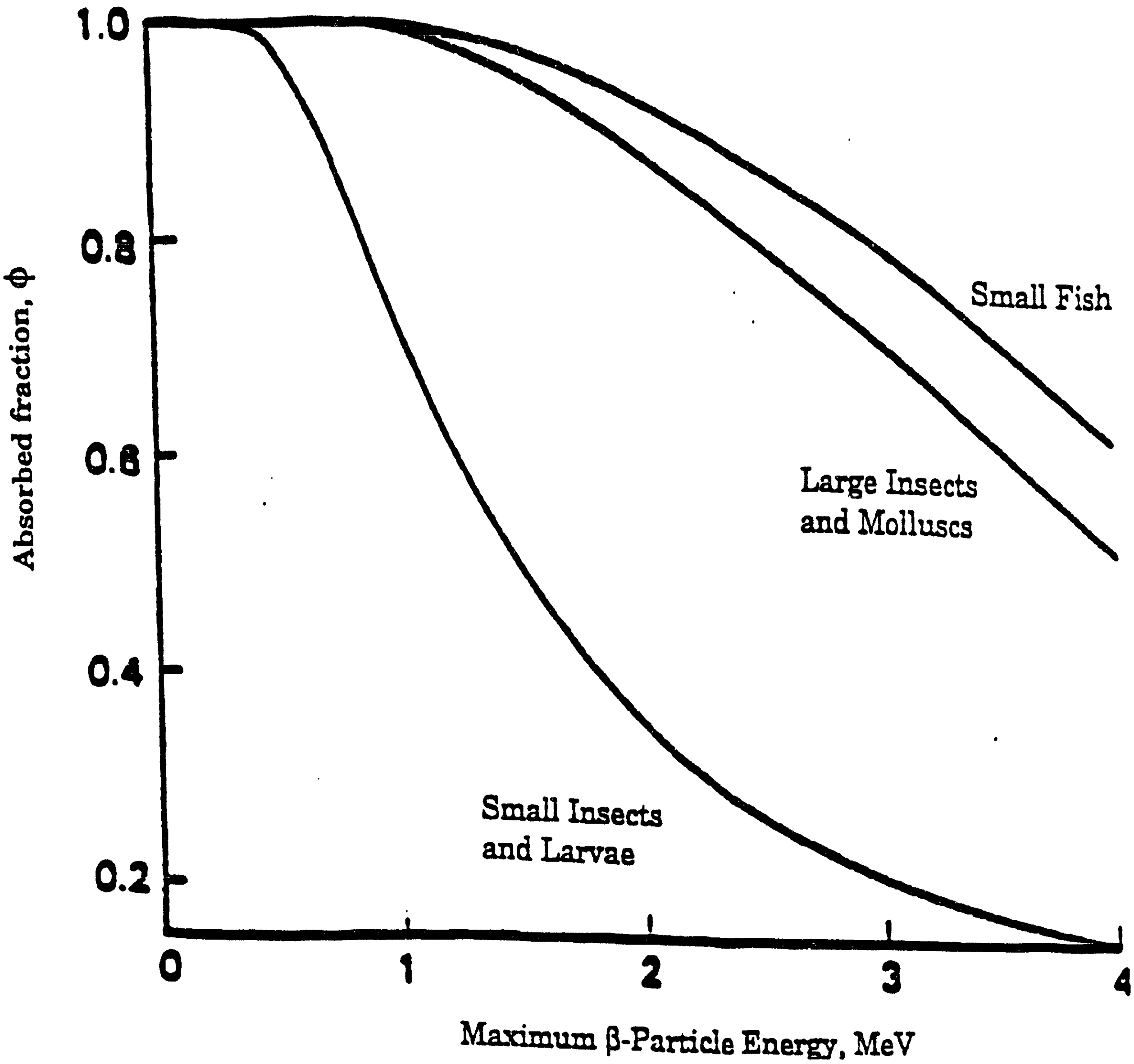

Fig. A.4 Absorbed fraction as a function of β-particle energy for three small geometries.

Table A.1. Average energies of selected beta and gamma emitters

Element	Biological Concentration Factor*	Radiological Half-life	Maximum beta energy (MeV)	Average beta energy (MeV)	Average gamma energy (MeV)
Antimony-125	15	2.7 y	6.12E-01	9.93E-02	4.30E-01
Barium-140 + Lanthanum-140	4	12.8 d, 40.22 h	3.21E+00	8.44E-01	2.49E+00
Cerium-141	30	33 d	5.80E-01	1.70E-01	7.61E-02
Cerium-144 + Praseodymium-144	30	284 d,17.3 m	3.30E+00	1.30E+00	5.25E-02
Cesium-134	30	2.06 y	1.45E+00	1.63E-01	1.55E+00
Cesium-137 + Barium-137m	2000	30 y, 2.6 min	1.17E+00	2.52E-01	5.96E-01
Chromium-51	--	27.8 d	--	3.86E-03	3.26E-02
Cobalt-60	300	5.26 y	1.48E+00	9.65E-02	2.50E+00
Europium-154	50	16 y	1.85E+00	2.88E-01	1.22E+00
Europium-155	50	1.81 y	2.47E-01	6.26E-02	6.05E-02
Hydrogen-3	1	12.3 y	1.80E-02	5.68E-03	--
Iodine-131	40	8.05 d	8.10E-01	1.90E-01	3.80E-01
Niobium-95	30	35 d	9.30E-01	4.40E-02	7.66E-01
Phosphorus-32	--	14.3 d	1.71E+00	6.95E-01	--
Potassium-40	10000	1.26E+09 y	1.32E+00	5.23E-01	1.56E-01
Ruthenium-103	10	39.6 d	7.10E-01	7.45E-02	4.68E-01
Ruthenium-106 + Rhodium-106	10	367 d, 30 s	3.58E+00	1.42E+00	2.01E-01
Sodium-24	--	15.0 h	4.17E+00	5.53E-01	4.12E+00
Strontium-90 + Yttrium-90	60	28.1 y, 64 h	2.79E+00	1.13E+00	1.69E-06
Technetium-99	20	2.12E+05 y	2.95E-01	1.01E-01	--
Uranium-237	10	6.75 d	2.48E+00	1.94E-01	1.42E-01
Zinc-65	--	245 d	--	6.87E-03	5.84E-01
Zirconium-95	300	65 d	1.13E+00	1.16E-01	7.39E-01

*The biological concentration factor (BCF) is the ratio of the concentration of the contaminant in freshwater fish to the concentration in water at steady-state conditions. Most BCFs were taken from IAEA Report No. 57 (1982) "Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases".

All average beta and gamma energies were obtained from ICRP Report No. 38. Radiations that contribute less than 0.1% of the energy per transformation are omitted. Average Beta Energies = beta particles, conversion electrons, and Auger radiations, including those for short-lived progeny. Average Gamma Energies = x-rays, gamma-rays, and photon radiations, including those for short-lived progeny.

Table A.2. Average energies of selected alpha emitters including those of naturally occurring alpha decay series

Element	Biological Concentration Factor*	Radiological Half-life	Average alpha & alpha recoil (MeV)	Maximum beta energy^ (MeV)	Average beta energy (MeV)	Average gamma energy (MeV)
Plutonium-239	30	2.41E+04 y	5.23E+00		6.65E-03	7.96E-04
Plutonium-240	30	6.54E+04 y	5.24E+00		1.06E-02	1.73E-03
Thorium Series						
Thorium-232	100	1.41E + 10 y	4.07E+00		1.25E-02	1.33E-03
Radium-228	50	5.75 y	--		1.69E-02	4.14E-09
Actinium-228		6.13 h	--		4.60E-01	9.30E-01
Thorium-228	100	1.91E+03 y	5.49E+00		2.05E-02	3.30E-03
Radium-224	50	3.64 d	5.78E+00		--	--
Radon-220		55 s	6.40E+00		8.19E-06	3.85E-04
Polonium-216	50	0.15 s	6.91E+00		1.61E-07	1.69E-05
Lead-212	300	10.64 h	--		1.75E-01	1.48E-01
Bismuth-212	10	60.6 m	2.22E+00	5.80E-01	4.69E-01	1.85E-01
Polonium-212 (64% yield)	50	305 μs	8.95E+00	2.25E+00	--	--
Neptunium Series						
Americum-241	30	458 y	5.57E+00		5.19E-02	3.24E-02
Neptunium-237	30	2.14E+06 y	4.84E+00		6.85E-02	3.43E-02
Protactinium-233	10	27.0 d	--		1.95E-01	2.03E-01
Uranium-233	10	1.59E+05 y	4.89E+00		6.08E-03	1.31E-03
Thorium-229	100	7.34E+03 y	4.95E+00		1.14E-01	9.54E-02
Radium-225	50	.14.8 d	--		9.48E-05	1.37E-02
Actinium-225		10.0 d	5.86E+00		2.17E-02	1.79E-02
Francium-221		4.8 m	6.41E+00		9.81E-03	3.10E-02
Astatine-217		0.032 s	7.19E+00		3.66E-05	3.08E-04
Bismuth-213	10	47 m	1.29E-01	1.39E+00	4.40E-01	1.33E-01
Polonium-213	50	4.2 μs	8.54E+00		--	--
Lead-209	300	3.3 h	--	6.35E-01	1.98E-01	--

Table A.2 (continued)

Element	Biological Concentration Factor[a]	Radiological Half-life	Average alpha & alpha recoil (MeV)	Maximum beta energy[A] (MeV)	Average beta energy (MeV)	Average gamma energy (MeV)
Uranium Series						
Uranium-238	10	4.51E+09 y	4.26E+00		1.00E-02	1.36E-03
Thorium-234	100	24.1 d	--		5.92E-02	9.34E-03
Protactinium-234m	10	1.17 m	--	2.29E+00	8.20E-01	1.13E-02
Uranium-234	10	2.47E+05 y	4.84E+00		1.32E-02	1.73E-03
Thorium-230	100	7.7E+04 y	4.74E+00		1.46E-02	1.55E-03
Radium-226	50	1.62E+03 y	4.86E+00		3.59E-03	6.47E-03
Radon-222		3.823 d	5.59E+00		1.09E-05	3.98E-04
Polonium-218	50	3.05 m	6.11E+00		1.42E-05	9.12E-06
Lead-214	300	26.8 m	--	6.70E-01	2.91E-01	2.48E-01
Astatine-218 (.02% yield)		2 s	6.82E+00		4.00E-02	6.72E-03
Bismuth-214	10	19.7 m	--	3.26E+00	6.48E-01	1.46E+00
Polonium-214	50	164 µs	7.83E+00		8.19E-07	8.33E-05
Lead-210	300	22.3 y	--		3.80E-02	4.81E-03
Bismuth-210	10	5.01 d	--	1.16E+00	3.89E-01	--
Polonium-210	50	138.4 d	5.40E+00		8.18E-08	8.50E-06
Actinium Series						
Uranium-235	10	7.04+08 y	4.47E+00		4.80E-02	1.54E-01
Thorium-231	100	25.5 h	--		1.63E-01	2.55E-02
Protactinium-231	10	3.28E+04 y	5.04E+00		6.28E-02	4.76E-02
Actinium-227		21.77 y	6.90E-02		1.56E-02	2.31E-04
Thorium-227	100	18.7 d	5.95E+00		4.57E-02	1.06E-01
Radium-223	50	11.43 d	5.75E+00		7.46E-02	1.33E-01
Radon-219		4.0 s	6.88E+00		6.30E-03	5.58E-02
Polonium-215	50	1.78 ms	7.52E+00		6.30E-06	1.76E-04
Lead-211	300	36.1 m	--	1.36E+00	4.54E-01	5.03E-02
Bismuth-211	10	2.15 m	6.68E+00			
Thallium-207	10000	4.79 m	--	1.52E+00	4.93E-01	2.21E-03

Table A.2 (footnotes)

*The biological concentration factor (BCF) is the ratio of the concentration of the contaminant in freshwater fish to the concentration in water at steady-state conditions. Most BCFs were taken from IAEA Report No. 57 (1982) "Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases".

^ Only maximum beta energies > 0.5 MeV are listed.

All average energies were obtained from ICRP Report No. 38. Radiations that contribute less than 0.1% of the energy per transformation are omitted. Average Beta Energies = beta particles, conversion electrons, and Auger radiations, including those for short-lived progeny. Average Gamma Energies = x-rays, gamma-rays, and photon radiations, including those for short-lived progeny.

Table A.3 Reproductive effects in fish and invertebrates in natural populations exposed to chronic irradiation.

Organism and life stage exposed	Exposure regime	Observation	Reference
Fish			
Gambusia affinis (mosquitofish)	White Oak Lake Oak Ridge, TN contaminated since 1943, 1960 dose rate ≥ 10 mGy d^{-1} falling to 3.5, 1.8 and 0.6 m Gy d^{-1} 1965, 1971, 1973, and 1975, respectively	Greater brood size and embryo mortality in fish	Blaylock 1969; Trabalka and Allen 1977; Blaylock and Frank 1980
Rutilus rutilus all life stages (Roach)	Ural Lakes (USSR) contaminated in the 1950s; early 1970s dose rate 7-15 mGy d^{-1}	Lower fecundity and delay in spawning	Voronina et al. 1974; Peshkov et al. 1978
Invertebrates			
Physa heterstropha all stages (pond snail)	White Oak Lake 6.5 mGy d^{-1}	Egg-capsule production reduced, but with more eggs per capsule, production rate comparable to to controls	Cooley 1973a

Table A.4 Reproductive effects in fish exposed to chronic irradiation under laboratory conditions

Organism and life stage exposed	Exposure regime	Observation	Reference
Fish			
Oncorhynchus tschawytscha embryos (chinook salmon)	5.0 - 475 mGy d^{-1} for 80 d	Gonadal development of smolts was retarded at ≥ 95 mGy d^{-1}	Bonham and Donaldson 1972
Poecilia reticulata 0-3-d neonates to adults	Tritiated water 100-410 mGy d^{-1} for 17 d	Male courtship activity reduced	Erickson 1973
Poecilia reticulata 0-3-d neonates to adults	40.8-305 mGy d^{-1}	Infertile at 40.8 and 96 mGy d^{-1} but fecundity unaffected	Purdom and Woodhead 1973
Poecilia reticulata 0-3-d neonates to adults	40.8-305 mGy d^{-1} up to 988 d	Fecundity reduced to 57, 52 and 3.5% of controls at 40.8, 96, and 305 mGy d^{-1}, respectively; smaller brood size all doses	Woodhead 1977
Oryzias Latipes adult males (medaka)	12-800 mGy d^{-1} for 60-120 d	Dose rates ≥65 mGy d^{-1} for 60 d increased sterility	Hyodo-Taguchi 1980
Ameca splendes young adults	185 mGy d^{-1} up to 244 d	Complete sterility after 190 d	Rackham and Woodhead 1984

Table A.5 Reproductive effects in invertebrates exposed to chronic gamma irradiation from a ^{60}Co source under laboratory conditions

Organism and life stage exposed	Exposure regime	Observation	Reference
Daphnia pulex all life stages (cladoceran)	217-721 mGy h^{-1} for 19 h d^{-1} for 20 to 35 d	Decreased birth rate at doses ≥ 4,610 mGy d^{-1}	Marshall 1962
Physa Heterostropha adult (pond snail)	240-6000 mGy d^{-1} for 168 d	Egg and egg-capsule production reduced progressively over dose rates 480-6,000 mGy d^{-1} 25°C temperature	Cooley and Miller 1971
Physa heterostsropha adult (pond snail)	240-1200 mGy d^{-1} for 98 d	Egg production and egg-capsule reduced progressively over dose range at 25°C but only at 1200 mGy d^{-1} at 15°C	Cooley 1973

Table A.6 Physiological and histological changes in fish and invertebrates exposed to chronic irradiation under laboratory conditions

Organism and life stage exposed	Exposure regime	Observation	Reference
Fish			
Salmo gairdnerii embryos (rainbow trout)	2 and 20 mGy d^{-1}	Lowered antibodies to *Chondrococcus columaris* disease in juveniles/ yearlings	Strand et al. 1973
Gambusia affinis adult (mosquitofish)	120-130 mGy d^{-1}	No hemopoietic damage after 37 d	Cosgrove et al. 1975
	360 or 720 mGy d^{-1}	mild hemopoietic atrophy in kidney and spleen in some fish after 128 d	
Gambusia affinis adult (mosquito fish)	312-1300 m Gd^{-1} 40 d	Testis atrophy at all dose rates; damage to other tissues not observed	Cosgrove and Blaylock 1973
Poecilia reticulata 0-3-d neonates to adult (guppy)	40.8-305 mGy d^{-1} up to 974 d	Oogenesis affected at all dose rates; damage appears earlier as dose rate increases	Woodhead 1977
Oryzia latipes adult males (medaka)	Tritiated water 10-210 mGy d^{-1} 30 d	Severe depletion of spermatogonia at ≥100 mGy d^{-1}	Hyodo-Taguchi and Egami 1977
Oryzia latipes adult males (medaka)	12-800 mGy d^{-1} for 120 d	Depletion of spermatogonia at ≥148 mGy d^{-1}	Hyodo-Taguchi 1980
Ameca splendens young adults	185 mGy d^{-1} up to 244 d	Spermatogenesis more radiosensitive than oogenesis	Rackham and Woodhead 1984
Invertebrates			
Physa heterostropha adult (pond snail)	240-1,200 mGy d^{-1} for 98 d	Partial atrophy of gonads in some snails at 1,200 mGy d^{-1}, but not at lower dose rates	Cooley 1973b

Appendix B

EXAMPLES SHOWING HOW TO CALCULATE RADIATION DOSE RATES TO AQUATIC ORGANISMS

Examples showing how to calculate radiation dose rates to aquatic organisms

The following examples are included to illustrate how to calculate radiation dose rates to aquatic organisms exposed to low levels of radionuclides in the environment. In all examples, it is assumed that a constant level of radioactivity is present and that the radioactivity in the organism is in equilibrium with that in the environment. The activity levels used in these examples are hypothetical and not intended to represent actual conditions. The energy values (MeV) used in the following examples were obtained from Tables A.1 and A.2. The values for phi (Φ) were obtained from Figs. A.1 through A.4.

Example 1: γ- and β-dose rates to a large fish

Given:	Isotope	$^{137}Cs + ^{137m}Ba$
	Geometry	Large fish
	Activity in organism	100 Bq/kg wet weight
	Water activity	Use BCF to calculate (Bq/L)
	Sediment activity	1500 Bq/kg wet weight

Equation (1) is used to calculate the internal γ-radiation dose rate as follows:

$$D_\gamma = 5.76 \times 10^{-4}\ E_\gamma\ n_\gamma\ \Phi\ C_o \qquad \mu Gy\ h^{-1}$$

substituting the average γ-radiation energy from Table A.1 for the E_γ n_γ terms and obtaining Φ from Fig. A.2 gives

$$D_\gamma = (5.76 \times 10^{-4})(0.596)(0.11)(100)$$

$$D_\gamma = 3.78 \times 10^{-3}\ \mu Gy\ h^{-1} \text{ internal } \gamma\text{-radiation.}$$

The activity of ^{137}Cs in the water is calculated from the activity in the fish using a biological concentration factor of 2000 (Table A.1). Equation (2) is used to calculate the external γ-radiation dose rate from water as follows:

$D_\gamma = 5.76 \times 10^{-4}\ E_\gamma\ n_\gamma\ (1\text{-}\Phi)\ C_w$ μGy h^{-1}

$D_\gamma = (5.76 \times 10^{-4})(0.596)(1\text{-}0.11)(100/2000)$

$D_\gamma = 1.53 \times 10\text{-}5$ μGy h^{-1} external γ-radiation from water.

Assuming the fish spends one-half its time near the sediment surface (R=0.5), the external γ-radiation dose rate from sediment is calculated as follows:

$D_\gamma = 2.88 \times 10^{-4}\ E_\gamma\ n_\gamma\ (1\text{-}\Phi)\ C_s\ R$ μGy h^{-1}

$D_\gamma = (2.88 \times 10^{-4})(0.596)(1\text{-}0.11)(1500)(0.5)$

$D_\gamma = 1.15 \times 10^{-1}$ μGy h^{-1} external γ-radiation from sediment.

The total γ-radiation dose rate to the fish in this scenario would be 1.18×10^{-1} μGy h^{-1} and the major contributor is sediment.

The internal dose rate from the β-radiation emitted by ^{137}Cs + ^{137m}Ba is calculated as follows:

$D_\beta = 5.76 \times 10^{-4}\ \overline{E}_\beta\ n_\beta\ C_o$ μGy h^{-1}

using the average energy for all β-decay spectra from Table A.1, the equation becomes

$D_\beta = (5.76 \times 10^{-4})(0.252)(100)$

$D_\beta = 1.45 \times 10^{-2}$ μGy h^{-1} internal β-radiation.

The β-dose rate from water and sediment is insignificant to an organism the size of a large fish. The dose rate from the γ- and β-radiation combined is 1.33×10^{-1} μGy h^{-1}.

Example 2: β-dose rate to mollusc

Given:	Isotope	^{90}Sr + ^{90}Y
	Geometry	Mollusc
	Activity in organism	100 Bq/kg wet weight
	Water activity	Use BCF to calculate (Bq/L)
	Sediment activity	167 Bq/kg wet weight

The energy of the γ-radiation from ^{90}Sr + ^{90}Y is insignificant compared to the β-radiation and can be ignored (Table A.1). The internal β-dose rate to molluscs is calculated as follows:

$$D_\beta = 5.76 \times 10^{-4}\ \bar{E}_\beta\ n_\beta\ \Phi\ C_o \qquad \mu Gy\ h^{-1}$$

using the average energy for all β-decay modes given in Table A.1 for $\bar{E}_\beta$ n_β and Φ for the β-particle maximum energy from Fig. A.4, the equation becomes

$$D_\beta = (5.76 \times 10^{-4})(1.13)(0.76)(100)$$

$$D_\beta = 4.95 \times 10^{-2}\ \mu Gy\ h^{-1} \text{ internal } \beta\text{-radiation.}$$

External β-radiation to molluscs from ^{90}Sr + ^{90}Y in water and sediment would be insignificant because the animal's shell would serve as an effective shield.

Example 3: α-dose rate to larvae

Given:	Isotope	^{239}Pu
	Geometry	Larva
	Activity in organism	100 Bq/kg wet weight

The internal α-radiation dose rate from ^{239}Pu is calculated as follows:

$$D_\alpha = 5.76 \times 10^{-4}\ E_\alpha n_\alpha\ C_o \qquad \mu Gy\ h^{-1} \qquad \text{Eq. (8)}$$

$$D_\alpha = (5.76 \times 10^{-4})(5.23)(100)$$

$$D_\alpha = 3.01 \times 10^{-1}\ \mu Gy\ h^{-1} \text{ internal } \alpha\text{-radiation.}$$

Only an internal α-dose rate from ^{239}Pu is considered because the γ- and β-radiations are very weak and α-radiation from external sources would not penetrate the outer covering of the larvae.

Example 4: dose rates to large fish from isotopes in an α-decay chain

Given:	Isotopes	^{226}Ra and short-lived progeny
	Geometry	Large fish
	Activity in organism	100 Bq of ^{226}Ra/kg wet weight
	Water activity	Use BCF to calculate (Bq/L)
	Sediment activity	1000 Bq of ^{226}Ra/kg wet weight

^{226}Ra is a member of the ^{238}U decay chain and it has a series of progeny with short half-lives (Table A.2). It is reasonable to assume that because of their short half-lives, these progeny will be present at the same activity level as ^{226}Ra. However, ^{226}Ra decays to ^{222}Rn which is a gas with a 3.8 day half-life. ^{222}Rn produced in water or surface sediment would escape to the atmosphere; therefore, the succeeding progeny would not be present in surface sediment or water unless other sources were available. In the example we assume that 30% of the ^{222}Rn produced within a fish remains in the fish tissue so that the activity level of the succeeding progeny will also be 30% of the ^{226}Ra.

Using average energies from Table A.2 and Equation (8), the internal α-dose rates for ^{226}Ra and its short-lived progeny are calculated as follows:

$$D_\alpha = 5.76 \times 10^{-4}\, E_\alpha n_\alpha C_o \qquad \mu\text{Gy h}^{-1}$$

Isotopes

^{226}Ra $D_\alpha = (5.76 \times 10^{-4})(4.86)(100) = 2.80 \times 10^{-1}$ μGy h^{-1}

^{222}Rn $D_\alpha = (5.76 \times 10^{-4})(5.59)(100 \times 0.30) = 9.66 \times 10^{-2}$ μGy h^{-1}

^{218}Po $D_\alpha = (5.76 \times 10^{-4})(6.11)(100 \times 0.30) = 1.06 \times 10^{-1}$ μGy h^{-1}

^{214}Pb D_α = no alpha

^{214}Bi D_α = no alpha

^{214}Po $D_\alpha = (5.76 \times 10^{-4})(7.83)(100 \times 0.30) = 1.35 \times 10^{-1}$ μGy h^{-1}

Total internal α-dose rate = 6.18×10^{-1} μGy h^{-1}

The internal dose rate from the γ-emitters with the highest energies is calculated as follows:

$$D_\gamma = 5.76 \times 10^{-4}\, E_\gamma\, n_\gamma\, \Phi\, C_o \qquad \mu Gy\ h^{-1} \qquad \text{Eq. (1)}$$

$$^{214}Pb\ D_\gamma = (5.76 \times 10^{-4})(0.248)(0.09)(100 \times 0.30) \qquad = 3.86 \times 10^{-4} \quad \mu Gy\ h^{-1}$$

$$^{214}Bi\ D_\gamma = (5.76 \times 10^{-4})(1.46)(0.07)(100 \times 0.30) \qquad = 1.77 \times 10^{-3} \quad \mu Gy\ h^{-1}$$

$$\textbf{Total internal } \boldsymbol{\gamma}\textbf{-dose rate} \qquad = \mathbf{2.16 \times 10^{-3}} \quad \boldsymbol{\mu}\mathbf{Gy\ h^{-1}}$$

The internal dose rate from the β-emitters with the highest energies is

$$D_\beta = 5.76 \times 10^{-4}\, \bar{E}_\beta\, n_\beta\, \Phi\, C_o \qquad \mu Gy\ h^{-1} \text{Eq. (5)}$$

$$^{214}Pb\ D_\beta = (5.76 \times 10^{-4})(0.291)(1)(100 \times 0.30) \qquad = 5.03 \times 10^{-3} \quad \mu Gy\ h^{-1}$$

$$^{214}Bi\ D_\beta = (5.76 \times 10^{-4})(0.648)(1)(100 \times 0.30) \qquad = 1.12 \times 10^{-2} \quad \mu Gy\ h^{-1}$$

$$\textbf{Total internal } \boldsymbol{\beta}\textbf{-dose rate} \qquad = \mathbf{1.62 \times 10^{-2}} \quad \boldsymbol{\mu}\mathbf{Gy\ h^{-1}}$$

As shown above, the α-dose rate is more than an order of magnitude greater than the dose rates from the γ- and β-emissions. Additionally, the relative biological effectiveness of a radiation is 20 times greater than γ- or β-radiation; consequently, the main concern from ^{226}Ra would be the α-dose.

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