ATTACHMENT F-1

METHODOLOGY FOR ESTIMATING DOSES TO NON-HUMAN BIOTA

UNSCEAR 2013 Report, Annex A, Levels and effects of radiation exposure due to the nuclear accident after the 2011 great east-Japan earthquake and tsunami, Appendix F (Assessment of doses and effects for non-human biota)

Notes

Contents

I. INTRODUCTION

1. In view of the great variety of plants and animals that could be exposed to radiation at any given site of concern, it has been argued [Pentreath, 1999] that an assessment could be reduced to manageable proportions by adopting reference exposure and dose models. This has been the approach advocated by the Committee [UNSCEAR, 2008], consistent with the International Commission on Radiological Protection [ICRP, 2008]. The organisms selected in the UNSCEAR 2008 Report form the basis for the exposure estimates made in the current assessment (table 1).

Table 1. List of organisms selected by the Committee for assessing exposures [UNSCEAR, 2008]

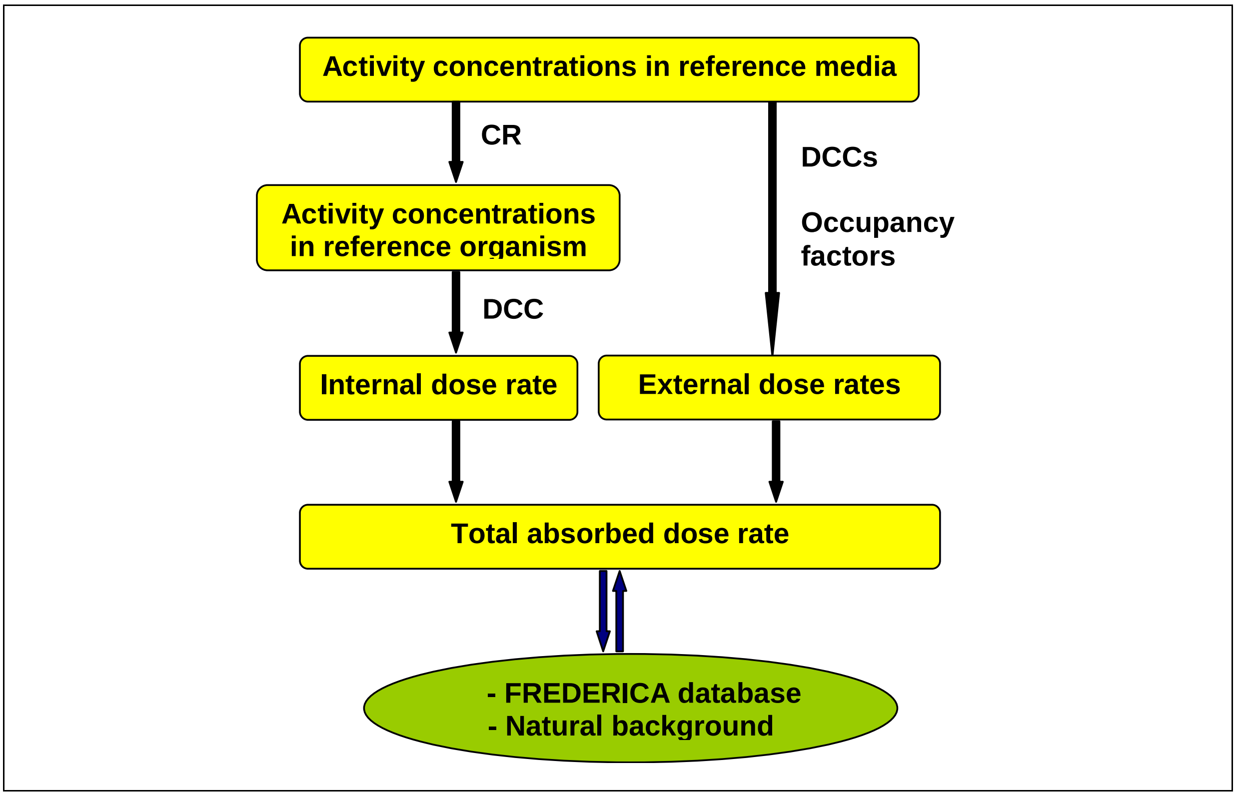
Earthworm/soil invertebrate	Rat/burrowing mammal	Bee/above ground invertebrate
Wildgrass/grasses, herbs and crops	Pine tree/tree	Deer/herbivorous mammal
Duck/bird	Frog/amphibian	Brown seaweed/macroalgae
Trout/pelagic fish	Flatfish/benthic fish	Crab/crustacean

2. The methodology that was used for both terrestrial and aquatic ecosystems was based around the ERICA Integrated Approach [Larsson, 2008] and consisted of the following steps: (a) select key species and radionuclides for analysis; (b) conduct the assessment preferably using actual radionuclide concentrations in the biota, measured over time and space from the first day of the accident; alternatively, (c) conduct an equilibrium assessment using concentration ratios (CRs) to infer radionuclide concentrations in biota from measured concentrations in media (soil and water), where appropriate; or (d) use kinetic models to calculate “dynamic” concentrations in biota based on measured concentrations in media (i.e. air, soil, freshwater, seawater, sediment); (e) enter the results into the ERICA Tool (dose

calculation module) [Brown et al., 2008], to project dose rate as a function of time from which integrated cumulative doses can be calculated. Once the dose rates were estimated, risk to non-human biota was assessed either deterministically or probabilistically.

3. The ERICA Integrated Approach was designed to provide guidance on impacts of radioactive material on the environment to ensure that decisions on environmental issues give appropriate weight to the exposure, effects and risks from ionizing radiation exposure. Emphasis was placed on protecting the structure and function of ecosystems from radionuclides [Larsson, 2008], and supporting software (the ERICA Tool) was developed to serve this purpose [Brown et al., 2008]. The part of the approach that deals with the exposure assessment (figure I) encapsulates the quantification of risk to organisms in the environment through the application of transfer and dosimetric models and, for screening purposes, the comparison of predicted dose rates with appropriately derived benchmarks.

Figure I. The ERICA Tool assessment



CR = concentration ratio, DCC = dose conversion coefficient (relating activity concentrations to dose rates); media is soil, water, sediment or air depending upon the ecosystem, available data and radionuclide under consideration. The FREDERICA (radiation effects) database is described in Copplestone et al. [2008].

4. The derivation of dose-rate can be split into two components, these being transfer and dose estimation. The two components are considered in the following sections.

II. TRANSFER

5. In many cases, activity concentrations in plants and animals needed to be derived from measured or estimated activity concentrations of radionuclides in environmental media (soil, water and/or sediments). Where possible, activity concentrations in plants and animals could be used directly in subsequent dose-rate calculations but the expectation was that such data would be quite limited compared to measured concentrations in environmental media. The approach

taken to simulating transfer was twofold. The first approach was to use a simple model based on concentration ratios where the assumption was that the activity concentrations in the body of a selected plant or animal were in equilibrium with the surrounding medium. Bearing in mind that equilibrium approaches have limited applicability in situations where environmental concentrations are changing rapidly with time [Coughtrey and Thorne, 1983; Brown et al., 2004], a second approach based upon kinetic models was also applied.

6. In the intermediate phase after the Fukushima-Daiichi Nuclear Power Station (FDNPS) accident (approximately the first two months), the fact that activity concentrations in various compartments within the ecosystems were unlikely to have been in equilibrium provided further support to the contention that it was more appropriate to model the activity concentrations in selected biota using kinetic models. For example, in the context of the FDNPS releases, Vives i Batlle [2011] considered that the activity concentrations in biota predicted by equilibrium-based models may deviate significantly from actual activity concentrations in the biota when activity concentrations were changing rapidly in time and space. For example, the application of concentration ratios had a tendency to produce an over-estimation in the initial phase when activity concentrations in media were increasing [Psaltaki et al., 2012]. Hence, a dynamic modelling approach was likely more appropriate to provide an indication of changing dose rates in the early stages of the accident. However, in view of the more involved parameterization required, the dynamic modelling work was limited to selected key species and a limited number of the radionuclides contributing most to dose.

7. The application of equilibrium models was essentially the approach that had been advocated by the Committee [UNSCEAR, 2008] for constant long-term releases of radioactive material, for deriving activity concentrations in terrestrial and aquatic organisms (with the exception of terrestrial flora where some consideration had also been given to deposition and interception of radionuclides).

A. Equilibrium approach to estimate activity concentrations in biota

8. Concentration ratios are used to derive activity concentrations in the ERICA Tool [Brown et al., 2008]. The approach was intended for assessing the impact of routine releases of radionuclides to the environment. By way of example, for terrestrial biota the whole-body concentration ratio, CR_{wo}, is defined as in equation (1)

$$CR_{wo} = \frac{A_{b,r}^{biota}}{A_{r}^{soil}} \qquad (1)$$

where:

$A_{b,r}^{biota}$ is the activity concentration of radionuclide *r* in the whole organism of biota "*b*" (Bq kg^{-1} fresh weight (f.w.));

A_{r}^{soil} is the activity concentration of radionuclide *r* in soil (Bq kg^{-1} dry weight (d.w.))

For aquatic organisms, activity concentrations in soil are replaced by those in water.

9. The transfer database in the ERICA Tool based upon Hosseini et al. [2008], Beresford et al. [2008] and supplemented by the more updated information provided in ICRP [2009] was used to derive activity concentrations in non-human biota for the UNSCEAR evaluation. Concentration ratios from ICRP [2009] are shown in tables 2–4.

Table 2. Concentration ratios for terrestrial ecosystem from ICRP [2009]

Element	*Organism*	*Concentration ratio (Bq kg^{-1} f.w. per Bq kg^{-1} d.w.)*
Cs	Pine tree/tree	7.5×10^{-2}
I	Pine tree/tree	5.3×10^{-2}
Te	Pine tree/tree	2.5×10^{-1}
Cs	Wildgrass/grasses,herbs and crops	8.6×10^{-1}
I	Wildgrass/grasses,herbs and crops	5.3×10^{-2}
Te	Wildgrass/grasses,herbs and crops	2.5×10^{-1}
Cs	Bee/above ground invertebrate	4.7×10^{-3}
I	Bee/above ground invertebrate	2.8×10^{-1}
Te	Bee/above ground invertebrate	3.8×10^{-2}
Cs	Earthworm/soil invertebrate	4.8×10^{-2}
I	Earthworm/soil invertebrate	1.4×10^{-1}
Te	Earthworm/soil invertebrate	3.8×10^{-2}
Cs	Rat/burrowing mammal	2.2×10^{-1}
I	Rat/burrowing mammal	4.0×10^{-1}
Te	Rat/burrowing mammal	2.1×10^{-1}
Cs	Deer/herbivorous mammal	1.6×10^{0}
I	Deer/herbivorous mammal	4.0×10^{-1}
Te	Deer/herbivorous mammal	2.1×10^{-1}
Cs	Duck/bird	2.2×10^{-1}
I	Duck/bird	4.0×10^{-1}
Te	Duck/bird	2.1×10^{-1}
Cs	Frog/amphibian	2.8×10^{-2}
I	Frog/amphibian	4.0×10^{-1}
Te	Frog/amphibian	2.1×10^{-1}

Table 3. Concentration ratios for freshwater ecosystem from ICRP [2009]

Element	*Organism*	*Concentration ratio (Bq kg^{-1} f.w. per Bq l^{-1})*
Cs	Trout/pelagic fish	2.7×10^{3}
I	Trout/pelagic fish	6.2×10^{1}
Te	Trout/pelagic fish	2.8×10^{2}
Cs	Frog/amphibian	1.6×10^{3}
I	Frog/amphibian	2.6×10^{2}
Te	Frog/amphibian	2.8×10^{2}
Cs	Duck/bird	4.4×10^{2}
I	Duck/bird	2.2×10^{2}
Te	Duck/bird	7.0×10^{2}

Table 4. Concentration ratios for marine ecosystem from ICRP [2009]

Element	*Organism*	*Concentration ratio (Bq kg^{-1} f.w. per Bq l^{-1})*
Cs	Flatfish/benthic fish	3.6×10^{1}
I	Flatfish/benthic fish	9.0×10^{0}
Te	Flatfish/benthic fish	1.0×10^{3}
Cs	Crab/crustacean	1.4×10^{1}
I	Crab/crustacean	3.0×10^{0}
Te	Crab/crustacean	1.0×10^{3}
Cs	Brown seaweed/macroalgae	1.2×10^{1}
I	Brown seaweed/macroalgae	1.4×10^{3}
Te	Brown seaweed/macroalgae	1.0×10^{4}

B. Dynamic approach to estimate activity concentrations in terrestrial flora and fauna

10. In view of the available data, it was most appropriate to split the modelling into flora (wild grass/grasses, herbs and crops and pine tree/tree) and fauna (deer/herbivorous mammal and rat/burrowing mammal) on the basis of the classifications given in UNSCEAR [2008].

11. Using a variant of the methodology given in UNSCEAR [2008], the activity concentration in flora could be estimated from the total deposition density using an expression accounting for interception by foliage, the direct deposition density onto soil, weathering losses of radionuclides from vegetation and uptake from soil to plant.

12. In the case of an acute deposition event, the radionuclide content on vegetation at time *t*, accumulated via direct deposition from the air, can be calculated (based on Brown et al. [2003]) as:

$$C^{air}_{flora,r} = \frac{f_{flora} \cdot D_{tot,r}}{b_{flora}} \cdot [e^{(-(\lambda_{flw,r} + \lambda_r) \cdot t)}] \qquad (2)$$

where:

$C_{flora,r}$ is the radionuclide activity concentration in flora from deposition (Bq kg^{-1} f.w.);
f_{flora} is the interception fraction for a given flora (dimensionless);
$D_{\mathrm{tot,r}}$ is the total deposition density of radionuclide r (Bq m^{-2});
$\lambda_{flw,r}$ is the weathering constant for a given flora for radionuclide r (d^{-1});
λ_r is the decay constant for radionuclide r(d^{-1});
b is standing biomass of the flora (kg m^{-2});
t is time (d).

13. For the same acute deposition, at time *t*, there is also a component of biota concentrations that arises from soil-to-plant transfer. In this case, an assumption was made that for this fraction of the concentration in the plant due to root uptake, equilibrium existed between the activity concentration in the plant and the soil.

$$C^{soil}_{flora,r} = \left[\frac{D_{tot,r} \cdot \left[(1 - f_{flora}) + f_{flora} \cdot (1 - e^{-\lambda_{flw,r} t}) \right] \cdot e^{-\lambda_r t}}{\rho_{soil} \cdot d_{soil}} \right] . CR_{flora,r} \qquad (3)$$

Where:

ρ_{soil} is the dry soil density (kg m^{-3} d m.);
d_{soil} is the depth of soil within which radionuclide r has become mixed (m);
$CR_{flora,r}$ is the soil-to-plant concentration ratio for radionuclide r (dimensionless).

All other parameters are described above for equation (2).

14. Finally, growth dilution may play an important role in determining activity concentrations in grass because the period of deposition considered, i.e. early spring, normally corresponds to substantial increases in vegetation biomass. A simple model was applied using information for grass yield at various calendar dates [Müller and Pröhl, 1993], assuming that the change in biomass with time was linear (table 5).

Table 5. Assumed change in biomass with season

Date	15 March	15 May	31 October
Yield (kg m^{-2} f.w.)	0.05	1.5	1.5

15. Application of this model also allowed for time-varying deposition rates to be considered. For this more complex situation, the problem could be solved numerically. There was an assumption in this model that a representative interception fraction *f* for a given flora type could be applied for the entire simulation period. Data compilations for agricultural systems in relation to this parameter [IAEA, 2010] indicate that the interception fraction depends on whether dry or wet deposition is occurring, the stage of development of the plant and plant type in question, the capacity of the canopy to retain water, elemental properties of the radionuclide, and other factors such as amount and intensity of rainfall in the case of wet deposition and particle sizes of the deposited material. The approach taken here was, therefore, arguably simplistic but — in view of the numerous uncertainties involved — it at least provided an indication of activity concentration levels following FDNPS releases and it attempted to model the dynamics of interception and loss from flora in contrast to approaches that consider soil-to-plant transfer only. In addition to the interception fraction, biomass (which clearly relates to the stage of development of the plant) also required further consideration as an important model parameter.

16. A map of vegetation coverage for Japan was available online [MOE, 2012]. Reference to map 564017 in this series, an area in proximity to Fukushima-Daiichi, showed natural/semi-natural ecosystems characterized by various plant communities including Korean hill/mountain cherry (*Prunus vercunda*), Japanese red pine (*Pinus densiflora*), red oak (*Quercus serrata*) and evergreen conifer plantations. Takahashi et al. [2002], provide information for a secondary deciduous broad-leaved forest in Japan where the total above- and below-ground biomass was given as 130 t ha^{-1} and the ratio of above- to below-ground biomass was 5.16. Clearly there would be differences in biomass relating to plant community type, species and age of forest stand but these data provide a reasonable indication for tree biomass for the subsequent calculations.

17. Empirical information on grass biomass in Fukushima Prefecture were not found for the dates of interest but generic information from published data could be used to provide indicative values. Although Schino et al. [2003] studied grasslands in mountainous areas of

central Italy, the work provides an indication of variations in grass biomass that can arise from seasonality and the presence of different species. The recorded range of grass biomass in this aforementioned study was approximately 60 to almost 700 g m^{-2} providing a useful context for selecting an appropriate biomass value for "wild grass/grasses, herbs and crops".

18. The growing season for Japanese pampas grass (*Miscanthus sinensis*) for sites in northern Japan is the period May to October [Shoji et al., 2012]. If this observation is considered typical for other grass species, it seems likely that the timing of the main deposition events following atmospheric radionuclide releases from the FDNPS, i.e. mid-March to early April, preceded any substantial new growth. Nonetheless, by mid-April it was possible to collect grass samples in areas affected by the Fukushima-Daiichi emissions, as evidenced by the supplementary material published by Yasunari et al. [2011], so it seems likely that some coverage of herbaceous vegetation would have been present at the time of atmospheric releases leading to a degree of interception, albeit to a relatively low degree, of deposited activity. This appears to be borne out by evidence from the early phase of the accident where it was possible to measure elevated concentrations of ^{131}I and ^{137}Cs in herbaceous "weeds" [NERH, 2011].

19. Yasunari et al. [2011] showed, using model simulations, that the highest deposition densities of radiocaesium downwind of FDNPS were clearly aligned with satellite-observed precipitation. There were numerous rainfall events during the emission period [Kato et al., 2012] as exemplified by a daily accumulated rainfall in excess of 20 mm in areas to the north-east of the FDNPS that occurred on 21 March 2011 [Yasunari et al., 2011].

20. Taking all of the above points into consideration, the grass biomass was likely to have been relatively low, as was the interception fraction. Best estimate values of 0.05 kg m^{-2} for biomass and 0.05 (dimensionless) for the interception fraction of radiocaesium on grass were selected.

21. The IAEA [2010] provides values for the mass interception fraction, f_B (defined as the interception fraction divided by biomass) of caesium deposited on grass ranging from 0.7 to 5.5, the values depending upon antecedent rainfall — the low value corresponding to a relatively high rainfall event. Furthermore, an f_B value of 1.1 was considered as typical for caesium deposited on grass following the Chernobyl accident. The selected values for interception fraction for caesium on grass and grass biomass for the model simulations corresponded to an f_B value of 1, a level which was reassuringly commensurate with the value expressing deposition from the Chernobyl accident.

22. Limitations to the concentration ratios used arose from an incompatibility between empirical data based on long-term steady-state conditions with a period directly following an accident. The CR values used (table 2) were based on empirical datasets from field investigations collated to avoid including data pertaining to the period directly following depositional events (e.g. global weapons test fallout and deposition following the Chernobyl accident for some radionuclides such as Cs, Pu, Sr and Am) and, thus, should omit values pertaining to deposition onto the surface of vegetation [Beresford et al., 2008]. These default CR data are generally assumed to correspond to, and thus are applicable for, a soil depth of 10 cm. There was thus an inconsistency with the observed distributions of radionuclides shortly following deposition. For example, Kato et al. [2012] reported that more than 86% of total radiocaesium and 79% of total ^{131}I were absorbed in the upper 2.0 cm in a soil profile from a cultivated area that had relatively high deposition density, sampled at the end of April 2011. This area was in proximity to the FDNPS site (less than 50 km distant, in a north-easterly direction). Furthermore, bioavailability of radiocaesium has been observed to decrease with time following its introduction to soils [Vidal et al., 1995] implying that CRs based upon long-

term post-depositional datasets might not appropriately reflect the transfer occurring in the early phase depositional environment. Indeed this contention is evidenced by reviews of published information on ^{137}Cs in the soil–plant system shortly after the Chernobyl accident [Fesenko et al., 2009]. Finally, the soil type, as defined by various soil properties, strongly influences transfer to plants [IAEA, 2010] and there will undoubtedly be differences in the soil types upon which the default data were based and the soil types in Japan for which the transfer parameters were applied.

23. Although some information exists on soil–to–grass transfer for the short term after accidents [Fesenko et al., 2009] and differs substantially from equilibrium transfer factors, these data are, by the author's own admission, insufficient for adequate transfer estimation. Furthermore, with the model constructed and parameterized in its current configuration, direct deposition dominates the total activity concentration in vegetation in the initial weeks. This means that although CR values are highly uncertain, their influence on the dose rates calculated was relatively unimportant.

24. The parameters were assigned different default values as shown in table 6. Two categories of flora — pine tree/tree and wild grass/grasses, herbs and crops — were considered.

Table 6. Parameters used to model activity concentrations in flora with time

Parameter	*Dependencies: flora, radionuclide*	*Value*	*Units and notes*	*References*
ρ_{soil}		1300	kg m^{-3}	Yasunari et al. [2011]
d_{soil}		0.05	m, Based on soil sampling information provided with deposition data	MEXT
f	Pine tree/tree, Cs Pine tree/tree, I	0.7 0.7	Dimensionless, IAEA present 50–90% deposition for global fallout and 50–60 year old pine stands	Bunzl and Schimmack [1989] IAEA [2010] IAEA [2010]
f	Wild grass/grasses, herbs and crops, Cs Wild grass/grasses, herbs and crops, I	0.05 0.03	f varies from 0.84 (dry deposition) to 0.027 (wet deposition, heavy rain) f_B for Chernobyl deposits = 0.7 (cf. 1.1. for Cs)	IAEA [2010] IAEA, 2010
b	Pine tree Wild grass/grasses, herbs and crops	11 0.05–1.5	kg m^{-2} kg m^{-2}	Takahashi et al. [2002] Table 5
$\lambda_{flw,r}$	Pine tree/tree, Cs Pine tree/tree, I	7.6×10^{-3} 7.6×10^{-3}	d^{-1}; IAEA [2010] considers a range of 7.6×10^{-3} to 2.8×10^{-2}	IAEA [2010]
$\lambda_{flw,r}$	Wild grass/grasses, herbs and crops, Cs Wild grass/grasses, herbs and crops, I	5.0×10^{-2} 7.0×10^{-2}	d^{-1}, Table VIII, p.37 [IAEA, 1996]	IAEA [1996]

25. Examples of kinetic models for terrestrial environments have been published in the open literature and one of these, the FASTer model, was selected for further application

[Beresford et al., 2010; Brown et al., 2003]. For herbivorous mammals, the data used for input could be those specifying the activity concentrations in grass as expressed above. Details were required regarding biokinetic parameters for various representative animals/fauna as described below.

$$\frac{dC_{r,a}}{dt} = \sum_{i=1}^{i=n} \left(x_i . AE_{r,i} . \frac{FMI}{M} . C_{r,i} \right) - C_{r,a} . \lambda_{r,a} \qquad (4)$$

Where :

x_i is the fraction of the diet associated with dietary component *i*;
$AE_{r,i}$ is the assimilation efficiency (dimensionless) for radionuclide *r* within dietary component *i*;
FMI/M is the ingestion rate per unit mass of animal (kg f.w. d^{-1} per kg f.w.);
$C_{r,i}$ is the activity concentration of radionuclide *r* in dietary component *i* (Bq kg^{-1} f.w.);
$C_{r,a}$ is the activity concentration of radionuclide *r* in the whole-body of the animal (Bq kg^{-1} f.w.);
$\lambda_{r,a}$ is the effective loss rate of radionuclide *r* from animal (d^{-1}) incorporating both excretion rate and physical decay of the radionuclide.

26. This model was applied to estimate the transfer to deer/herbivorous mammals and rat/burrowing mammals. Fresh matter ingestion rates (FMI) were derived using allometric relationships of the form given in equation (5) as shown in table 7. The animal masses that were used in these derivations were extracted from ICRP [2008].

$$FMI = a.M^b \qquad (5)$$

Where:

a is the multiplication constant in the allometric relationship for fresh matter intake by the animal (kg d^{-1});
b is the exponent in the allometric relationship for fresh matter intake by the animal (relative units);
M is the mass of the animal (kg).

Table 7. Estimated fresh matter ingestion rates, FMI, for the various animals selected for study

Organism	*FMI (kg/d)*	*Comments and references*
Deer/herbivorous mammal	6.3×10^{0}	Mass = 245 kg [ICRP, 2008] FMI for herbivores (kg d^{-1}) = $0.1995M^{0.628}$ from Nagy [2001]
Rat/burrowing mammal	8.4×10^{-2}	Mass = 0.314 kg [ICRP, 2008] FMI for Rodentia (kg d^{-1}) = $0.2296M^{0.864}$ from Nagy [2001]

27. Similarly, $\lambda_{r,a}$ the effective loss rate of radionuclide, *r*, from animal, *a*, can be derived using allometric relationships (table 8) and the animal masses specified above (table 7)

Table 8. Allometric equations used to derive effective loss rates (d^{-1}) for studied animals from the mass of animals (kg) [Brown et al.,2003]

Element	*Allometric equations*
Caesium	$\lambda_{r,a} = \frac{\ln 2}{18.36 M^{0.24}}$
Iodine	$\lambda_{r,a} = \frac{\ln 2}{16.7 M^{0.13}}$

28. The various parameters required in the model runs are thus specified in table 9.

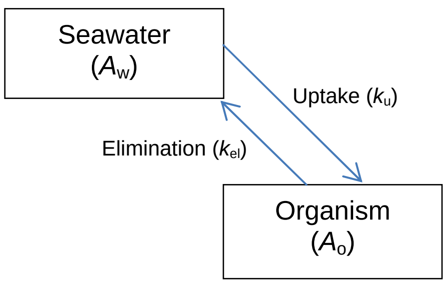
Table 9. Parameters used in dynamic model runs

Parameter	*Dependencies: fauna, flora, chemical element*	*Value*	*Units*	*Notes (references)*
x_i	Grass (deer) Tree (deer)	0.5 0.5	dimensionless dimensionless	
AE	Deer, Cs Deer, I	1 1 1	dimensionless dimensionless dimensionless	
FMI/M	Deer	2.6×10^{-2}	kg f.w. d^{-1} per kg	(FMI/M)
$\lambda_{r,a}$	Deer, Cs Deer, I	1.01×10^{-2} 2.03×10^{-2}	d^{-1} d^{-1}	Table 8; Mass = 245 kg Table 8; Mass = 245 kg

C. Dynamic approach to estimate activity concentrations in aquatic organisms

29. A relatively simple biokinetic model based on first-order exchange kinetics between the medium and the organism can be defined for the aquatic ecosystem using only the biological half-life of elimination and the CR. The simplest approach, which was used here, was to assume a two-compartment first-order kinetic model with constant rates of uptake and excretion from water (k_u and k_{el}), as shown in figure II.

Figure II. Simple two-compartment biokinetic model



30. The kinetic rates k_u and k_{el} can be simply written in terms of the biological half-life of elimination after uptake from water ($T_{B1/2}$), the length of time required for all combined physiological processes to cause the loss of half of the bioaccumulated radionuclide from an organism. Hence:

$$k_{el} = \frac{\ln 2}{T_{B1/2}} \text{ and } k_u = \frac{\ln 2}{T_{B1/2}} \frac{m}{V} CF \qquad (6)$$

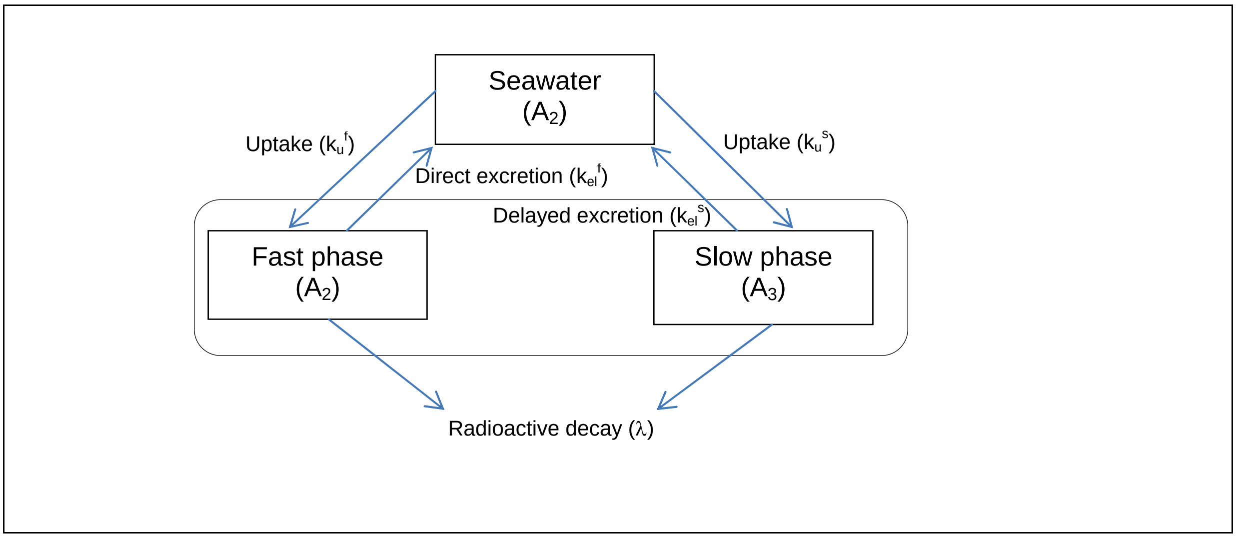
31. The above explanation is somewhat simplified because radionuclide turnover by marine organisms can be multiphasic, i.e. with an initial fast release followed by a slower, longer term release, with a certain point in time marking the transition from one phase to another. There are many instances of this, such as the time-dependence of technetium and plutonium in European lobsters [Olsen and Vives i Batlle, 2003; Swift, 1992]; ^{241}Am and ^{237}Pu in mussels [Guary and Fowler, 1981] or the typical multiphasic release curve representing the depuration of ^{131}I from *Littorina littorea* [Wilson et al., 2007; Wilson et al., 2005].

32. An adequate compromise is a three-compartment model which makes the working simplification that marine organisms absorb radionuclides mainly from the surrounding seawater, allowing one to relate their radioactivity concentration to their environment. Such a model includes direct exchange with seawater of specific activity A_w through both a fast compartment of activity A_f and a slow compartment of activity A_s:

$$\frac{dA_w}{dt} = \mathrm{k}_{\mathrm{el}}^{\mathrm{f}} A_f + k_{el}^{s} A_s - (\mathrm{k}_{\mathrm{u}}^{\mathrm{f}} + \mathrm{k}_{\mathrm{u}}^{\mathrm{s}}) A_w$$
$$\frac{dA_f}{dt} = \mathrm{k}_{\mathrm{u}}^{\mathrm{f}} A_w - \mathrm{k}_{\mathrm{el}}^{\mathrm{f}} A_f ; \; \frac{dA_s}{dt} = \mathrm{k}_{\mathrm{u}}^{\mathrm{s}} A_w - k_{el}^{s} A_s \qquad (7)$$

33. In this way, the model incorporates biphasic release, being sufficiently simple to be solved analytically without recourse to numerical computation. The model (figure III) requires relatively few parameters, rendering the approach practical for use in radiological assessments [Vives i Batlle et al., 2008].

Figure III. Basis of a dynamic model for the transfer of radionuclides to marine biota [Vives i Batlle et al., 2008]



34. There is the more advanced ECOMOD radioecological approach, which describes the dynamic processes of radionuclide migration in aquatic biota as radioactive tracers of stable analogous elements involved in growth and metabolism of organisms [Kryshev and Ryabov, 2000; Kryshev and Sazykina, 1986; Sazykina, 2000]. The ECOMOD modelling approach was

developed to calculate non-equilibrium, non-linear processes in aquatic ecosystems, combining radioecological and ecological equations.

35. This approach considers that, for a growing organism of total mass $M(t)$, the radionuclide activity concentration y (Bq kg^{-1}, fresh weight) in a given tissue changes according to the following equation [Kryshev and Ryabov, 2000]:

$$\frac{dy}{dt} = -\left(\lambda + \varepsilon_a \frac{B_r}{M} + \frac{1}{M}\frac{dM}{dt}\right) y + \frac{Q_1^A}{Q_0^A}\left(\frac{1}{M}\frac{dM}{dt} + \varepsilon_a \frac{B_r}{M}\right) X(t) \tag{8}$$

where:

λ is the radioactive decay constant (s^{-1});

ε_a is a proportionality coefficient between the rate of biological loss of a radionuclide from a target tissue and the general metabolic rate of the organism;

B_r is the metabolic rate (kg s^{-1});

$X(t)$ is the radionuclide activity concentration in food (Bq kg^{-1}, fresh weight) or in water (Bq L^{-1}) when bioassimilation occurs directly from water at time t;

Q_1^A, Q_0^A are the stable element concentrations in a given tissue and in food or water (mg kg^{-1}), respectively.

36. In the simplest case where $X(t) = X = \text{const}$; $y(0) = 0$, assuming that the mass of the organism is constant and neglecting radioactive decay, the solution represents the classical form of the radionuclide bioaccumulation curve:

$$y(t) = X \frac{Q_1^A}{Q_0^A}\left[1 - \exp\left(-\varepsilon_a \frac{B_r}{M} t\right)\right]$$

For the depuration case, in which $X(t) = 0$, $y(0) = Y_0$, the solution gives the simple monophasic depuration curve:

$$y(t) = y(0) e^{-\varepsilon_a \frac{B_r}{M} t}$$

III. DOSES TO BIOTA

37. Once activity concentrations in environmental media and biota had been collated and calculated, dose rates were derived through application of the ERICA Tool. The basic underlying equations (equations 9 and 10) use activity concentration data in order to derive absorbed dose rates due to internal (D_{int}) and external (D_{ext}) exposure (in units of µGy h^{-1}). The total absorbed dose rate is the sum of these components, through the application of dose conversion coefficients (DCCs).

$$\dot{D}^b_{\text{int}} = \sum_i C_i^b * \text{DCC}^b_{\text{int},i} \tag{9}$$

Where:

C_i^b is the average concentration of radionuclide i in the reference organism b (Bq kg^{-1} fresh weight);

$DCC^{b}_{int,i}$ is the radionuclide-specific dose conversion coefficient (DCC) for internal exposure defined as the ratio between the average activity concentration of radionuclide *i* in the organism *j* and the dose rate to the organism *b* (µGy h^{-1} per Bq kg^{-1} fresh weight).

$$\dot{D}^{b}_{ext} = \sum_{z} v_z \sum_{i} C^{ref}_{zi} * DCC^{b}_{ext,zi} \tag{10}$$

Where:

v_z is the occupancy factor, i.e. fraction of the time that the organism *b* spends at a specified position *z* in its habitat;
C_{zi}^{ref} is the average concentration of radionuclide *i* in the reference media at a given location *z* (Bq kg^{-1} fresh weight or dry weight (soil or sediment) or Bq L^{-1} (water));
$DCC^{j}_{ext,zi}$ is the dose conversion coefficient for external exposure defined as the ratio between the average activity concentration of radionuclide *i* in the reference media corresponding to the location *z* and the dose rate to organism *b* (µGy h^{-1} per Bq kg^{-1} fresh weight or Bq L^{-1}).

38. The DCCs used correspond to those reported in ICRP [2009]. Occupancy factors for organisms were selected such that they might characterize simplified yet realistic exposure geometry (table 10).

Table 10. Source target exposure geometry assumed for selected organisms

Organism	*Exposure geometry assumption*
Earthworm/soil invertebrate	In soil, volumetric source
Wildgrass/grasses, herbs and crops	On soil, volumetric source
Duck/bird	1.On soil, volumetric source 2. At water–air interface, aquatic source
Trout/pelagic fish	In water column, aquatic source
Rat/burrowing mammal	In soil, volumetric source
Pine tree/tree	On soil, volumetric source
Frog/amphibian	1.On soil, volumetric source 2. In water column, aquatic source
Flatfish/benthic fish	At water–sediment interface, aquatic source
Bee/above ground invertebrate	On soil, volumetric source
Deer/herbivorous mammal	On soil, volumetric source
Brown seaweed/macroalgae	At water–sediment interface, aquatic source
Crab/crustacean	At water–sediment interface, aquatic source

39. Weighted total dose rates (in µGy h^{-1}) are derived through the application of weighting factors (dimensionless) for alpha, low beta and high beta–gamma radiation (equations 11 and 12).

$$DCC_{int} = wf_{low\beta} \cdot DCC_{int,low\beta} + wf_{\beta+\gamma} \cdot DCC_{int,\beta+\gamma} + wf_{\alpha} \cdot DCC_{int,\alpha} \tag{11}$$

$$DCC_{ext} = wf_{low\beta} \cdot DCC_{ext,low\beta} + wf_{\beta+\gamma} \cdot DCC_{ext,\beta+\gamma} \tag{12}$$

Where:

wf = weighting factors for various components of radiation (low β, $\beta + \gamma$ and α);
DCC = dose conversion coefficients in µGyh^{-1} per Bq L^{-1} or Bq kg^{-1}.

40. Default radiation weighting factors of 10 for alpha radiation, 1 for low energy beta and 1 for (high energy) beta and gamma radiation were applied in this assessment in line with those applied in UNSCEAR [2008].

41. The dosimetric calculation underpinning the derivation of DCCs is dealt with in detail elsewhere [Ulanovsky and Pröhl, 2006; Ulanovsky et al., 2008]. Radioactive progeny were included in the DCCs for their parent if their half-lives were shorter than 10 days. DCCs for internal exposure were derived assuming a homogeneous distribution of the radionuclide in the organism; the error introduced by this assumption was, in view of the assessment goals, considered to be of minor significance [Gómez-Ros et al., 2008].

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