Ensuring robust radiological risk assessment for wildlife: insights from the International Atomic Energy Agency EMRAS and MODARIA programmes

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Abstract

In response to changing international recommendations and national requirements, a number of assessment approaches, and associated tools and models, have been developed over the last *circa* 20 years to assess radiological risk to wildlife. In this paper, we summarise international intercomparison exercises and scenario applications of available radiological assessment models for wildlife to aid future model users and those such as regulators who interpret assessments. Through our studies, we have assessed the fitness for purpose of various models and tools, identified the major sources of uncertainty and made recommendations on how the models and tools can best be applied to suit the purposes of an assessment. We conclude that the commonly used tiered or graded assessment tools are generally fit for purpose for conducting screening-level assessments of radiological impacts to wildlife. Radiological protection of the environment (or wildlife) is still a relatively new development within the overall system of radiation protection and environmental assessment approaches are continuing to develop. Given that some new/developing approaches differ considerably from the more established models/tools and there is an increasing international interest in developing approaches that support the effective regulation of multiple stressors (including radiation), we recommend the continuation of coordinated international programmes for model development, intercomparison and scenario testing.

1. Introduction

In response to changing international recommendations on the need to ensure that the environment (i.e. wildlife) is not adversely impacted by radioactivity (ICRP 2007; IAEA 2006), a number of assessment approaches, and associated tools and models, have been developed over the last *c*. 20 years (see IAEA 2021). Some of the tools and models implementing these approaches are now routinely being used in radiological impact assessments (e.g. Doering & Bollhöfer 2016; Doering et al. 2019; Nedveckaite et al. 2011; Vandenhove et al. 2013; Li et al. 2015; Posiva, 2014; Jaeschke et al. 2013; Allott & Copplestone 2009).

The International Atomic Energy Agency (IAEA) has coordinated four international collaboration programmes (EMRAS, EMRAS II, MODARIA I and MODARIA II; http://www-ns.iaea.org/projects/modaria/modaria2.asp) with the aim of improving Member States' capabilities for protection of the environment by comparing and validating models being used as part of the regulatory process of authorisation and compliance monitoring of authorised releases of radionuclides. To achieve this, the authors have conducted several intercomparison exercises and evaluations of fitness for purpose of models and approaches (IAEA 2012; Aramrum et al. 2019; Beaugelin-Seiller 2014, 2016; Beaugelin-Seiller et al. 2016; Beresford et al. 2008a; Beresford et al. 2009, 2010a; Johansen et al. 2012; Stark et al. 2015; Vives i Batlle et al. 2007a, 2011, 2016; Yankovich et al. 2010a). The IAEA programmes have also led to the establishment of databases on radionuclide transfer to wildlife (Copplestone et al. 2013; IAEA 2014) and biological half-lives for wildlife (Beresford et al. 2015a,b). In this paper, we discuss the lessons learnt from these model evaluations to assist assessors and regulators in conducting and interpreting assessments.

2. Model Selection

2.1 Overview of available modelling tools

Most of the assessment tools developed over the last 20 years have a common structure (IAEA 2012; Beresford et al. 2008b):

1) A set of hypothetical organisms that are typical of broad wildlife types (e.g. fish, mammal, riparian animal, terrestrial plant, etc., rather than representing specific species) are considered and these are typically represented by ellipsoidal geometries to allow for simplified dose calculations.
2) Simplified ecosystems are considered, typically 'terrestrial', 'freshwater' and 'marine'.
3) Simplified environmental exposure geometries are considered (e.g. for terrestrial ecosystems: in soil, on soil, in air) with occupancy factors being used to describe the amount of time an organism spends in each.
4) The activity concentration in organisms may be entered directly into the model if data are available, or estimated using an equilibrium concentration ratio ($CR_{wo\text{-}media}$) which relates the homogenous fresh mass (FM) activity concentration in the whole-organism to that in the relevant environmental medium (i.e. soil (dry mass, DM), water or air).
5) Equilibrium distribution coefficients (K_d) are used to relate water to dry mass sediment activity concentrations if either of these media concentrations is missing from the input data.

6) Unweighted absorbed dose rates are estimated from whole-organism activity concentrations (internal dose rate) and environmental media activity concentrations (external dose rate) via the application of pre-calculated dose conversion coefficients (DCC[1]) (e.g. μGy h^{-1} per Bq kg^{-1} fresh soil for the external dose rate to a terrestrial organism).

7) Weighted dose rates are estimated through the application of radiation weighting factors for α, low-energy β, high-energy β and γ radiations, reflecting the relative biological effectiveness of these different types of radiation.

Whilst the tools have a similar structure, the default parameter values assumed especially $CR_{wo\text{-}media}$ and K_d, can vary widely, impacting on model predictions (e.g. Beresford et al. 2008a,b, 2010a; Johansen et al. 2012; Yankovich et al. 2010a; Stark et al. 2015); this is discussed further below. Commonly used tools such as RESRAD-BIOTA and the ERICA Tool allow the user to enter their own $CR_{wo\text{-}media}$ and K_d values rather than using default parameters.

Table 1 provides an overview of freely available assessment tools and models that we can recommend based upon our experiences; the key advantages of each are highlighted. The models listed include: (i) those designed to conduct 'complete' assessments, often following a tiered (or graded) approach that starts with a simplistic conservative screening assessment and extends to more realistic assessments with increasing data and resource requirements; and (ii) those for specific elements of the assessments (e.g. dosimetry or for specific radionuclides). Worldwide, the two most commonly used assessment tools are the ERICA Tool (Brown et al. 2008, 2016) and RESRAD-BIOTA (USDoE 2004), both of which implement a tiered assessment approach and are freely available. However, they both lack some functionality which might be required for particular assessment purposes (e.g. assessment of dose from radon) for which the other models listed in Table 1 can be used.

Table 1. Freely available assessment tools and models which we can recommend for assessment of the radiological risk to wildlife based on our experience of application and intercomparison exercises.

Model	**Comment**	**Availability/Documentation**
ERICA Tool	Supports a tiered assessment approach. Allows additional radionuclides and organisms to be added; user can replace some of the default parameter values.	Software available from: http://www.erica-tool.eu/ Underlying reports: https://wiki.ceh.ac.uk/display/rpemain/ERICA+reports Special issue of Journal of Environmental Radioactivity: https://www.sciencedirect.com/journal/journal-of-environmental-radioactivity/vol/99/issue/9 Comprehensive integral Help file
RESRAD-BIOTA	Supports a tiered (or graded) assessment approach. Simple food chain models can be created and contaminated water as a source of intake by terrestrial animals can be modelled. User can replace	Software available from: https://resrad.evs.anl.gov/codes/resrad-biota/ User guide and other publications: https://resrad.evs.anl.gov/documents/ Special issue of Journal of Environmental Radioactivity: https://www.sciencedirect.com/journal/journal-of-environmental-radioactivity/vol/66/issue/1

[1]In some approaches these are referred to as dose conversion factors (DCFs) or dose coefficients (DCs).

	some of the default parameter values.	
Radon dose calculator*	MSExcel™ spreadsheet model which enables doses from ^{220}Rn, ^{222}Rn and short-lived progeny to be estimated (a functionality missing from current versions of the ERICA Tool and RESRAD-BIOTA)	Version (15/4/20): https://radioecology-exchange.org/content/radioecology-models Paper describing methodology: Vives i Batlle et al. (2017)
Ar – Kr – Xe dose calculator*	MSExcel™ spreadsheet model which enables estimation of doses from noble gases (important components of releases from nuclear reactors) to be estimated (a functionality missing from current versions of the ERICA Tool and RESRAD-BIOTA)	Download via: https://radioecology-exchange.org/content/radioecology-models Paper describing methodology: Vives i Batlle et al. (2015)
BiotaDC	Tool enabling dose coefficients to be calculated for the ICRP Reference Animals and Plants for a range of exposure geometries according to methodology used in ICRP Publication 136.	On-line tool: http://biotadc.icrp.org/ Methodology presented in: ICRP (2017)
EDEN2	Estimates DCC or exposure from Bq per unit mass or volume. Enables greater flexibility than default approaches in tiered assessment tools (e.g. heterogeneous soil/sediment contamination profiles).	Code can be requested via: http://www.irsn.fr/EN/Research/Scientific-tools/Computer-codes/Pages/The-EDEN-computer-code-Elementary-Dose-Evaluation-for-Natural-environment-2368.aspx Paper presenting the approach: Beaugelin-Seiller et al. (2006)
D-DAT	A dynamic tool for the assessment of radiation doses to marine biota.	Download via: https://radioecology-exchange.org/content/radioecology-models Papers describing methodology: Vives i Batlle et al. (2008) and Vives i Batlle (2016)

*The next release of the ERICA Tool will contain models for Rn-isotopes and other noble gases based on Vives i Batlle et al. (2015, 2017).

2.2 Outside of the typical assessment models

2.2.1 Noble gases

Noble gas radioisotopes (e.g. ^{85}Kr, ^{41}Ar) are significant components of the release from nuclear power plants (e.g. Copplestone et al. 2010). However, the commonly used assessment models do not currently consider immersion dose from contaminated air masses (although air immersion dose coefficients are available in ICRP (2017)).

In human dose assessment, the contribution from noble gases is small (Smith 2013) because they are not taken up significantly and immersion dose represents the most significant exposure pathway. Because of the large percentage contribution of noble gases (*circa* 85%;

Copplestone et al. 2010) to the overall release, some regulators recommend determination of their contribution to wildlife dose (e.g. Environment Agency, 2012). As immersion exposure is not considered within the commonly used assessment tools, such as RESRAD-BIOTA or the ERICA Tool, an additional model has been developed (Vives i Batlle et al. 2015; see Table 1). To estimate total exposures in an assessment, the outputs of this bespoke model for noble gases needs to be combined with outputs from the more generic assessment tools. The spreadsheet model accompanying Vives i Batlle et al. (2015), which is an update of the 'R&D128' methodology initially developed for the England and Wales Environment Agency (Copplestone et al. 2001), should not be used for any radionuclides other than noble gases as parameter values for the other radionuclides are out of date.

A similar methodology as that used for assessing noble gas exposure has also been developed for assessing dose rates from ^{222}Rn and ^{220}Rn exposure (Vives i Batlle et al. 2012, 2017; see Table 1). The natural background dose rates from ^{222}Rn to burrowing animals can be considerable (10s of μGy h^{-1}) (MacDonald and Laverock 1998; Beresford et al. 2012), dominating total exposure from natural background sources (Beresford et al. 2008c; 2012). When estimating radon dose rates, consideration may need to be given to burrow air exchange rates (see Doering & Bollhöfer 2016).

2.2.2 Acute releases

The available assessment tools assume equilibrium between radionuclide activity concentrations in organisms and environmental media. However, there are scenarios whereby equilibrium cannot be assumed. For example, after a short-term pulse release of radionuclides into the aquatic environment, released radionuclides may be rapidly cleared from the area under assessment because of dispersion. However, the activity concentration in most organisms for many radionuclides along the dispersion path will increase, and subsequently decrease, gradually with time. This is because the biological half-life of radionuclides in organisms is often in the range of 10s of days (Beresford et al. 2015a). Consequently, $CR_{wo\text{-}water}$ values are unlikely to be representative in the days to months after an acute release. Soon after the release, CRs are likely to overestimate the activity concentrations in organisms. Conversely, after peak water activity concentrations have declined, CRs will underestimate organism activity concentrations (e.g. Takata et al. 2019).

If the timeframe of interest is long (e.g. years or decades of planned authorised discharges, involving continuous releases or gradual changes in discharge concentrations), then the $CR_{wo\text{-}media}$ approach is likely to be sufficient. However, if unplanned release scenarios involving abrupt changes in discharge concentrations are being modelled, then the $CR_{wo\text{-}media}$ approach may be inadequate and dynamic models of radionuclide transfer to biota might be a better assessment tool. This is especially true for organisms that respond slowly to a change in ambient radionuclide activity concentration (Vives i Batlle et al. 2008). Whilst a number of dynamic models have been developed (see Vives i Batlle et al. 2016), most of these are not freely available; an exception is the D-DAT model (see Table 1).

As an example comparison of equilibrium and dynamic models, Figure 1 compares predicted ^{137}Cs and ^{131}I activity concentrations in benthic fish close to the Fukushima NPP using the conventional $CR_{wo\text{-}water}$ model (ERICA Tool) and the D-DAT dynamic model (from Vives i Batlle et al. 2016). It can be seen that the equilibrium model predicts activity concentrations to rise and subsequently decline more rapidly than the dynamic model. This is a consequence of the latter incorporating biological half-lives, which result in a reduced rate of uptake and loss compared to the instantaneous equilibrium of the $CR_{wo\text{-}water}$ model. The difference is less pronounced for ^{131}I where the rate of loss is dominated by the physical decay of the isotope ($T_{1/2} \approx 8$ d). However, by the end of the simulation period, the two model types give more

comparable results for both radionuclides with much of the residual variability likely being due to differences in the $CR_{wo\text{-}water}$ values used (the dynamic D-DAT model uses $CR_{wo\text{-}water}$ to predict equilibrium radionuclide activity concentrations).

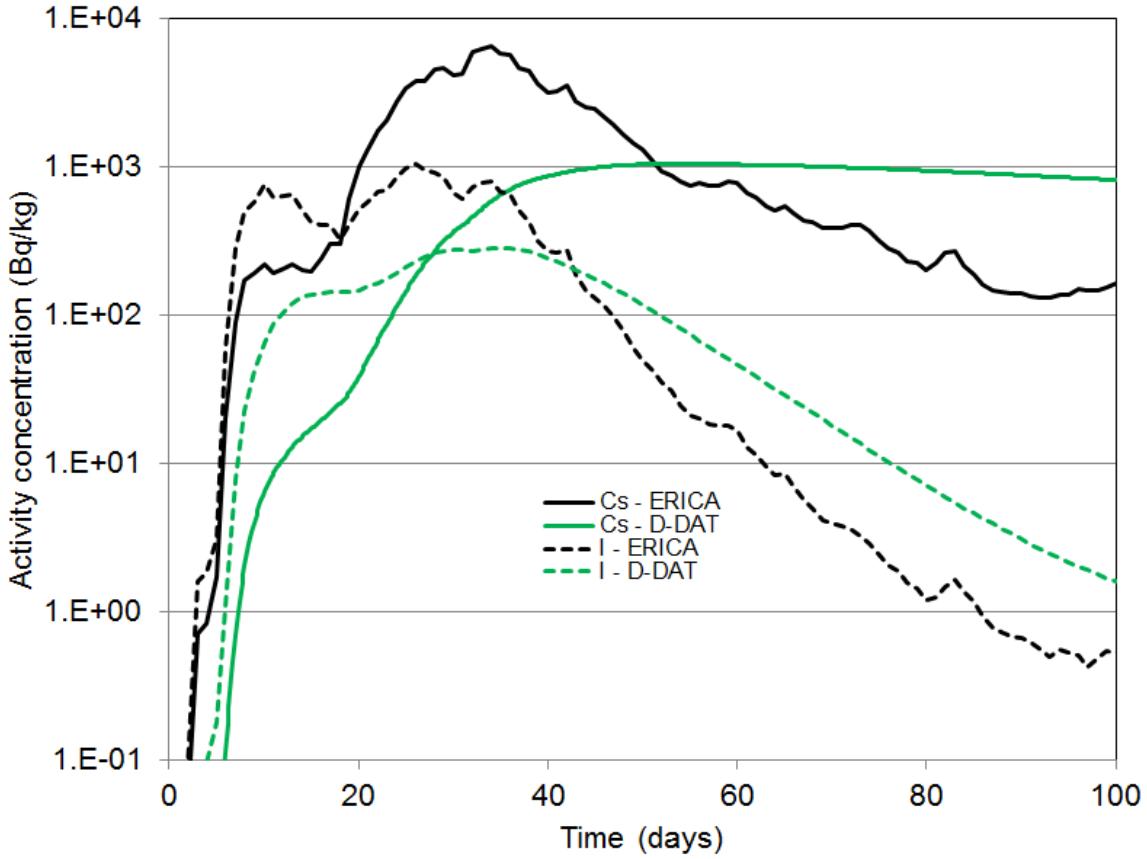

Figure 1. Predicted changes in ^{131}I and ^{137}Cs activity concentrations in the whole body (fresh mass) of a benthic fish (adapted from Vives i Batlle et al. 2016). The predictions used modelled water activity concentrations with time '0' being the 11th March 2011; the assessment site was assumed to be located 30 m north of the Fukushima Daiichi drainage channels (37° 25' 51" N, 141° 2' 3" E).

The Fukushima Daiichi accident highlighted the desire to be able to estimate the exposure of wildlife as a consequence of accidental releases (e.g. see assessments by Garnier-Laplace et al. 2011; Vives i Batlle et al. 2014; Strand et al. 2014). Furthermore, the ICRP (2014) has proposed a system of environmental protection encompassing the assessment of emergency exposure situations, for which dynamic models would presumably be required. However, the dynamic models initially applied to the marine ecosystem following the Fukushima accident did not predict well the rate of decline in radiocaesium activity concentrations in some fish species resulting in considerable under prediction of radionuclide activity concentrations 2-3 years after the accident (Johansen et al. 2015). Furthermore, our comparison of models for marine ecosystems showed orders of magnitude variation between outputs of different dynamic models for some radionuclides and organism types (Vives i Batlle et al. 2016). There is a need to improve the available dynamic models and, where possible, validate them. Our published database of biological half-life data for wildlife (in marine, freshwater and terrestrial ecosystems) may help in improving model parameterisation (Beresford et al. 2015a;b). There are dynamic models for aquatic systems, which can be relatively easily adapted to conduct wildlife exposure assessments (as demonstrated in Vives i Batlle et al. 2016); however, for terrestrial ecosystems, model development is required.

3. Radionuclide transfer

3.1 Concentration ratios

For radiological environmental assessments, a default $CR_{wo\text{-}media}$ value is needed in most models for each radionuclide and organism combination. These parameters are used when sufficient site-specific data (measured activity concentrations in organisms or site specific $CR_{wo\text{-}media}$ values) are not available. They are also used to calculate media screening levels in RESRAD-BIOTA and the ERICA Tool for application in the initial screening tier; media screening values are activity concentrations in environmental media that would result in a predicted dose rate equal to the model's screening dose rate benchmark for a given default organism (see Beresford et al. 2010b).

Concentration ratios are derived empirically by measuring the radionuclide activity concentration in the whole organism relative to the activity concentration in the appropriate medium (generally soil or water). In the case of small animals, this can be relatively easily achieved by homogenising the whole sampled animal or, for certain radionuclides, through live-monitoring (e.g. Bondarkov et al. 2011); however, for larger animals this is not practicable. Two methods are commonly used to address this. The first method is to measure the activity concentration in one or more tissue types and then use conversion factors for tissue specific to whole organism activity concentrations (Yankovich et al. 2010b). However, this relies upon conversion factors having already been determined for animals of a similar type and there is potentially considerable uncertainty associated with such conversion factors (currently values are based on few data for some types of organisms and a number of radionuclides). A second approach is to subsample the major tissue types of an organism and to analyse these separately. With a measurement of the contribution of each tissue type to the whole-organism mass, it is then possible to estimate the whole body activity concentration (e.g. see approaches used by Barnett et al. (2014, 2020) to estimate whole body activity concentrations for a medium sized deer and Yankovich (2009) for fish). This second approach also provides additional data from which conversion factors, as used in the first method described, can be estimated.

Our evaluations demonstrated that the transfer components of the models contributed most to the observed variation in estimated dose rates (IAEA 2012; Beresford et al. 2008a,b, 2010a; Johansen et al. 2012; Stark et al. 2015; Vives i Batlle et al. 2011; Yankovich et al. 2010a); $CR_{wo\text{-}media}$ values for a given organism-radionuclide may vary over four-orders of magnitude (IAEA 2014). This large variation is commonly observed in the CR datasets for all ecosystems for most radionuclides and organism types. Where comparatively little variation is observed, this is often because there are few reported $CR_{wo\text{-}media}$ values. For instance, in Beresford et al. (2008a), we report that predictions by a number of models for the transfer of ^{99}Tc to wildlife were similar. However, the similarity in predicted ^{99}Tc activity concentrations was because of a lack of Tc data, which meant that the different models obtained their default $CR_{wo\text{-}media}$ values from the same limited sources or by using the same assumptions.

Given the high uncertainty of the transfer component of models, much work has been conducted to collate $CR_{wo\text{-}media}$ values internationally, through the 'Wildlife Transfer Database' (http://www.wildlifetransferdatabase.org/ (WTD)); Copplestone et al. 2013), which now forms the basis of some models' databases (e.g. Brown et al. 2016) and international handbooks/compilations (ICRP 2009, IAEA 2014). The WTD was initially built upon the empirical datasets used in the original ERICA Tool transfer databases (Hosseini et al. 2008; Beresford et al. 2008d) but has subsequently been extensively expanded (Copplestone et al. 2013) with data being continually added and subjected to quality assurance (e.g. Brown et al. 2016; Hirth et al. 2017).

Within the WTD and IAEA (2014), the $CR_{wo\text{-}media}$ values are categorised into wildlife groups such as mammals, fish, birds with the option to categorise into subgroups where appropriate, for example:

Bird: bird – carnivorous, bird – herbivorous, bird – omnivorous

Fish: fish – benthic feeding, fish – forage, fish - piscivorous

Mammal: mammal – carnivorous, mammal – herbivorous, mammal – marsupial, mammal - omnivorous, mammal – *Rangifer* spp.

These additional sub-categories were provided in case, for instance, feeding habits lead to differences in CRs. IAEA (2014) presents recommended $CR_{wo\text{-}media}$ for these wildlife group subcategories where there are sufficient data. However, analyses to date, have shown that we do not have the statistical justification for separating the major categories into these subcategories (Wood et al. 2013); Wood et al. also highlighted problems with the approach used to estimate summarised geometric mean and standard deviation values in IAEA (2014).

Data are not available for many radionuclide-organism combinations. For instance, only *circa* 50% of the required *circa* 1500 $CR_{wo\text{-}media}$ values for the ERICA Tool (v1.3) were based on measured data (Brown et al. 2016). To address this, a number of extrapolation approaches have been developed (IAEA 2014; Brown et al. 2013). For example, data for a 'similar reference organism' may be used (e.g. using data for mammals if data for birds are unavailable) or $CR_{wo\text{-}media}$ values from a similar ecosystem may be used (e.g. comparatively highly saline estuarine environments may provide surrogate data for marine systems). In some models, 'extrapolated' default $CR_{wo\text{-}media}$ values can be highly conservative and contribute to the variation observed in model outputs (Beresford et al. 2008a).

3.2 Allometric models

At higher assessment levels for some organisms, RESRAD-BIOTA has the option to use a 'kinetic-allometric' approach alternative to the $CR_{wo\text{-}media}$ model (Higley et al. 2003). Allometric scaling relates biological parameters to body mass (M) with the dependency of a biological variable (Y) on mass (M) typically characterised as: $Y=aM^b$ where a and b are constants. Whilst the US DoE RESRAD-BIOTA model (USDoE 2004) uses allometric relationships to define, for instance, dry matter food intake rate or water ingestion rate, they also present allometric relationships describing the biological half-lives of 16 radionuclides in animals. We have proposed an approach whereby allometric relationships of biological half-life can be derived when data are not available to parameterise such relationships (Beresford & Vives i Batlle 2013; Beresford et al. 2016a). For homeostatic organisms, we proposed that biological half-life ($T_{1/2b}$) can be estimated as:

$$T_{1/2b} = \frac{ln2}{a_I f_1} CR_{org-diet} M^{0.25}$$

where: a_I is the multiplicand in the allometric model describing the dry matter intake rate of food; f_1 is the fractional gastrointestinal; $CR_{org\text{-}diet}$ is the ratio between the activity concentrations in the whole-organism (fresh mass) and the diet (dry matter) absorption coefficient; and *0.25* is the assumed allometric scaling constant (b value) in the allometric model describing radionuclide biological half-life. The approach gave reasonable predictions when compared to blind test $T_{1/2b}$ datasets for a number of radionuclides (Beresford et al. 2016).

Subsequently, we proposed that the equation for homeostatic organisms could be made applicable to (poikilothermic) reptiles by taking into account the lower metabolic rate of

these organisms and replacing 0.25 with values more applicable for reptiles (see Beresford & Wood, 2014).

3.3 External deposition on vegetation surfaces

A fundamental difference has been identified between human and environmental radiological assessments with regard to the way in which activity concentrations in vegetation are derived when assessing aerial discharges (Copplestone et al. 2010; Beresford et al. in preparation). For human radiological assessments, the activity concentrations in vegetation (e.g. crops, leafy vegetables and pasture for domestic animals) are derived from root uptake (typically using a *CR* approach) and a 'direct' component, which accounts for the initial deposition onto, and interception by, vegetation surfaces. It could be argued that if the collated $CR_{wo\text{-}media}$ databases were sufficiently robust, they would include data for such release scenarios. However, the $CR_{wo\text{-}media}$ databases are typically dominated by 'existing' (e.g. much of the Cs, Sr, and Pu data originates from post-Chernobyl studies) rather than 'planned' exposure situations (Copplestone et al. 2013). During the preparation of IAEA (2014), we actively removed data for vegetation on a site receiving deposition from an operating plant because these values were extreme high outliers compared to the rest of the data distribution. Furthermore, $CR_{wo\text{-}media}$ values are increasingly being derived from stable isotope studies using approaches, such as ICP-MS (Beresford 2010), and these data do not include any aerial deposition from release sources.

A modelling assessment presented in Brown et al. (2003) demonstrated that ignoring the direct deposition pathway may significantly underestimate transfer and hence exposure. Figure 2 demonstrates that for radionuclides with low soil-to-plant *CR* values, modelling transfer to plants by assuming only root uptake leads to underestimation of the activity concentrations in herbivorous mammals compared to when deposition to vegetation surfaces is included in the model parameterisation. There are few field studies with defined radionuclide source terms and release patterns which allow a definitive conclusion to be drawn about the importance of these pathways. One study that potentially demonstrates the effect of direct deposition is that described by Copplestone et al. (1999), conducted in a coniferous woodland close to the Sellafield site (UK). For this site, it was suggested that radionuclide activity concentrations in vegetation were largely the consequence of continued deposition to vegetation surfaces (rather than root uptake).

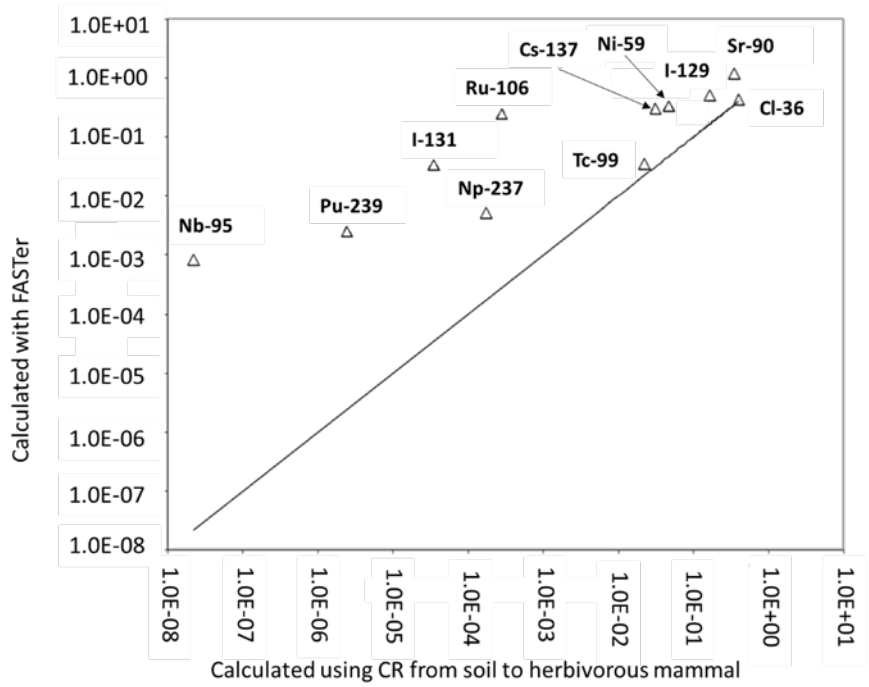

Figure 2. Activity concentrations (Bq kg^{-1} FM) in herbivorous mammals 50 years after a continuous constant deposition (1 Bq m^{-2} y^{-1}) onto a semi-natural ecosystem, calculated with the computer programme FASTer (Avila et al. 2004), which includes a direct external contamination pathway, with root uptake modelled using $CR_{wo\text{-}soil}$ (reproduced from Brown et al. 2003).

To be consistent with the assessment method used for humans, direct deposition would need to be included in radiological environmental impact assessments and existing models could be adapted to do this. The IAEA are doing this in the development of screening models (updating the SRS-19 models (IAEA, 2001) for both human and environmental radiological impact assessments (IAEA in-press, in-preparation)). These models involve the use of mass interception factors (i.e. the fraction of deposited activity intercepted by vegetation per unit mass) and accumulation factors, which account for the continuous deposition of a contaminant onto vegetation and also the continuous loss of the contaminant through 'weathering' processes and physical decay. There is a need to consider which organisms would be affected by including direct deposition on vegetation. For instance, herbivorous mammals would be predicted to have higher activity concentrations than using a $CR_{wo\text{-}soil}$ value alone and, therefore, we could also expect carnivorous mammals to have higher predicted activity concentrations, whereas $CR_{wo\text{-}soil}$ are likely sufficient for earthworms.

3.4 Distribution coefficients

Model default distribution coefficients (K_d) for aquatic systems are often taken from IAEA (2004) for marine values and IAEA (2010) for freshwater values. Until recently, there has been considerably less consideration of improving K_d datasets compared to the efforts devoted to compiling $CR_{wo\text{-}media}$ values. However, an updated compilation and assessment of freshwater K_ds is now available (Boyer et al. 2018; Tomczak et al. 2019) and work is ongoing to improve the marine database (Kelleher et al. *submitted*). As for $CR_{wo\text{-}media}$ values, it may be necessary to extrapolate K_d values when there are no data for a specific ecosystem. In the

past, there has been some application of marine K_d values in freshwater ecosystems (e.g. Hosseini et al. 2008); this should be avoided, as there is no evidence to support this approach (the chemical speciation and colloidal association of many radionuclides in seawater is different in saline solution compared with freshwater).

4. Dosimetry - what matters and what does not matter?

In two intercomparison exercises (Vives i Batlle et al. 2007a, 2011), we compared estimates by a number of models including the commonly used ERICA Tool and RESRAD-BIOTA. The comparisons considered unweighted internal and external dose rates under assumptions of unit organism and environmental media activity concentrations, respectively. Generally, internal dose rates compared well between the models, with 70% of predictions being within ±20% (Vives i Batlle et al. 2011). External dose rate estimates were more variable, though 90% fell within an order of magnitude of one another. Differences could generally be readily explained by: the number of progeny (or daughter products) included in the parent nuclide DCC; source-target geometry differences; rounding errors; database used to source radionuclide energy and yield information; assumptions on skin/fur shielding; and assumed media densities. However, for some models and radionuclides, there were instances where these factors combined to cause systematic differences in the estimated dose rate (Vives i Batlle et al. 2011). Variation in external dose estimates tended to be greatest for radionuclides such as ^{3}H, ^{14}C and α-emitters (Vives i Batlle et al. 2007a). However, whilst variation was considerable, the DCC values tended to be low, for instance, the 'on soil' external ^{3}H DCC for the ICRP Reference Rat geometry (ICRP 2008) ranged from 0 to $5x10^{-6}$ μGy h^{-1} per Bq kg^{-1} soil. Whilst variation in estimates was large, it is highly unlikely this will add much to the overall variation in total estimated dose rates to any meaningful degree (because the DCCs are relatively low and hence the external dose rates will be low whichever model is used).

Subsequent application of the models to case study scenarios (Beresford et al. 2010a; Goulet et al. *submitted*; Johansen et al. 2012; Stark et al. 2015; Yankovich et al. 2010a) demonstrated that typically, variation between the dosimetry components of models contributed little to the overall variation in total dose estimates. The variation was dominated by differences in predicted biota activity concentrations (which determined the consequent variation in total dose rates estimated).

4.1 Dosimetry and organism size

The available approaches tend, for pragmatic reasons, to consider a limited number of organisms (i.e. geometries) (e.g. Brown et al. 2008; ICRP 2008; USDoE 2004). The geometries need to have defined sizes (dimensions and mass) within the assessment tools and some organisms requiring assessment will have sizes outside the ranges included in the tools. This leads to questions as to whether the available organisms can be used to represent species being assessed. Figure 3, compares weighted absorbed dose rates to a selection of the default marine organism geometries within the ERICA Tool (v1.3; Brown et al. 2016), assuming, for each of the four selected radionuclides, either 1 Bq L^{-1} in water or 1 Bq kg^{-1} fresh mass in the organism to estimate internal and external absorbed weighted dose rates, respectively. The masses of the geometries range over eight orders of magnitude. However, with the exception of ^{90}Sr, the external dose varies across the geometries by less than one order of magnitude. The external dose predictions for ^{90}Sr differ by a factor of approximately 100. This is because, ^{90}Sr, and its short-lived progeny ^{90}Y, are pure beta emitters, emissions are consequently not very penetrating, and as the organism size increases, the relatively small absorbed energy is diluted across a larger mass. However, variation in the external ^{90}Sr absorbed dose across all organisms, in the range >10 g to nearly 200 kg, varies by about one

order of magnitude only. Similarly, there is comparatively little variation in the predicted internal dose rates across the organisms. The largest difference is observed for the high-energy gamma-emitter ^{60}Co where comparatively more emissions will escape from the smallest organisms. Consequently, selecting the closest predefined geometry to represent an organism within an assessment is likely to be acceptable (this conclusion has subsequently been supported by (Charrasse et al. 2019)).

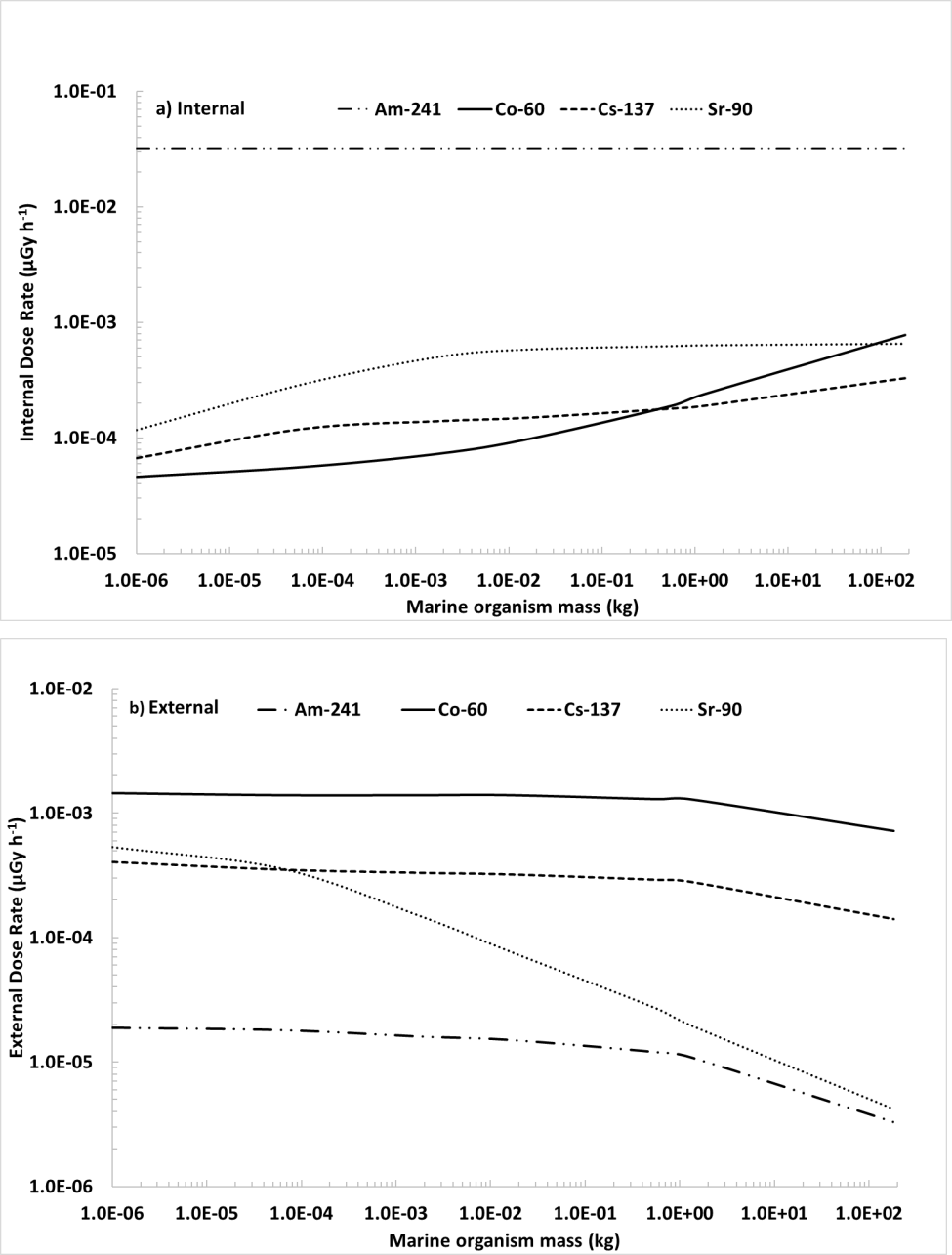

Figure 3. Change in absorbed weighted doses of ^{241}Am, ^{60}Co, ^{137}Cs and ^{90}Sr with increasing mass of default marine organism within the ERICA Tool assuming (a) 1 Bq kg^{-1} (FM) in the organism for internal exposure or (b) 1 Bq L^{-1} in water for external exposure. Organisms were modelled as spending 100% of their time in the water column and assuming the ERICA Tool default radiation weighting factors (10 for alpha, 3 for low energy beta and 1 for other beta and gammas).

4.2 Characterisation of uncertainty in internal dosimetry

As already discussed, in the available assessment tools, a set of simplified dosimetric phantoms, typically ellipsoids, are used to represent different types of wildlife. This approach makes it possible to quickly and easily estimate radiation dose rate to a homogeneous geometry.

In contrast, voxel modeling utilises advanced imaging technologies to generate realistic and detailed dosimetric phantoms to calculate radiation dose to individual organs via Monte Carlo modelling (Higley et al. 2015). This approach is used in human dosimetry. A number of voxel models have recently been developed for different species of wildlife (Caffrey et al. 2013, 2015a,b; Ruedig et al. 2014; Kinase 2008; Dogdas et al. 2007; Stabin 2006). The voxel models are not suggested for regulatory purposes, however, they provide a mechanism by which we can assess if the ellipsoid assumptions are fit for purpose.

The dosimetric systems used within assessment tools for wildlife rely upon three major assumptions (e.g. ICRP 2008):

1. That any organism can be represented by a simplified dosimetric phantom;
2. That for dosimetric purposes, four-component human tissue (composed of H, C, N and O) adequately mimics real tissue; and
3. That assuming a homogeneous distribution of radionuclides within an organism's body is not a large source of uncertainty.

Each of these assumptions introduces some uncertainty into the dosimetric calculation. We have used voxel models for terrestrial and aquatic organisms to quantify these uncertainties and the results of this are discussed below.

4.2.1 Geometry assumption: simple vs. voxel

Differences between voxel phantom dose rates for individual organs (D_{voxel}) and dose rates for whole-organisms represented by ellipsoids ($D_{ellipsoid}$) have been calculated for the following radionuclides: ^{14}C, ^{36}Cl, ^{60}Co, ^{90}Sr, ^{131}I, ^{134}Cs, ^{137}Cs, and ^{210}Po in three aquatic organisms (flatfish, trout, and crab), and for $^{134,\ 137}Cs$, $^{238,239}Pu$, ^{239}Pu, $^{239,240}Pu$, ^{90}Sr, and ^{241}Am for a terrestrial mammal (rabbit). Estimated dose rates were calculated separately for each organ considered in the voxel models and the ratio of dose rates ($D_{voxel}/D_{ellipsoid}$), termed the K-value, was calculated. In most cases, the ellipsoidal models provided conservative estimates of organ dose rates (i.e. $K<1$).

For the rabbit, dose rates for alpha-emitting radionuclides are identical for each method because full energy absorption in source tissue is assumed in both cases. For beta-emitting ^{90}Sr, K-values were in the range 0.8-0.9. For highly penetrating photon radiations (e.g., ^{134}Cs and ^{137}Cs), K-values ranged from 0.42 to 0.64, indicating the ellipsoidal phantoms over-predict organ dose rates for the rabbit (Caffrey et al. 2015a).

Similarly, the dose rates calculated via the two methods generally agreed within a factor of two to three for the three aquatic organisms investigated. K-values ranged from 0.36 (^{60}Co) to 1.08 (^{14}C) (Ruedig et al. 2015), again suggesting that the ellipsoid assumption is generally conservative, and hence, fit for purpose in regulatory assessments. Potentially the evaluation presented here overestimates the difference between doses that would be calculated by ellipsoid and voxel models as we have not included consideration of organ-to-organ (or 'crossfire') contributions (Caffrey et al. 2015a).

4.2.2 Tissue composition and density assumption

In most geometric and voxel models created to date, human tissue composition and density values have been used in lieu of biologically accurate values for non-human biota. This has

raised questions regarding variable tissue composition and density effects on the fraction of radioactive emission energy absorbed within tissues. These assumptions were tested using the same rabbit model as discussed above. Results indicate that the variation in composition between mammalian tissue types made little difference to the fraction of emissions absorbed. Furthermore, comparison of more variable tissue densities (e.g. heart, liver, bone etc.) showed little difference in fractions of energy absorbed (Caffrey et al. 2015a; Ruedig et al. 2015).

4.2.3 Homogeneity assumption

Typically, the dosimetric approaches assume that radionuclides are homogeneously distributed throughout the organism of interest. For some radionuclides (e.g., ^{137}Cs, ^{3}H) this assumption approximates reality (Yankovich et al. 2010b). However, in some cases, radionuclides may be heterogeneously partitioned. Examples are radioiodine accumulating in the thyroid of animals and bone seeking radionuclides such as ^{90}Sr, ^{226,228}Ra and ^{239}Pu. This heterogeneous distribution may lead to doses to specific organs which are considerably higher than those to homogenous ellipsoids. For instance, the dose rate to the thyroid (thyroid mass 1.5g) of a roe deer (body mass, 18.2 kg), due to ^{131}I uptake, may be a factor of 10,000 higher than the whole organism dose rate based upon a homogeneous distribution of ^{131}I (Farhana & Ganie 2010). Iodine-131 is probably the most extreme example. If the main radionuclide in an assessment is likely to accumulate in a specific organ, then the assessor may wish to consider how these findings will influence their interpretation of the results. Furthermore, if conducting dose-effect studies with these type of radionuclides, then voxel phantoms should potentially be used to estimate doses and consideration will need to be given to how the dose-response relationship is interpreted.

4.2.4 Conclusions for regulation - application of simple ellipsoid models

Sources of uncertainty in the standard dosimetric methodologies used in wildlife assessments are summarised in Table 2. Based upon evaluations summarised above, the continued use of simplified geometric models as the basis for the majority of regulatory dose assessments seems appropriate. The uncertainty arising due to the use of a simplified geometry (relative to a voxel geometry) is relatively minor when compared with other sources of uncertainty within wildlife dosimetry (e.g. prediction of radionuclide transfer to organisms). An exception may be the few cases where most of the organism's activity is located in one relatively small organ (e.g. the radioiodine example given above). Voxel models may be particularly useful in scenarios where accurate (as opposed to conservative) estimates of dose rates are necessary, such as in aiding the interpretation of wildlife dose-effects studies.

Table 2. Sources and quantification of uncertainty within the standard assumption of a homogenous geometry.

Assumption of ellipsoid dosimetric model	**Alternative approach**	**Maximum potential uncertainty introduced by using assumption versus alternative**
Simplified phantom geometry	Voxel phantom geometry	< 3x Assumption is conservative
Four-part tissue composition	Complex tissue composition	< 1.5x Assumption is not conservative

Homogeneous distribution of radionuclides, whole-organism dose rate	Heterogeneous distribution of radionuclides, organ specific dose rate	< 10000x (for accumulating organ) Assumption is not conservative

4.3 Uncertainties in external exposure estimates

4.3.1 Heterogeneity in radionuclide soil/sediment profile distributions

Homogeneous distribution of radionuclides in soils and sediments is typically assumed in assessment models and tools. However, in reality, significant variation in radionuclide activity concentrations with depth may be observed for soils and sediments. We investigated the effect of this vertical heterogeneity on organism exposure by determining the dose rates to various soil- and sediment-dwelling organisms externally exposed to a set of radionuclides (Beaugelin-Seiller 2014). The soil or sediment compartment was described with an increasing level of complexity, from the usual single uniform compartment through to a multilayer representation (using the EDEN2 model, Beaugelin-Seiller et al. 2006). Resultant total external dose rates varied by up to three orders of magnitude compared to a homogenous distribution assumption. However, this depended upon the exposure situation, i.e., the organism, its location and the radionuclides considered. The assumption of a homogeneous distribution within soil or sediment was not consistently conservative. However, the homogenous assumption is suitable for the purpose of a conservative screening-level risk assessment (e.g. fit for regulatory purpose), as long as it is combined with the maximal activity concentration reported in the soil or sediment profile being considered. The use of radionuclide activity profiles in soils or sediments is a potential refinement for higher tier assessment or may be useful when interpreting field effects studies.

4.3.2 Soil and sediment water

Soil and sediment activity concentrations are required inputs of the various models to estimate external exposure rates and the activity concentrations in organisms. To determine organism activity concentrations, the required input for the application of the $CR_{wo\text{-}media}$ is soil dry mass activity concentrations (IAEA 2014), whereas for the determination of dose rate the DCC values are often applied to fresh mass soil activity concentrations (Brown et al. 2008). External DCC values were estimated to vary by a factor of about 1.5 over realistic ranges of soil moisture (dry to saturated) (Beaugelin-Seiller 2016). Consequently, this adds little uncertainty to the estimation of exposure in comparison with uncertainties in transfer parameters.

However, while the external DCC values should be applied to soil fresh mass activity concentrations, assessors typically input dry mass soil activity concentrations. Generally, the impact of this on the external dose rate will be relatively minor and it is a conservative assumption appropriate for screening level assessments. However, for ecosystems with high soil moisture contents, the uncertainty added to the assessment by inputting soil activity concentrations on a dry mass basis could be considerable. For instance, in an assessment of wetlands using the ERICA Tool (which allows the soil dry mass percentage to be added), the estimated external ^{137}Cs dose rates using the reported soil dry matter content (of 10%) were an order of magnitude lower than estimates using the default soil dry matter content (Stark et al. 2015); in effect, converting to a fresh mass soil activity concentration resulted in an order of magnitude decrease in the soil activity concentration.

4.4 Radioactive decay/progeny

Radioactive decay can generate new, 'daughter', radionuclides. Short-lived progeny are most often included in the DCC of their parent (Vives i Batlle et al. 2007a); secular equilibrium between the parent radionuclide and the decay products (daughters) is assumed. Depending upon the model, different threshold (cut-off) values are used. For example, in the current version of the ERICA Tool (Brown et al. 2016), DCC values include progeny with half-lives of ≤10 days. For RESRAD-BIOTA, the cut-off is 100 years for the first two assessment levels and can be reduced to 180 days at the highest assessment level. A potential limitation of this approach is that progeny are assumed to have the same behaviour regarding transfer as their parents; the conservatism of this assumption is unclear. Users need to be aware of which progeny are included in the parent radionuclide DCC for the approach that they are using.

Users also need to take care not to consider separately progeny already integrated in the parent DCCs, as this would overestimate the radiological risk (see Vives i Batlle et al. 2007a); this is possible to do in some of the models (including RESRAD-BIOTA and the ERICA Tool). Theoretically, the ideal would be to consider each progeny separately, which is the approach used in the recent methodology described by ICRP (2017). However, this requires either measurements of all of the progeny and/or $CR_{wo\text{-}media}$ values for them. Some of the progeny are elements for which $CR_{wo\text{-}media}$ would otherwise not be required and for which there are few or no data available. The implication of the uncertainties this adds to the estimation of dose has yet to be evaluated.

5. Estimating exposure in contaminated environments: animal – environment interactions

Currently, in assessments, dose rates to animals may be predicted using point of capture media activity concentrations or media activity concentrations averaged across either an assessment site or an assumed home range for a species. Some field studies may even simply relate observations of purported radiation effects to ambient dose rate measurements using handheld detectors (see discussion in Beaugelin-Seiller et al. 2020). For the assessment of non-radiological contaminants, the application of animal movement models to estimate exposure have been proposed (Cairns & Niederlehner 1996; Forbes & Calow 2012).

To our knowledge, prior to the MODARIA programmes, only one study had attempted to evaluate how well assessment tools and standard assumptions predicted external exposure in the field; Beresford et al. (2008e) attached thermoluminescent dosimeters (TLDs) to mice and voles at three sites in the Chernobyl Exclusion Zone comparing results to predictions from the ERICA Tool. The Chernobyl small mammal study suggested that assuming the average ^{137}Cs activity concentration in soil across the assumed home range gave adequate external dose rate predictions (compared to results from the TLDs attached to the animals). To further test the applicability of the approaches used in assessments, we have conducted two studies: one on reindeer (*Rangifer tarandus tarandus*) in Norway and the other on Eurasian elk (*Alces alces*) in Sweden.

In Aramrum et al. (2019), we describe a study conducted in an upland area of Norway which received comparatively high deposition of ^{137}Cs from the 1986 Chernobyl accident. Free-ranging reindeer were fitted with collars onto which dosimeters and GPS units were mounted. The dosimeters were recovered from 12 animals approximately 11 months after being fitted. Live-monitoring data were available for the animals, as were spatial datasets of soil ^{137}Cs and natural radionuclide activity concentrations. External dose rates were estimated using the ERICA Tool from: (i) soil activity concentrations averaged over the whole ranging area of the herd; and (ii) soil activity concentrations for areas that the reindeer were known to have visited based on the GPS-tracking data. The average ^{137}Cs dose rate to the animals estimated for areas they were known to be have visited was approximately twice that estimated for the entire range

area. Collared reindeer mostly occupied areas with the highest ^{137}Cs soil concentrations, as these were correlated with their favoured habitats. Whilst estimating external exposure, assuming that the animals ranged over the whole area available to them (i.e. as may be assumed for an assessment) would underestimate external doses by a factor of about two, internal doses calculated from live-monitoring data were approximately an order of magnitude higher than the external doses. The external dose (^{137}Cs and natural background exposure) estimated using the GPS-tracking data were in reasonable agreement with dose measurements from the collar mounted dosimeters.

In our second study, data were made available for an area of Sweden in which long-term studies of the behaviour of Eurasian elk had been conducted (Singh et al. 2012; Allen et al. 2016). The data included: GPS-tracking locations for moose 2006-2007; habitat and topography spatial datasets; and a ^{137}Cs deposition surface from post-Chernobyl aerial surveys. Habitats were defined as 'unsuitable', 'suitable' and 'preferred' for elk; areas with a slope >10° were also defined as unsuitable. These data were used to compare the following exposure modelling approaches (see IAEA 2021):

i) *Conventional approach* – Estimations of exposure were made using a spatially adjusted mean soil activity concentration for: the entire assessment area; 'suitable' locations within the assessment area; and 'preferred' locations.
ii) *Mass-balanced food-web approach* – A mass-balanced food web model (Christensen et al. 2014) was applied in a spatial context.
iii) *Individual-based movement (IBM) approach* – a simplistic stochastic-Lagrangian approach was implemented in Goldsim Dynamic Monte Carlo Simulation Software (Goldsim™; https://www.goldsim.com/web/home/) to model spatially-variable exposures to individual elk based on random-walk movement biased by known habitat and terrain slope preferences (defined as for the mass-balance food-web approach).

From a comparison of the three exposure modelling approaches, we concluded that for screening level assessments, the conventional approach, taking into account habitat/terrain preferences is sufficient for external exposure estimation; external exposure estimates were broadly comparable for the three approaches. The conventional approach was the simplest to apply and could be adapted to readily take into account habitat/terrain preferences. However, the IBM approach has the potential to estimate variability within a population and, therefore, may be useful for higher tier assessments and to inform the interpretations of field studies on radiation effects.

In addition to the estimation of external exposure, the conventional and IBM approaches were both used to estimate internal exposure. The conventional approach used an elk-specific $CR_{wo\text{-}soil}$ value extracted from the Wildlife Transfer Database (Copplestone et al. 2013) to estimate the ^{137}Cs activity concentration of elk. The IBM approach used a simplified intake-retention model in which the internal dose over time related to the soil-to-vegetation-to-elk uptake of ^{137}Cs along a 'foraging pathway' across the variably-contaminated landscape. Internal dose was estimated to be 20-30 times higher than the external dose. Predictions from the IBM approach demonstrated that internal dose rate will respond to spatial changes in soil contamination more slowly than external dose rate, which responds instantaneously. The slower response of internal dose rate is the consequence of organism uptake and loss rates of radionuclides being influenced by the radionuclide's biological half-life for the organism under consideration.

Encouragingly, there has been reasonable agreement between dosimeter estimates and model predictions for all studies now available where model estimates have been compared to the

results of dosimeters attached to animals (reindeer - Aramrum et al. 2019; snakes - Gerke et al. 2020; small mammals - Beresford et al. 2008e).

We can conclude that the 'conventional approach' of averaging soil activity concentrations over an appropriate area is suitable for screening-level assessments. However, the elk and reindeer studies, and papers published on wolves (Hinton et al. 2019) and wild hogs (Gaines et al. 2005) demonstrate the potential influence of habitat utilisation on the exposure of animals. Therefore, beyond screening-level assessments, the application of organism-specific knowledge of habitat and terrain preferences should be considered when estimating exposure. It may be necessary to use different spatial extents (and hence, spatially-averaged media activity concentrations) for different organisms within an assessment area. The importance of spatial behaviour relative to the spatial resolution and pattern of contaminant data may vary between different organisms. For instance, as noted above, averaging ^{137}Cs activity concentrations in soil across an assumed home range gave adequate predictions of external exposure (compared to the results of TLDs attached to mice and vole species) at three sites in the Chernobyl Exclusion Zone (Beresford et al. 2008e). The difference in conclusions reached between the small mammal study and the findings of the elk and reindeer studies discussed above may be due to the limited spatial variation in soil contamination over smaller ranging areas; whilst variable, soil ^{137}Cs activity concentrations demonstrated no significant spatial trend at any of the three Chernobyl sites (Beresford et al. 2008e).

The focus of the studies discussed here was the estimation of external exposure, however, habitat and terrain utilisation will also impact on internal exposure. Depending upon how animals utilise their habitats, the areas contributing most to internal dose may not be the same as those contributing to external dose (e.g. animals such as badgers and foxes may feed in areas relatively distant to their burrow). Hinton et al. (2019) suggest their results of external dose rate measurements and internal ^{137}Cs activity concentrations for Chernobyl wolves demonstrate differences in total ranging and foraging areas. In the reindeer and elk examples discussed above, internal dose dominated total exposure. However, the relative contributions of external and internal exposure to total dose will depend upon ecosystem characteristics determining radionuclide transfer, animal species and radionuclide.

Consideration of the importance of animal-environment interaction on animal exposure has, to date, been restricted to terrestrial ecosystems. Attention should also be given to aquatic environments (e.g. spatial heterogeneity in sediment activity concentrations would impact on dose rates to mobile benthic organisms).

6. Making the assessment tools work for you

The available models have limitations with respect to, for instance, the default ecosystems, organisms considered, exposure geometries, default radionuclides and input requirements. This section explores how some of these issues can be managed such that more confidence can be placed in the assessment outcomes.

6.1 Conducting an initial screening-level assessment for a missing radionuclide

The ERICA Tool and RESRAD-BIOTA use tiered assessment approaches but include a limited set of default radionuclides. Whilst radionuclides can be added for higher-level assessments in the ERICA Tool, they cannot be added for the initial screening level (Tier 1). Tier 1 in the ERICA Tool simply compares input media concentrations to pre-defined values, termed Environmental Media Concentration Limits (EMCLs), representing the soil, water or sediment activity concentrations giving rise to the screening dose rate to the most exposed reference organism. Whilst it is not possible to add radionuclides at Tier 1, it is possible to use the ERICA Tool to estimate EMCL values for additional radionuclides. This requires

$CR_{wo\text{-}media}$ and, in the case of aquatic ecosystems, K_d values, with associated probability distribution functions for the new radionuclide. If the ERICA Tool is run in probabilistic mode (Tier 3) with inputs of 1 Bq per unit media, an EMCL value can be calculated by dividing the screening dose rate by the highest estimated 95th percentile dose rate for any default organism.

This approach could be used to derive screening media activity concentrations for application in approaches other than the ERICA Tool. For example, it has been applied to derive soil and water 'radiological guideline values' for a uranium mining site in Australia (Doering & Bollhöfer 2016; Doering et al. 2019).

6.2 Dealing with organisms in different environments

Assessment tools, such as RESRAD-BIOTA and the ERICA Tool, consider generic ecosystem types (e.g. 'terrestrial', 'freshwater', 'marine', 'riparian'). This means that assessors have to make decisions on how to cope with assessments for different environments (e.g. estuaries or saltmarshes), migratory species (e.g. birds, marine mammals) and species that spend time in different environments (e.g. frogs), as examples.

For some ecosystem types not included within the models, it may be possible to simply re-parameterise the $CR_{wo\text{-}media}$ values. For instance, if an estuarine system was being assessed, then the marine ecosystem could be re-parameterised using appropriate values (e.g. data for species living in estuaries can be found in the Wildlife Transfer Database (Copplestone et al. 2013)).

For organisms which may inhabit more than one ecosystem type (e.g. a seal, duck or frog), model runs for each ecosystem should be performed (e.g. freshwater and terrestrial ecosystems for a frog). Outputs of model runs can be combined by taking into consideration the time spent in each environment and where the organism feeds. For example, in the case of the seal, all of the internal exposure would originate from feeding in the marine environment, but there would be an external exposure contribution when the seal is resting in the terrestrial environment. However, if the user were considering an organism such as an otter (*Lutra lutra*), they may assume that the otter feeds in both the aquatic and terrestrial environment and fractionate the internal dose according to the diet assumed. RESRAD-BIOTA allows dose rates to riparian animals (e.g. an otter) to be estimated, though transfer parameters and simple diets would need to be defined.

6.3 Exposure geometries

There are limitations in most assessment models on the organism and exposure geometries which can be considered. For instance, the ERICA Tool allows birds in the terrestrial ecosystem to be 'on soil' or 'in the air', whereas mammals can be 'on soil' or 'in soil'. However, assessments might be required for birds that burrow (e.g. puffins (*Fratercula arctica*)) or flying mammals (i.e. bats). While it may appear that the ERICA Tool could not model these types of organisms, the bat could be modelled as a bird (organism density assumptions in the model are the same for both mammals and birds) if an external dose rate in air is required. Similarly, a dose rate for a puffin in burrows could be estimated by modelling it as a burrowing mammal. Johansen et al. (2012) presents our model intercomparison exercise for an area containing contaminated waste trenches and gives examples of how non-standard organism exposure geometries such as, a lizard (*Varanus varius*) that spends some time in trees and also trees growing directly over buried wastes (some models, e.g. the ERICA Tool, only have default organisms representative of above ground plant parts).

6.4 Representing organisms of interest

Specific protected or keystone species may require consideration within an assessment; these species may be defined within legislation (e.g. Copplestone et al. 2003) or by stakeholders.

RESRAD-BIOTA and the ERICA Tool make highly conservative assumptions in their initial screening tier, and hence, it is likely that results would be applicable to protected species. This assumes that existing $CR_{wo\text{-}media}$ databases or assumed *CR* values would encompass those applicable for the protected species and we note that RESRAD-BIOTA does not cover all organism types (e.g. aquatic plants).

At higher assessment tiers, it is likely that default geometries would be applicable to the protected species of interest, especially taking into account the discussion above about the comparative lack of effect of size on the dose calculation. However, both tools give the opportunity to create new organisms, if desired. Consideration would need to be given as to whether the default $CR_{wo\text{-}media}$ values were appropriate for the species under consideration. If the species of interest is protected, it is unlikely that whole-organism activity concentration or transfer data could be obtained directly (as for most radionuclides, this would involve killing the organism). It is possible that $CR_{wo\text{-}media}$ values could be derived from analysis of similar unprotected organisms (Beresford et al. 2016a). If no suitable data are available, IAEA (2014) provides summarised values for organism sub-groups (e.g. herbivorous mammal rather than the generic mammal category that may be included as a default within the model). However, the quantity and representativeness of the data is such that it is not recommended that the sub-group values are used in preference to the generic values (Wood et al. 2013). RESRAD-BIOTA gives the option of creating simple food chain models that could be used as an alternative approach to assess protected species.

For some radionuclides, it is possible to use taxonomic models of transfer to predict activity concentrations in families or genera, which may encompass the species of interest (Beresford et al. 2013, 2016a; Beresford and Willey 2019). Currently, such models have been demonstrated for Cs in freshwater fish and Pb in terrestrial organisms, with Søvik et al. (2017) suggesting it will be possible to establish models for a wider range of radionuclides (Cs, Sr, U and Se). However, we note that whilst the predictive power of these initial models was good for freshwater fish and terrestrial organisms, it was poor for Cs in marine organisms (Brown et al. 2019).

Some organisms have very different life stages, which may involve living in different environments for periods of time (e.g. tadpole-frog, insects). For exposure assessment in such cases, different geometries could be created in tools, such as RESRAD-BIOTA or ERICA, to represent the various life-stages. However, currently, there are few $CR_{wo\text{-}media}$ data for non-adult life-stages.

6.5 Carbon-14 and tritium – model inputs

In some approaches specific activity models are used to predict ^{3}H and ^{14}C activity concentrations in terrestrial organisms (e.g. Beresford et al. 2008a; IAEA 2014). These approaches relate organism activity concentrations to input air concentrations, not soil activity concentrations as is the case for most radionuclides (Beresford et al. 2008a); this is similar to approaches for estimating ^{3}H and ^{14}C transfer to human foodstuffs (IAEA 2010, 2014). However, in some instances, assessors will have soil but not air concentrations available (see example in Stark et al. 2015). Guidance is needed for assessors on how to estimate an air activity concentration from a soil concentration.

The underlying assumption of a simple specific activity model is that the ratio of the concentrations of radioactive and stable isotopes is the same in all environmental

compartments. Therefore, in the case of ^{14}C, if the soil activity concentration is known, ^{14}C air concentrations can be approximated for input into assessments as:

$$^{14}C_{air} = \frac{^{14}C_{soil} \times 0.2}{Soil\ Carbon}$$

where $^{14}C_{soil}$ is the activity concentration of ^{14}C in soil (Bq kg^{-1} DM); $^{14}C_{air}$ is the activity concentration of ^{14}C in air (Bq m^{-3}); *Soil Carbon* is the concentration of stable carbon in soil (g kg^{-1} DM); and *0.2* is the typical stable carbon content of air (g m^{-3}) (IAEA 2014).

For ^{3}H, the assumption can be made that the activity concentration in air moisture (C_{AM}, Bq m^{-3}) will be equal to that in soil water (C_{SW}, Bq m^{-3}). The concentration of ^{3}H in air (C_{air}, Bq m^{-3}) can then be estimated as:

$$C_{air} = C_{AM} \times H_A$$

where H_A is the absolute humidity (kg m^{-3}). Typical H_A values for different climates are presented in IAEA (2019), which also presents a methodology for estimating H_A from relative humidity, if known. This approach will give an approximation of the ^{3}H concentration in air which can be input into models. However, it should be noted that root uptake may be the dominant source of ^{3}H in plants at sites with contaminated soil and groundwater (Evenden et al. 1998; Yim & Caron 2006).

7. Coping with missing data – suggested best practice approach

In the course of the various model intercomparisons described above, we encountered the need to deal with missing data (e.g. for a specific progeny within a decay chain). This was most notable for the scenarios that considered Canadian lakes impacted by U mining and processing industries (Goulet et al. *submitted*; Beaugelin-Seiller et al. 2016). Consequently, we defined some 'best practice guidance' to aid future assessors, as summarised below:

- If only water activity concentrations are available for an aquatic assessment, to calculate sediment activity concentrations, a best estimate K_d value should be used, for instance, a mean value calculated for similar sites or a value selected from an up-to-date review (e.g. Boyer et al. 2018; Tomczak et al. 2019; Kelleher et al. *submitted*).
- If only sediment activity concentrations are available for an aquatic assessment, to calculate water activity concentrations, a best estimate K_d value derived as above should be used.
- If media activity concentrations are lacking for radionuclides within a decay chain for an aquatic assessment, secular equilibrium in the same media should be assumed, preferably with the closest member in the decay chain for which data are available. For the ^{238}U and ^{232}Th decay chains, radon and thoron gas will escape; ^{210}Po and ^{210}Pb activity concentrations in media can be assumed to be 80% of the ^{226}Ra activity concentrations (UNSCEAR 2000). However, some knowledge of the assessment site will be required when determining which radionuclides should be assumed to be in

secular equilibrium as some processes may impact on this (e.g. see case of phosphate-fertiliser plant considered in Vandenhove et al. (2015)).

- If sufficient whole-organism activity concentrations are available for the organisms being assessed, then these should be used in the assessment; consideration will need to be given to the amount of data available versus the quantity and provenance of $CR_{wo\text{-}media}$ values from compilations (i.e. the Wildlife Transfer Database (http://www.wildlifetransferdatabase.org/ (Copplestone et al. 2013)).
- If whole-organism activity concentrations for a given species are not available for an assessment site, data for a similar species at the same site should be used.
- If no measured data for a given species are available at a site, the whole-organism activity concentration should be predicted using $CR_{wo\text{-}media}$ values, preferably from measurements made previously at the assessment site, or at similar, site(s); the assessor would need to determine if sufficient measurements are available to justify using a site specific rather than a generic $CR_{wo\text{-}media}$ value (see Sheppard 2005; Wood et al. 2009).
- If no relevant $CR_{wo\text{-}media}$ values are available, appropriate values should be obtained from the Wildlife Transfer Database (i.e. for the organism-radionuclide combination being considered).
- If neither whole-organism activity concentrations nor relevant $CR_{wo\text{-}media}$ values are available for a specific radionuclide-organism combination, then extrapolation approaches, such as those described by Beresford et al. (2016a) and Brown et al. (2013, 2016), should be used.

8. Conclusions

Since *circa* 2000, a number of models and tools to assess the radiological risk to wildlife have been developed and are now being used in assessments worldwide. Through the studies discussed above, we have assessed the fitness for purpose of a number of these models and tools, and highlighted areas of highest uncertainty. Based on our evaluations, we are able to recommend a number of tools/models for undertaking radiological assessments for wildlife which are freely available to users (Table 1). Although not discussed above, a number of our evaluation exercises (e.g. Johansen et al. 2015; Goulet et al. *submitted*; Stark et al. 2015) demonstrated the potential contribution of 'assessor uncertainty' (Wood et al. 2009) to the total uncertainty in model predictions. Choices made by assessors (e.g. regarding parameter selection and model application) can significantly affect model outcomes and must be appropriately justified. It is therefore recommended that assessors: (i) clearly document and justify all decisions made within an assessment, including the provenance of all data/parameter values used; (ii) follow the best practice guidance provided here; and (iii) consider undertaking one of the available training courses on the use of dose assessment models in environmental radiation protection to ensure that they are undertaking assessments based on the most up-to-date knowledge.

Some of these models and tools can be used to undertake tiered or graded assessments, beginning with conservative screening tiers and progressing, if required, to more refined assessments requiring increasing amounts of data. Others allow more functionality for specific aspects of assessments (e.g. dosimetry, transfer or dynamic modelling) or fill gaps in the capabilities of the tiered/graded assessment tools (e.g. enabling the assessment of noble gases). Although there are many simplifications in the available assessment models, and also large uncertainties with respect to radionuclide transfer to organisms, we conclude that the commonly used tiered/graded assessment tools (e.g. RESRAD-BIOTA (USDoE 2004) and

the ERICA Tool (Brown et al. 2016)) are generally fit for purpose for conducting screening-level assessments. As discussed above (see section 6), the available models have greater utility than may first appear.

However, radiological protection of the environment (or wildlife) is still a relatively new development and assessment approaches are continuing to develop. The ICRP has refined its approach to estimating dose conversion coefficients (which the ICRP now refers to as 'dose coefficients' (*DC*)). The new approach includes assessment-specific consideration of the contribution of radioactive progeny to the *DC* of parent radionuclides (i.e. progeny may not be included in the parent *DC* but modelled separately) and an extended set of exposure geometries (ICRP 2017). With respect to the Wildlife Transfer Database (Copplestone et al. 2013), $CR_{wo\text{-}media}$ values continue to be added, comprehensive compilations of post-Fukushima accident studies are becoming available (IAEA 2020) and regional $CR_{wo\text{-}media}$ compilations are being produced (Hirth et al. 2017).

The forthcoming IAEA approach for the assessment of the impact of radioactive discharges to the environment, replacing (IAEA 2001), adopts aspects of the revised ICRP dosimetry methodology for estimating dose coefficients. It also includes exposure pathways not present in the existing screening-level models (e.g. land irrigation, application of sewage sludge to land) and adopts approaches used for human food chain modelling including interception of aerially released radionuclides by vegetation surfaces (see section 3.3).

Furthermore, to address data gaps and uncertainties in the $CR_{wo\text{-}media}$ approach, novel approaches to estimating the transfer of radionuclides to wildlife are being developed and tested (Beresford et al. 2013, 2016a; Beresford & Wiley 2019; Beresford & Vives i Batlle 2013; Brown et al. 2019; Søvik et al. 2017; Vives i Batlle et al. 2007b). Some new/developing assessment approaches described above differ considerably from the more established models/tools that we have tested. There is also increasing international interest in developing assessment approaches and associated models/tools that support the effective regulation of multiple stressors (e.g. radiological and non-radiological contaminants (Beaumelle et al. 2017; Vandenhove et al. 2018). Therefore, as environmental assessment approaches continue to develop, we recommend the continuation of coordinated international programmes for model development, intercomparison and scenario testing, as we have described in this paper. Such programmes would also contribute to the provision of training in this developing area of radiation protection, given that many countries have only recently started to adapt to revised international recommendations to ensure that wildlife are protected from releases of radioactivity into the environment.

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