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Background exposure rates of terrestrial wildlife in England and Wales

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Abstract

It has been suggested that, when assessing radiation impacts on non-human biota, estimated dose rates due to anthropogenically released radionuclides should be put in context by comparison to dose rates from natural background radiation. In order to make these comparisons, we need data on the activity concentrations of naturally occurring radionuclides in environmental media and organisms of interest. This paper presents the results of a study to determine the exposure of terrestrial organisms in England and Wales to naturally occurring radionuclides, specifically ^{40}K, ^{238}U series and ^{232}Th series radionuclides. Whole body activity concentrations for the reference animals and plants (RAPs) as proposed by the ICRP have been collated from literature review, data archives and a targeted sampling campaign. Data specifically for the proposed RAPs is sparse. Soil activity concentrations have been derived from an extensive geochemical survey of the UK. Unweighted and weighted absorbed dose rates were estimated using the ERICA Tool. Mean total weighted whole-body absorbed dose rated estimated for the selected terrestrial organisms was in the range 6.9×10^{-2} - 6.1×10^{-1} μGy h^{-1}.

Keywords: Terrestrial biota; ERICA; ^{40}K; ^{238}U series; ^{232}Th series; Reference animals and plants.

1. Introduction

There are now a number of models/software tools available for use in environmental impact assessments to estimate the exposure of non-human biota (i.e. wildlife) to radioactivity (Beresford et al., in-press). It has been suggested that, when assessing radiation impacts on non-human biota, estimated dose rates due to anthropogenically released radionuclides should be put in context by comparison to dose rates from natural background radiation normally experienced by animals and plants (Pentreath, 1999; 2002). This concept is being considered further by the International Commission on Radiation Protection (ICRP, 2005). The use of background exposure rates to put results of assessments into context is also recommended in the interpretation of results from the ERICA Tool (Beresford et al., 2007a; Brown et al., this issue).

In order to make these comparisons, we need comprehensive data on the activity concentrations of naturally occurring radionuclides in environmental media (i.e. soil, water and sediments) and organisms of interest. Brown et al. (2004) presented a review of such data for European marine organisms concluding that, whilst information for a few organisms (namely marine mammals and seabirds) were lacking, the ranges in dose rates from natural radionuclides to most marine organisms could be readily estimated. However, the same authors reported that there were insufficient data to conduct assessments for many freshwater organisms of interest. There is also an acknowledged lack of data for naturally occurring radionuclides concentrations in terrestrial wildlife (e.g. Jones et al., 2003). In their assessment of the exposure of terrestrial organisms to naturally occurring radionuclides Gómez-Ros et al. (2004) used no data for wild terrestrial species, instead their assessment was based upon reported activity concentrations in agricultural crops and livestock.

This paper presents the results of a study to determine the exposure of terrestrial organisms in England and Wales to naturally occurring radionuclides, specifically ^{40}K, ^{238}U series and ^{232}Th series (see Table 1). This suite of radionuclides includes the principal contributors to background radiation exposure. Initial results from this work (Beresford et al., 2007b) were used to estimate the background dose rates to terrestrial organisms for inclusion within the ERICA Tool (available from www.project.facilia.se/erica/download.html).

2. Materials and Methods

2.1 Proposed ICRP reference animal and plants

The ICRP has recently suggested a set of reference animals and plants (RAPs) as points of reference for assessing radiation effects in non-human species (ICRP, 2005). With respect to terrestrial ecosystems, the proposed list of reference animals and plants comprises: Reference Deer; Reference Rat; Reference Duck; Reference Frog; Reference Bee; Reference Earthworm; Reference Pine Tree; Reference Wild Grass. An objective of the work described in this paper was to categorise available data for naturally occurring radionuclides in England and Wales on the basis of these RAPs.

2.2 Review of available data

The first stage of the study was to determine the availability of data on natural radionuclide activity concentrations in terrestrial RAPs. Whilst the focus of this review was England and Wales, available data for all of the United Kingdom were collated as it soon became evident that there would be few data if the review was restricted to England and Wales alone. Available sources of data for natural radionuclide activity concentrations in biota included: (i) 'grey' literature; (ii) refereed journal papers; (iii) in-house databases.

The RIFE (Radioactivity in Food and the Environment) reports are published annually by UK agencies and report radionuclide activity concentrations throughout the United Kingdom (predominantly close to sites discharging radionuclides to the environment). These reports provided activity concentrations for some naturally occurring radionuclides in relevant biota such as game and plant species. With the exception of grass, sampled biota are the edible components of species eaten by man. Data were compiled from RIFE reports presenting the results of sampling conducted between 1995 and 2004 (MAFF, 1996; MAFF and SEPA, 1997; 1998; 1999; FSA and SEPA, 2000; 2001; 2002; EA et al., 2003; 2004; 2005).

To review the refereed literature, 88 variations of keyword searches were performed using *Web of Knowledge* (http://wok.mimas.ac.uk/) and in house radioecological bibliographical databases. Whilst this identified a large number of references (>7,000 references) to review, only 5 yielded usable data (more details of the literature review can be found in Beresford et al., 2007b).

The Centre for Ecology & Hydrology (CEH) has previously conducted many analyses of relevant biota types for anthropogenic radionuclides. These samples were obtained from studies conducted into the deposition and behaviour of radiocaesium deposited following the 1986 Chernobyl accident, and assessments of the transfer of radioactivity discharged from the Sellafield reprocessing plant (Horrill et al., 1988; Horrill and Mudge, 1990). Whilst previously not always reported, the gamma analysis of these samples yielded results for ^{40}K and these data were collated from existing databases, or the original analytical reports.

In the preliminary stages of the work it was recognised that much of the available data were not specific to the proposed ICRP RAPs. Therefore, available data were attributed to appropriate RAPs as follows:

- Reference Duck – any species of wild bird
- Reference Frog – any species of amphibian
- Reference Bee – any species of flying insect
- Reference Earthworm – any species of earthworm
- Reference Pine Tree – any species of tree
- Reference Wild Grass – any species of wild grass or herb, and any data recorded as 'grass'

Because the ICRP recommend two mammalian RAPs, data were collated for 'any wild mammalian species' and separately for deer (data for any species of wild deer) and rat. Although this categorisation has been used to attribute data to RAPs the availability of data specifically for the RAP as defined by ICRP is also briefly discussed.

2.2.1 <u>Availability of data</u>

Table 2 summarises the availability of data by RAP and radionuclide (data sources are also identified). The most abundant data were available for ^{40}K; the majority of which was sourced from available CEH databases; no data were identified for ^{228}Th. Many of the data for mammals and birds were from Scotland reflecting the focus of post Chernobyl studies; little data originated from Wales. No data were available for: Frog, Bee or Earthworm nor specifically for the proposed Reference Rat. All of the data allocated to Pine Tree were predominantly for edible fruits of deciduous trees.

2.3 Sampling and analyses

On the basis of the comparative lack of available data, a targeted sampling and measurement campaign to provide data for Duck, Bee, Earthworm, Pine Tree and Frog was conducted. Where possible, existing sample archives available to CEH were utilised in combination with targeted sampling. Samples obtained comprised:

- A mixed sample (1-2 g dry weight) of flying insects (predominantly moths) from each of the eight terrestrial Environmental Change Network sites in England and Wales (see www.ecn.ac.uk)
- Five samples of lodgepole pine (*Pinus contorta*) trunk
- 20 samples of grey heron liver (*Ardea cinerea*) obtained from CEH's Predatory Birds Monitoring Scheme sample archive (see http://pbms.ceh.ac.uk/)
- Six earthworm samples
- Four rabbits (*Oryctolagus cuniculus*)
- One toad (*Bufo* spp.) obtained from CEH sample archives
- Five mallards (*Anas platyrhynchos*)

Because of the paucity of existing data from Wales all rabbit samples, four mallards, three samples of lodgepole pine and two earthworm samples were obtained from the principality; nine of the heron liver samples selected for analyses had also been collected from Wales.

2.3.1 Sample preparation and analyses

Samples of lodgepole pine trunk were first washed before being ashed at 450^{o}C.

Earthworm were kept on wet tissue paper for *circa* 2 days to allow evacuation of their guts. Samples were freeze dried and homogenised prior to analyses. Insect samples were supplied dry, these were homogenised prior to analyses.

Mallards were plucked and their gastrointestinal tract removed; the liver was removed for separate analyses. Rabbits were skinned and their claws and gastrointestinal tract removed. Both rabbit and mallard carcasses were washed prior to ashing at 450^{o}C. Herons and mallard liver samples were freeze dried before homogenisation.

Depending upon sample size, ashed and dried samples were accurately weighed into either 25 ml petri dishes or 150 ml plastic containers. These were sealed and stored for 25 days to allow secular equilibrium to be established. They were subsequently analysed by gamma spectrometry on hyper-pure germanium detectors to determine gamma-emitting radionuclide activity concentrations; two day count times were used for all samples and resultant spectra were analysed using Canberra Genie-VMS. The detectors were calibrated for the energy range 59.5 to 1836 keV using certified reference solutions added to materials of various densities (samples in this study were analysed using a dried ground vegetation efficiency calibration). Due to the small sample size, the toad sample was homogenised and analysed fresh (using an aqueous solution efficiency calibration); the sample was then ashed prior to determination of total U and Th concentrations.

All samples were finely powdered prior to analyses to determine total U and Th activity concentrations. Where available, 0.5 g of sample was digested with 10 ml of concentrated HNO_3 and 2 drops of HF in a microwave oven, dried down and taken up so that the final digest was in 10 ml of 10% HNO_3. Samples were analysed using ICP-MS (detection limits for samples analysed were typically of the order 10^{-2} mg kg^{-1} FW for Th and 10^{-3} mg kg^{-1} FW for U). Specific activities of 4.07 Bq ^{232}Th mg^{-1} Th and 12.21 Bq ^{238}U mg^{-1} U respectively were used to estimate radionuclide activity concentrations from total U and Th determinations. An assumption of secular equilibrium between ^{238}U and ^{234}U would appear to be valid for biological samples (see data presented by Eisenbud and Gessel, 1997) enabling ^{234}U activity concentrations to be estimated. Equilibrium between the parents and other members of the two decay chains could not be assumed for biological samples because of differing environmental and biological behaviours.

2.4 Derivation of soil activity concentrations

For over 35 years the British Geological Survey (BGS) have been conducting geochemical surveys of the UK. This work (referred to as the G-BASE project (Johnson and Breward, 2004; Johnson, 2005)) includes determinations of K, U and Th concentrations in soils. The estimates of ^{40}K, ^{238}U and ^{232}Th series radionuclides in soils in England and Wales have been based upon this extensive data source supplemented with data from other BGS projects. The derivations of soil activity concentrations are briefly described here; full details can be found in Beresford et al. (2007b) (which also describes derivation of stream sediment and water activity concentrations using data from the G-BASE project).

The G-BASE sampling procedures are detailed in Johnson et al. (2005). Soil samples are collected at an optimum density of one every 2 km^2; each sample represents a composite of material taken from five holes distributed within an area of approximately 20 m x 20 m. The extent of available data is illustrated for K in Figure 1; coverage for K is similar to that for U and slightly better than that for Th (the total number of samples which have been analysed is: K – 28694, U – 33627, Th - 24567).

As the G-BASE survey has been on-going for many years there have been changes in both analytical procedures and sampling (either surface (5-20 cm) or subsurface (35-50 cm) soils have been analysed in different areas). Both these issues have required the database to be normalised for the purposes of this work. This process is fully described in Beresford et al. (2007b). Where data existed for both surface and subsurface soils the values were compared. This demonstrated a good linear relationship between surface and subsurface K data (R^2=0.67), the subsurface values being generally higher and the regression between the two was used to convert the surface soils to equivalent subsurface values where needed. For U data the two values were so close to a 1:1 relationship that the data could be combined without transformation. In the case of Th there was a lack of appropriate data for comparison. As a result, we did not feel that transformation of the Th data was justified and it was assumed that, as for U, Th concentrations were essentially comparable at both depths.

As can be seen in Figure 1 coverage of England and Wales by the G-BASE survey is not yet complete. To provide a more complete coverage geological extrapolation utilising the strong link between soil geochemistry and geology was applied. Simplified bedrock and superficial geology codes based on BGS 1: 50 000 scale digital geological maps (see Appleton, 2005) were attributed to each soil sample location. Geometric means for each element were calculated for each 1 km grid square and parent material (bedrock plus superficial geology) polygon from the nearest 5 soil sample values for that parent. These data were then used to compute geometric means for each 25 km^2 (5 x 5 km) grid square using area-weighted geometric mean values for each parent material found in the square (Figure 1 presents data on this basis). This involved summing the products of the mean element content for each 1 km grid square/parent material polygon (derived from the 5 nearest data points on that parent material) and the area of that polygon and dividing the sum of those products by the total area of the 25 km^2 grid square:

$$GM_{5km} = \sum_{1}^{n} \frac{(\overline{X}_1 Area_1) + (\overline{X}_2 Area_2) + ...(\overline{X}_n Area_n)}{25 km^2}$$

Where $\overline{X}$ is the geometric mean (GM) for a 1 km grid square/parent material polygon.

Figure 2 presents an example (Th) of the resultant extrapolated surface. Specific activities of ^{40}K, ^{232}Th and ^{238}U of 31.6 Bq g^{-1} K, 4.07 Bq mg^{-1} Th and 12.21 Bq mg^{-1} U respectively were used to estimate the activity concentrations of the three radionuclides in soil from stable element concentrations.

3. Results

3.1 Biota

Potassium-40 was detectable in the majority of samples. The activity concentrations of ^{238}U and ^{232}Th series gamma-emitting radionuclides were generally below detection limits (typically of the order 10^2 Bq kg^{-1} for isotopes of interest) or had large analytical errors. Consequently, these results and have not been used in the subsequent analyses presented below (N.B. all data identified within the initial data review stage of this work had been determined by radiochemical separation and subsequent alpha-spectroscopy). Approximately 50% of the 75 samples analysed by ICP-MS had total U concentrations in excess of detection limits; only 20 % of samples had measurable total Th concentrations.

Table 3 summarises activity concentrations of naturally occurring radionuclides in biota in the United Kingdom, combining data from the literature review with those obtained during this study. The data are presented as fresh weight (FW) whole-body activity concentrations as these are required for subsequent dose estimations. Some of the compiled data were for specific tissues (e.g. much of the collated data were available for tissues relevant to the human foodchain) rather than whole-body. To estimate whole-body activity concentrations from tissue-specific values, the assumptions used to derive default transfer databases for the ERICA Tool (see Beresford et al., this issue) were used as described below.

Potassium is relatively uniformly distributed within organisms and tissue specific ^{40}K values can therefore be taken to be representative of whole-body activity concentrations. To convert ^{210}Pb and ^{210}Po activity concentrations reported in the meat of mammals it was been assumed that: meat concentrations are representative of all soft-tissues; soft tissues comprise 90 % of whole-body live-weight; and 10 % and 40 % of the whole-body burdens of Pb and Po respectively will be in soft tissues. Activity concentrations of U and Th isotopes in meat are assumed to be representative of whole-body concentrations. There was no relationship between either Th or U concentrations in the liver of ducks analysed within this study and those in the remainder of the carcass. Therefore, concentrations determined in the liver of heron samples were taken to be representative of whole-body concentrations. A dry matter content of 25% was assumed to convert collated dry matter activity concentrations in both flying insects and plant materials to fresh weight values; for all other dry or ash to fresh weight conversion sample specific measurements were available. To estimate the mean values presented in Table 3, a value of half the minimum detectable activity was assumed for those samples determined to below detection limits.

The most abundant data are for ^{40}K followed by ^{232}Th, ^{234}U and ^{238}U (Table 3); there are relatively few data for ^{210}Pb, ^{210}Po and ^{226}Ra for some RAPs.

Estimated uranium isotopes and ^{232}Th activity concentrations are comparatively high in earthworms compared to the other animals considered. This may be the consequence of residual soil within the gut. The summary values for Wild Grass in Table 3 do not include data from the vicinity of the Springfields plant (north-west England) which manufactures reactor fuel elements and produces uranium hexafluoride. Uranium-234 and ^{238}U activity concentrations in grass samples from around Springfields (EA, 2003) were up to 50 Bq kg^{-1} (FW) (i.e. approximately two-orders of magnitude greater than the mean of the remaining data). Because of the nature of the available data many are from monitoring activities close to sites discharging radionuclides. The database was inspected to investigate if sites other than Springfields skew the data, but no evidence of this was found.

3.2 Soil

Table 4 presents a summary of ^{40}K, ^{238}U and ^{232}Th soil activity concentrations in England and Wales. The summary statistics are for all of the 25 km^2 grid squares for which we have predicted values (*circa* 6200 depending on the extent of coverage for each element). Where values based on data are available these have been used to derive the summary values, otherwise predictions from the geological interpolation described above have been used.

4. Estimation of dose rates

To estimate the exposure of the selected organisms to naturally occurring radionuclides in England and Wales, the probabilistic routines within the ERICA Tool were used (Brown et al., this issue). Dimensions for RAPs have been proposed for phantom dose modelling assuming solid ellipsoids (ICRP, 2005). The ERICA Tool includes dose conversion coefficients (DCCs) for geometries corresponding to adult stages of all of the ICRP's proposed RAPs. The DCCs relate whole-body activity concentrations in biota, or media (soil in the case of terrestrial assessments), to absorbed dose (μGy h^{-1} / Bq kg^{-1} fresh weight) (see Ulanovsky et al., this issue for description of the derivation of DCC values). Daughter products with a physical half-life of less than 10 days are assumed to be in secular equilibrium with the parent radionuclide (e.g. the DCC for ^{228}Th includes contributions from ^{222}Rn, ^{218}Po, ^{214}Pb, ^{214}Bi and ^{214}Po (see Table 1) and inputs for these short-lived radionuclides are not required).

In the case of ^{238}U series radionuclides it can be assumed that activity concentrations of ^{234}U, and the intervening decay products (see Table 1), are in equilibrium with those of ^{238}U in soils to provide estimates of soil activity concentrations. Thereafter, assumptions of equilibrium may not be valid because of the different chemical properties and environmental behaviours of the different elements. Consequently in order to be able to conduct a complete assessment of dose rates activity concentrations of ^{230}Th, ^{226}Ra, ^{210}Pb and ^{210}Po needed to be derived. A limited number of values (n=18-80 depending upon radionuclide) are available for UK soils from predominantly routine monitoring data (Green et al., 2002; Ham et al., 1998; MAFF, 1993, 1994, 1995, 1996; MAFF and SEPA, 1997, 1998, 1999; FSA and SEPA, 2000, 2001, 2002; EA et al., 2003, 2004, 2005). As some of these data are reported with values for ^{238}U, it was possible to estimate ratios of the four radionuclides to ^{238}U of: 1.5 for ^{230}Th, 1.3 for ^{226}Ra, 2.0 for ^{210}Pb and 1.9 for ^{210}Po. These ratios have been used to estimate soil activity concentrations in England and Wales for the four radionuclides from the ^{238}U values we have derived (values are presented in Table 4). All ^{232}Th series radionuclides are likely to be in approximate equilibrium (Olley et al., 1996).

To enable probabilistic estimation of dose rates using the ERICA Tool the mean and standard deviations as presented in Tables 3 and 4 were entered for biota and soil activity concentrations respectively; log-normal distributions were assumed for all input data. It was assumed that the dry matter content of soil was 100 % as information on soil moisture content was not available from G-BASE. If activity concentrations for a given organism-radionuclide were not available they were estimated from input soil concentrations using the ERICA Tool's default transfer parameters. The ERICA Tool uses concentration ratios, defined as the ratio of the whole-body activity concentration (Bq kg^{-1} fresh weight) to the activity concentration in soil (Bq kg^{-1} dry weight) to estimate activity concentrations in biota if data are unavailable (see Beresford et al., this issue). As data were limited for Frog (one sample only), whole-body activity concentrations were estimated from soil concentrations; the ERICA Tool does not contain transfer values for ^{40}K hence the same activity concentration as for Duck (Table 3) was assumed (this being the highest of the two available mean values for vertebrate RAPs). Given the lack of data specific to the proposed RAPs Deer and Rat, data for all terrestrial mammals (predominantly rabbit (*Oryctolagus cuniculus*)) were used for both organisms.

Both unweighted and weighted absorbed whole-body dose rates have been estimated, weighted values were derived using the default radiation weighting factors provided in the ERICA Tool of 3 and 10 for low beta and alpha radiations respectively. For the purpose of estimating external dose rates it was assumed that:

Earthworms spend 100 % of their time in soil; Rat and Frog spend 50 % of their time in soil and 50% on the soil surface; all other animals spend 100 % of their time on the soil surface. Tables 5-7 present estimated external, weighted internal and unweighted internal, absorbed doses rates for each RAP-radionuclide combination; Table 8 compares total weighted and unweighted absorbed dose rates for each RAP.

For all organisms ^{40}K, ^{226}Ra, ^{228}Th and ^{228}Ra (in descending order) dominated the external dose rates (Table 5); dose rates from the other radionuclides were typically at least two orders of magnitude lower than from these four. In the case of the animal RAPs, external dose rates were highest, although of the same order, for those organisms modelled as spending some time underground (Earthworm, Rat and Frog).

Potassium-40 was consistently one of the most important contributors to internal absorbed dose rates (Table 6). Variation in the relative contribution of other radionuclides appeared to be dependent on the origin of the biota activity concentrations used in the dose assessment, (i.e. if actual whole-body activity concentration had been used or if activity concentrations had been estimated using the ERICA Tool default concentration ratios (indicated on Table 6)). This is most noticeable for ^{226}Ra which was predicted to be a major contributor to the internal dose for Duck, Frog, Bee and Earthworm based on whole-body activity concentrations estimated using concentration ratios. For Rat and Deer, for which data were available (Table 3), the estimated dose rates due to ^{226}Ra were more than one-order of magnitude lower than those estimated for ^{40}K. For some organisms ^{228}Th was predicted to be the major contributor to internal weighted dose rate, all calculations for this radionuclide were based on whole-body activity concentrations estimated using the ERICA Tool default concentration ratios. Weighted internal dose rates to Earthworms estimated from U-isotopes were comparatively high compared to most other RAPs.

The estimated unweighted absorbed whole-body dose rates (Table 7) resulting from those radionuclides with α-emissions (see Table 1) were approximately an order of magnitude lower than the weighted dose rate estimates (Table 6). This difference was to be expected given that an α-radiation weighting factor of 10 was used to estimate the weighted dose rates.

5. Discussion

5.1 Soil activity concentrations

The concentrations of naturally occurring radionuclides determined from the G-BASE dataset (Table 4) are in agreement with more limited datasets available for England and Wales (Green et al., 2002; Ham et al., 1998; MAFF, 1996; MAFF and SEPA, 1997, 1998, 1999; FSA and SEPA, 2000, 2001, 2002; EA et al., 2003, 2004, 2005). The comprehensive geospatial datasets derived here for soil activity concentrations in England and Wales are within the ranges of reported values for other European countries (The Radiation Protection Authorities in Denmark, Finland, Iceland, Norway and Sweden, 2000; Anagnostakis et al., 2005; UNSCEAR, 2000; Uyttenhove, 2005). For instance, data presented in UNSCEAR (2000) for European soils gives median activity concentrations of: 465 Bq ^{40}K kg^{-1}, 32 Bq ^{238}U kg^{-1}, 34 Bq ^{226}Ra kg^{-1}

and 30 Bq ^{232}Th kg^{-1}(UNSCEAR present fresh weight activity concentrations assuming an 81 % dry matter content of soil). However, the extremes of ranges reported in most large scale surveys are greater than reported here for England and Wales. This will, to a large extent, be a consequence of using spatial averaging over 5x5 km^2 rather than results for individual sample points.

Activity concentrations of ^{230}Th, ^{226}Ra, ^{210}Pb and ^{210}Po had to be estimated using relatively small datasets to define ratios of the isotopes to ^{238}U in UK soils. Consequently, there is a greater uncertainty associated with these estimates than for the other radionuclides. However, as noted above the estimated values appear to be realistic compared to other data available for the UK and Europe.

5.2 Estimated dose rates

The external dose rates presented in this paper are likely to be conservative as a consequence of having assumed 100% dry matter content for soil. However, the impact of this on the overall estimate of total absorbed dose rate is unlikely to be large because the assumption of 100% dry matter content for soil will only affect the calculation of external dose rates. The input parameter for the ERICA Tool is soil dry weight activity concentrations and the concentration ratios used within the Tool relate dry weight activity concentrations in soil to fresh weight activity concentrations in biota. As a result, the whole-body activity concentrations estimated using the ERICA Tool and resultant internal dose rates will not be overestimated.

The mean total weighted whole-body absorbed dose rate estimated for selected terrestrial organisms was in the range 6.9×10^{-2} to 6.1×10^{-1} μGy h^{-1} (Table 8). This is at least an order of magnitude lower than any benchmark dose rate currently being used in environmental assessments (Hingston et al., 2007). They are however, similar to dose rates estimated in an assessment of a terrestrial ecosystem close to the Sellafield reprocessing plant (Wood et al., this issue).

Uranium isotopes and ^{232}Th activity concentrations determined in earthworms were comparatively high (Table 3) resulting in the highest internal dose rates estimated for any RAP for these radionuclides (Tables 7-8). This may have been the consequence of residual soil within the gut. Similarly, it is likely that a component of the activity concentrations reported for both plant RAPs will have been associated with external contamination (a number of the radionuclides considered have low soil-plant transfers and adherent soil is likely to contribute to the measured activity concentration).

As noted above, the contributions of ^{226}Ra and ^{228}Th to the estimated internal dose rate appear to be comparatively high for some organisms when whole-body activity concentrations are calculated using the ERICA Tool concentration ratios. Estimated dose rates for those organisms for which data are available for ^{226}Ra appear to indicate that some of the higher estimates for other organisms are likely to be overestimated (^{226}Ra is a relatively minor contributor to the total internal dose rate estimated for both mammal RAPs for which data are available). The majority of the default concentration ratios available in the ERICA Tool for radionuclides of U, Th and Ra either originate from studies in North America or the former Soviet Union, or are based upon a guidance methodology used to select default values when specific organism-radionuclide concentration ratios are not available (see Beresford et al. (this issue) for details).

As yet there is no consensus on radiation weighting factors to apply for assessments of biota. A comparison of unweighted dose rates with those estimated using a weighting factor for α-emissions of 10 (Table 8) gives an indication of the influence of the weighting factor used to the overall dose rate predicted.

A potential route of exposure we have not considered is inhalation of ^{222}Rn by burrowing animals. Macdonald and Laverlock (1998) suggest that dose rates to the lung of burrowing animals may be in excess of 57 μGy h^{-1} in southeastern Manitoba (Canada). The soil gas concentrations these authors quote are in the range of those reported for typical soils in some European countries (The Radiation Protection Authorities in Denmark, Finland, Iceland, Norway and Sweden, 2000).

5.3 Applicability of data to European terrestrial ecosystems

As noted above the ERICA Tool contains data derived from the initial report of this work (Beresford et al., 2007b) to enable users to put results of their assessments of anthropogenic activities into context with natural exposure rates (Brown et al., this issue). We therefore need to consider if these data derived for the United Kingdom are of value for assessments in Europe.

There is an acknowledged lack of data for naturally occurring radionuclides and terrestrial wildlife (e.g. Jones et al., 2003) especially from ecosystems subject to typical exposure rates (i.e. not the result of studies of sites impacted as a result of (e.g.) uranium mining activities). Given that potassium is an essential element under homeostatic control, the ^{40}K activity concentrations reported here should be similar to other areas within Europe. Similarly, as the soil activity concentrations for ^{238}U and ^{232}Th series radionuclides estimated here for England and Wales are broadly comparable to those of other European countries it could be anticipated that whole-body activity concentrations (and consequently estimated absorbed dose rates) may be generally applicable to most of Europe. An exception may be reindeer which can ingest lichen with comparatively high concentrations of ^{210}Po and ^{210}Pb (Gómez-Ros et al., 2004; Beresford et al., this issue). However, if assessments of this species were required, data tend to be available (e.g. Beresford et al., 2005). Given the lack of other data, the data presented here should give an indicative estimate of natural background exposure to wildlife in most European countries with the exception of those areas known to have enhanced levels of naturally occurring radionuclides. It is however worth noting that much of the data compiled here for wildlife in the United Kingdom originated in the grey literature (e.g. routine monitoring reports); it is possible that data for other European countries exists in similar publications.

The absorbed dose rates reported in Beresford et al. (2007b) and currently (December 2007) quoted within the ERICA Tool as typical of terrestrial ecosystems are not the same as those reported here. The earlier work did not estimate absorbed dose rates probabilistically and reported values are averages for different administrative areas of England and Wales. Furthermore, dose rate estimates were based upon available biota activity concentrations (i.e. concentration ratios and soil activity concentrations were not used to provide missing values) and soil activity concentrations for ^{40}K and those radionuclides of the ^{238}U and ^{232}Th series which could be considered in equilibrium with the series parent (i.e. soil concentrations of ^{210}Pb, ^{210}Po, ^{230}Th, or ^{226}Ra were not estimated). We recommend that, if the ERICA Tool is to continue to rely upon this work (at the moment there appears to be little alternative) to provide estimates of natural exposure rates to wildlife, the existing values are replaced with those reported here.

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Table 1. Primary decay schemes of ^{238}U and ^{232}Th (adapted from Whicker and Schultz 1982).

Uranium-238			Thorium-232		
Radionuclide	**Half-life**	**Radiation**	**Radionuclide**	**Half-life**	**Radiation**
^{238}U	$4.5x10^{9}$ y	α,γ	^{232}Th	$1.4x10^{10}$ y	α,γ
^{234}Th	24 d	β,γ	^{228}Ra	6.7 y	β,γ
^{234}Pa	1.2 min	β,γ	^{228}Ac	6.1 h	β,γ
^{234}U	$2.5x10^{5}$ y	α,γ	^{228}Th	1.9 y	α,γ
^{230}Th	$8.0x10^{4}$ y	α,γ	^{224}Ra	3.6 d	α,γ
^{226}Ra	$1.6x10^{3}$ y	α,γ	^{220}Rn	55 s	α,γ
^{222}Rn	3.8 d	α,γ	^{216}Po	0.16 s	α,β
$^{218}Po^{*}$	3.1 min	α,β	^{212}Pb	11 h	β, γ
^{214}Pb	27 min	β,γ	$^{212}Bi^{*}$	61 min	α,β,γ
$^{214}Bi^{*}$	20 min	α,β,γ	^{212}Po	$3.0x10^{-7}$ s	α
^{214}Po	$1.6x10^{-4}$ s	α	^{208}Pb	Stable	None
^{210}Pb	19 y	β,γ			
^{210}Bi	5.0 d	α,β,γ			
^{210}Po	138 d	α,γ			
^{206}Pb	Stable	None			

*Less frequent branching decays not shown.

Table 2. Numbers of observations for which ^{40}K, and ^{232}Th and ^{238}U series radionuclide activity concentrations in UK biota were compiled from the literature/in-house sources; data sources are indicated.

RAP	^{40}K	^{210}Po	^{210}Pb	^{226}Ra	^{230}Th	^{232}Th	^{234}U	^{238}U
Duck	27 (a,e)				5 (j-m,s)	5 (j-m,s)		
Pine tree	2 (a)	5 (c)	5 (c)	5 (c)	5 (c)	5 (c)	9 (b,c)	9 (b,c)
Wild grass	91 (e,f,n,p-r)	45 (c,d,j-r,t)	46 (c,d,j-r,t)	40 (c,d,j-r,t)	4 (c,d)	46 (c,d,j-r,t)	55 (b-d,g-m,o-s)	55 (b-d,g-m,o-s)
All mammals*	145 (a,e,n,p-r)	32 (c,j-r)	30 (c,j-r)	24 (c,j-p,r)	2 (c)	30 (c,j-r)	4 (c,r)	3 (c,r)
Deer	72 (a,e)	1 (k)						

*All mammals includes available Deer data in addition to that for other mammalian species.
No data were available for terrestrial RAPs not shown on the table.

Data sources:
a. Fulker et al. (1995)
b. Green et al. (1999)
c. Green et al. (2002)
d. Ham et al. (1998)
e. Horrill et al. (1988) (data collated for this study, previously unpublished)
f. Horrill and Mudge (1990)
g. Environment Agency (2001a)
h. Environment Agency (2001b)
i. Environment Agency (2003)
j. MAFF (1996)
k. MAFF and SEPA (1997)
l. MAFF and SEPA (1998)
m. MAFF and SEPA (1999)
n. FSA and SEPA (2000)
o. FSA and SEPA (2001)
p. FSA and SEPA (2002)
q. EA et al. (2003)
r. EA et al. (2004)
s. EA et al. (2005
t. Smith et al. (2001)

Table 3. Summary of natural radionuclide activity concentrations in biota within the UK.

		Whole-body activity concentration (Bq kg^{-1}) FW							
RAP		^{40}**K**	^{210}**Po**	^{210}**Pb**	^{226}**Ra**	^{230}**Th**	^{232}**Th**	^{234}**U**	^{238}**U**
Wild grass	Mean	270	2.54	3.81	0.33	2.7×10^{-2}	0.12	0.38	0.36
	SD	122	2.4	3.16	0.29	3.0×10^{-2}	0.17	0.45	0.40
	Min	40.6	8.5×10^{-2}	0.18	3.7×10^{-2}	2.8×10^{-3}	$<5.5\times10^{-2}$	$<2.0\times10^{-2}$	$<2.0\times10^{-2}$
	Max	708	13	17	1.70	7.0×10^{-2}	0.81	2.9	2.3
	n	91	45	46	40	4	46	55	55
Pine tree	Mean	35.2	0.19	0.10	7.8×10^{-2}	5.9×10^{-3}	5.2×10^{-3}	7.6×10^{-3}	7.3×10^{-3}
	SD	43.7	0.19	4.9×10^{-2}	8.5×10^{-2}	6.0×10^{-3}	4.6×10^{-3}	7.4×10^{-3}	7.4×10^{-3}
	Min	7.79	3.4×10^{-2}	3.5×10^{-2}	4.6×10^{-3}	$<4.8\times10^{-3}$	$<4.7\times10^{-3}$	1.1×10^{-3}	7.7×10^{-4}
	Max	135	0.5	0.16	0.18	1.4×10^{-2}	1.4×10^{-2}	2.6×10^{-2}	2.6×10^{-2}
	n	8	5	5	5	5	11	11	11
Earthworm	Mean	32.0	n/a	n/a	n/a	n/a	0.68	0.74	0.74
	SD	21.4					0.46	0.39	0.39
	Min	<7.5					0.24	0.36	0.36
	Max	54.6					1.41	1.44	1.44
	n	6					6	6	6
Bee	Mean	66.4	n/a	n/a	n/a	n/a	$<3.1\times10^{-2}$	1.4×10^{-2}	1.4×10^{-2}
	SD	73.4						6.6×10^{-3}	6.6×10^{-3}
	Min	<38.6					$<3.1\times10^{-2}$	$<6.1\times10^{-3}$	$<6.1\times10^{-3}$
	Max	240					$<3.1\times10^{-2}$	2.4×10^{-2}	2.4×10^{-2}
	n	8					8	8	8
Frog	Mean	<16.1	n/a	n/a	n/a	n/a	1.8×10^{-2}	3.4×10^{-2}	3.4×10^{-2}
	n	1					1	1	1
Duck	Mean	106	n/a	n/a	n/a	9.1×10^{-3}	3.3×10^{-2}	1.7×10^{-2}	1.6×10^{-2}
	SD	32.5				6.2×10^{-3}	3.1×10^{-2}	2.4×10^{-2}	2.4×10^{-2}
	Min	<26				$<5.0\times10^{-3}$	$<1.9\times10^{-3}$	$<6.2\times10^{-3}$	$<6.2\times10^{-3}$
	Max	173				1.5×10^{-2}	0.18	0.13	0.13
	n	40				5	30	25	25
All mammals	Mean	103	8.4×10^{-2}	0.43	2.1×10^{-2}	2.9×10^{-3}	2.3×10^{-3}	4.7×10^{-3}	5.3×10^{-3}
	SD	20.2	8.9×10^{-2}	0.41	2.8×10^{-2}	1.4×10^{-3}	2.3×10^{-3}	3.1×10^{-3}	4.4×10^{-3}
	Min	36.3	$<1.8\times10^{-2}$	$<9.9\times10^{-2}$	1.3×10^{-3}	1.8×10^{-3}	$<9.0\times10^{-4}$	4.2×10^{-4}	3.3×10^{-4}
	Max	178	0.5	1.62	0.14	3.9×10^{-3}	1.2×10^{-2}	1.1×10^{-2}	1.5×10^{-2}
	n	153	32	30	24	2	38	12	11

n/a = no available data

Table 4. Estimated activity concentrations of natural radionuclides in soils of England and Wales; summary statistics are over all 25 km^2 grid squares (n≈6200) for which there are data or predicted values.

	Bq kg^{-1} (DM)						
	^{40}K	**^{232}Th**^{a}	**^{238}U**^{b}	**^{230}Th**	**^{226}Ra**	**^{210}Pb**	**^{210}Po**
Mean	514	32	25	37	32	50	47
SD	152	9.5	10	15	13	20	19
Min	94	2.9	1.5	2.2	1.9	2.9	2.8
Max	1205	109	77	115	99	153	145

[a]All radionuclides within the ^{232}Th decay series (see Table 1) are assumed to have the same activity concentration as ^{232}Th in subsequent calculations.
[b]Uranium-234 and intervening decay products (see Table 1) are assumed to have the same activity concentration as ^{238}U in subsequent calculations.

Table 5. Estimated external whole-body dose rates for terrestrial RAPs due to ^{40}K, ^{238}U-series and ^{232}Th series radionuclides; mean 5th and 95th percentile predictions are presented.

RAP	Absorbed dose rate µGy h^{-1}										
	^{40}K	**^{238}U**	**^{234}Th**	**^{234}U**	**^{230}Th**	**^{226}Ra**	**^{210}Pb**	**^{210}Po**	**^{232}Th**	**^{228}Ra**	**^{228}Th**
Wild grass	1.5E-2	2.5E-6	1.2E-4	3.4E-6	5.3E-6	1.1E-2	2.0E-5	7.9E-8	3.5E-6	6.1E-3	8.9E-3
	(0.89-2.2)E-2	(1.3-4.3)E-6	(0.56-2.1)E-4	(1.6-6.9)E-6	(2.6-9.3)E-6	(0.51-1.8)E-2	(0.92-3.5)E-5	(3.8-14)E-8	(2.1-5.5)E-6	(3.7-9.3)E-3	(5.4-14)E-3
Pine tree	1.2E-2	1.7E-7	9.3E-5	4.4E-7	1.7E-6	8.6E-3	6.4E-6	6.5E-5	6.7E-7	4.8E-3	7.3E-3
	(0.73-1.8)E-2	(0.86-3.0)E-7	(4.4-17)E-5	(2.1-7.6)E-7	(0.82-2.9)E-6	(4.2-15)E-3	(3.0-11)E-6	(3.1-11)E-8	(3.9-11)E-7	(2.9-7.4)E-3	(4.4-11)E-3
Earthworm	4.1E-2	3.0E-6	2.8E-4	4.2E-6	8.0E-6	2.9E-2	2.9E-5	2.1E-7	4.5E-6	1.6E-2	2.5E-2
	(2.5-6.1)E-2	(1.5-5.2)E-6	(1.3-5.0)E-4	(2.0-7.1)E-6	(3.9-14)E-5	(1.4-5.0)E-2	(1.4-5.3)E-5	(1.0-3.7)E-7	(2.6-7.0)E-6	(0.98-2.5)E-2	(1.5-3.9)E-2
Bee	1.6E-2	1.2E-6	1.2E-4	1.7E-6	2.7E-6	1.1E-2	1.4E-5	7.9E-8	1.4E-6	6.1E-3	9.3E-3
	(0.93-2.3)E-2	(0.62-2.2)E-6	(0.55-2.1)E-4	(0.83-3.0)E-6	(1.3-4.8)E-6	(0.54-1.9)E-2	(0.67-2.6)E-5	(3.8-14)E-8	(0.81-2.2)E-6	(3.7-9.3)E-3	(5.6-14)E-3
Frog	2.8E-2	2.1E-6	2.0E-4	2.9E-6	5.1E-6	2.0E-2	2.1E-5	1.4E-7	2.9E-6	1.1E-2	1.7E-2
	(1.7-4.2)E-2	(1.1-3.6)E-6	(0.93-3.6)E-4	(1.4-5.0)E-6	(2.5-9.0)E-6	(0.96-3.4)E-2	(0.99-3.8)E-5	(0.69-2.5)E-7	(1.7-4.6)E-6	(0.66-1.7)E-2	(1.0-2.7)E-2
Duck	1.5E-2	1.2E-6	1.2E-4	1.7E-6	2.7E-6	1.1E-2	1.4E-5	7.9E-8	1.4E-6	6.1E-3	9.3E-3
	(0.87-2.2)E-2	(0.60-2.1)E-6	(0.55-2.1)E-4	(0.81-2.9)E-6	(1.3-4.7)E-6	(0.53-1.9)E-2	(0.64-2.5)E-5	(3.8-14)E-8	(0.80-2.2)E-6	(3.7-9.3)E-3	(5.6-14)E-3
Deer	8.4E-3	2.5E-7	5.5E-5	4.2E-7	9.5E-7	5.7E-3	3.8E-6	4.0E-8	4.1E-7	3.1E-3	5.1E-3
	(5.0-13)E-3	(1.3-4.3)E-7	(2.6-10)E-5	(2.0-7.1)E-7	(4.6-17)E-7	(2.8-10)E-3	(1.8-6.8)E-6	(1.9-7.0)E-8	(2.4-6.5)E-7	(1.9-4.8)E-3	(3.1-8.0)-3
Rat	2.7E-2	1.8E-6	2.0E-4	2.7E-6	4.7E-6	1.9E-2	2.0E-5	1.4E-7	2.7E-6	1.1E-2	1.6E-2
	(1.6-4.1)E-2	(0.92-3.2)E-6	(0.92-3.5)E-4	(1.3-4.6)E-6	(2.3-8.3)E-6	(0.92-3.3)E-2	(0.92-3.5)E-5	(0.67-2.4)E-7	(1.5-4.1)E-6	(0.64-1.6)E-2	(0.99-2.6)E-2

Table 6. Estimated weighted internal whole-body dose rates for terrestrial RAPs due to ^{40}K, ^{238}U-series and ^{232}Th series radionuclides; mean 5th and 95th percentile predictions are presented.

	Absorbed dose rate µGy h^{-1}										
RAP	**^{40}K**	**^{238}U**	**^{234}Th**	**^{234}U**	**^{230}Th**	**^{226}Ra**	**^{210}Pb**	**^{210}Po**	**^{232}Th**	**^{228}Ra**	**^{228}Th**
Wild grass	7.9E-2	8.7E-3	4.5E-4	1.0E-2	7.2E-4	4.6E-2	8.8E-4	7.5E-2	2.5E-3	3.4E-4	2.7E-1
	(3.4-15)E-2	(1.3-26)E-3	(0.25-16)E-4	(0.15-30)E-2	(1.2-21)E-4	(1.1-13)E-2	(2.0-23)E-4	(1.5-21)E-2	(0.24-8.2)E-3	(0.33-11)E-3	(0.16-9.9)E-1
Pine tree	1.2E-2	1.9E-4	1.4E-5	2.1E-4	1.6E-4	1.1E-2	2.6E-5	5.8E-3	1.2E-4	1.3E-5	6.6E-3
	(0.18-3.5)E-2	(0.29-5.0)E-4	(0.22-4.1)E-5	(0.38-6.1)E-4	(0.28-4.4)E-4	(0.18-3.2)E-2	(1.1-5.2)E-5	(1.0-17)E-3	(0.26-3.0)E-4	(0.17-3.9)E-5	(1.1-18)E-3
Earthworm	9.4E-3	1.7E-2	8.9E-5	2.1E-2	8.9E-3	3.7E-1	3.3E-4	3.9E-3	1.6E-2	8.0E-4	5.0E-2
	(2.8-21)E-3	(0.70-3.2)E-2	(0.35-28)E-5	(0.78-4.2)E-2	(0.35-30)E-3	(0.16-12)E-1	(0.23-11)E-4	(0.18-13)E-3	(0.48-3.6)E-2	(0.35-25)E-4	(0.23-15)E-2
Bee	1.8E-2	3.3E-4	7.3E-5	3.8E-4	9.1E-3	3.8E-1	6.5E-4	4.0E-3	7.2E-4	7.4E-4	5.2E-2
	(0.27-5.1)E-2	(1.5-6.3)E-4	(0.27-23)E-5	(1.7-7.3)E-4	(0.32-29)E-3	(0.16-12)E-1	(2.8-12)E-4	(0.17-13)E-3	(0.36-21)E-4	(0.31-24)E-4	(0.22-12)E-2
Frog	3.4E-2	2.9E-4	4.4E-6	3.4E-4	3.8E-4	1.6E-1	1.5E-3	4.0E-3	2.7E-4	3.7E-4	2.2E-3
	(2.0-5.4)E-2	(0.16-9.1)E-4	(0.18-16)E-6	(0.16-11)E-4	(0.17-12)E-4	(0.31-5.0)E-1	(0.01-5.4)E-3	(0.17-13.0)E-3	(0.12-8.4)E-4	(0.21-11)E-3	(0.12-7.1)E-3
Duck	3.6E-2	3.9E-4	4.9E-6	4.8E-4	2.2E-4	1.5E-1	8.2E-4	3.9E-3	7.7E-4	4.2E-4	2.3E-3
	(2.2-5.8)E-2	(0.35-14)E-4	(0.23-15)E-6	(0.51-18)E-4	(0.01-8.5)E-4	(0.07-4.8)E-1	(0.17-37)E-4	(0.19-14)E-3	(1.4-20)E-4	(0.18-13)E-4	(0.10-6.9)E-3
Deer	4.0E-2	1.3E-4	1.5E-6	1.2E-4	7.9E-5	2.9E-3	1.0E-4	2.5E-3	5.2E-5	5.8E-4	7.1E-4
	(2.8-5.3)E-2	(0.28-3.2)E-4	(0.12-4.8)E-6	(0.39-2.7)E-4	(3.4-15)E-5	(0.34-9.1)E-3	(0.21-2.8)E-4	(0.40-7.1)E-3	(0.89-14)E-5	(0.60-18)E-4	(0.65-23)E-4
Rat	3.5E-2	1.3E-4	1.5E-6	1.3E-4	7.8E-5	2.9E-3	1.1E-4	2.7E-3	5.5E-5	2.7E-4	7.5E-4
	(2.5-4.6)E-2	(0.29-3.2)E-4	(0.12-5.3)E-6	(0.42-3.2)E-4	(3.3-15)E-4	(0.35-9.0)E-3	(0.21-2.9)E-4	(0.45-8.2)E-3	(0.94-15)E-5	(0.32-7.8)E-4	(0.67-23)E-4

Shaded cells identify organism-radionuclide combinations for which whole body activity concentrations were estimated using the ERICA Tool default concentration ratio values.

Table 7. Estimated unweighted internal whole-body dose rates for terrestrial RAPs due to ^{40}K, ^{238}U-series and ^{232}Th series radionuclides; mean 5th and 95th percentile predictions are presented.

	Absorbed dose rate μGy h^{-1}										
RAP	**^{40}K**	**^{238}U**	**^{234}Th**	**^{234}U**	**^{230}Th**	**^{226}Ra**	**^{210}Pb**	**^{210}Po**	**^{232}Th**	**^{228}Ra**	**^{228}Th**
Wild grass	8.0E-2	8.4E-4	4.5E-4	1.1E-3	6.9E-5	4.7E-3	8.2E-4	7.5E-3	2.8E-4	3.3E-4	2.9E-2
	(3.5-15)E-2	(1.4-26)E-4	(0.27-17)E-4	(0.13-3.0)E-3	(1.1-21)E-4	(1.0-12)E-3	(2.0-19)E-4	(1.6-20)E-3	(0.29-9.1)E-4	(0.32-11)E-4	(0.18-10)E-2
Pine tree	1.3E-2	1.8E-5	1.4E-5	2.2E-5	1.6E-5	1.2E-3	2.4E-5	5.9E-4	1.1E-5	1.2E-5	6.9E-4
	(0.18-4.3)E-2	(0.32-5.1)E-5	(0.21-4.3)E-5	(0.42-6.0)E-5	(0.31-4.4)E-5	(0.18-3.5)E-3	(1.1-4.6)E-5	(1.1-16)E-4	(0.25-3.1)E-5	(0.16-3.9)E-5	(1.1-20)E-4
Earthworm	9.3E-3	1.8E-3	9.1E-5	2.0E-3	9.0E-4	4.1E-2	3.4E-4	4.1E-4	1.5E-3	7.8E-4	5.5E-3
	(2.3-21)E-3	(0.72-3.7)E-3	(0.29-31)E-5	(0.80-4.0)E-3	(0.30-29)E-4	(0.23-13)E-2	(0.25-13)E-4	(0.20-13)E-4	(0.45-3.5)E-3	(0.47-24)E-4	(0.20-17)E-3
Bee	1.8E-2	3.4E-5	7.1E-5	4.0E-5	8.6E-4	4.2E-2	6.3E-4	4.0E-4	6.8E-5	7.1E-4	5.2E-3
	(0.28-5.0)E-2	(1.5-6.7)E-5	(0.33-22)E-5	(1.7-7.5)E-5	(0.44-27)E-4	(0.15-13)E-2	(2.9-12)E-4	(0.18-13)E-4	(0.37-21)E-4	(0.26-22)E-4	(0.27-16)E-3
Frog	3.4E-2	2.9E-5	4.2E-6	3.4E-5	3.8E-5	1.7E-2	1.7E-3	4.2E-4	2.8E-5	3.5E-4	2.3E-4
	(2.0-5.3)E-2	(0.14-9.8)E-5	(0.25-13)E-6	(0.14-11)E-5	(0.21-12)E-5	(0.07-5.2)E-2	(0.02-5.8)E-3	(0.23-13)E-4	(0.19-8.9)E-5	(0.16-11)E-4	(0.14-6.4)E-4
Duck	3.6E-2	3.6E-5	4.7E-6	5.0E-5	2.1E-5	1.8E-2	8.0E-4	4.1E-4	7.7E-5	4.3E-4	2.3E-4
	(2.1-5.7)E-2	(0.35-12)E-5	(0.20-16)E-6	(0.50-17)E-5	(0.01-9.4)E-5	(0.06-6.1)E-2	(0.18-31)E-4	(0.20-13)E-4	(1.6-20)E-5	(0.19-14)E-4	(0.10-6.8)E-4
Deer	4.0E-2	1.2E-5	1.5E-6	1.3E-5	7.7E-6	3.2E-4	1.0E-4	2.5E-4	5.5E-6	5.3E-4	7.4E-5
	(2.8-5.3)E-2	(0.30-3.0)E-5	(0.12-4.9)E-6	(0.41-2.7)E-5	(3.2-15)E-6	(0.38-11)E-4	(0.20-2.9)E-4	(0.40-7.8)E-4	(0.98-15)E-6	(0.58-18)E-4	(0.74-24)E-5
Rat	3.5E-2	1.3E-5	1.3E-6	1.3E-5	7.6E-6	3.1E-4	1.0E-4	2.7E-4	5.4E-6	2.7E-4	6.8E-5
	(2.5-4.7)E-2	(0.30-3.2)E-5	(0.11-4.6)E-6	(0.41-3.0)E-5	(3.3-15)E-6	(0.35-11)E-4	(0.20-2.8)E-4	(0.43-8.3)E-4	(0.90-15)E-6	(0.30-8.3)E-4	(0.63-23)E-5

Shaded cells identify organism-radionuclide combinations for which whole body activity concentrations were estimated using the ERICA Tool default concentration ratio values.

Table 8. A comparison of total weighted and unweighted whole-body absorbed dose rates for terrestrial RAPs due to ^{40}K, ^{238}U-series and ^{232}Th series radionuclides; mean 5th and 95th percentile predictions are presented.

	Absorbed dose rate μGy h^{-1}	
RAP	**Weighted**	**Unweighted**
Wild grass	5.3E-1	1.7E-1
	(2.3-13)E-1	(0.99-2.7)E-1
Pine tree	6.9E-2	4.9E-2
	(4.1-11)E-2	(3.2-7.8)E-2
Earthworm	6.1E-1	1.7E-1
	(2.2-15)E-1	(1.1-2.8)E-1
Bee	5.1E-1	1.1E-1
	(1.1-14)E-1	(0.54-2.1)E-1
Frog	2.8E-1	1.3E-1
	(1.2-6.4)E-1	(0.93-1.8)E-1
Duck	2.4E-1	9.8E-2
	(0.88-5.7)E-1	(6.4-15)E-2
Deer	6.9E-2	6.3E-2
	(5.5-8.7)E-2	(5.0-7.9)E-2
Rat	1.2E-1	1.1E-1
	(0.93-1.4)E-1	(0.88-1.3)E-1

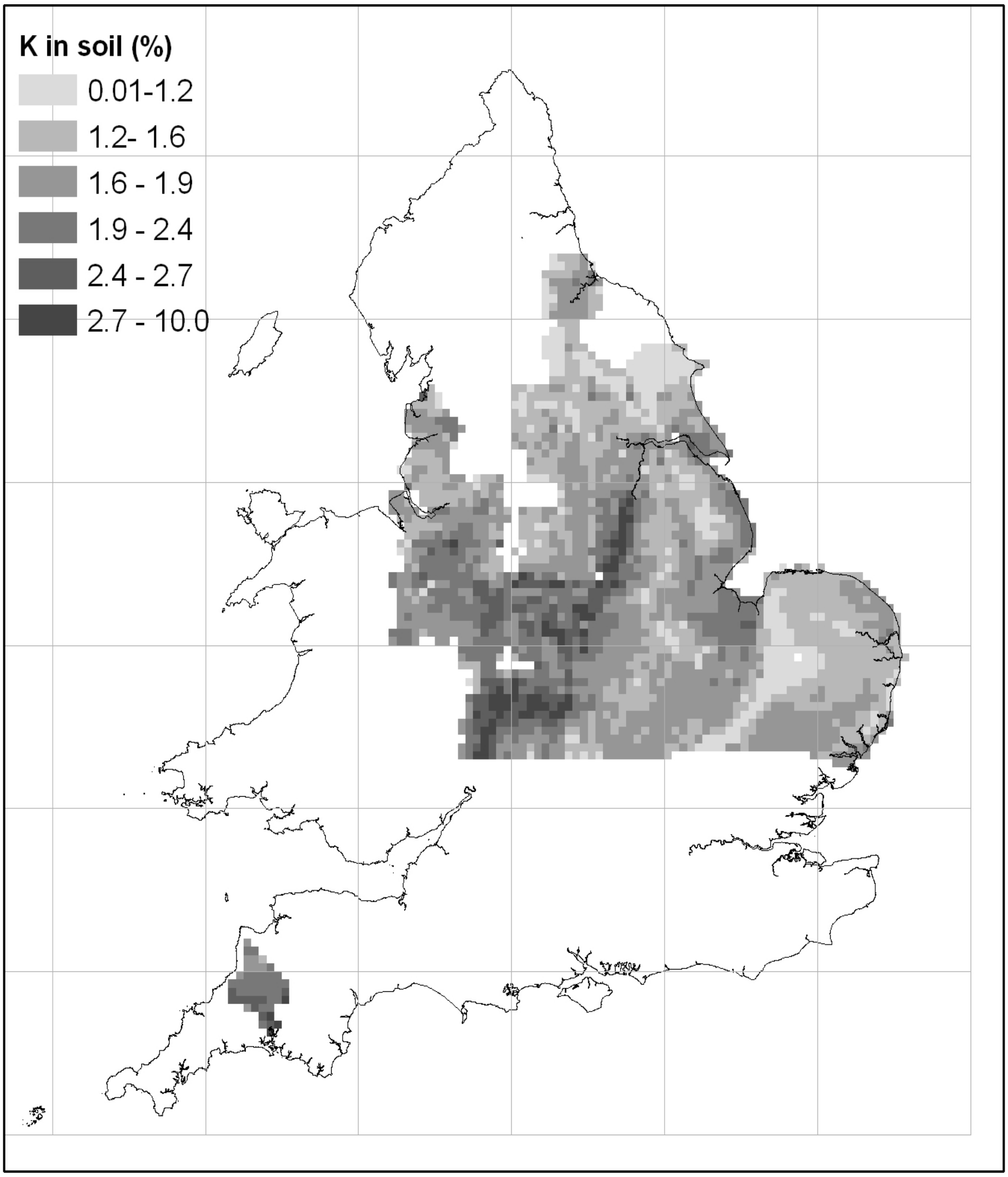

Figure 1. Concentrations of K in soils presented as geometric mean of measurements within 5x5 km squares (presented as percentage dry weight).

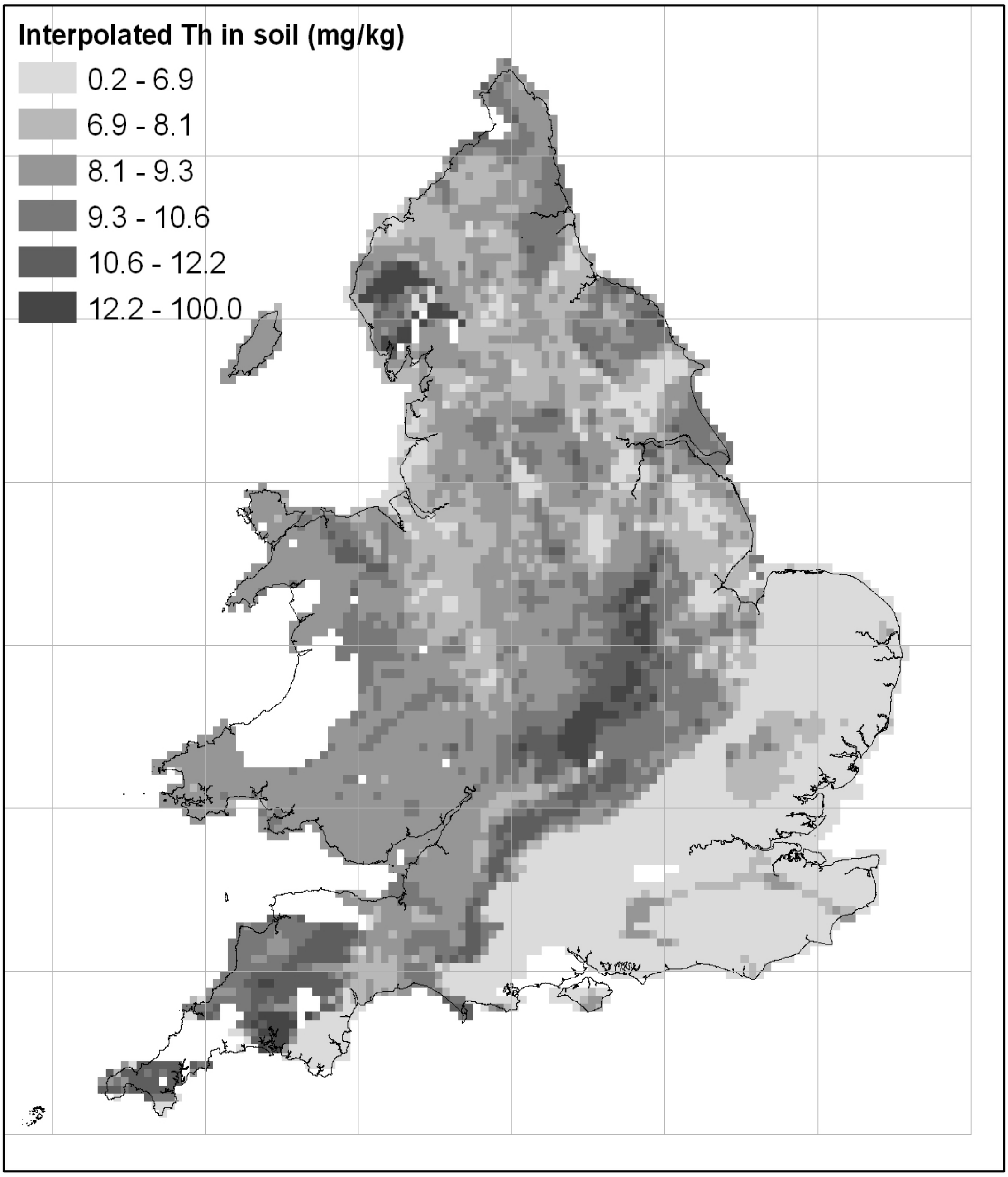

Figure 2. Concentrations of Th in soils derived by geological extrapolation (mg kg^{-1} dry weight).