Conference or Workshop Item

Beresford, N.A.; **Barnett, C.L.**; **Potter, E.D.**; Vives I Batlle, J.; Barlow, T.S.; Copplestone, D.. 2011 Radon-222 dose rates to burrowing mammals. [Lecture] In: *Radioecology & Environmental Radioactivity - Environmental & Nuclear Renaissance, Hamilton, Ontario, Canada, June 19-24, 2011*. (Unpublished)

RADON-222 DOSE RATES TO BURROWING MAMMALS

BERESFORD, N.A.[1+], BARNETT, C.L.[1], POTTER, E.D.[1], VIVES I BATLLE, J.[2], BARLOW T.S.[3] & COPPLESTONE D.[4]

[1]Centre for Ecology & Hydrology, Lancaster Environment Centre, Library Av. Bailrigg, Lancaster, LA1 4AP, United Kingdom
[2] Belgian Nuclear Research Centre, Mol, Belgium
[3]British Geological Survey, United Kingdom
[4]University of Stirling, United Kingdom
[+]nab@ceh.ac.uk

1. INTRODUCTION

Estimates of absorbed dose rates as a consequence of exposure of wildlife to natural background radionuclides are required to put results of assessments conducted for releases of radionuclides from licensed sites into context. There have been recent review papers in which estimated dose rates to marine, freshwater and terrestrial wildlife (specifically the ICRP Reference Animals and Plants (ICRP 2008)) from ^{40}K and radionuclides in the ^{238}U and ^{232}Th series have been presented (Beresford et al. 2008; Hosseini et al. 2010). Average estimated weighted absorbed dose rates to all organisms considered were in the region of 1 μGy h^{-1}. However, there is to date, only one study published in the refereed literature which estimates dose rates to burrowing mammals as a consequence of inhalation of ^{222}Rn (Macdonald & Laverock 1998). The results of this study suggested that dose rates from ^{222}Rn may be an order of magnitude greater than those received from ^{40}K, and ^{238}U and ^{232}Th series radionuclides. However, the study was conducted in an area of Canada with radon-rich soils and the results may not be typical for most areas.

Here we report provisional results from a study 2009-2010 to estimate the ^{222}Rn to burrowing mammals at seven sites in northwest England.

2. MATERIALS & MTHODS

The studies used passive detectors of the type used to monitor household ^{222}Rn concentrations, which were placed in artificial burrows, and the dosimetric method recently developed for the Environment Agency (Vives i Batlle et al. 2008).

2.1 Field studies

Seven sites were selected in northwest England with the aid of the British Geological Survey. Using data for household radon potentials, where radon potential is the percentage of homes in an area above the radon Action Level (AL) (Miles et al. 2007), the sites were chosen to give a probable range of gas radon concentrations in soil. Details of the seven sites are presented in Table 1. Sites 1, 6 and 7 (i.e. those with the highest potential) were all located in areas of limestone.

At each site, three artificial burrows were put into place; the burrows were located within an area of approximately 100 m^2 at each site. These constituted an approximately 1.2 m length of 10 cm diameter perforated plastic land drainage pipe buried at a depth of *circa* 50 cm for most of the length with one end open at the ground surface (Figure 1). A small flap was cut into the top of the pipe at the end to be buried to allow the passive detector to be inserted; the location of the flap was marked by a cane prior to the tube being buried. The open end was covered by wire poultry fencing to prevent animals accessing the burrows[1]. Soil was packed over the burrow and tamped down such that no gaps were visible in the soil. Because of the shallow nature of the soil at Site 1 it was only possible to locate one burrow to the required depth.

---

[1]During the course of the study rodent damage to the polythene bag holding the detector was observed at a number of the sites. To stop this, balls of the poultry wire were placed in the tube either side of the detector; this was effective in preventing any further damage.

Approximately one week later the sites were revisited and a passive detector fitted into the tube at the buried end. The soil was again packed back over the tube and tamped down (as it was on each subsequent sampling occasion).

Sites 1 -5 were initiated in July 2009, work at Sites 6 and 7 began somewhat later in August and October 2009 respectively. Detectors were replaced at 4-6 week intervals until June 2010. The detectors used were obtained from the UK Health Protection Agency who also analysed the exposed detectors. The manufacture and analysis of the detectors is described in Ibrahimi & Miles (2008). For this work, a moisture-resistant variant of the detectors was used, which comprised the standard detector heat-sealed in 200 µm thick polyethylene (Miles et al. 2009).

Soil samples (0-10 cm) were taken close to each artificial burrow, dried, homogenised and weighed into plastic containers for subsequent gamma-analyses. The containers were sealed and allowed to reach secular equilibrium for 25 days prior to counting using hyper-pure Ge-detectors with spectral analyses performed using the Canberra Apex-Gamma software package.

Table 1. Summary information for the study sites.

Site	Habitat	Rn potential	Latitude	Longitude	Month started
Site 1	Deciduous woodland	10-30% >AL	54 12 32 9	002 51 58 6	July 2009
Site 2	Pasture	1-3 %>AL	54 14 48 9	002 59 33 3	July 2009
Site 3	Deciduous woodland	<1 %>AL	54 14 45 6	002 59 33 6	July 2009
Site 4	Coniferous woodland	<1 %>AL	54 23 17 0	003 10 51 0	July 2009
Site 5	Scrub	<1 %>AL	54 19 08 5	003 02 14 3	July 2009
Site 6	Pasture	10-30 %>AL	54 26 39 0	002 31 02	Aug. 2009
Site 7	Deciduous woodland	>30 %>AL	54 13 59 1	003 01 30 7	Oct. 2009

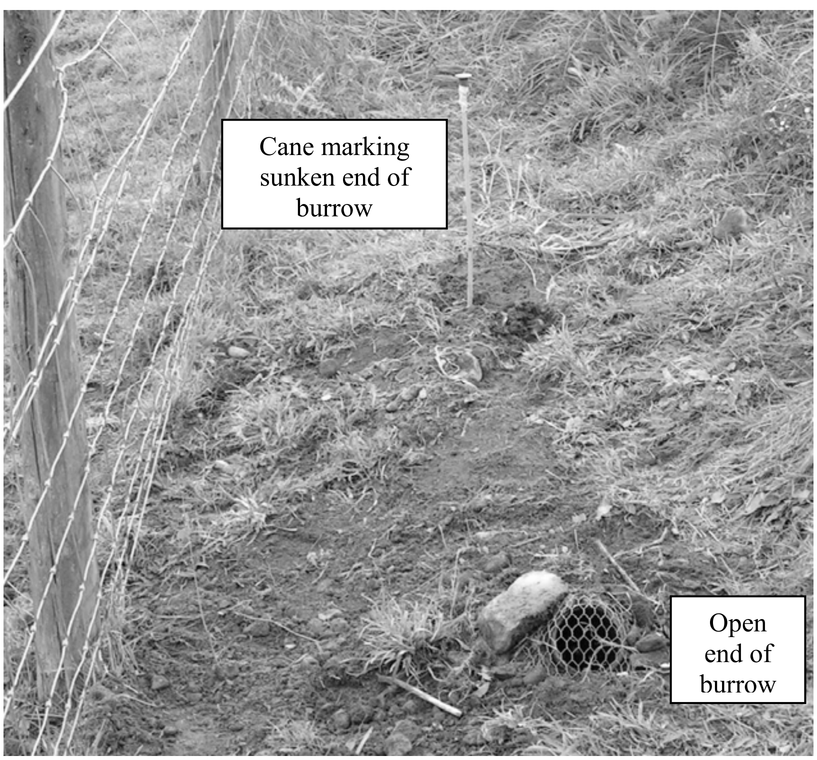

Figure 1. Artificial burrow at Site 2 immediately after being set-up.

2.2 Dosimetric methodologies

For this study we adapted the wholebody dose conversion coefficients (DCCs) for ^{222}Rn in equilibrium with short-lived daughter radionuclides (^{218}Po, ^{218}At, ^{214}Pb and ^{214}Bi) estimated by Vives i Batlle et al (2008). This uses an allometric method to scale parameters for the respiratory system, and consequently the DCCs for different animals, according to the following equation:

$$DPURn_{WB} = F_U R_{WF}^{\alpha} \left(D_P^{\alpha} A_{BR} \right) M^{B_{BR}-1}$$

Where: D_P^{α} = 5.54 × 10^{-9} J Bq^{-1} is the potential α-energy per Bq activity of the short-lived radon daughters in secular equilibrium F_U is a unit conversion factor (3.6 × 10^{9} μGy h^{-1} per Gy s^{-1}); R_{wf}^{α} is the radiation weighting factor for α-radiation; and $A_{BR,}$ and B_{BR} are the base and the exponent of the allometric formulae for breathing rate, (4.83E-08 and -2.37E-01, respectively).

Vives i Batlle et al. (2008) derived DCCs for a range or organisms, here we use those presented for a 'small rodent' with an assumed mass of 21 g (Table 2). The radiation weighting factor used for α-energies to calculate the weighted DCC values in Table 2 was 20. An equilibrium factor (*F, i.e.* the ratio of the equilibrium equivalent concentration of radon to the actual radon concentration) of 1.0 was used. In reality this is higher than that observed in environmental measurements and a more realistic value is likely to be about half this value. The annual mean value of *F* in open air has been determined to be in the range: 0.4-0.6 (Keller et al. 1984; Wenbin et al. 1990). Changing the *F* value in the derivation of the DCC values of Vives i Battle et al. would result in a proportional change to the DCCs estimated by Vives i Batlle et al. (i.e. a value of *F* of 0.5 would result in DCC values of one half of those presented in Table 2).

Table 2. Weighted and unweighted DCC values for a rodent from Vives i Battle et al. (2008) (μGy h^{-1} per Bq m^{-3}).

Weighted			Unweighted		
Internal	**External**	**Total**	**Internal**	**External**	**Total**
8.69E-03	8.48E-04	8.86E-03	4.35E-04	8.48E-04	6.05E-04

For comparison dose rates as a consequence of other naturally occurring radionuclides and anthropogenic radionuclides detected in soil samples (see below) were calculated using the ERICA Tool (Brown et al. 2008). The ERICA Tool has a default radiation weighting factor for α-energies of 10. As this has previously been used to estimate background dose rates in the UK (Beresford et al. 2008) we have, for comparison, used a value of 10 in this assessment. As can be seen in Table 2 the weighted internal DCC is simply the unweighted value times 20 (the radiation weighting factor used by Vives i Batlle et al.). Consequently a DCC for ^{222}Rn can be easily modified for a different radiation weighting factor assumption. The weighted (total) DCC value we have used for the rodent assuming a radiation weighting factor of 10 and a value of *F* of 0.5 is 2.6x10^{-3} μGy h^{-1} per Bq m^{-3}.

3. RESULTS AND DISCUSSION

Radon-222 concentrations as determined from the detectors were reported in kBq m^{-3} h, these were converted into ^{222}Rn kBq m^{-3} by dividing by the total exposure time in the burrow. Mean ^{222}Rn concentrations as measured in the artificial burrows over the sampling period are presented in Figure 2. Based on the radon potential values (Table 1) it may have been expected that Sites 1, 6 and 7 would have the highest burrow ^{222}Rn concentrations. Whilst this was the case for Sites 1 and 7, Site 6 consistently had amongst the lowest concentrations, whereas Site 2 for some sample periods had higher concentrations than may have been anticipated. More data would be required before we could comment on the application of spatial databases of radon potential could be used to predict dose rates to burrowing animals.

As is apparent from the Figure 2 at a number of sites the lowest ^{222}Rn concentrations were observed during the winter. However, this was not consistent across all seven sites. The degree of within site variation in ^{222}Rn activity concentrations varied between sites with coefficients of variation ranging from <20 to >100 %.

Dose rates estimated using the approach described above are summarised across the measurement periods in Table 3. For comparison, McDonald and Laverock (1998) derived dose rates to burrow mammals for ^{222}Rn using a similar experimental protocol in Canada and estimated average dose rates to small burrowing mammals of approximately 80 μGy h^{-1}.

Soil activity concentrations of ^{232}Th series radionuclides and ^{40}K were similar to the UK average value reported in Beresford et al. (2008). Consequently for comparative purposes we will use the weighted wholebody dose rate we estimated in Beresford et al. for the ICRP (2008) Reference Rat geometry for ^{232}Th and ^{238}U series radionuclides and ^{40}K in the UK of 0.12 μGy h^{-1}. In addition the study area has received ^{137}Cs from the 1957 Windscale accident, weapons tests and the 1986 Chernobyl accident (Wright et al. 2003), consequently ^{137}Cs was measurable in soil samples from all the sites with activity concentrations ranging from 25 (Site 6) to 130 Bq kg^{-1} (Site 5) dry weight. The wholebody absorbed dose rates resulting from these ^{137}Cs soil activity concentrations estimated using the ERICA Tool ranges from 1.7x10^{-2} to 8.9 x10^{-2} μGy h^{-1}. The ^{222}Rn wholebody dose rate estimates presented in Table 3 are a 10 to 100 times higher than dose rates as a consequence of ^{137}Cs, ^{40}K and ^{232}Th and ^{238}U series radionuclides.

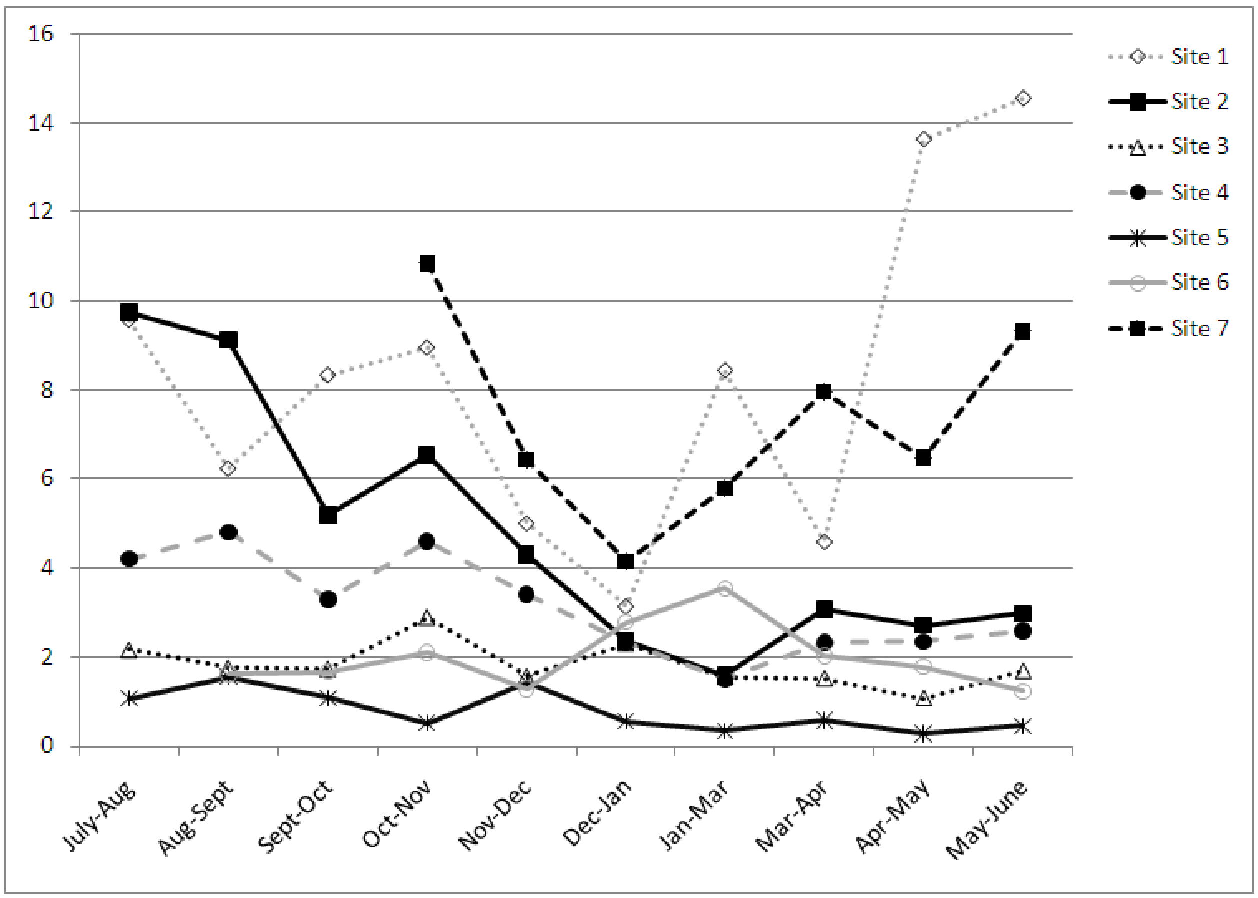

Figure 2. Arithmetic mean ^{222}Rn concentrations determined in the artificial burrows (kBq m^{-3}); sampling period was 2009-2010. Note result for site one is based upon one detector per measurement period; all other sites are mean of three measurements.

Table 3. Summarised dose rates (µGy h^{-1}) estimated for the rodent geometry of Vives i Batlle et al. (2008).

Site	Mean	Minimum	Maximum
Site 1	21.4	8.1	37.8
Site 2	12.4	4.1	25.3
Site 3	4.8	2.8	7.5
Site 4	8.2	3.9	12.5
Site 5	2.1	0.7	4.1
Site 6	5.2	3.2	9.2
Site 7	18.9	10.8	28.2

There are some uncertainties associated with the calculations presented here including the values of radiation weighting factor and *F* used (see above). The radiation weighting factor for α-energies used here is lower than that of 20 used by some models for wildlife assessment models. If this higher weighting factor were used then the dose rates estimates presented in Table 3 would double. Uncertainty in the value of *F* used may result in the estimated dose rates presented in being under or overestimated by a factor of two. We have adapted the DCC values presented by Vives i Batlle et al. (2008) for a rodent geometry. These authors also present values for a number of other animals (geometries), the range of those appropriate to burrowing mammals would result in ^{222}Rn dose rates of *circa* 30 % (larger mammal) to 120 % (smaller mammal) of those presented in Table 3.

Accepting these uncertainties it is likely that dose rates to burrowing mammals as a consequence of ^{222}Rn exposure are often likely to be in excess of some of the dose rates being suggested as no-effects levels. For instance, Garnier-Laplace et al. (2010) derived a generic predicted no effects dose rate for incremental (i.e. above background) exposure of 10 µGy h^{-1}. The ICRP (2008) proposed that the dose rate band in which effects could be expected to be observed for mammals is 4-40 µGy h^{-1}. There is a need to put such advised dose rates better into context with background dose rates, including exposure to ^{222}Rn.

It should be noted that few animals spend 100 % of their time underground and consequently they will not be subjected to the dose rates presented in Table 3 all of the time.

ACKNOWLEDGEMENTS

This work was co-funded by the Centre for Ecology & Hydrology and the Environment Agency of England and Wales. We are grateful to Fero Ibrahimi of the Health Protect Agency for analyses of the passive detectors and advice. We also thank Cathy Scheib and Dave Jones of the British Geological Survey for their help and advice, and Natural England and the Forestry Commission for access to three of the study sites used.

REFERENCES

Beresford, N.A., Barnett, C.L., Jones, D.G., Wood, M.D., Appleton, J.D., Breward, N., Copplestone D. 2008. Background exposure rates of terrestrial wildlife in England and Wales. J. Environ. Radioact., 99, 1430-1439.

Brown, J.E., Alfonso, B., Avila, R., Beresford, N.A., Copplestone, D., Pröhl, G., Ulanovsky A. 2008. The ERICA Tool. J. Environ. Radioact., 99, 1371-1383.

Garnier-Laplace, J., Della-Vedova, C., Andersson, P., Copplestone, D., Cailes, C., Beresford, N.A., Howard, B. J., Howe, P., Whitehouse, P. 2010. A multi-criteria weight of evidence approach for deriving ecological benchmarks for radioactive substances. J. Radiol. Prot., 30, 215-233.

Hosseini, A., Beresford N.A., Brown, J.E., Jones, D.G., Phaneuf, M., Thørring H., Yankovich, T. 2010. Background dose-rates to reference animals and plants arising from exposure to naturally occurring radionuclides in aquatic environments J. Radiol. Prot., 30, 235-264.

Ibrahimi, F., Miles, J. 2008. Performance review of a slide scanner based automated counting system for PADC radon detectors. Radiation Measurements, 43, S395-S400.

ICRP (International Commission on Radiation Protection). 2008. Environmental Protection: the Concept and Use of Reference Animals and Plants. Annals of the ICRP, 108. Oxford:Pergamon Press.

Keller, G., Folkerts, K.H., and Muth, H. 1984. Special aspects of the Rn-222 and daughter product concentrations in dwellings and in the open air. Rad. Prot. Dosim., 7, 151-154.

MacDonald, C.R., Laverock, M.J. 1998. Radiation exposure and dose to small mammals in radon-rich soils. Arch. Environ. Contam. Toxicol., 35, 109-120.

Miles, J.C.H., Appleton, J.D., Rees, D.M., Green, B.M.R., Adlam, K.A.M., Myers, A.H. 2007. Indicative atlas of radon in England and Wales. HPA-RPD-033. Chilton:Health Protection Agency.

Miles, J., Ibrahimi, F., Birch, K. 2009. Moisture-resistant passive radon detectors. J. Radiol Prot., 29, 269-271.

Vives i Batlle. Jones, S.R., Copplestone, D. 2008. Dosimetric approach for biota exposure to inhaled radon daughters. Environment Agency Science Report, SC060080. Bristol: Environment Agency.

Wenbin, M., Yan, J., Chanzhu, Q. and Weidao, Z. 1990. Investigation of Indoor and Outdoor Radon and its Daughters in the Region of Lianing, People's Republic of China. Rad. Prot. Dosim., 30, 37-40.

Wright, S.M., Smith, J.T., Beresford N.A. , Scott, W.A. 2003. Monte-Carlo prediction of changes in areas of west Cumbria requiring restrictions on sheep following the Chernobyl accident. Radiat. Environ. Biophys., 42, 41-47.