APPENDIX M

CONTENTS

APPENDIX M1

Verification of radiation related figures in the Draft EIS

M1.1 AIM

The aim of this technical report is to provide additional information to complement the information provided in the Olympic Dam Supplementary EIS.

The report is in four parts, covering the following aspects:

M1.2 Additional information on radiation estimates for the open pit

M1.3 Additional information on radon estimates for the tailings storage facility

M1.4 Additional radiation studies

M1.5 Assumptions in estimating environmental radon concentrations.

M1.2 ADDITIONAL SUPPORT INFORMATION ON RADIATION ESTIMATES FOR THE OPEN PIT

The dose assessment for the open pit is based on a number of assumptions and calculations. The core assumptions were as follows:

- ore uranium grade – 500 ppm
- mineralised basement (non-ore) uranium grade – 70 ppm
- sedimentary cover uranium grade – 5 ppm
- average uranium grade of entire pit (40 years) – 100 ppm
- average uranium grade of all basement material (including ore) – 200 ppm
- open pit dimensions – diameter 4 km and depth 1.2 km nominal size (a larger diameter would result in no significant change in conclusions), volume approximately 6 km^3.

M1.2.1 GAMMA DOSE TO MINERS

Gamma dose rates were estimated using the formula of Thomson and Wilson (1980), which quotes 65 μSv/h for each per cent of uranium grade (i.e. 65 μSv/h/%U). The highest doses would be expected from a worker spending the whole working year (2,000 hours) on ore with a uranium grade of 500 ppm or 0.05%. For this case the dose rate would be 3.25 μSv/h, and the annual dose (2,000 hours per year) would be 6.5 mSv.

In practice, it would be expected that the actual gamma dose rates in the proposed Olympic Dam pit would be much lower than observed in other mines with higher ore grades. For example, at Ranger uranium mine in the Northern Territory the ore grade is almost five times higher than the Olympic Dam grade, however the reported maximum annual gamma doses at Ranger are approximately 4 mSv (refer Table S2, Appendix S of the Draft EIS).

In addition, the current underground workers at Olympic Dam, such as production drillers, raise drillers and production charge-up crews, receive an average dose of approximately 3 mSv/y from gamma radiation even though they spend most of their work time in ore-bearing areas.

It would be expected that because of the different exposure geometry (that is, the workers in underground mines are surrounded by ore, while in an open pit, the workers are generally exposed from only one direction), open pit workers would receive lower exposures than underground workers.

Thus the calculated exposure is conservative when compared to actual measurements.

From a practical perspective, another important factor is that many workers would be inside heavy mining equipment which provides some shielding from gamma radiation, resulting in lower measured gamma radiation rates than those calculated.

Based on these factors, a realistic conservative estimate of the average gamma dose rate for pit miners was made and these were provided in Table S2 of Appendix S of the Draft EIS.

M1.2.2 PIT DUST LEVELS AND POTENTIAL DOSES FROM DUST

Dust exposure estimates for open pit workers were based on dust concentrations observed in other open pit uranium mines. This provided the estimate of the average radiation dose of 0.1 mSv/y from dust. Conservatively, it was recognised that under certain low ventilation conditions the pit could become very dusty, and the probable maximum dust concentration was estimated from the results of dust monitoring in open cut coal mines. The 99th percentile dust concentration was 3 mg/m^3 (refer Section 2.2.3 of Appendix S of the Draft EIS) and this was adopted as the probable maximum concentration.

Doses were then determined through the standard dose conversion method outlined in ARPANSA (2005).

Calculating the potential dose from a dust concentration is straightforward, and an example is as follows:

- for 500 ppm ore, the activity of each radionuclide is 6.2 Bq/g (the specific activity of pure uranium-238 is 12,400 Bq/g, and 500 ppm equates to 0.05% of 12,400, giving 6.2 Bq/g for each radionuclide)
- the intake is calculated by assuming a standard breathing rate of 1.2 m^3/h for 2,000 hours (2,400 m^3 per year) (Draft EIS, Appendix S), which results in an annual intake of 45 Bq of each radionuclide (assuming a dust concentration of 3 mg/m^3 for 500 ppm uranium in ore)
- for an annual intake of 45 Bq for each radionuclide, there would be a total of 215 alpha disintegrations per second (dps) for the five long-lived alpha-emitting radionuclides
- using the conversion factor 7.2 μSv/alpha dps (ARPANSA 2005), the resulting annual dose is 1.6 mSv.

Additional conservative assumptions in these calculations include the following:

- an implied activity median aerodynamic diameter (AMAD) of 1 μm, which is a conservative size classification
- 'S' (slow) lung absorption class used for each radionuclide, which is the most restrictive class with the exception of Th-230 (the S class for Th-230 is justified because the ore is held within an insoluble matrix and therefore it is the solubility of this matrix rather than the solubility of the isotope that is likely to determine the retention in the lung)
- no allowance for any reduction in dust inhalation due to respiratory protection (either personal or from filtration in cab air-conditioning).

The radiological hazard from occupational and environmental exposure to radioactive dust is low and one of the reasons is that the uranium ore grade is relatively low.

M1.2.3 RADON IN THE PIT

Radon emanation from a solid depends on its surface area. This means that emanation rates are based on the area expressed as m^2. The primary radon parameters used in the Draft EIS were the emanation rate for ore and broken ore as follows:

- ore – 2.5 Bq/m^2/s (for a uranium grade of 500 ppm)
- broken ore – five times the unbroken ore level.

The Draft EIS assumed that uranium ore grade and radon emanation rate were directly proportional.

The assessment of radon in the pit is based on a number of factors, including measured emanation rates from Olympic Dam, theoretical emanation rates, estimates of modelled and measured pit ventilation rates and the effects of atmospheric conditions that might affect radon levels. These are explained below.

Measurements of radon emanation

Previous measurements of radon emanation at Olympic Dam show emanation rates as follows:

- 1–3 Bq/m^2/s for ore (Sonter et al. 2002)
- 2.5 Bq/m^2/s for ore, and 0.52 Bq/m^2/s for 'underground tunnels' (in non-ore) (BHP Billiton operational radiation monitoring results)
- 0.75 Bq/m^2/s as an average of all underground surfaces (ore and non-ore) (Akber et al. 2001).

When ore is broken by blasting or crushing (or both) more fissures will be introduced, and this would be expected to increase the rate of diffusion and hence the emanation rate. However, there appears to be a lack of general measurements in the literature on the emanation of radon from broken ore.

It is difficult to measure the radon emanation rate from broken rock. Measurement usually involves fixing an airtight container to the sampling area to collect any radon that emanates from the sampling area. If a small sampling area (and therefore a small container) is used then it is often only effectively 'sealed' onto one rock, and so collects radon only from that. If a large area sampler is used, it is difficult to completely seal the sampler to the matrix of broken rock, and radon may leak out between the interstices. Both of these problems are likely to lead to an underestimation of the true emanation rate.

Theoretical estimates of radon emanation

Measurements are available for the radon release from unbroken ore but, as noted, there is a lack of data for broken ore, or for broken waste rock. Therefore a theoretical assessment of radon emanation from broken rock was used. The method is described as follows.

There are three basic factors determining the rate of radon release (emanation rate) from the surface of some material:

- the rate of generation of radon in the material (radon generation)
- the fraction of radon generated that escapes from the mineral grain into the interstitial spaces of the rock or ore (emanation coefficient)
- the fraction of the radon that is able to diffuse through the interstitial spaces to the surface (diffusion coefficient).

For broken ore, the factors were calculated as follows.

Radon generation

The radon generated in a material is directly related to the radium content. If the radium activity is A_{Ra}, then A_{Ra} atoms of radon will be formed each second. The number of atoms of radon (N_{Rn}) is related to the activity of radon (A_{Rn}) by the standard decay equation $A= \lambda N$, where λ is the decay constant for radon (2.1×10^{6} /s). Therefore the rate of radon production (in Bq/s) is equal to the radium activity (in Bq) multiplied by 2.1×10^{-6}.

The radium content of the material can be determined from the uranium grade, with the activity of radium-226 in the material being equal to the activity of uranium-238. The specific activity of uranium-238 is 12.4 MBq/kg, and therefore 100 ppm ore has 1,240 Bq/kg of radium-226.

Radon escape into pore space (known as the emanation coefficient (ε))

Radon is formed in the mineral grain that contains the radium. The diffusion of radon out of the solid mineral grain is extremely slow, and so only those atoms of radon formed very close to the grain surface are likely to escape.

The fraction of radon getting out of the mineral grain into the pore space is referred to as the emanation coefficient. Mudd (2008) reports 0.1 to 0.3 for waste rock and low-grade ore. A typical value for soil is approximately 0.2.

Diffusion through pore spaces (known as the diffusion coefficient)

Once the radon is out of the mineral grain, it then diffuses through the interstices of the host material.

The rate of diffusion in a material is referred to as the diffusion coefficient D_e (in m^2/s). Diffusion coefficients for soils are generally about 10^{-6} while rock (non-porous) is about 10^{-7}. Concrete can be about 10^{-9}, while saturated mud can be about 10^{-10}.

The radon diffusion coefficient in free air (D_0) is 1.1×10^{-5}, and this can be taken as an absolute upper limit on diffusion through a porous medium.

Determining the amount of radon emanating from a material (radon emanation rate)

The emanation rate at the surface of a material depends on the radon generation rate (proportional to radium concentration or ore grade), emanation coefficient (ε) and the diffusion coefficient (D).

The rate of radon emanation (J) from a material (Mason et al. 1982; Yu et al. 1993; UNSCEAR 2000) can be written as:

$J = \varepsilon \lambda \rho R (D/\lambda)1/2$

Where:

ε is the emanation coefficient

λ is the (radon) decay constant

ρ is the density of the material

R is the radium content of the material, and

D is the diffusion coefficient in the medium.

Typical values of emanation coefficients (ε) are about 20%. If this value is substituted into the above equation, using the experimentally determined emanation rate of 2.5 $Bq/m^2/s$ for (unbroken) Olympic Dam ore, then the resulting diffusion coefficients are 1.2×10^{-7}. Breaking the ore cannot give a diffusion constant of greater than 1.1×10^{-5} (the value in free air). The worst-case increase in diffusion coefficient is thus less than a factor of 100 (from 1.2×10^{-7} to 1.1×10^{-5}).

That is, the increase in diffusion coefficient must be somewhere between 1 (no increase: radon diffuses at the same rate as unbroken rock) and 100 (radon diffuses through broken rock at the same rate that it diffuses through free air). As the emanation rate depends on the square root of the diffusion coefficient ($D^{1/2}$), an increase in diffusion coefficient of a factor of 100 corresponds to an increase in emanation rate of 10. Possible increases thus range from a factor of 1 (no increase) to 10 (diffusion as in free air). A figure of 5 was selected as a reasonable representation in the increase in emanation rate from broken ore, compared to unbroken ore.

Radon concentrations in the proposed pit (under normal ventilation conditions)

To calculate the subsequent radon concentrations, it is necessary to estimate the time that air spends in the pit.

The ventilation rate under normal (non-inversion) conditions was obtained from an expression developed by Thompson (1994).

$\tau = 33.8\ (V/U_rLW)*(0.7\cos(\theta) + 0.3)$

Here, τ is the residence time (s), V is the pit volume (m^3), Ur is the wind velocity (m/s), L and W are the pit length and width (m), and θ is the angle of the wind to the pit long axis. The pit was assumed to be circular, so the cos (θ) term was set to 1. This expression generated ventilation rates (that is, the number of times the air in the pit changes) in the range 0.028 to 2.8 air changes/hour for wind speeds from 0.1 to 10 m/s.

The equilibrium radon concentration was then calculated as follows:

$[Rn] = E_R / (PV \times VR)$

Where:

[Rn] is the radon concentration

E_R the radon generation rate (Bq/h)

PV is the pit volume

VR is the number of air changes per hour.

Resulting radon concentrations ranged from 1.76 to 176 Bq/m^3 for wind speeds from 10 m/s down to 0.1 m/s.

The average wind speed at Olympic Dam is approximately 3.7 m/s (Kinhill 1997). A conservative wind speed of 2 m/s was adopted for this calculation, giving a radon concentration in the pit of 8.8 Bq/m^3.

Radon concentrations – atmospheric inversion conditions

During cooler nights, stable conditions can result in temperature inversions (refer Section 8.3.5 of the Draft EIS for reference to inversions).

Calculating radon concentrations under inversion conditions requires information on the radon emanation rate, the height of the inversion above the emanating surface, and the length of time the inversion persists. To assess potential annual doses, the number of inversions occurring each year is also required.

It was assumed that the inversion forms at a height of 100 m above the pit floor. Inversions at this height, or lower, are common on the surface, however there was little information on the properties of inversions within the pit at the time of writing the Draft EIS.

The air quality modelling (Chapter 13 and Appendix L of the Draft EIS) indicated that when inversions formed on the surface, they may penetrate 200 to 350 m into the pit, which means they would be approximately 800 m above the pit base. The modelling was limited to 400 m below the surface.

Recent work by Hibberd (2010), as outlined in Section 26.2.1 of the Supplementary EIS, indicate that the assumption that inversions occur at 100 m above the pit floor is likely to be conservative.

The area that is emanating radon into the volume below the inversion is the area of the base, and the walls to 100 m height. The base and walls were assumed to be entirely ore, with an emanation rate of 2.5 Bq/m2/s. While some ore will be broken, and thus have a higher emanation rate, some of the base and walls is also likely to be barren material and have a lower emanation rate; therefore, an average of 2.5 $Bq/m^2/s$ was used.

The combination of the floor area and inversion height define a volume into which radon is being introduced from the floor, and the walls below the inversion. The radon concentration [Rn] after time T will thus be:

$[Rn] = P \times T / (A \times H)$

Where:

P is the radon production rate (Bq/m^2)

A the area of the base (m^2)

H the inversion height (m).

For the Draft EIS assessment, it was assumed that the inversion persists for 12 hours, from just after sunset to just after dawn.

After 12 hours, the radon concentration would reach approximately 1,700 Bq/m^3, corresponding to a radon decay product concentration (assuming equilibrium) of 9.3 μJ/m^3. Therefore, the average concentration during the 12-hour period would be 850 Bq/m^3 (radon) and 4.6 uJ/m^3 (radon decay products).

To determine the frequency of inversions in the pit, surface inversion data were used. One approach was to use the frequency of very stable atmospheric conditions (Pasquil stability class F) which occur on 26% or 95 nights in the year (calculated from Table 8.5 in Section 8.5 of the Draft EIS). It was then assumed that the proportion of surface inversions would be the same as the proportion of inversions occurring in the pit. This is almost certainly an overestimate, as there are a number of factors, such as geothermal heating, heat from heavy machinery and differential solar heating of the pit walls, all of which are likely to promote convection or air movement within the pit.

Dose assessment for pit workers

In this calculation (under both the normal and atmospheric inversion conditions) it is assumed that the radon decay products were in equilibrium with radon. This is the most conservative assumption.

The dose conversion factor recommended in ARPANSA 2005 (1.4 mSv per mJ.h/m^3) is then used to calculate the dose from 2,000 hours exposure to the average radon decay product concentration. The calculated dose is 0.14 mSv/y.

In addition to exposure from normal conditions, workers would be exposed to higher concentrations during inversions. The average radon decay product concentration during inversion conditions, over the 12-hour period, was calculated to be 4.6 μJ/m^3. Using the dose conversion factor of 1.4 mSv per mJ.h/m^3 recommended in ARPANSA 2005, the dose resulting from this exposure is 78 μSv per 12-hour shift.

A mine worker would not be likely to be present on all nights when inversions occurred. Typically, on a standard roster, they would work one night in four. Thus, a worker would be exposed to inversion conditions on about 25 nights a year, and receive a total dose of about 1.8 mSv from nights worked in these atmospheric conditions.

The total annual dose was therefore calculated to be approximately 2 mSv, consisting of 1.8 mSv during inversions and 0.14 mSv from 'normal' ventilation conditions.

Note that there is some minor overlap. Doses have been calculated for a complete year for routine ventilation but the inversion contribution has been calculated in effect for an additional 26% of nights. Against that, the build-up of radon decay product concentration during inversions has been calculated from zero, whereas it would typically start from the average normal ventilation concentration. These corrections will be small and their net effect even smaller. It should also be noted that it was assumed that all workers are working within 100 m of the pit base during the inversion conditions (which would not be the case in practice). In addition, it should be noted that no allowance has been made for the use of respiratory protection, meaning that actual doses would be much lower than predicted.

M1.3 ADDITIONAL INFORMATION ON RADON ESTIMATES FOR THE TAILINGS STORAGE FACILITY (TSF)

Table S5 in Appendix S of the Draft EIS indicated the emanation rate for radon from the expanded tailings would be 0.5 Bq/m^2/s. Based on this emanation rate and the exposed surface area of the tailings, the amount of radon emanating from the tailings was calculated to be 13.9 MBq/s for the completed TSF system. The emanation would peak at this level as TSF cells were progressively added until the facility reached its final design size.

Additional information on the emanation rate for radon from the TSF is provided here.

The primary source of the emanation rate used in the Draft EIS is based on actual measurements from the work of Akber et al.(2001). The work notes that the arithmetic mean of radon emanation from tailings was 0.48 Bq/m^2/s; the arithmetic mean is the figure that would be used to represent the average emanation rate for the whole surface. For the purposes of the Draft EIS estimates, this figure was then rounded up to 0.5 Bq/m^2/s and used as the basis for the expansion calculations.

Akber et al. (2001) noted that during sampling it is important to prevent the release of radon that is effectively trapped in permeable layers beneath the surface, which would lead to an overestimate of the radon emanation rate. They provided evidence of this effect occurring by comparing results of measurements where the sampling device (an activated charcoal can) was pushed into the tailings. This resulted in a significant 'burst' of radon from lower levels within the tailings, leading to an increase in measured emanation rates by a factor of six. This situation is not representative of long-term emanation from the tailings, however these figures were conservatively used in calculating the final averages.

Akber et al. (2001) also reviewed other earlier radon emanation rate monitoring results from Olympic Dam tailings, which gave results with an arithmetic mean of approximately 0.5 Bq/m^2/s. In particular, they noted the results of extensive sampling from December 1993 to February 1994, in which the arithmetic mean of the sampling results was 0.53 Bq/m^2/s and broader sampling from June 1993 to May 1994 (reported in 1994 Environmental Radiation Report) which reported a mean of 0.44 Bq/m^2/s.

In Olympic Dam Operation (ODO) 1996, results of radon emanation measurements are provided for the period 1988 to 1996, with the mean of the results being reported as 1.12 Bq/m^2/s. Akber and others also note that ODO 1997 uses a figure of 1.27 Bq/m^2/s.

The earlier measured results (before 1994) are higher than the later results (after 1994), because the ore grade has declined over the life of the Olympic Dam mine. BHP Billiton metallurgical plant records show that the average uranium ore grade until the end of 1993 was 1,110 ppm. After this period, the average dropped to 690 ppm.

Supporting evidence showing the reduction in ore grade on the radionuclide composition of the process material can also be seen directly in the results of routine radionuclide monitoring. Table 1 shows the concentration of radium-226 (in Bq/g) in the process feed material.

Table 1: Change in ^{226}Ra levels in process feed material with time

Sampling period	^{226}Ra concentration (Bq/g)
March 1989	15
November 1993	15
February 1994	6.9–8.2
May 1994	7.5
March 1995	6.8

Comparison of the emanation rates between mines is problematic. Mudd (2008) and Sonter et al. (2002) noted that factors such as Ra-226 characteristics, moisture content and type of tailings (i.e. acid, alkaline or neutral) need to be considered. In these cases, direct measurements of emanation provide the most appropriate results (Mudd 2008).

During 2010, additional radon emanation from tailings monitoring from the existing Olympic Dam TSF was undertaken by BHP Billiton radiation safety personnel using charcoal cups. A total of 31 samples were obtained, giving an average of 0.3 Bq/m^2/s. This figure is consistent with the Draft EIS figure and confirms that a figure of 0.5 Bq/m^2/s for radon emanation from tailings is appropriate.

M1.4 ADDITIONAL RADIATION STUDIES

The following additional radiation studies have been conducted since the submission of the Draft EIS.

M1.4.1 GAMMA RADIATION MEASUREMENTS

The Draft EIS provided estimates of gamma radiation from the side of a rail wagon containing copper concentrate and were based on first principle estimates. Additional work was conducted during the Supplementary EIS to verify these estimates and involved obtaining a bulk sample of copper concentrate and performing gamma radiation measurements.

The gamma measurements were undertaken by BHP Billiton radiation safety personnel on a bulk sample of approximately 20 tonnes of material, which simulates the contents of a rail wagon. The estimated levels (from Appendix E, section E4.10.2 of the Draft EIS) and the measured levels can be seen in Table 2.

Table 2: Comparison of estimated and measured gamma rates from copper concentrate

	Gamma rate (estimated) μSv/h	Gamma rate (measured) μSv/h
Level at 0.5 m	5	1.5
Level at 5 m	0.8	0.7

The results show that the estimated gamma levels used in the Draft EIS were conservative.

M1.4.2 RADON EMANATION MEASUREMENTS

In Appendix E (section E4.10.2) of the Draft EIS, an estimate of radon release from copper concentrate is provided and is based on the assumption that all radon contained in the concentrate is released during loading operations. It was recognised that this is a significant overestimate because much of the radon would remain within the grains of the concentrate. However, in the absence of analytical data, this conservative radon release estimate was used in the Draft EIS to estimate potential doses from radon decay products to residents of Darwin EIS.

To better quantify the release of radon from copper concentrate, a sample was tested in the South Australian Environment Protection Authority radon chamber in 2010. The aim of the work was to better quantify the radon release levels, which would provide more accurate information for dose assessment from exposure to radon decay products. The work was undertaken by BHP Billiton radiation safety personnel.

The results of the work show that less than 5% of the total contained radon is released from the copper concentrate.

Consequently, doses from exposure to radon decay products, from radon emanating from the copper concentrate, for the residents of Darwin is estimated to be one twentieth of the levels provided in the Draft EIS.

M1.4.3 RADON EMANATION FROM SOUTH AUSTRALIA

In Chapter 26 of the Supplementary EIS, an estimate of radon emanation from the whole of South Australia is provided. This estimate is based on broad assumptions, as follows:

- the total surface area of South Australia is approximately 1,000,000 km^2
- the average radon emanation rate for the state was taken to be 0.01 Bq/m2/s, based on pre-operational monitoring at Olympic Dam, which is considered to be conservative
- the total amount of radon emanation from the expanded project is estimated to be 100 MBq/s.
- The total amount of radon emanation from the state is approximately 10,000 MBq/s

Based on these figures, the expanded Olympic Dam operation would contribute about 1% to the current natural emanation levels in South Australia.

M1.5 ASSUMPTIONS IN ESTIMATING ENVIRONMENTAL RADON CONCENTRATIONS

In the Draft EIS, environmental radon concentrations arising from the proposed operation were determined from the air quality modelling (see Appendix S, section S2.5.2 and S3.1).

The basic radon-related input information to the modelling is in Table 3 and 4.

Table 3: Assumptions used in the estimates of point source radon emissions

Details	Radon emanation rate ($Bq/m^2/s$)	Uranium grade (ppm)	Reference/Notes
Ore	2.5	500	Sonter et al. (2002), Akber (2001)
Broken ore	12.5	500	Five times broken rock
Waste rock	0.25	50	Proportional to uranium grade
Broken waste rock	1.25	50	Five times broken rock
Haul roads	0	0	Assuming haul roads constructed of limestone
Tailings	0.5	NA	Akber et al. (2001)

Table 4: Assumptions used in the estimate of radon emissions from broad area sources

Source	Radon emanation (TBq/y)	Assumptions (Note that all emanation based on Akber et al. (2001) unless indicated otherwise)
Underground exhaust air	105	Assumes underground operations continue at current rate, with a 10% increase to account for additional development
Open pit emanation	124	Emanation from surface area of exposed ore
	146	Emanation from surface area of exposed non-ore (overburden, non-mineralised rock)
Processing ore stockpile	14	Assumes 3 x 100,000-tonne ore stockpiles with a 0.4 x volume factor for broken rock
ROM ore stockpile	26	Based on a 240 m x 240 m x 20 m stockpile
Waste rock stockpile	2,601	Based on estimated surface area of RSF at Year 40
Concentrators	29	Assumes combined existing and proposed concentrators
Tailings storage facility	520	Assumes emanation rate based on Akber et al. (2001) and TSF cell configuration as per Draft EIS, with existing cells, 10-hectare total supernatant area, and walls built of waste rock.

The revised radon contour plot is shown in Figure 1.

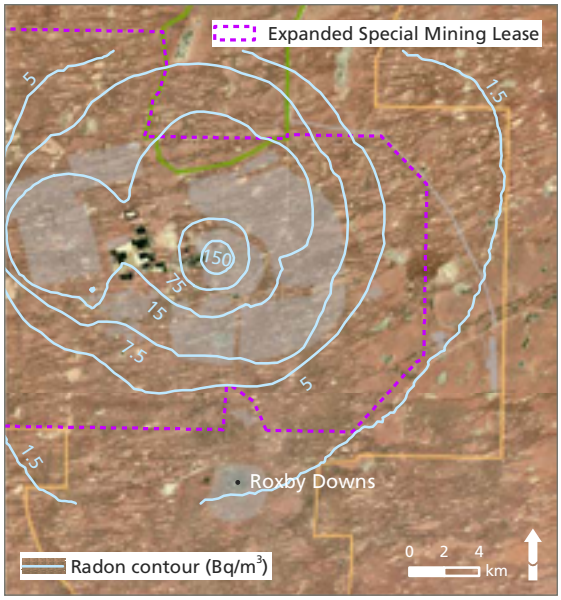

Figure 1 Contour plot of revised modelled radon concentrations (above natural background levels)

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APPENDIX M2

Initial geochemistry survey: East Arm Wharf and Alice Springs, Northern Territory

Olympic Dam EIS Port of Darwin

BHP Billiton

21 September 2010

Final Report: Initial Geochemistry Survey

East Arm Wharf and Alice Springs, Northern Territory

Final Report: Initial Geochemistry Survey

East Arm Wharf and Alice Springs, Northern Territory

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Table of Contents

1.0 Introduction

BHP Billiton is proposing to expand the existing Olympic Dam copper, uranium, gold and silver mine and processing plant, including associated infrastructure at the Olympic Dam site in South Australia.

The Darwin Transport Option proposes to transport copper concentrate by rail to Darwin and load ore from a dedicated facility onto concentrate ships for export.

AECOM Australia Pty Ltd (AECOM) was commissioned by BHP Billiton to undertake an initial geochemical sampling program of marine sediment and airborne dust at East Arm Wharf and Darwin Harbour, and radon monitoring at East Arm Wharf and Alice Springs. This report on the Initial Geochemistry Survey presents the results of the sediment and air sampling in relation to national guidelines, as well as a description of the approach taken for each component.

1.1 Scope of Works

The Scope of Works of the Initial Geochemistry Survey is as per the Proposal for Consulting Services “Baseline Contaminant Study- Port of Darwin Transport Open” dated 25 May 2009 and the Briefing on the Project dated 11 August 2009.

The Scope of works was originally defined as:

- The collection of three sediment samples from the load out facility at the East Arm Wharf and three samples nearby in Darwin Harbour, at three depths (0.0-0.3, 0.5 and 1.0 m) per sample location (i.e. total of 18 samples).
- The laboratory analysis of sediment samples for heavy metals, including total uranium (U) and thorium (Th) and long-lived radionuclides, specifically radium-226 (Ra-226), lead-210 (Pb-210), thorium-230 (Th-230) and polonium-210 (Po210).
- The collection of airborne dust samples from the East Arm Wharf area for a period of three months, using a High Volume Air Sampler (HVAS).
- The laboratory analysis of dust samples for heavy metals, including total U and Th and long-lived radionuclides, specifically, Ra-226, Pb-210, Th-230 and Po-210.
- The use of track etch detectors (TEDs) to measure average radon concentrations, at East Arm Wharf and at six sites in Alice Springs, as close to the railway line as possible.
- The review of existing relevant data, preparation of a draft, interim final and final report.

Changes to the scope as a result of unexpected or adverse field conditions are as follows:

- The sampling method for the seabed sediments was modified during the program as result of the sediments being very soft and loose along the berth of East Arm Wharf. A grab sampler was used instead of the piston sampler as originally proposed. The grab sampler could only retrieve surface sediments, which resulted in the sampling program being modified to include only 18 surface samples and no depth samples. The sample sites were distributed across the western face of the East Arm Wharf and behind the sand island to the south of the wharf.
- A total of 31 attempts were made to retrieve suitable sediment samples. Due to high tides scouring parts of the sea-floor bare of sediments, or the presence of corals/rocky substrates, only 16 successful samples were retrieved (including the QA/QC).
- After an assessment of the distribution and field texture analysis of the sediment in each sample location, it was determined that 13 samples (including QA/QC) would be analysed for heavy metals, as they would be representative of the surroundings, and 3 samples would be held by the laboratory to be analysed at request, if considered applicable. A total of six samples (including the QA/QC) were analysed in the laboratory for radionuclides, with one sample approximately at either end of the berth at East Arm Wharf, which would be representative of the radionuclide concentrations for the general wharf area, and three located at South Shell Island, which was identified to be representative of the background radionuclide concentrations in sediments. The remaining 10 samples were held for possible radionuclide analysis if considered applicable.
- The HVAS was installed for a total of 107 days from 31 July 2009 to 17 November 2009, with a total of 343.3 sampling hours.

Sediment samples SED_EA_02 and its QA/QC duplicate SED_EA_SPLIT were reanalysed due to unacceptable Relative Percentage Differences, as discussed in Section 2.3

2.0 Methodology

2.1 Airborne Sampling

Airborne sampling was conducted in accordance with the following guidelines:

- AS/NZS 3580.1.1: 2007. *Methods for Sampling and Analysis of Ambient Air*.
- AS/NZS 3580.9.3: 2003. *Methods for Sampling and Analysis of Ambient Air. Method 9.3: Determination of suspended particulate matter – Total suspended particulate matter (TSP) – High volume sampler gravimetric method*.

2.1.1 East Arm Wharf

AECOM collected two airborne dust samples using a HVAS installed within the Darwin Port Corporation facility at East Arm Wharf, Berrimah, shown in **Plate P-1**. The location of the HVAS is presented in Figure F1 (in the Figures Section).

The HVAS was placed on site on 30 July 2009. An issue was immediately identified with the HVAS equipment, with power failures and spikes in supply causing the equipment to switch off automatically. Over a period of 2 months, recurrent power failures resulted in a total sample run time of just over 180 hours (out of a possible 1392 hours).

The original HVAS equipment was replaced with an alternative unit on 25 September 2009, which over the course of the following 2 months, due to further frequent, periodic power outages, recorded the balance of the sample run hours, 159.1 hours out of a possible 1272 hours.

Over a period of 107 days, the total sampling hours were 343.3. Calibration of the HVAS flow rate was undertaken approximately fortnightly.

The first batch of filter papers was sent to the laboratory (Australian Radiation Services) for analysis after 184.2 sample hours at 70 m^3/hr (12,894 m^3 of air sampled). The second batch was sent after 159.1 sample hours at 70 m^3/hr (11,137 m^3 of air sampled).

Table 1 outlines the analytical suite for the airborne sampling undertaken using the HVAS at East Arm Wharf.

Table 1: Analytical Suite for Airborne Sampling

Sample ID	Date	Run Hours	Analysis	
			Metals[1]	Radionuclides[2]
Month01 (Sample 1)	31/07/09 – 13/10/09	184.2	✓	✓
Month02 (Sample 2)	13/10/09 – 17/11/09	159.1	✓	✓

Notes:

1. Metals: Sb, As, Ba, Be, Cd, Ca, Cr, Co, Cu, La, Pb, Mn, Hg, Ni, Ag, Th, U, V, Zn
2. Th-234, Th-230, Ra-226, Pb-210, Po-210

Radon was monitored using track-etch detectors (TED) according to AS/NZS 3580.1.1:2007. All detectors were sent to Australian Radiation Services for analysis of radon levels.

AECOM installed six TEDs at East Arm Wharf (**Plate P-2**) and one background sample in the AECOM Darwin office located at 17-19 Lindsay Street, Darwin from 31 July 2009 to 6 November 2009. The locations of the TEDs in Darwin and East Arm Wharf are presented in

Figure F1 (in the Figures Section).

2.1.2 Alice Springs

AECOM installed six TEDs in Alice Springs and one background sample was collected approximately 9 km south-east of the Alice Springs Town Centre for the period between 21 August 2009 and 20 November 2009. The locations of the TEDs in Alice Springs are presented in **Figure F2** (in the Figures Section).

2.2 Sediment Samples

Sediment sampling at East Arm Wharf was conducted on 24 and 25 of August 2009 in accordance with the *Handbook for Sediment Quality Assessment* (Simpson *et al*., 2005) and the *National Assessment Guidelines for Dredging* (Commonwealth of Australia, 2009).

Sampling was undertaken on the lowest tides for the month, with a 0.68 metre high tide on 24 August and a 0.88 metre high tide on 25 August. Depth to sediment on the sampling days was between approximately 6 to 10m.

A Dormer Piston Sampler was initially deployed with the intention of obtaining three samples at three different locations across the western face of East Arm Wharf, and three samples at three different locations running parallel to North Shell Island, located approximately 350 m south-east of the wharf.

After deployment at the first intended location, it was identified that there was little to no sediments located in the area of sampling, as a result of maintenance dredging programs, and indicated that only a thin layer of recent deposited sediment was present in the nominated sampling areas.

As the surface sediment layer provides information on the most recently deposited sediment materials, and can be used to determine the spatial variation in sediment properties and the distribution of potential analytes of interest (metals and LLNR's) (Simpson *et al.,* 2005), the program was developed that using a grab sampler would still provide adequately representative samples.

The number of samples, and sample locations in the original scope of work was modified to reflect the change in sampling methodology.

A total of 16 sediment samples (including QA/QC) were obtained. The samples were placed in a clean (decontaminated) container, photographed, and sub-samples were placed in plastic, labelled, zip-lock bags and stored on ice for shipment to ship to the laboratory. The locations of the sediment samples taken from the East Arm area are presented in **Figure F1** (in the Figures Section).

Table 2 outlines the analytical suite for sediment samples obtained at East Arm Wharf.

Table 2: Analytical Suite for Sediment Samples

Sample ID	Date	Time	Analysis	
			Metals[1]	Radionuclides[2]
SED_EA_01	24/08/2009	1523	H	H
SED_EA_02	24/08/2009	1610	✓	✓
SED_EA_SPLIT	24/08/2009	1610	✓	✓
SED_EA_03	24/08/2009	1624	✓	H
SED_EA_04	24/08/2009	1635	H	H
SED_EA_05	24/08/2009	1646	✓	✓
SED_EA_06	24/08/2009	1705	✓	H
SED_EA_07	25/08/2009	1343	✓	H
SED_EA_08	25/08/2009	1354	✓	H
SED_EA_10	25/08/2009	1401	✓	H
SED_EA_11	25/08/2009	1420	✓	H
SED_EA_14	25/08/2009	1517	✓	H
SED_EA_18	25/08/2009	1650	H	H
SED_EA_19	25/08/2009	1610	✓	✓
SED_EA_20	25/08/2009	1550	✓	✓
SED_EA_21	25/08/2009	1556	✓	✓

Notes:

'H' indicates samples held by laboratory for possible future analysis.

1. Metals: Sb, As, Ba, Be, Cd, Ca, Cr, Co, Cu, La, Hg, Mn, Ni, Pb, Ag, Th, U, V, Zn
2. Th-234, Th-230, Ra-226, Pb-210, Po-210

In summary, 31 attempts produced 16 primary sediment samples, including 1 QA/QC sample. Metals analysis was carried out on 13 samples, including the QA/QC sample, and analysis for radionuclides was carried out on 6 samples, including the QA/QC sample.

2.3 QA/QC

All works completed as part of the Initial Geochemistry Survey at East Arm Wharf and Alice Springs were conducted in accordance with standard AECOM environmental sampling protocols. All fieldwork was performed under the direct supervision of the AECOM Fieldwork Coordinator/Project Manager. All AECOM fieldwork personnel responsible for undertaking the fieldwork program had been trained on previous projects and were well experienced in the relevant fieldwork protocols.

The essential elements of the field QA/QC program are:

- The use of experienced personnel: Field work was completed by trained AECOM engineers/scientists with previous experience in contaminated site assessment, field sampling techniques and health and safety issues
- Record Keeping: Full records of all field activities were maintained on standard field logging sheets
- Sample Collection: New nitrile gloves were worn during sediment sampling, and replaced between each sample collected. All samples were collected in general accordance with the Handbook for Sediment Quality Assessment (Simpson *et al*., 2005) and the National Ocean Disposal Guidelines for Dredged Material (Environment Australia, 2009)
- Sample Labelling: A unique sample number was used for each sample to clearly specify the sample origin, preservation standards and analytical requirements.
- Chain of Custody: Chain of Custody procedures were required for all sample transfers. Chain of Custody sheets listed sample numbers; date of sample collection and analyses required and were signed by each person transferring and accepting custody of the samples. The collected sediment samples were transferred to approved sampling containers and then placed in cool storage prior to transfer to the laboratories
- Decontamination: All equipment used in the sampling process was decontaminated using Decon 90, a phosphate free detergent, followed by rinsing with de-ionised water, prior to mobilisation and between sampling locations to reduce the risks of cross contamination.
- Intra-laboratory Field Duplicates: Intra-laboratory Field Duplicates refer to blind field split samples analysed by the laboratory. Intra-laboratory field duplicates provide information on analysis precision and sample heterogeneity. Relative percentage differences (RPDs) are used to assess precision.

To check accuracy with the sampling and analytical process, a field quality control sample was obtained as listed in **Table 3**.

Table 3: Quality Assurance and Quality Control Sample

Sample Type	Primary Sample	Duplicate Sample	Date
Field Duplicate Sample	SED_EA_02	SED_EA_SPLIT	24/08/2009

Results of the QA/QC assessment are discussed in **Section 4.3**

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3.0 Desktop Review of Pre-Development Data

3.1 Heavy Metals

A pilot air quality study (CSIRO, 2001) was carried out from March to December in 2000 to provide measurements of selected National Environment Protection Measure (NEPM) components, from two sites in Darwin; the Charles Darwin University Campus (formerly referred to as Northern Territory University (NTU) and approximately 13 km to the north of East Arm), and the CSIRO facility at Berrimah (approximately 10 km north-east of East Arm).

The 2001 CSIRO study has been included in this review as it is the only publicly available air quality data relevant to this investigation. The data obtained for the Berrimah site are presented as representative of an urban location.

Measurements undertaken in the study included airborne mass for particles with aerodynamic diameter less than 10 µm (PM10), airborne lead, zinc and iron (in PM10), NO_2, SO_2 and ozone.

Atmospheric lead loadings at the Berrimah site varied markedly throughout the sampling period, as shown in **Figure 1,** with the highest reading of approximately 11 ng/m^3 recorded in September. This concentration falls well below the National Environment Protection Measure for Ambient Air Quality (NEPC, 1998) guideline of 500 ng/m^3 (0.5 µg/m^3) for 24 hour average lead loading (CSIRO, 2001).

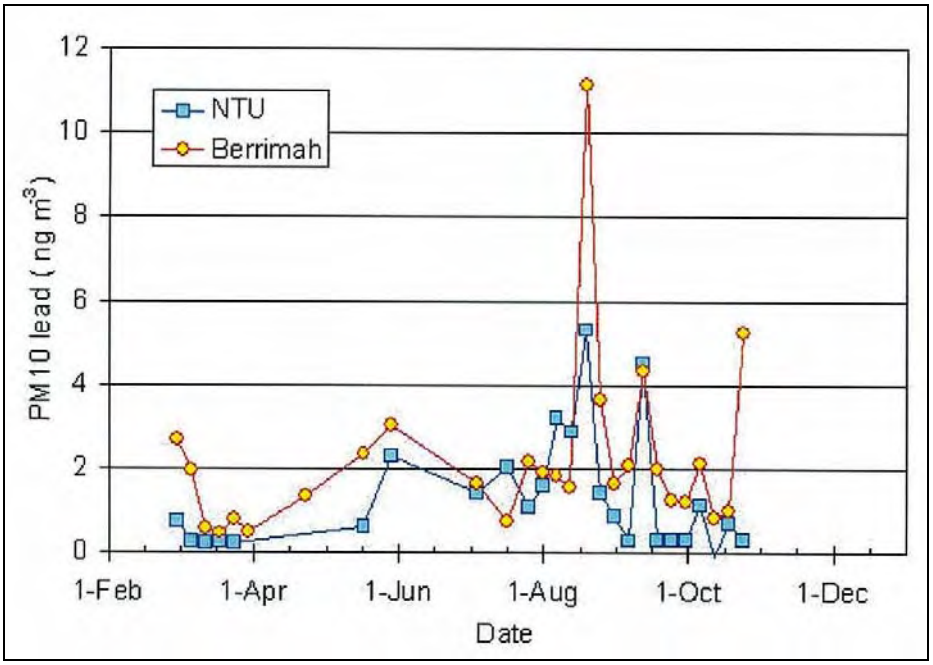

Figure 1: Atmospheric Lead Loadings in PM10

Source: CSIRO (2001)

Atmospheric zinc loadings at the Berrimah site generally remained below 100 ng/m^3 throughout the sampling period, with one high sample recorded in March (CSIRO, 2001), as shown in **Figure 2**. There is currently no NEPM for Ambient Air Quality guideline for zinc.

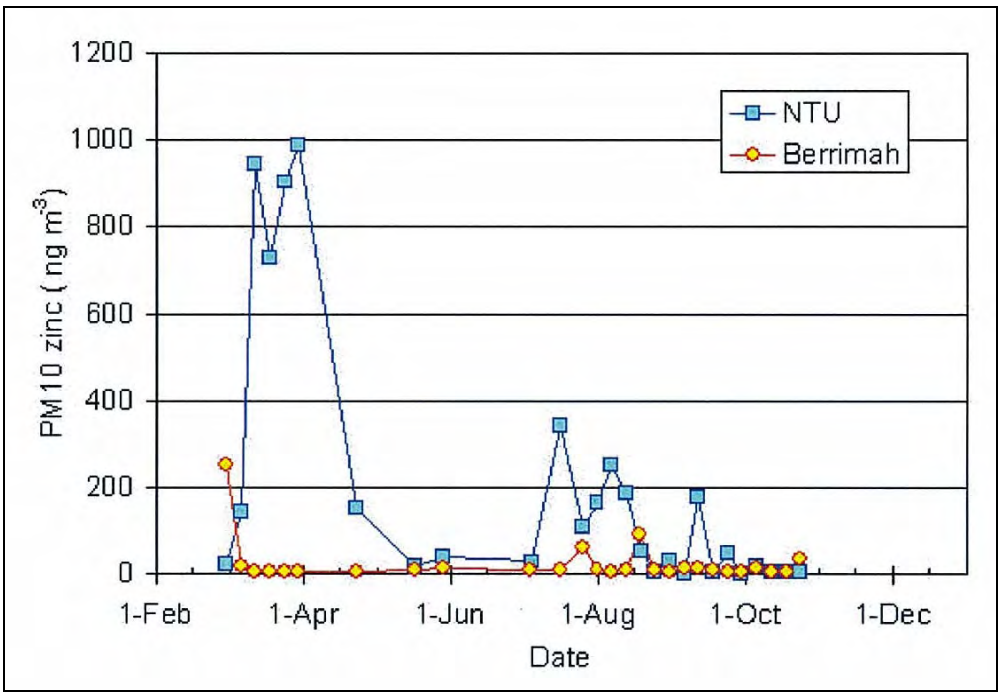

Figure 2: Atmospheric Zinc Loadings in PM10

Source: CSIRO (2001)

3.2 Radionuclides

The Australian Radiation Protection and Nuclear Safety Agency (ARPANSA) have issued technical reports summarising the monitoring of radioactivity of waters, soils and foodstuffs as well as the atmospheric monitoring of natural and man-made radionuclides.

Table 4 summarises the data for average lead-210 activity concentrations in air across Australia between 1996 and 2001.

Table 4: Average Lead-210 Activity Concentrations in Air in Darwin

Year	1996	1997	1998	1999	2000	2001
Lead-210 mBq/m^3	0.72	0.39	0.36	0.48	0.36	0.41

Note:

Hardege, 2008 (approximations only, raw data not available)

Samples are collected by continuous sampling of the air over a period of one week, at average flow rates of 80m^3 per hour and are measured by high-resolution gamma-ray spectrometry for no less than 2 hours.

Results are reported in millibecquerel per cubic metre (mBq/m^3).

Figure 3 reproduced from Hardege (2008) shows monitoring data for lead-210 in a number of capital cities from 1996 to 2006. **Table 4** presents the activity concentrations of lead-210 in Darwin from 1996 to 2001 estimated from the bar chart (**Figure 3**).

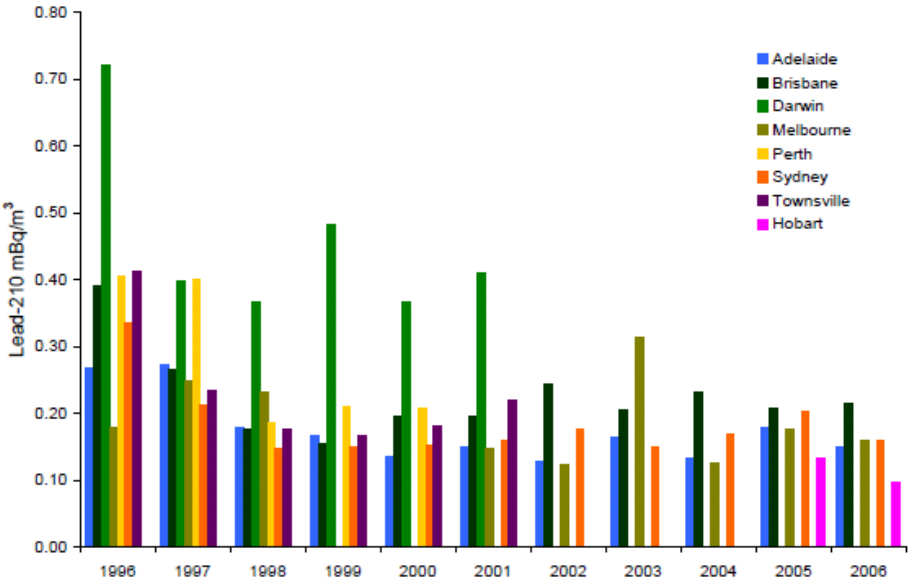

Figure 3: Annual average lead-210 activity concentrations across Australia 1996-2006

Source: Hardege, 2008

3.3 Sediment Quality

Many reports exist summarising various studies undertaken to assess the sediment quality in Darwin Harbour; however, few were undertaken prior to the development of East Arm Wharf. Three studies are summarised in this section that allow for comparison with the AECOM Initial Geochemistry Survey results.

No data were available on the radionuclide concentrations of sediments in Darwin Harbour.

3.3.1 Heavy Metals

The most comprehensive study undertaken to date to characterise for heavy metal content of the sediments in Darwin Harbour was undertaken by the Northern Territory Department of Natural Resources, Environment, the Arts and Sport (NRETAS) in 1993, and was reported by Fortune in 2006. The study was undertaken to assess the spatial distribution of Darwin Harbour sediment grainsizes and heavy metal content, and represents baseline data prior to infrastructure development and dredging of the East Arm Port Facility. The East Arm area was one of the seven areas of Darwin Harbour examined in the study, with the locations of the sampling sites within this study area presented in **Figure F3** (in the Figures Section). A summary of results from the East Arm sediment samples is presented in **Table 5**.

Table 5: Extract of Results from Fortune (2006) Study.

Site	Concentrations of Selected Metals within Marine Sediments (mg/kg)							
	As	Cd	Hg	Cu	Pb	Mn	Ni	Zn
127	39.6	<0.8	<0.025	5.1	15.1	240	10.9	24
140	10.6	<0.8	0.027	3.7	<5.6	304	8.1	38
141	8.5	<0.8	<0.025	6.4	6.1	258	13.1	58
142	20.4	<0.8	0.029	7.1	<5.6	311	12.4	42
142	30.8	<0.8	<0.025	6.6	10.8	307	14	40
145	10.9	<0.8	0.029	1.6	<5.6	232	5.5	16.5
151	8.8	<0.8	0.032	3.5	<5.6	268	9.4	36
152	11.2	<0.8	0.033	3.3	5.9	306	8.6	39
153	10.6	<0.8	0.031	4.3	<5.6	256	9.8	46
154	13.1	<0.8	<0.025	4.6	7.5	117	9.2	30
155	9.4	<0.8	0.03	4.1	<5.6	261	9.5	34
156	5.3	<0.8	<0.025	5.6	<5.6	247	12.4	45
157	10.7	<0.8	0.033	4.7	<5.6	275	11.4	42
158	11.4	<0.8	<0.025	4.4	6.3	269	10	34
159	9.6	<0.8	<0.025	4.7	<5.6	250	11.3	34
161	73.8	<0.8	<0.025	5.6	13.1	229	9.6	26
163	12.3	<0.8	0.031	5.6	7.1	143	12.5	42
165	27.1	<0.8	<0.025	10.1	8.8	299	16.3	50
167	10.8	<0.8	<0.025	4.8	<5.6	216	11.9	42
168	16.2	<0.8	<0.025	6.6	7.7	141	12.4	34
169	24.5	<0.8	<0.025	6.8	<5.6	136	14.4	28
171	11.6	<0.8	<0.025	6	<5.6	222	14.9	45
173	10.7	<0.8	<0.025	3.3	<5.6	257	8.7	29
177	29.1	<0.8	<0.025	3	<5.6	456	6.4	14.2
179	70.9	<0.8	<0.025	4.5	6.5	220	10.5	22
181	101.4	<0.8	0.036	16.8	29.2	333	18.9	37
183	11.1	<0.8	<0.025	4.1	5.9	82	8.2	19.4
185	178.9	<0.8	<0.025	18.9	49.9	676	34	73
187	18.1	<0.8	<0.025	7.4	9.9	123	10.4	22
Min	5.3	0.0	0.0	1.6	5.9	82.0	5.5	14.2
Max	178.9	0.0	0.036	18.9	49.9	676.0	34.0	73.0
Mean	27.8	<0.8	0.031	6.0	12.7	256.3	11.9	35.9
StdDev	36.8		0.003	3.7	11.9	111.6	5.1	12.6

Note: Results are directly copied from Fortune (2006) Appendix 2: Heavy Metal and Arsenic Concentrations (mg/kg), Pb and Cd corrected- erroneously high values due to interference. Mean values appear to have been reported as the arithmetic averages of the value that exceeded the limit of quantification.

Arsenic concentrations within the harbour area ranged from 5 to 179 mg/kg. Samples taken between East and Lee Points, the Fannie Bay area, and in the East and West Arms had concentrations greater than 100mg/kg. Average concentration in samples taken from East Arm was 27.8mg/kg, and the highest concentration was 178.9mg/kg. It was assessed that the sample sites with elevated arsenic concentrations have no known anthropogenic arsenic sources (Fortune, 2006).

The highest concentration of zinc in the East Arm area (73 mg/kg) was below the screening level of 200 mg/kg, as determined by the *National Assessment Guidelines for Dredging* (Commonwealth of Australia, 2009). Average concentration of zinc in the analysed samples was 35mg/kg.

One sediment sampling location had lead concentrations equal to the screening level of 50 mg/kg (Commonwealth of Australia, 2009); however, the average concentration was 12.7 mg/kg. Fortune (2006) concluded stormwater is a major source of anthropogenic lead to the harbour, increasing load by 90 times. The location with the highest lead concentration was upstream of the East Arm wharf, at the Elizabeth River Bridge, and receives runoff from Mitchell Creek, which is fed by stormwater drains from Palmerston (Fortune, 2006).

Concentration of nickel at one site within the East Arm study area was elevated (34 mg/kg against screening level of 21mg/kg); however, the average concentration in the area was 11.9 mg/kg. Concentrations of Cd, Cu and Hg were all below the screening levels (Commonwealth of Australia, 2009).

Lead and nickel were found to be elevated in many of these sites, including the East Arm study area, however Fortune (2006) states that concentrations of these metals are low when compared to those found in other estuaries of Australia and overseas. Arsenic was found to be elevated against screening levels at all sites in Darwin Harbour, and more notably in the main body of the Harbour to East Arm. Fortune (2006) concluded these concentrations are an indication of local geology rather than anthropogenic sources.

Generally, elevated metals were found in areas receiving stormwater, although apart from localised elevations in metal concentration, primarily in the City area, this study found no evidence of extensive contamination of sediments (Fortune, 2006).

More recent data is presented in Esslemont (1997), which summarises a study that was undertaken in 1996 to establish baseline data for heavy metal concentrations in the sediments of harbour precincts. The study provides an assessment of Cd, Cu, Cr, Pb, Ni and Zn prior to the establishment of the East Arm Port facility at various locations in the Darwin Harbour, including North Shell Island, which post-development has been incorporated into the East Arm Wharf.

The location of the North Shell sampling site is presented in **Figure F3** (in the Figures Section), and a summary of the concentrations of selected metals within the fine and coarse fractions of marine sediments from these sites are presented in **Table 6**.

Table 6: Extract of results from Esslemont (1997).

Site	Concentrations of Metals (mg/kg)					
	Cd	Cu	Cr	Pb	Ni	Zn
North Shell Island (coarse)	1.0	1.8	7.1	13.3	6.3	5.3
North Shell Island (fine)	0.8	3.1	23.6	13.5	10.6	15.4

The most recent study, with results available at the time of the production of this report, was undertaken in 2004 by URS with the purpose of establishing baseline contaminant data prior to the development of the Darwin City Waterfront Redevelopment, and included the sampling of a reference site located in the Elizabeth River approximately 6 km upstream from the East Arm Wharf.

The location of the Elizabeth River site is presented in **Figure F3** (in the Figures Section). A summary of results from this site is presented in Table 7.

Table 7: Extract of results from URS (2004).

Site	Concentrations of Selected Metal Ions within Marine Sediments (mg/Kg)						
	As	Cd	Cr	Pb	Ni	Zn	Mn
DWREF	31	<1	29	9	5	9	181

Fortune (2006) and URS (2004) studies concluded that the elevated arsenic concentrations, when compared against the screening level (Commonwealth of Australia, 2009) for arsenic of (20mg/kg) can be attributed to the natural weathering bedrock within the Darwin Harbour catchment, as the reference site was located remote from potential anthropogenic inputs of arsenic (URS, 2004).

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4.0 Results

4.1 Air Quality

4.1.1 High Volume Air Sampler

The laboratory results for elemental concentrations and radionuclides in air from the program of air sampling using the High Volume Air Sampler (HVAS) from July to November at East Arm Wharf, the mean concentrations of lead and zinc results from previous studies, as well as applicable guidelines, are presented in **Table 8**.

The concentrations are expressed in micrograms per cubic metre averaged by the laboratory over the period of air sampling for metals, or micro-Becquerels per cubic metre averaged by the laboratory over the period of air sampling for radionuclides.

Guidelines provided below are sourced from:

- New South Wales Department of Environment, Climate Change and Water (DECCW) (formerly Department of Environment and Conservation) *Approved Methods for the Modelling and Assessment of Air Pollutants in New South Wales* (DEC, 2005)
- National Environment Protection Council *National Environment Protection Measure for Ambient Air Quality Guidelines* (NEPC, 1998)

All guidelines presented below are from DEC (2005) except where indicated otherwise. No guidelines are currently available for Be, Ca, Co, La, Th, U, and V, and radionuclides in air.

Table 8: Analytical Results- Pollutant Concentration in Air

	Units	Guidelines	Month 1	Month 2	CSIRO (2001)[2]
Metals					
Antimony	µg/m^3	9	0.0008	0.0002	
Arsenic	µg/m^3	0.09	0.007	0.005	
Barium	µg/m^3	9	0.89	1.77	
Beryllium	µg/m^3	-	0.0003	0.0002	
Cadmium	µg/m^3	0.018	0.00006	0.0003	
Calcium	µg/m^3	-	18.3	19.1	
Chromium	µg/m^3	9 (Cr III)	0.014	0.010	
Cobalt	µg/m^3	-	0.014	0.02	
Copper	µg/m^3	18 (Dust)	4.5	9.4	
Lanthanum	µg/m^3	-	0.010	0.006	
Lead	µg/m^3	0.5[1]	0.018	0.015	0.011
Manganese	µg/m^3	18	2.1	2.1	
Mercury	µg/m^3	1.8 (Inorganic)	0.00001	0.00001	
Nickel	µg/m^3	0.18	0.014	0.013	
Silver	µg/m^3	1.8 (Metal)	0.003	0.0021	
Thorium	µg/m^3	-	0.002	0.0013	
Uranium	µg/m^3	-	0.002	0.0031	
Vanadium	µg/m^3	-	0.023	0.014	

	Units	Guidelines	Month 1	Month 2	CSIRO (2001)[2]
Zinc	µg/m^3	90 (Oxide Fumes)	0.27	2.5	0.260
U-238 Series					
Th-234	µBq/m^3		<70	160 ± 50	
Th-230	µBq/m^3		<150	<150	
Ra-226	µBq/m^3		43 ± 18	79 ± 20	
Pb-210	µBq/m^3		840 ± 130	650 ± 150	
Po-210	µBq/m^3		140 ± 15	180 ± 20	

Notes:

[1] NEPM Ambient Air Quality Guidelines (NEPC, 1998) 1 year averaging period

[2] Maximum lead and zinc loadings observed over 12 month sampling period (CSIRO, 2001)

DECCW '*Approved Methods for the Modelling and Assessment of Air Pollutants in New South Wales*' (DEC, 2005) 1 hour average period

Elemental Concentrations in Air (metals) are expressed in microgram (µg) per cubic metre of air sampled. Radioactivity concentrations are expressed in microBecquerel (µBq) per cubic metre sampled.

Reported concentrations of metals are an average for the three filters submitted to the laboratory for the sampling period.

Less than (<) values indicate the limit of detection for each isotope and radionuclide for the measurement system

The reported uncertainty in each radioactivity concentration in air is the expanded uncertainty calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.

The laboratory results for the HVAS air samples collected at East Arm Wharf show that all monitored metals met NSW DECCW (DEC, 2005) and NEPM (NEPC, 1998) assessment criteria where available.

The lead results for the HVAS air samples collected at East Arm Wharf of 0.018 µg/m^3 for Month 1 and 0.015 µg/m^3 for Month 2 were similar to those collected in the CSIRO Pilot Study of Air Quality in Darwin (CSIRO, 2001) of 0.011 µg/m^3.

The laboratory analysis reports are provided in **Appendix A.**

4.1.2 Track Etch Detectors

The laboratory results from the radon monitoring undertaken at Alice Springs and East Arm Wharf, as well as control sites, between July and November 2009 are presented. In **Table 9**, with concentrations reported as the reported as the Average Radon Concentration over Exposure Period (ARCOEP)

Table 9: Analytical Results- Track Etch Detectors

Site	Detector Number						
East Arm Wharf	00000005	0000006	0000010	0000011	0000015	0000016	0000020
ARCOEP (Bq/m^3)	21.5 ± 10.5	15.5 ± 10.0	6.5 ± 9.0	17.0 ± 8.0	12.0 ± 8.5	29.0 ± 8.5	26.0 ± 9.5
Alice Springs	0000012	0000013	0000014	0000017	0000018	0000019	**0000021**
ARCOEP (Bq/m^3)	16.3 ± 5.0	13.4 ± 5.0	12.2 ± 5.0	15.0 ± 5.0	12.4 ± 5.0	32.5 ± 6.0	26.7 ± 8.0

Notes:

Radon concentrations are in Becquerel (Bq) per cubic metre of air. One Becquerel equals one nuclear transformation per second.

The reported uncertainty in each result is the expanded uncertainly calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.

Exposure period for East Arm Wharf was 31 July 2009 – 6 November 2009

Exposure period for Alice Springs was 21 August – 20 November 2009.

The results of radon monitoring are consistent with outdoor concentrations which are reported to range from 1 to more than 100 Bq/m^3 (UNSCEAR, 2000), with typical values of 10 Bq/m^3

The laboratory analysis reports are provided in **Appendix A.**

4.2 Sediment Geochemistry

The laboratory results from sediment sampling undertaken in July 2009 are presented in **Table 10**, as well as the mean concentrations of heavy metals in sediments from previous studies, as outlined in the previous section. Also presented is the sediment screening levels (Commonwealth of Australia, 2009), and average crustal abundances data (Bowen, 1979).

The laboratory analysis reports for sediment samples are provided in **Appendix A**.

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Table 10: Analytical Results- Sediment Samples

Analytes	Units	Screening Levels [1]	Average Crustal Abundance [2]	SED_EA_02	SED_EA_SPLIT	SED_EA_03	SED_EA_05	SED_EA_06	SED_EA_07	SED_EA_08	SED_EA_10	SED_EA_11	SED_EA_14	SED_EA_19	SED_EA_20	SED_EA_21	Fortune (2006)	Esslemont (1997)	URS (2004)
METALS (total)																			
Antimony*	mg/kg	2	0.2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2			
Arsenic	mg/kg	20	1.8	11	11	10	11	10	9	10	9	10	9	14	59	13	27.8		31
Barium	mg/kg	#	425	43	45	66	32	5	8	9	8	12	28	6	9	5			
Beryllium*	mg/kg	#	2.8	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2			
Cadmium*	mg/kg	1.5	0.2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<0.8	1.8	<1
Calcium	%	#		12	12	7.1	7.6	22	16	11	9	8.7	9.4	17	25	17			
Chromium	mg/kg	80	100	32	33	16	16	8	15	18	18	18	16	18	42	14		30.7	29
Cobalt	mg/kg	#	25	8	8	12	6	3	4	5	5	4	4	5	7	4			
Copper	mg/kg	65	55	38	38	36	54	2	8	10	11	21	16	4	4	3	5.9	4.9	5
Lanthanum	mg/kg	#	30	15	15	11	9	11	10	11	10	10	9	14	18	11			
Lead	mg/kg	50	12.5	9	10	16	10	5	7	8	8	9	11	9	12	6	6.5	26.8	9
Manganese	mg/kg	#	950	380	410	1800	590	180	180	220	180	220	240	250	350	200	256		181
Mercury*	mg/kg	0.15	0.08	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	0.01		
Nickel	mg/kg	21	75	10	10	6	6	5	5	6	6	7	5	6	6	5	11.8	16.9	5
Silver*	mg/kg	#	0.07	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2			
Thorium*	mg/kg	#	10	6	5	<2	<2	<2	<2	<2	<2	<2	<2	<2	5	<2			
Uranium	mg/kg	#	2.7	1	1	<2	<2	<2	<2	<2	<2	<2	<2	2	2	2			
Vanadium	mg/kg	#	135	34	35	19	21	12	20	23	24	23	20	24	90	20			
Zinc	mg/kg	200	70	29	30	37	60	9	19	22	20	40	87	16	12	13	35	20.7	9

Analytes	Units	Screening Levels [1]	Average Crustal Abundance [2]	SED_EA_02	SED_EA_SPLIT	SED_EA_03	SED_EA_05	SED_EA_06	SED_EA_07	SED_EA_08	SED_EA_10	SED_EA_11	SED_EA_14	SED_EA_19	SED_EA_20	SED_EA_21	Fortune (2006)	Esslemont (1997)	URS (2004)
RADIONUCLIDES																			
Th-234	Bq/kg	-		45±16	43±15		39±12							33±12	27±6	38±8			
Th-230	Bq/kg			54±6	35±4		30±4							36±5	39±4	31±4			
Ra-226[4]	Bq/kg			19±2	22±3		17±2							18±2	12±1	15±2			
Pb-210[4]	Bq/kg			65±15	78±18		52±16							39±15	34±9	27±2			
Po-210[4]	Bq/kg			54±5	59±9		35±4							20±3	40±5	24±3			
Total Radioactivity Concentrations																			
	Bq/kg	35,000[3]																	

Notes:

1. *National Assessment Guidelines for Dredging* (Commonwealth of Australia, 2009).
2. *Bowen (1979)Environmental Chemistry of the Elements*
3. *Maximum sum of gross alpha and gross beta*

 # No guideline is available as the quality of the data used for guideline development was poor or because guidelines developed using different methods were inconsistent. Environment Australia (2009) recommends comparing concentrations to ambient baseline levels for sediments of comparable grainsize in the vicinity of the disposal area.

**The Limit of Reporting used by the laboratory was greater than that request*

As per the scope of works, samples were collected from the load out facility at the East Arm Wharf and nearby South Shell Island in Darwin Harbour as follows:

Load out facility at East Arm Wharf

- SED_EA_02
- SED_EA_03
- SED_EA_05
- SED_EA_07
- SED_EA_08
- SED_EA_10
- SED_EA_11
- SED_EA_14

South Shell Island

- SED_EA_06
- SED_EA_19
- SED_EA_20
- SED_EA_21

The locations are shown in **Figure F1** (in the Figures Section).

Plates P-3 to P-11 (in the Plates Section) depict the type of material collected for various samples.

The median, range and mean concentrations of each metal and radionuclide at these sampling sites are presented in **Table 11**.

Table 11: Average Concentrations for Defined Sediment Sample Areas

Analytes	Units	Guideline[1]	Load out facility at East Arm Wharf			South Shell Island		
			Median	Range	Mean	Median	Range	Mean
METALS (total)								
Antimony	mg/kg	2	<2	-	<2	<2	-	<2
Arsenic	mg/kg	20	10	9-11	10	13.5	10-59	24
Barium	mg/kg	#	25.75	8-66	26	5.5	5-9	6
Beryllium	mg/kg	#	<2	-	<2	<2	-	<2
Cadmium	mg/kg	1.5	<2	-	<2	<2	-	<2
Calcium	%	#	9.4	7.1-16	10	19.5	17-25	20
Chromium	mg/kg	80	18	15-32	19	16	8-42	21
Cobalt	mg/kg	#	5	4-12	6	4.5	3-7	5
Copper	mg/kg	65	21	8-54	24	3.5	2-4	3
Lanthanum	mg/kg	#	10	9-15	11	12.5	11-18	14
Lead	mg/kg	50	9	7-16	10	7.5	5-12	8
Manganese	mg/kg	#	240	180-1800	476	225	180-350	245
Mercury	mg/kg	0.15	<2	-	<2	<2	-	<2
Nickel	mg/kg	21	6	5-10	6	5.5	5-6	6
Silver	mg/kg	#	<2	-	<2	<2	-	<2
Thorium	mg/kg	#	6	<2-6	6	5	<2-5	<2
Uranium	mg/kg	#	<2	-	<2	2	<2-2	2
Vanadium	mg/kg	#	23	19-34	23	22	12-90	37
Zinc	mg/kg	200	37	19-87	39	12.5	9-16	13
RADIONUCLIDES (4 samples analysed for radionuclides at East Arm Wharf and 1 sample at South Shell Island)								
U-238 Series								
Th-234	Bq/kg		-	39-45	42	-	27-38	33
Th-230	Bq/kg		-	30-54	42		31-39	35
Ra-226	Bq/kg		-	17-19	18	-	12-18	15
Pb-210	Bq/kg		-	52-65	58.5	-	27-39	33
Po-210	Bq/kg		-	35-54	44.5	-	20-40	28

Notes:

1. *National Assessment Guidelines for Dredging* (Commonwealth of Australia, 2009).

\# *No guideline is available as the quality of the data used for guideline development was poor or because guidelines developed using different methods were inconsistent. Environment Australia (2009) recommends comparing concentrations to ambient baseline levels for sediments of comparable grainsize in the vicinity of the disposal area.*

The data presented in **Table 10** and **Table 11** were compared to average crustal abundances for metals (excluding radionuclides). The element enrichments were identified using the Geochemical Abundance Index (GAI) based on method developed by Förstner et al (1993).

The GAI quantifies an assay result for a particular element in terms of the average crustal abundance of that element. The GAI (based on a log-2 scale) is expressed in 7 integer increments (viz. 0 to 6). A GAI of 0 indicates that the content of the element is less than, or similar to, the average crustal abundance; a GAI of 3 corresponds to a 12-fold enrichment above the average crustal abundance; and so forth, up to a GAI of 6 which corresponds to a 96-fold, or greater, enrichment above average-crustal-abundances.

The data indicates that there is slight (6- to 24-fold) enrichment in arsenic in most samples from this study and a 6-fold enrichment above average crustal abundance for marine sediments in the Esslemont (1997) sampling.

All other metals analysed in this and the Esslemont (1997) study reported concentrations at or below average crustal abundance.

Comparison of metal concentrations with dredge spoil disposal guidelines demonstrate that most of samples returned metal concentrations below Guideline limits. Sample SED_EA_20 from East Arm reported a 3-fold arsenic concentration enrichment compared to average crustal abundance for marine sediments.

There are no reported data on average crustal abundance of radionuclides for marine sediments. There are no guidelines for specific radionuclide concentrations for disposal of dredge spoil.

4.3 Quality Control

A field duplicate sediment sample was prepared in the field by splitting the primary field sample. The duplicate samples were labelled so that they could not be linked to their respective primary samples. The duplicate samples and their corresponding primary sample are presented in the **Table 3**.

The relative percentage difference (RPD) between primary and duplicate sample analysis results is used to measure the representativeness and/or precision of duplicate samples. The RPD is calculated from the absolute difference between results of the duplicate pair divided by the mean value of the duplicate pair.

$$RPD\ (\%) = 100 \times (D1-D2) / ((D1+D2) / 2)$$

where: D1 = primary sample analysis

D2 = duplicate sample analysis.

The AS 4482.1-2005 (*Guide to the sampling and investigation of sites with potentially contaminated soil – Part 1: Non volatile and semi-volatile compounds*) states that the typical RPD which can be expected from acceptable field duplicate sample is 30-50% of the mean concentration of the analytes where at least one result is greater than 10 times laboratory LOR. RPD values exceeding this range and criteria were considered unacceptable.

The initial report issued by the laboratory was assessed and RPDs calculated for the primary and secondary sediment samples (SED_EA_02 and SED_EA_SPLIT). RPDs were calculated, and ranged from 14.6% to 169.6%. The calculated RPDs for arsenic, barium, cobalt, copper, lead and manganese were considered unacceptable (Standards Australia, 2005), and the laboratory was contacted immediately to initiate the re-analysis of the two samples. The final report provided to AECOM indicated that RPDs were within acceptable limits.

The duplicate sample results and RPDs for both the initial and re-analysis are presented in **Table 12**

Table 12: Relative Percentage Difference Calculations for QA/QC Samples

Analytes	Units	INITIAL ANALYSIS			RE-ANALYSIS		
		SED_EA_02	SED_EA_SPLIT	RPD	SED_EA_02	SED_EA_SPLIT	RPD
Antimony	mg/kg	<2	<2	0	<2	<2	0
Arsenic	mg/kg	15	11	30.7%	11	11	0
Barium	mg/kg	210	22	162.0%	43	45	4.5%
Beryllium	mg/kg	<2	<2	0	<2	<2	0
Cadmium	mg/kg	<2	<2	0	<2	<2	0
Calcium	%	130000	110000	16.6%	12	12	0
Chromium	mg/kg	22	19	14.6%	32	33	3.0%
Cobalt	mg/kg	18	5	113.0%	8	8	0
Copper	mg/kg	68	32	72%	38	38	0
Lanthanum	mg/kg	14	11	24%	15	15	0
Lead	mg/kg	26	9	97.14%	9	10	10.5%
Manganese	mg/kg	3900	320	169.6%	380	410	7.5%
Mercury	mg/kg	<2	<2	0	<2	<2	0
Nickel	mg/kg	9	7	25%	10	10	0
Silver	mg/kg	<2	<2	0	<2	<2	0
Thorium	mg/kg	<2	<2	0	6	5	18.1%
Uranium	mg/kg	<2	<2	0	1	1	0
Vanadium	mg/kg	29	23	23.0%	34	35	2.9%
Zinc	mg/kg	28	24	15.3%	29	30	3.4%

4.3.1 Precision and Accuracy

Overall, the precision and accuracy of the laboratory analysis data was satisfactory. Laboratory LOR was increased for 7 metals due to matrix interferences such as excess salinity.

Raised LORs were applied to the following analytes:

- As, from 0.1mg/kg to 2mg/kg
- Sb from 0.05mg/kg to 2mg/kg
- Be from 0.05mg/kg to 2mg/kg
- Cd from 0.01mg/kg to 2mg/kg
- Hg from 0.01mg/kg to 2mg/kg
- Ag from 0.01mg/kg to 2mg/kg
- Th from 0.2mg/kg to 2mg/kg

5.0 Summary

BHP Billiton is proposing to expand the existing Olympic Dam mine and processing plant, and proposes to transport copper concentrate by rail to Darwin and load ore from a dedicated facility onto concentrate ships for export.

AECOM has undertaken an initial geochemistry survey of marine sediment and airborne dust at East Arm Wharf and Darwin Harbour, and radon monitoring at East Arm Wharf and Alice Springs for BHP Billiton, and has presented the results of the sediment and air sampling in relation to national guidelines.

The geochemical study was undertaken in accordance with the scope of works as per the Proposal for Consulting Services “Baseline Contaminant Study- Port of Darwin Transport Open” dated 25 May 2009 and the Briefing on the Project dated 11 August 2009, and included:

- the collection of 16 sediment samples (including a QA/QC sample), with the analysis of 13 sediment samples for 13 heavy metals analysis and five radionuclides analysis, including the QA/QC from the load out facility (the berth) at the East Arm Wharf and from South Shell island nearby in Darwin Harbour
- the intermittent collection of airborne dust samples from East Arm Wharf for a period of 343.3 hours using a High Volume Air Sampler (HVAS)
- the measurement of average radon concentrations at East Arm Wharf and Alice Springs.

This report included a review of existing relevant data and a brief analysis of the laboratory results in comparison to relevant guidelines.

A summary of the results is provided:

- Air quality concentrations for metals were within guideline concentrations for the period of monitoring undertaken at the East Arm Wharf.
- The radon concentrations measured at Alice Springs and East Arm Wharf were within the range of global averages for ambient air quality (UNSCEAR, 2000).
- The concentrations of heavy metals in marine sediments sampled, apart from arsenic, were below the screening level determined by Commonwealth of Australia (2009).
- Sediment sample SED_EA_20 exceeded the screening level for arsenic.
- Arsenic concentrations are generally elevated in Darwin Harbour due to the natural weathering bedrock in the catchment (Fortune, 2006 and URS, 2004).
- The radionuclide concentrations in marine sediments were documented with a limited number of samples for Th-234, Th-230, Ra-226, Pb-210, Po-210.
- There are no guideline concentrations for specific radionuclides for marine sediments, or for dredge spoil; however the total radioactivity count, based on the radionuclides analysed is below the screening levels as per Commonwealth of Australia (2009) for dredge spoil disposal.

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6.0 References

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Figures

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LEGEND

- HVAS
- Track Etch Detectors

Sediment Sample Locations

- East Arm Wharf
- South Shell Island

Data sources:
Base Data: GoogleEarth(c) 2009

Location of Sediment Samples, HVAS and Track Etch Detectors: East Arm Wharf

BHP Billiton
Olympic Dam Expansion EIS
Baseline Geochemistry Study

East Arm Wharf and Alice Springs, Northern Territory

Figure
F1

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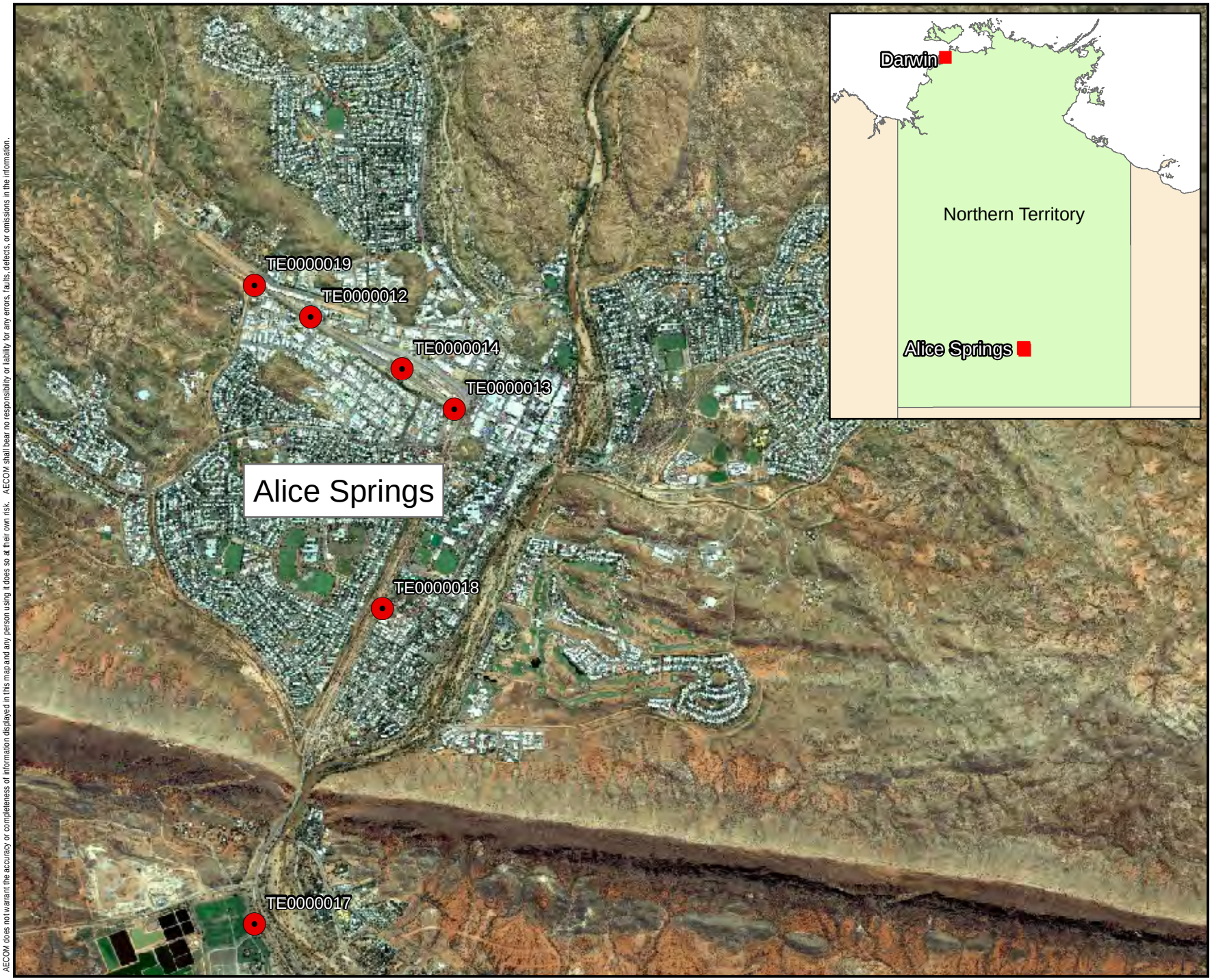

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LAST MODIFIED CW - 03 March 2010

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LEGEND

Track Etch Detectors

Data sources:
Base Data: GoogleEarth(c) 2009

Location of Track Etch Detectors: Alice Springs

BHP Billiton
Olympic Dam EIS
Baseline Geochemistry Study

East Arm Wharf and Alice Springs, Northern Territory

Figure
F2

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LEGEND

- Esslemont (1997)
- CSIRO (2001)
- URS (2004)
- Fortune (2006)

Data sources:
Esslemont (1997)
URS (2004)
Fortune (2006)
All are approximations only

Locations of Sampling Sites: Historical Data

BHP Billiton
ODX EIS
Baseline Geochemistry Study

East Arm Wharf and Alice Springs, Northern Territory

Figure
F3

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Plates

Plate P-Error! Unknown switch argument.**: High Volume Air Sampler, East Arm Wharf**

Plate P-2: Track Etch Detector, East Arm Wharf

Plate P-3: East Arm Wharf Sediment Sample SED_EA_02 24/08/2009

Plate P-4: East Arm Wharf Sediment Sample SED_EA_08 25/08/09

Plate P-5: East Arm Wharf Sediment Sample SED_EA_08 25/08/09

Plate P-6: East Arm Wharf Sediment Sample SED_EA_11 25/08/09

Plate P-7: East Arm Wharf Sediment Sample SED_EA_14 25/08/09

Plate P-8: East Arm Wharf Sediment Sample SED_EA_19 25/08/09

Plate P-9: East Arm Wharf Sediment Sample SED_EA_20 25/08/09

Plate P-10: East Arm Wharf Sediment Sample SED_EA_21 25/08/09

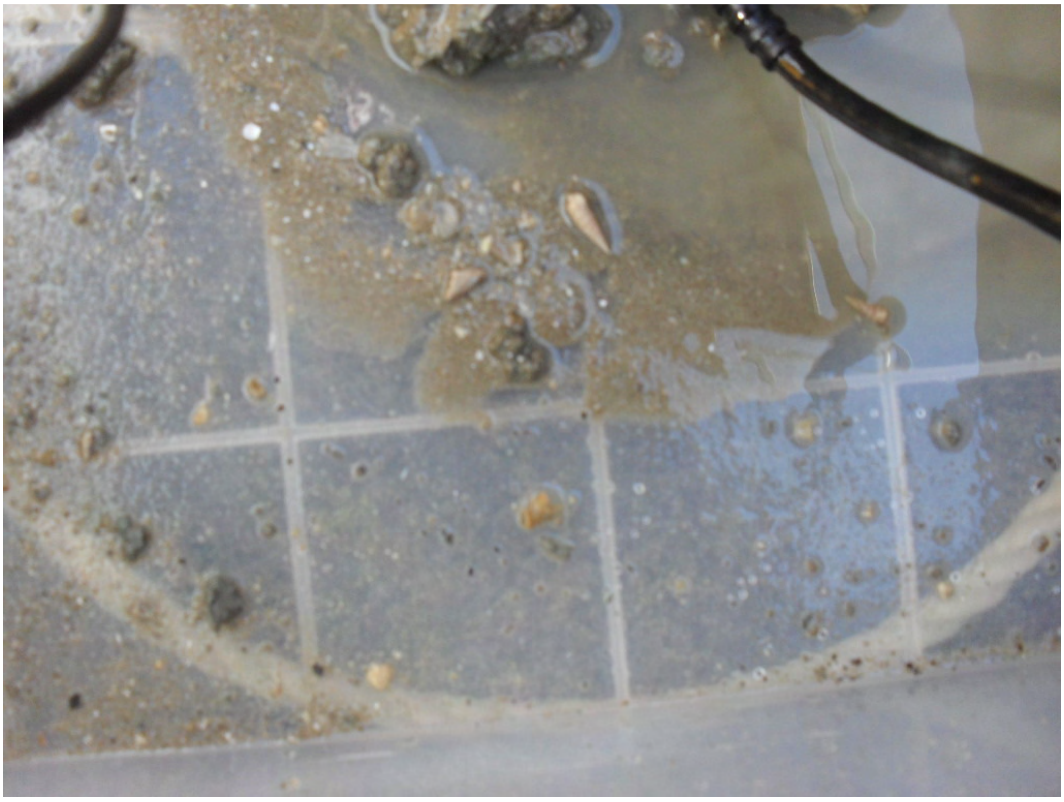

Plate P-11: East Arm Wharf Sediment Sample SED_EA_21 25/08/09

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Appendix A

Analytical Results

Australian Radiation Services Pty Ltd
PO Box 3103, Nunawading, Victoria, 3131
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Email: info@australian-radiation-services.com.au
Web Site: www.australian-radiation-services.com.au
ABN 66 006 528 267

Keeping people safe.

Radioactivity Analysis Report

To: Aecom - Darwin
P.O. Box 3175
Darwin NT 0801

Report No.: 10-111-R1

Contact: Mr. Jorge Martinez

Sample description: High volume air sampler filters

Number of samples: Two batches of 3 filters (Month 01 and Month 02)

Submission date: 14th October and 18th November 2009

Analysis required: Determination of radionuclide and metal concentrations.

Analytical method: Filters were weighed to determine the mass of dust per filter and subdivided by weight for the determination of metals, and radionuclides. Portions were submitted to sub-contracting laboratories for the determination of metals by ICPMS and for polonium-210 analysis by alpha particle spectrometry. Other radionuclides were determined by high resolution gamma ray spectrometry.

Analysis performed by: Malcolm Cooper (radionuclides), Leeder Consulting (ICPMS) and National Radiation Laboratory, New Zealand (polonium-210). ,

---

AUSTRALIAN RADIATION SERVICES PTY. LTD.

Report prepared by: Dr. Malcolm Cooper
Consultant Environmental Scientist

Signed: 

Reviewed by: Mr. Colin Foley
Health Physicist

Signed: 

Date: 18th December 2009

Results:

A. Radioactivity Concentrations in Dust ($Bq.g^{-1}$)

Note:

a) Radioactivity concentrations are expressed in becquerel (Bq) per gram of dust. One becquerel equals one nuclear transformation per second.
b) Less than (<) values indicate the limit of detection for each radionuclide for the measurement system.
c) The reported uncertainty in each result is the expanded uncertainty calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.

	Client Sample ID (ARS Lab. ID)	
Radionuclide	Month 01 (10-111-02/04/09)	Month 02 (10-111-03/06/08)
Naturally-occurring uranium (U-238) series		
Thorium-234	< 0.5	0.8 ± 0.3
Radium-226	0.35 ± 0.15	0.38 ± 0.10
Lead-210	6.7 ± 1.0	3.1 ± 0.6
Polonium-210	1.2 ± 0.1	0.85 ± 0.09
Naturally-occurring thorium (Th-232) series		
Radium-228	0.33 ± 0.06	0.13 ± 0.10
Thorium-228	0.25 ± 0.07	0.11 ± 0.05
Other naturally-occurring radionuclides		
Potassium-40	2.5 ± 1.5	2.4 ± 1.0
Beryllium-7	37 ± 5	29 ± 3

B. Radioactivity Concentrations in Air ($\mu Bq.m^{-3}$)

Note:

a) Radioactivity concentrations are expressed in micro-becquerel (µBq) per cubic metre of air sampled. One becquerel equals one nuclear transformation per second.
b) Less than (<) values indicate the limit of detection for each radionuclide for the measurement system.
c) The reported uncertainty in each result is the expanded uncertainty calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.
d) Air sample volumes provided by the client.

	Client Sample ID (ARS Lab. ID)	
Radionuclide	Month 01 (10-111-02/04/09)	Month 02 (10-111-03/06/08)
Naturally-occurring uranium (U-238) series		
Thorium-234	< 70	160 ± 50
Radium-226	43 ± 18	79 ± 20
Lead-210	840 ± 130	650 ± 150
Polonium-210	140 ± 15	180 ± 20
Naturally-occurring thorium (Th-232) series		
Radium-228	41 ± 8	30 ± 20
Thorium-228	30 ± 7	23 ± 12
Other naturally-occurring radionuclides		
Potassium-40	310 ± 190	500 ± 130
Beryllium-7	4500 ± 200	6000 ± 400

C. Elemental Concentration in Dust ($\mu g.g^{-1}$)

Note: a. Concentrations are expressed in micro-gram (µg) per gram of dust on the filter.
b. Reported concentrations are an average for the three filters submitted for the sampling period.
b. Less than (<) values indicate the limit of detection for each isotope for the measurement system.

Analyte	Elemental Concentration ($\mu g.g^{-1}$)[a,b,c]	
	Client Sample ID (ARS ID)	
	Month-01 (10-111-02/04/09)	Month-02 (10-111-03/06/08)
antimony	6.6	0.93
arsenic	59	26
barium	7200	8500
beryllium	2.3	1.1
cadmium	0.5	1.4
calcium	148000	92000
chromium	110	50
cobalt	110	97
copper	37000	45000
lanthanum	80	29
lead	145	75
manganese	16700	9900
mercury	0.06	0.06
nickel	110	65
silver	23	10.3
thorium	17	6.3
uranium	18	15
vanadium	185	68
zinc	2200	11900

D. Elemental Concentration in Air (µg.m^{-3})

Note:
a. Concentrations are expressed in microgram (µg) per cubic metre of air sampled.
b. Reported concentrations are an average for the three filters submitted for the sampling period.
b. Less than (<) values indicate the limit of detection for each isotope for the measurement system.
c. Air sample volumes provided by the client.

Analyte	Elemental Concentration (µg.m^{-3})[a,b,c]	
	Client Sample ID (ARS ID)	
	Month-01 (10-111-02/04/09)	Month-02 (10-111-03/06/08)
antimony	0.0008	0.0002
arsenic	0.007	0.005
barium	0.89	1.77
beryllium	0.0003	0.0002
cadmium	0.00006	0.0003
calcium	18.3	19.1
chromium	0.014	0.010
cobalt	0.014	0.02
copper	4.5	9.4
lanthanum	0.010	0.006
lead	0.018	0.015
manganese	2.1	2.1
mercury	0.00001	0.00001
nickel	0.014	0.013
silver	0.003	0.0021
thorium	0.002	0.0013
uranium	0.002	0.0031
vanadium	0.023	0.014
zinc	0.27	2.5

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Radioactivity Analysis Report

To: Aecom - Darwin
P.O. Box 3175
Darwin NT 0801

Report No.: 09-1683-R2

Contact: Mr. Jorge Martinez

Sample description:	Radon track etch detectors
Number of detectors:	Fourteen
Issue date:	24th June 2009
Submission date:	12 November 2009 (Batch 1) and 30th November 2009 (Batch 2)
Analysis required:	Determination of average radon concentration in air over exposure period.
Analytical method:	Radon exposure determined by alpha particle track counting after chemical etching of CR 39 detector.
Measurment performed by:	External laboratory,

AUSTRALIAN RADIATION SERVICES PTY. LTD.

Report prepared by: Dr. Malcolm Cooper
Consultant Environmental Scientist

Signed: 

Reviewed by: Mr. Colin Foley
Health Physicist

Signed: 

Date: 23rd December 2009

Results:

Average radon in air concentrations ($Bq \cdot m^{-3}$) using track etch detectors

Note:

a) Radon concentrations are in becquerel (Bq) per cubic metre of air. One becquerel equals one nuclear transformation per second.
b) Less than (<) values indicate the limit of detection for the measurement system.
c) The reported uncertainty in each result is the expanded uncertainty calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.

Batch 1 – Exposure period 31/7/09 to 6/11/09

Detector no.	Exposure period		Detector location / Comments	Average radon concentration over exposure period ($Bq \cdot m^{-3}$)
	Start date	End date		
0000005	31 July 2009	6 November 2009	EAW - 4 Project #B4416	21.5 ± 10.5
0000006	31 July 2009	6 November 2009	Office (Control)	15.5 ± 10.0
0000010	31 July 2009	6 November 2009	EAW - 5 Project #B4416	6.5 ± 9.0
0000011	31 July 2009	6 November 2009	EAW - 3 Project #B4416	17.0 ± 8.0
0000015	31 July 2009	6 November 2009	EAW - 6 Project #B4416	12.0 ± 8.5
0000016	31 July 2009	6 November 2009	EAW - 2 Project #B4416	29.0 ± 8.5
0000020	31 July 2009	6 November 2009	EAW - 1 Project #B4416	26.0 ± 9.5

Batch 2 – Exposure period 21/8/09 to 20/11/09

Detector no.	Exposure period		Detector location / Comments	Average radon concentration over exposure period ($Bq \cdot m^{-3}$)
	Start date	End date		
0000012	21 August 2009	20 November 2009	Alice Springs Freight Terminal	16.3 ± 5.0
0000013	21 August 2009	20 November 2009	Alice Springs Freight Terminal	13.4 ± 5.0
0000014	21 August 2009	20 November 2009	Alice Springs Freight Terminal	12.2 ± 5.0
0000017	21 August 2009	20 November 2009	Showgrounds	15.0 ± 5.0
0000018	21 August 2009	20 November 2009	St. Johns	12.4 ± 5.0
0000019	21 August 2009	20 November 2009	Power & Water	32.5 ± 6.0
0000021	21 August 2009	20 November 2009	Bill Low	26.7 ± 8.0

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Analysis Report (Interim)

To: Aecom - Darwin
P.O. Box 3175
Darwin NT 0801

Report No.: 10-385-R1

Contact: Mr. Jorge Martinez

Sample description: Marine Sediment

Number of samples: Thirteen

Submission date: 31st August 2009

Analysis required: Determination of the concentration of selected metals.

Analytical method: A portion of each sample was submitted to an external analytical laboratory for analysis of selected metals by ICPMS.

Analysis performed by: Subcontracted laboratory (Leeder Consulting)

AUSTRALIAN RADIATION SERVICES PTY. LTD.

Report prepared by: Ms. Genelle Jones
Radiochemist

Signed:

Reviewed by: Dr. Malcolm Cooper
Consultant Environmental Scientist

Signed:

Date: 1st October 2009

Results:

Note: a. Concentrations are expressed in milli-gram (mg) per kilogram (kg).
b. Less than (<) values indicate the limit of detection for each isotope for the measurement system.

Analyte	Elemental Concentration ($mg \cdot kg^{-1}$)[a,b]				
	Client Sample ID (ARS ID)				
	SED_EA_20 (10-385-02)	SED_EA_06 (10-385-03)	SED_EA_19 (10-385-04)	SED_EA_21 (10-385-05)	SED_EA_07 (10-385-06)
antimony	< 2	< 2	< 2	< 2	< 2
arsenic	59	10	14	13	9
barium	9	5	6	5	8
beryllium	< 2	< 2	< 2	< 2	< 2
cadmium	< 2	< 2	< 2	< 2	< 2
calcium	250000	220000	170000	170000	160000
chromium	42	8	18	14	15
cobalt	7	3	5	4	4
copper	4	2	4	3	8
lanthanum	18	11	14	11	10
lead	12	5	9	6	7
manganese	350	180	250	200	180
mercury	< 2	< 2	< 2	< 2	< 2
nickel	6	5	6	5	5
silver	< 2	< 2	< 2	< 2	< 2
thorium	5	< 2	< 2	< 2	< 2
uranium	2	< 2	2	2	< 2
vanadium	90	12	24	20	20
zinc	12	9	16	13	19

Analyte	Elemental Concentration ($mg \cdot kg^{-1}$)[a,b]				
	Client Sample ID (ARS ID)				
	SED_EA_08 (10-385-07)	SED_EA_10 (10-385-08)	SED_EA_11 (10-385-09)	SED_EA_05 (10-385-11)	SED_EA_03 (10-385-13)
antimony	< 2	< 2	< 2	< 2	< 2
arsenic	10	9	10	11	10
barium	9	8	12	32	66
beryllium	< 2	< 2	< 2	< 2	< 2
cadmium	< 2	< 2	< 2	< 2	< 2
calcium	110000	90000	87000	76000	71000
chromium	18	18	18	16	16
cobalt	5	5	4	6	12
copper	10	11	21	54	36
lanthanum	11	10	10	9	11
lead	8	8	9	10	16
manganese	220	180	220	590	1800
mercury	< 2	< 2	< 2	< 2	< 2
nickel	6	6	7	6	6
silver	< 2	< 2	< 2	< 2	< 2
thorium	< 2	< 2	< 2	< 2	< 2
uranium	< 2	< 2	< 2	< 2	< 2
vanadium	23	24	23	21	19
zinc	22	20	40	60	37

Analyte	Elemental Concentration ($mg \cdot kg^{-1}$)[a,b]		
	Client Sample ID (ARS ID)		
	SED_EA_02 (10-385-14)	SED_EA_14 (10-385-15)	SED_EA_SPLIT (10-385-16)
antimony	< 2	< 2	< 2
arsenic	15	9	11
barium	210	28	22
beryllium	< 2	< 2	< 2
cadmium	< 2	< 2	< 2
calcium	130000	94000	110000
chromium	22	16	19
cobalt	18	4	5
copper	68	16	32
lanthanum	14	9	11
lead	26	11	9
manganese	3900	240	320
mercury	< 2	< 2	< 2
nickel	9	5	7
silver	< 2	< 2	< 2
thorium	< 2	< 2	< 2
uranium	< 2	< 2	< 2
vanadium	29	20	23
zinc	28	87	24

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18th November 2009

Mr. Jorge Martinez
Aecom – Darwin
P.O. Box 3175
Darwin NT 0801

Sample analysis of marine sediment samples submitted to ARS – August 2009

Dear Jorge

Please find with this letter the re-analysis report (Report No. 10-385-R3) for the metals re-analysis of marine sediment samples submitted to Australian Radiation Services Pty. Ltd. (ARS) on the 31st August 2009.

If you have any questions regarding the report, please do not hesitate to contact me.

Best regards
Australian Radiation Services Pty. Ltd.

Ms. Genelle Jones
Radiochemist

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Analysis Report

To: Aecom - Darwin
P.O. Box 3175
Darwin NT 0801

Report No.: 10-385-R3

Contact: Mr. Jorge Martinez

Sample description: Marine sediment

Number of samples: Two

Submission date: 31st August 2009

Analysis required: Determination of the concentration of selected metals – repeat analysis of selected samples

Analytical method: A portion of each sample was submitted to an external analytical laboratory for analysis of selected metals by ICPMS.

Analysis performed by: Sub-contracted laboratory (Leeder Consulting)

AUSTRALIAN RADIATION SERVICES PTY. LTD.

Report prepared by: Dr. Malcolm Cooper
Consultant Environmental Scientist

Signed: 

Reviewed by: Ms. Genelle Jones
Radiochemist

Signed: 

Date: 18th November 2009

Results:

Elemental Concentrations ($mg \cdot kg^{-1}$)

Note: a. Concentrations are expressed in milligram (mg) per kilogram (kg).
b. Less than (<) values indicate the limit of detection for each isotope for the measurement system.

	Client Sample ID (ARS ID)	
Analyte	SED_EA_02 (10-385-14)	SED_EA_SPLIT (10-385-16)
antimony	< 2	< 2
arsenic	11	11
barium	43	45
beryllium	< 2	< 2
cadmium	< 2	< 2
calcium	120000	120000
chromium	32	33
cobalt	8	8
copper	38	38
lanthanum	15	15
lead	9	10
manganese	380	410
mercury	< 2	< 2
nickel	10	10
silver	< 2	< 2
thorium	6	5
uranium	1	1
vanadium	34	35
zinc	29	30

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Radioactivity Analysis Report

To: Aecom - Darwin
P.O. Box 3175
Darwin NT 0801

Report No.: 10-2016-R2

Contact: Mr. Jorge Martinez

Sample description: Marine Sediment and high volume air filters

Number of samples: Six sediment, two filters

Submission date: 31st August 2009 (sediments)

Analysis required: Determination of radionuclide concentrations, including thorium-230.

Analytical method: a. Pre-treatment – Each sediment sample was dried and milled to a uniform particle size. A sub-sample was contained in a standard geometry for measurement by gamma spectrometry. An additional portion of the sample was submitted to an external laboratory for polonium-210 and thorium-230 analysis. Air filters – as described in previous report,10-111-R1.

b. Gamma ray-emitting radionuclides were determined by high resolution gamma ray spectrometry. Polonium-210 and thorium-230 were determined by alpha spectrometry.

AUSTRALIAN RADIATION SERVICES PTY. LTD.

Report prepared by: Dr. Malcolm Cooper
Consultant Environmental Scientist

Signed: 

Reviewed by: Mr. Michael Gilhen
Health Physicist

Signed: 

Date: 3rd May 2010

Results:

A. Radioactivity concentrations in sediments (Bq•kg^{-1})

Note:

a) Radioactivity concentrations in sediment are expressed in becquerel (Bq) per kilogram of dry solid. Radioactivity concentrations in air filters are expressed in terms of becquerel per gram of dust on the filter. One becquerel equals one nuclear transformation per second.
b) Less than (<) values indicate the limit of detection for each radionuclide for the measurement system.
c) The reported uncertainty in each result is the expanded uncertainty calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.

	Client Sample ID (ARS Lab. ID)			
Radionuclide	**SED-EA-02 (10-385-14)**	**SED-EA-05 (10-385-11)**	**SED-EA-19 (10-385-03)**	**SED-EA-SPLIT (10-385-16)**
Naturally-occurring uranium (U-238) series				
Thorium-234	45 ± 16	39 ± 12	33 ± 12	43 ± 15
Thorium-230	54 ± 6	30 ± 4	36 ± 5	35 ± 4
Radium-226	19 ± 2	17 ± 2	18 ± 2	22 ± 3
Lead-210	65 ± 15	52 ± 16	39 ± 15	78 ± 18
Polonium-210	54 ± 5	35 ± 4	20 ± 3	59 ± 9
Naturally-occurring thorium (Th-232) series				
Radium-228	25 ± 5	29 ± 4	24 ± 4	24 ± 4
Thorium-228	29 ± 4	27 ± 2	26 ± 2	27 ± 4
Other naturally-occurring radionuclides				
Potassium-40	305 ± 30	260 ± 25	180 ± 20	300 ± 30

	Client Sample ID (ARS Lab. ID)	
Radionuclide	**SED-EA-20 (10-385-02)**	**SED-EA-21 (10-385-05)**
Naturally-occurring uranium (U-238) series		
Thorium-234	27 ± 6	38 ± 8
Thorium-230	39 ± 4	31 ± 4
Radium-226	12 ± 1	15 ± 2
Lead-210	34 ± 9	27 ± 2
Polonium-210	40 ± 5	24 ± 3
Naturally-occurring thorium (Th-232) series		
Radium-228	17 ± 2	22 ± 2
Thorium-228	21 ± 2	27 ± 3
Other naturally-occurring radionuclides		
Potassium-40	111 ± 10	185 ± 15

B. Radioactivity Concentrations in Air (μBq.m^{-3})

Note:

a) Radioactivity concentrations are expressed in micro-becquerel (μBq) per cubic metre of air sampled. One becquerel equals one nuclear transformation per second.
b) Less than (<) values indicate the limit of detection for each radionuclide for the measurement system.
c) The reported uncertainty in each result is the expanded uncertainty calculated using a coverage factor of 2, providing a level of confidence of approximately 95%.
d) Air sample volumes provided by the client.

	Client Sample ID (ARS Lab. ID)	
	Month 01	Month 02
Radionuclide	**(10-111-02/04/09)**	**(10-111-03/06/08)**
Naturally-occurring uranium (U-238) series		
Thorium-234	< 70	160 ± 50
Thorium-230	< 150	< 150
Radium-226	43 ± 18	79 ± 20
Lead-210	840 ± 130	650 ± 150
Polonium-210	140 ± 15	180 ± 20
Naturally-occurring thorium (Th-232) series		
Radium-228	41 ± 8	30 ± 20
Thorium-228	30 ± 7	23 ± 12
Other naturally-occurring radionuclides		
Potassium-40	310 ± 190	500 ± 130
Beryllium-7	4500 ± 200	6000 ± 400

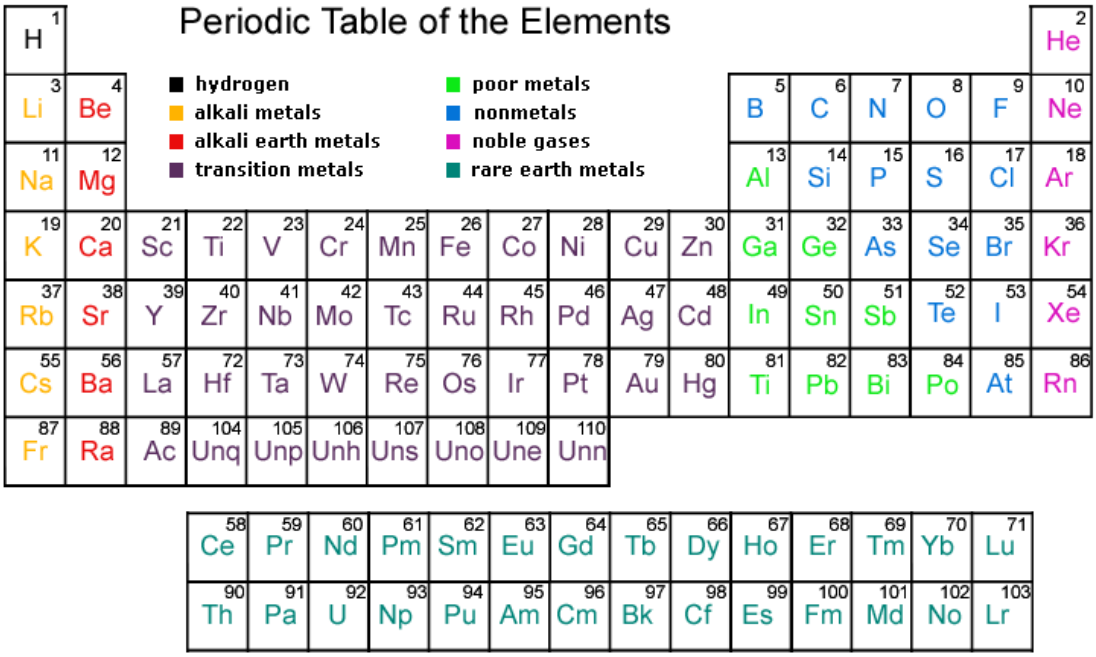
APPENDIX M3

Statistical analysis of radionuclide survey data

Rho Environmetrics Pty Ltd
PO 366 Highgate
South Australia 5064

ρE

Statistical Analysis of Radionuclide Survey Data



Ray Correll

Prepared for Dr Barbara Radcliffe
Olympic Dam EIS

September 2010
Report 29/10

Disclaimer

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Acknowledgement

Rho Environmetrics gratefully acknowledges the cooperation given by the Olympic Dam EIS staff, and particularly the comments provided by Dr Barbara Radcliffe on a draft of this report.

Summary

Radionuclide data from the Olympic Dam area have been analysed, and it is shown that concentrations of Th-234 and Ra-226 are higher nearer the plant than further away.

Two statistical models are presented together with 3 methods of handling below-detection values – parameter estimates differ between the models but the significance is consistent. Th-234 and Ra-226 appear to be potential contaminants. Be-7 concentrations are depressed near the plant and although they are related to distance, Be-7 is not considered a contaminant.

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Introduction

Olympic Dam EIS has collected samples of foliage and analysed them for both metal and radionuclide content. This report considers data from analysis of *Acacia aneura* (mulga) from four transects radiating in cardinal directions from the treatment plant. In common with the previous report on heavy metals (Correll, 2010), this report considers two models for the relationship between radionuclide concentration and distance – a negative exponential model and a linear model based on the inverse of distance from the plant (Correll, 2010).

Many of the observations of the radionuclides were below the quantification level (QL). This posed problems in the analysis particularly as the majority of those measures were at distances up to 100 km from the potential source of concentration. The <QL data rendered estimating a background level difficult. Three different methods of estimating the <QLs and the most satisfying results were obtained using a Cu surrogate method for Th-234, Ra-226 Where the surrogate method was not feasible and there were values <QL, the simple substitution of QL/2 has been used for testing. An alternative maximum likelihood method was found to give a comparable results and these are included in an appendix.

The radionuclides analysed were Th-234, Ra-226, Pb-210, Ra-228, K-40 and Be-7. Of these, two radionuclides (Th-234 and Ra-226) provided indication of potential impact and discussion has been concentrated on those two analytes.

Data

The data included analyses of seven radionuclides in *Acacia aneura* foliage which had been collected at varying distances up to a nominal 100 km north, east, south and west of the Olympic Dam facility.

Distance data (calculated from GPS co-ordinates) were supplied for both radionuclide and metal data.

In many cases data were assessed as being below the quantifiable limit (<QL). This implies that that sample had a concentration between zero and the QL. In the other cases an exact estimate or quantification of the concentration was available.

Quality control samples were in two forms

- Field duplicates (two series of 10 samples taken 10 km south of the facility and 10 taken 10 km west)
- Check samples sent to a second laboratory to provide a measure of the quality of the assays.

The data for the assessment of laboratory variation were provided on a dry weight basis but the other data were on a wet weight basis. The wet/dry ratio was supplied enabling all data to be expressed on a dry weight basis.

Statistical Methods

Handling < QL

Values <QL occurred in Th-234, Th-228, Ra-228 and Ra-226. As explained in Appendix C, the <QL values for Th-234 and Ra-226 were replaced using the surrogate method and the Th-228 and Ra-228 were replaced using the QL/2 method. There were no <QL values for Pb-210, K-40 and Be-7.

Statistical models of concentration from a point source

There are many statistical models that may be fitted to the data – these include an exponential decay from the source and an inverse model: both these models offer an estimate of the background level.

Exponential decay model

This model has the form

$$\ln(Y_d) = a + b \times \exp(- k \times d)$$

where Y_d is a measure of the concentration d km from the source;
a is a background concentration (log scale);
b is a potential increase in concentration near the source (log scale);
d is the distance from the plant (km); and
k is parameter determining the rate of decline away from the source.

Inverse model

The inverse model has the form

$$\ln(Y_d) = a + b / d$$

where Y_d is a measure of the concentration d km from the source;
a is a background concentration (log scale); and
b reflects the potential increase in concentration near the source (log scale).

This model is linear in 1/distance and is therefore easier to fit and more stable than the exponential decay model, at least for moderate distances. However b has no immediate interpretation as its units are concentration × distance – it can be better understood as the increase in log concentration at a distance 1 km from the source. The model is unstable for small values of d and undefined at the source.

The exponential decay and inverse distance models were fitted to estimates as obtained by half QL and surrogate methods using the package using R statistical software (http://www.r-project.org) either using the ‘lm’ or ‘nls’ function. Standard errors of fitted values were obtained using the ‘predict’ function for the inverse model or by bootstrapping (described below) for the exponential decay model.

Bootstrapping is a non-parametric estimation of estimating confidence intervals. The technique involves repeatedly re-sampling with replacement to form a series of samples of the same size of the original sample. These ‘bootstrap’ samples are then used to form an empirical distribution of parameters (e.g. predicted concentration at 5

km). That empirical distribution can then be used to obtain confidence intervals of the parameter. The technique is described by Efron and Tibshirani (1993).

Results

Tabular presentation

Inverse model

A summary of the inverse model substitution for <QL values) is given in Table 1. The parameters *a* and *b* are estimates of the parameters of the inverse model as described on page 2. If the parameter *b* is constrained to be zero, the model reduces to the background parameter *a* or mean of the log concentration. That model is referred to as the null model. The change in the sum of squares when *b* is included in the model provides an F value (on 1 degree of freedom in this case as both *b* was estimated, and 51 degrees of freedom) providing a test of significance for the full model as compared to the null model.

Table 1 shows that there was an increase in the concentration of Th-234 and Ra-226 near the plant but a decrease in the concentration of Be-7 with increasing proximity. There was no statistically significant effect of distance on Pb-210, Ra-228, Th-228 and K-40.

Table 1 Parameter estimates and test of significance for inverse model for seven radionuclides. Significant increase with proximity to the plant has been indicated with red shading and significant decrease with proximity to the plant has been indicated with green shading. The parameters *a* and *b* refer to the $\log_e$ background concentration and the potential increase near the plant (Bq/kg dry weight) at a distance of 1 kilometre. Details of the models are described in the Statistical Methods section

	Th-234	Ra-226	Pb-210	Ra-228	Th-228	K-40	Be-7
Full model							
a	0.43	0.20	4.27	0.07	0.15	5.68	5.04
b	5.05	4.17	0.25	-0.86	-0.59	0.35	-1.00
s of sqs	6.96	7.78	6.23	23.03	21.26	3.38	4.69
df	51	51	51	51	51	51	51
Null model							
Mean (*a*)	1.15	0.79	4.31	-0.05	0.07	5.73	4.90
s of sqs	49.80	36.98	6.34	24.27	21.84	3.59	6.37
df	52	52	52	52	52	52	52
F	314.16	191.50	0.86	2.76	1.39	3.15	18.34
p	0.000	0.000	0.36	0.10	0.24	0.082	0.000

Exponential model

A summary of the exponential model substitution for <QL values) is given in Table 2. The parameters *a*, *b* and *k* are estimates of the parameters of the exponential decay model as described on page 2. If the parameter *b* is constrained to be zero, the model reduces to the background parameter *a* or mean of the log concentration. That model

is referred to as the null model. The change in the sum of squares provides an F value (on 2 degrees of freedom in this case as both *b* and *k* were estimated, and 50 degrees of freedom) providing a test of significance for the full model as compared to the null model.

Table 2 shows that there was an increase in the concentration of Th-234 and Ra-226 near the plant but a decrease in the concentration of Be-7 with increasing proximity.

There was no statistically significant effect of distance on Pb-210, Ra-228, Th-228 and K-40.

Comparison of models

Although the two models have a very different philosophy, the conclusions obtained were the same – there was increasing concentration near the plant for Th-238 and Ra-226, and decrease with proximity for Be-7 and no statistically significant effect of distance on Pb-210, Ra-228, Th-228 and K-40. There were differences in the estimation of the background between the models.

Table 2 Parameter estimates and test of significance for exponential model for seven radionuclides. Significant increase with proximity to the plant has been indicated with red shading and significant decrease with proximity to the plant has been indicated with green shading. The parameters *a*, *b* and *k* refer to the $\log_e$ background concentration and the potential increase near the plant (Bq/kg dry weight) and *k* is the rate of decline per kilometre. Details of the models are described in the Statistical Methods section

	Th-234	Ra-226	Pb-210	Ra-228	Th-228	K-40	Be-7
Full model							
a	0.57	0.40	4. 3	0.06	0.17	5.7	5.1
b	4.5	5.50	6.0	-0.86	-0.50	2.3	-0.74
k	0.30	0.50	2.0	0.28	0.17	1. 4	0.14
s of sqs	7.41	8.15	5.98	22.62	20.97	3.28	4.16
df	50	50	50	50	50	50	50
Null model							
Mean (*a*)	1.15	0.79	4.3	-0.05	0.06	5.72	4.90
sd	0.98	0.84	0.35	0.68	0.04	0.26	0.35
s of sqs	49.80	36.98	6.34	24.27	21.84	3.59	6.37
df	52	52	52	52	52	52	52
F	142	88	1.49	1.82	1.04	2.35	13.3
p	0.000	0.000	0.234	0.172	0.361	0.106	0.000

Graphical presentations

This section of the report provides diagrammatic presentations of key analyses. Boxplots have been included for all 7 radionuclides because of their simplicity and because they do not require any distributional assumptions.

Thorium 234

The results from Th-234 data using surrogate substitution for <QL data are shown in

Figure 1 to Figure 3. The data at a distance further than 10 km from the plant are dominated by samples that had levels below the quantifiable limit.

Figure 1 is a boxplot comparison and shows quite clearly that there were higher concentrations nearer the source than further away – there was little overlap between the 0-10 and 10-50 km groups of samples. There was strong evidence ($p < 0.001$) of a focus of elevated Ra-226 levels near the plant with declining values with distance (Table 1 and Table 2) as indicated by the F test.

The blue line shown in Figure 2 shows the inverse model fits the high values near plant very well. The exponential decay model (Figure 3) shows a similar pattern but approaches an asymptote more rapidly. The fit away from the plant is basically background estimate (parameter *a*).

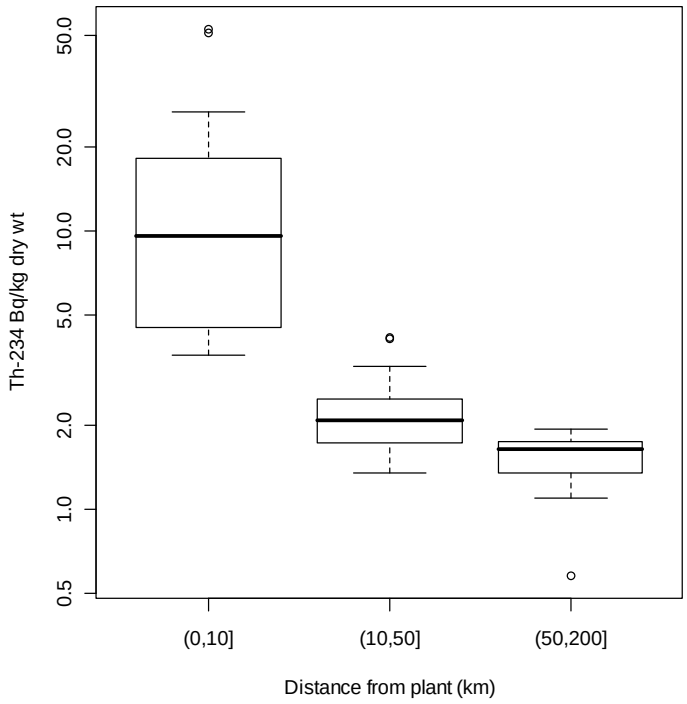

Figure 1 Boxplot of Th-234 classified into groups defined as 0 - 10 km, 10 - 50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

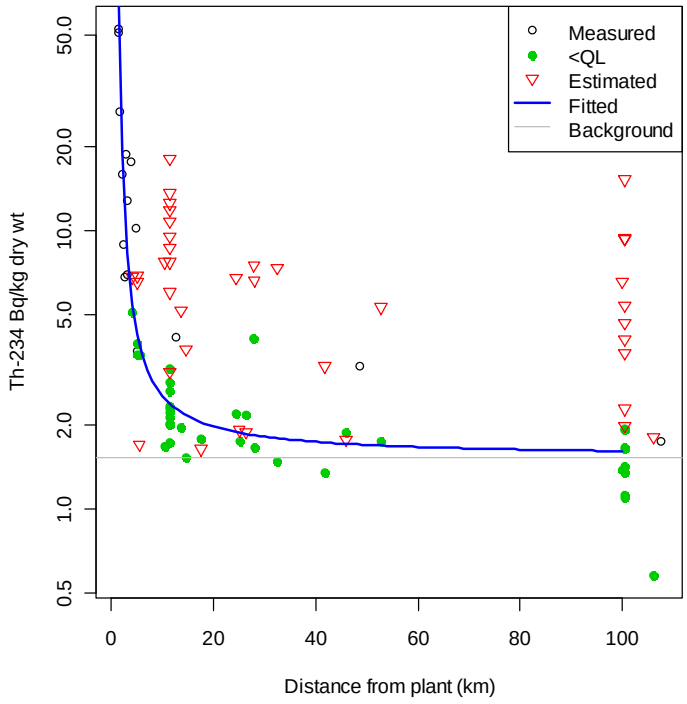

Figure 2 Effect of increasing distance from the plant on Th-234 concentrations using the inverse model and surrogate substitution for <QL values. The green dots were obtained using the surrogate method and the red triangles indicate quantifiable limit, suggesting that the true value is less than that amount

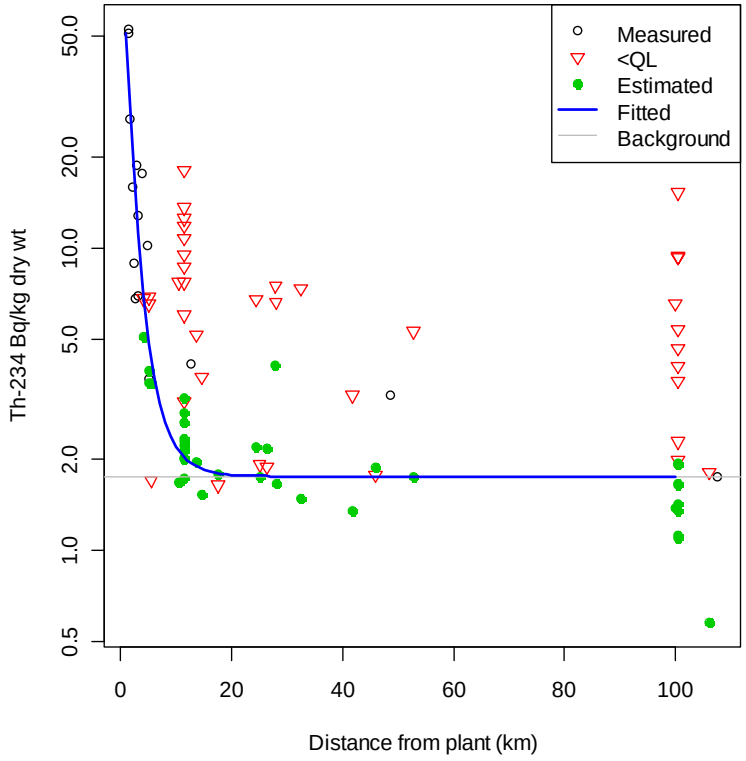

Figure 3 Effect of increasing distance from the plant on Th-234 concentrations using the exponential decay model and surrogate substitution for <QL values. The green dots were obtained using the surrogate method and the red triangles indicate quantifiable limit, suggesting that the true value is less than that amount

Radium 226

The boxplot Figure 4, like that for Th-234, showed a strong trend between the distance classes. There was strong evidence ($p < 0.001$) of a focus of elevated Ra-226 levels near the plant with values declining with distance (Table 1 and Table 2) as indicated by the F test.

The blue line shown in Figure 5 shows the inverse model fits the high values near plant very well. The exponential decay model (Figure 6) shows a similar pattern but approaches an asymptote more rapidly. The fit away from the plant is basically background estimate (parameter *a*).

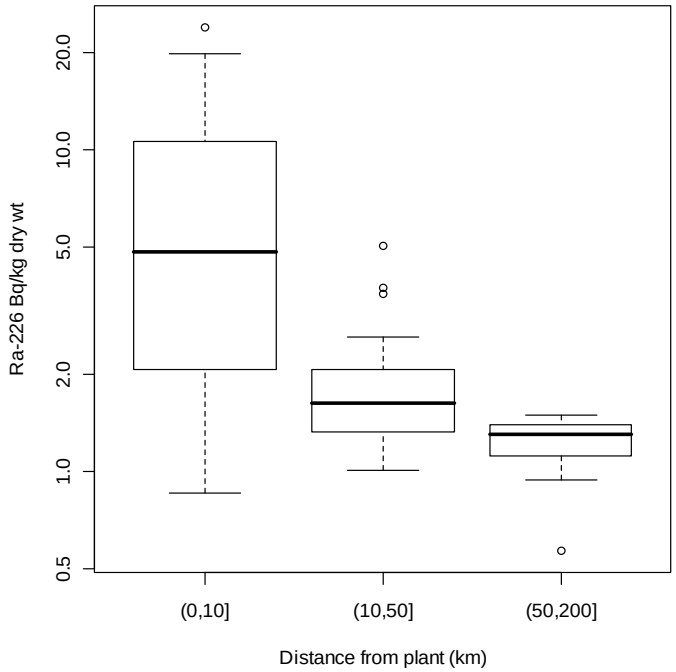

Figure 4 Boxplot of Ra-226 classified into groups defined as 0 - 10 km, 10 - 50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

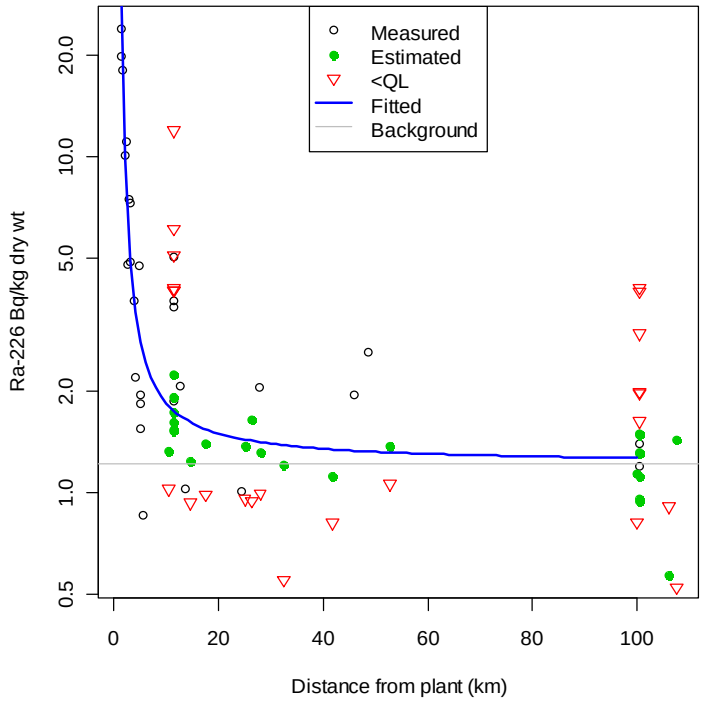

Figure 5 Effect of increasing distance from the plant on Ra-226 concentrations using the inverse model and surrogate substitution for <QL values. The green dots were obtained using the surrogate method and the red triangles indicate quantifiable limit, suggesting that the true value is less than that amount

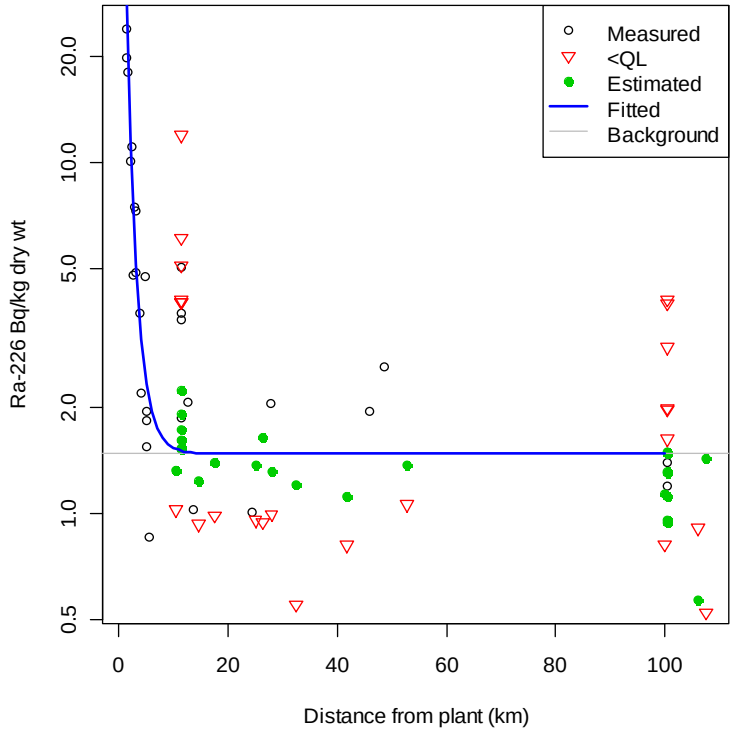

Figure 6 Effect of increasing distance from the plant on Ra-226 concentrations using the exponential decay model and surrogate substitution for <QL values. The green dots were obtained using the surrogate method and the red triangles indicate quantifiable limit, suggesting that the true value is less than that amount

Pb-210

Figure 7 shows boxplots of the Pb-210 data. That plot shows that there is a large range of variability in the data but it does not support any trend with distance.

There was no evidence of changing levels of Pb-210 with distance from the plant from either from the inverse or the exponential decay models (Table 1, Table 2 and Figure 8). These data were all above the quantifiable limit and the reported laboratory error was small (< 10%). Although these data offer no evidence of an influence of the site on the level of Pb-210 there was considerable uncertainty in the estimates.

Other analyses were performed (results not presented in this report) included fitting a trend to the northing and easting data, but there were no statistically significant effects found.

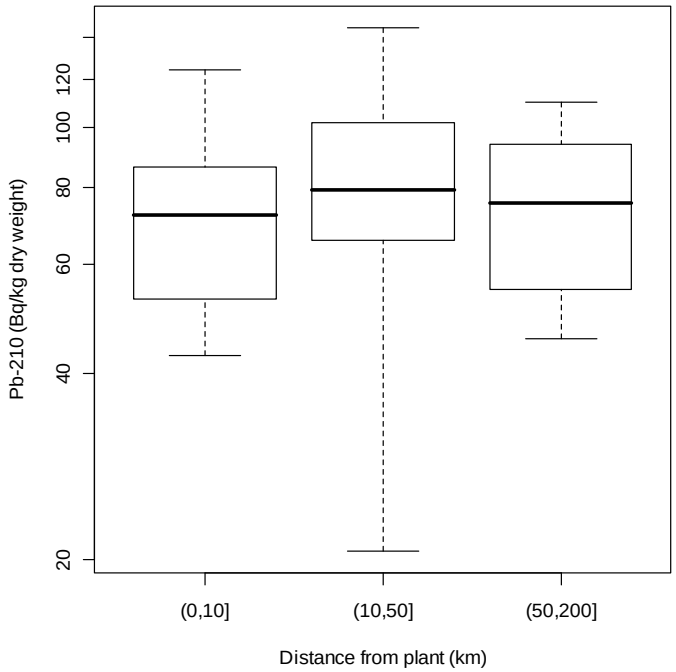

Figure 7 Boxplot of Pb-210 classified into groups defined as 0 - 10 km, 10 - 50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

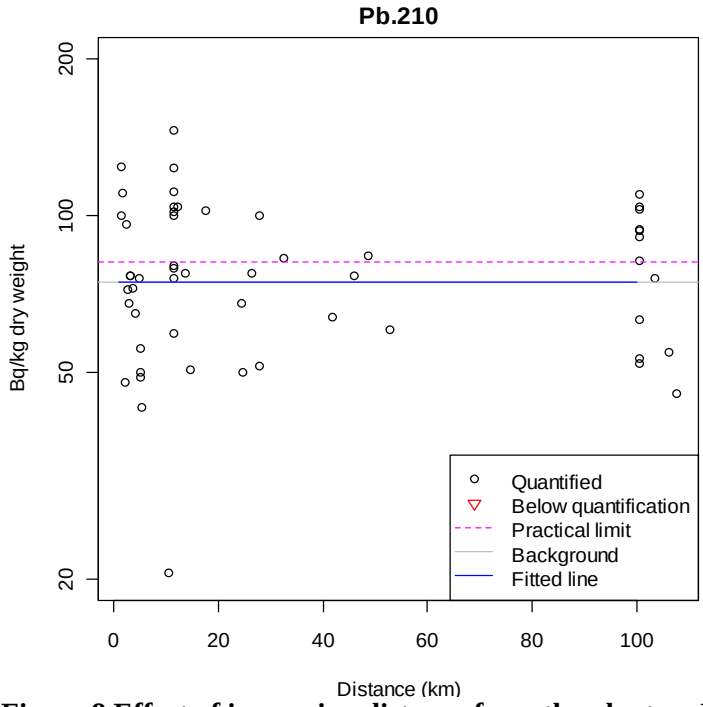

Figure 8 Effect of increasing distance from the plant on Pb-210 concentrations

Ra-228

There was no statistically significant effect found by using the inverse or the exponential decay models (Table 1 and Table 2). There was however a suggestion from the boxplots (Figure 9) that there was a decrease in concentration with proximity to the site.

An examination of Figure 10 shows that the influential points were often substitutions where the values were <QL. Little inference can therefore be drawn from the Ra-228 data.

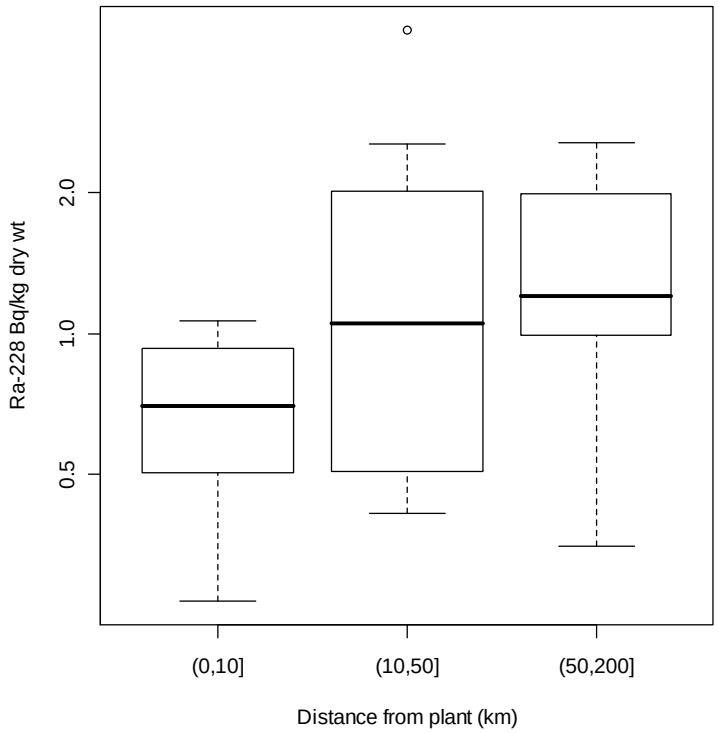

Figure 9 Boxplot of Ra-228 classified into groups defined as 0 - 10 km, 10 -50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

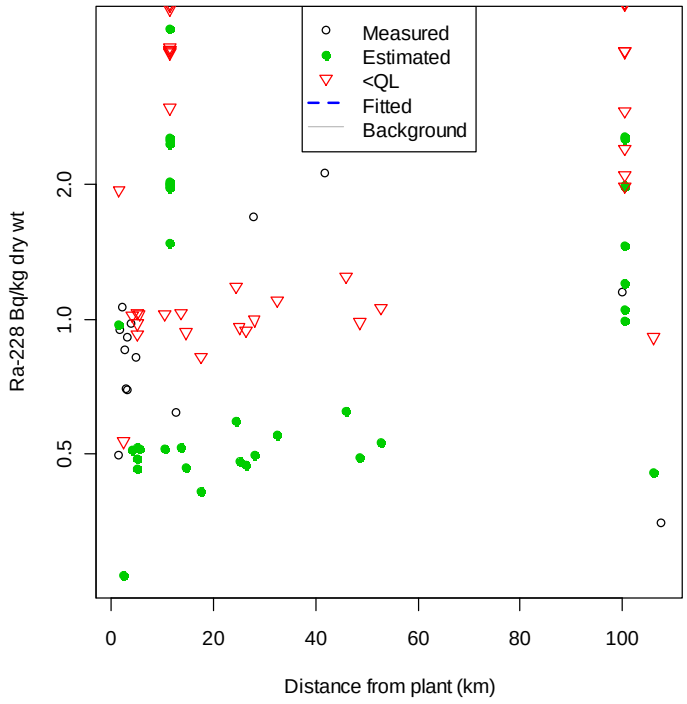

Figure 10 Effect of increasing distance from the plant on Ra-228 concentrations. The green dots were obtained using half the QL method and the red triangles indicate quantifiable limit, suggesting that the true value is less than that amount

Th-228

The boxplot of the Th-228 data (Figure 11) had a suggestion of decreasing concentrations with proximity to the site. There were several high values which were more than 10 km from the plant. Statistical analysis of the data (Table 1 and Table 2) indicated no significant trends. Both the inverse (Figure 12) and the exponential decay (Figure 13) models showed a decreased concentration nearer the plant, but the trend was not statistically significant.

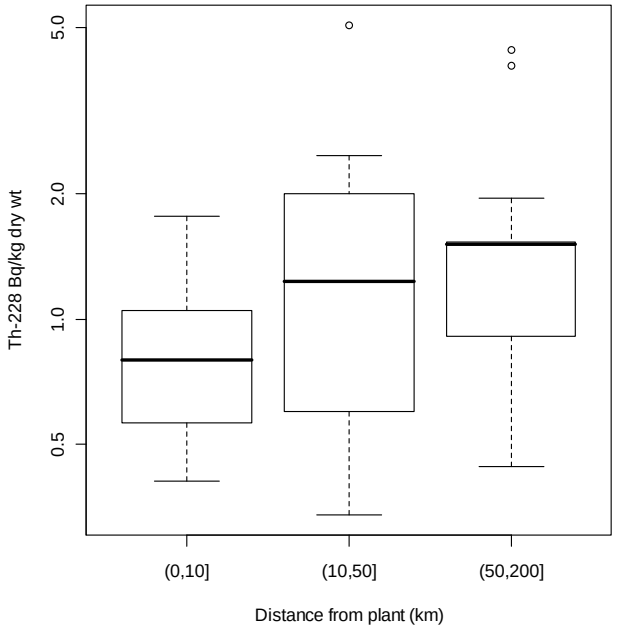

Figure 11 Boxplot of Th-228 classified into groups defined as 0 - 10 km, 10 - 50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

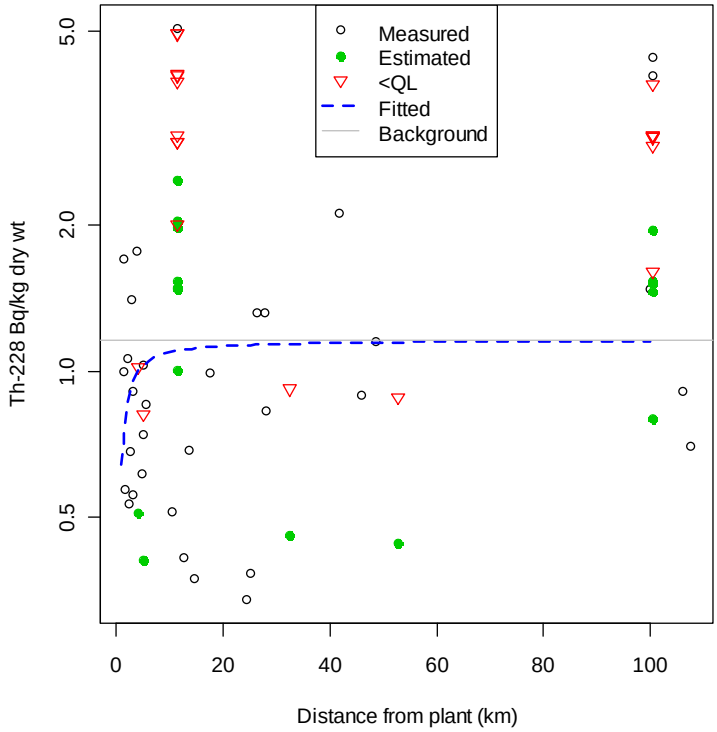

Figure 12 Effect of increasing distance from the plant on Th-228 concentrations using the inverse model and 0.5 QL substitution for <QL values. The fitted line has been shown as broken because it did not differ significantly from the null model

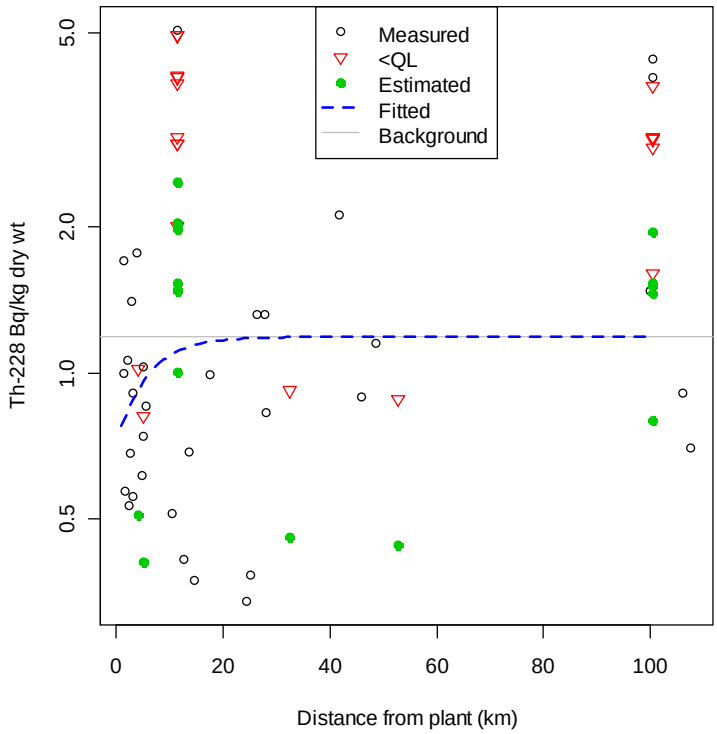

Figure 13 Effect of increasing distance from the plant on Th-228 concentrations using the exponential decay model and 0.5 QL for <QL values. The fitted line has been shown as broken because it did not differ significantly from the null model. The green dots were obtained using the half QL method and the red triangles indicate quantifiable limit, suggesting that the true value is less than that amount

K-40

There was no statistical evidence of changing levels of K-40 with distance from the plant ((Table 1 and Table 2). In common with Pb-210, all values were higher than the QL. A boxplot representation of the data (Figure 14) showed no consistent trend. The boxplot showed that there was one high value near the plant. This very high value had both a high wet weight reading (310) and a very high wet/dry ratio of 2.43.

Neither the inverse (Figure 15) nor exponential decay (Figure 16) models was statistically significant.

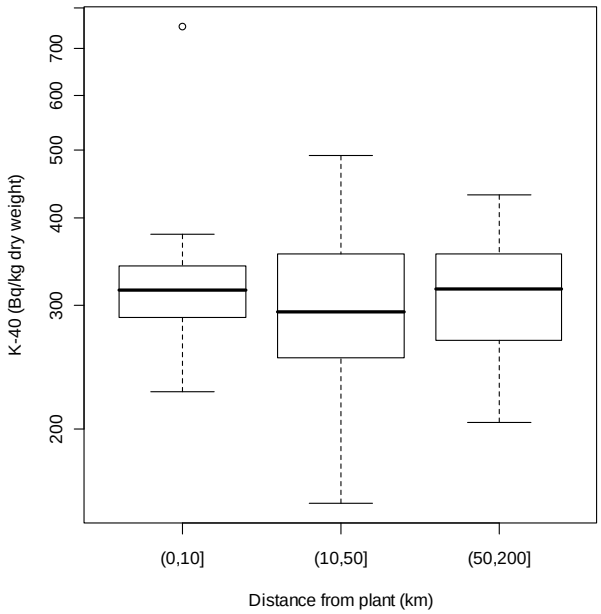

Figure 14 Boxplot of K-40 classified into groups defined as 0 - 10 km, 10 - 50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

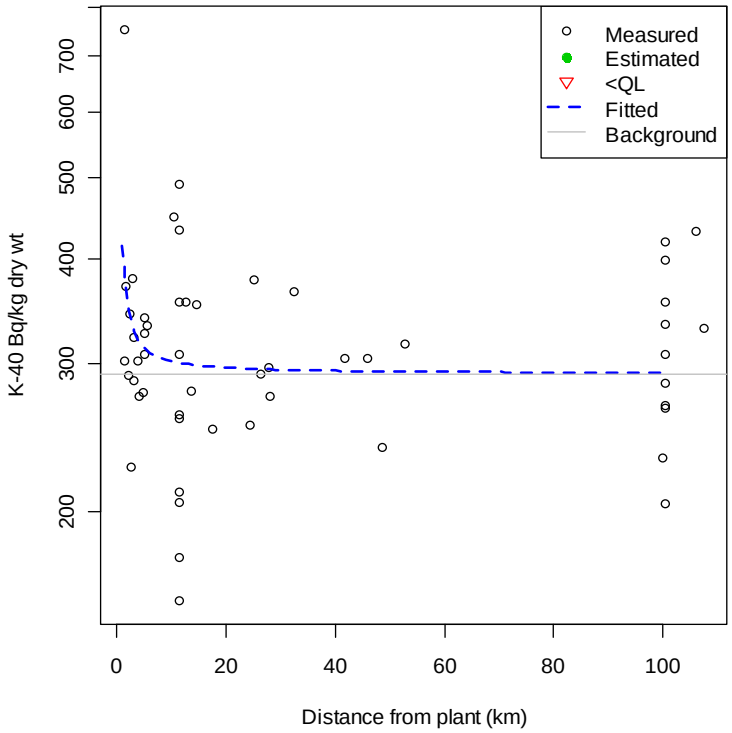

Figure 15 Effect of increasing distance from the plant on K-40 concentrations using the inverse model. The fitted line has been shown as broken because it did not differ significantly from the null model

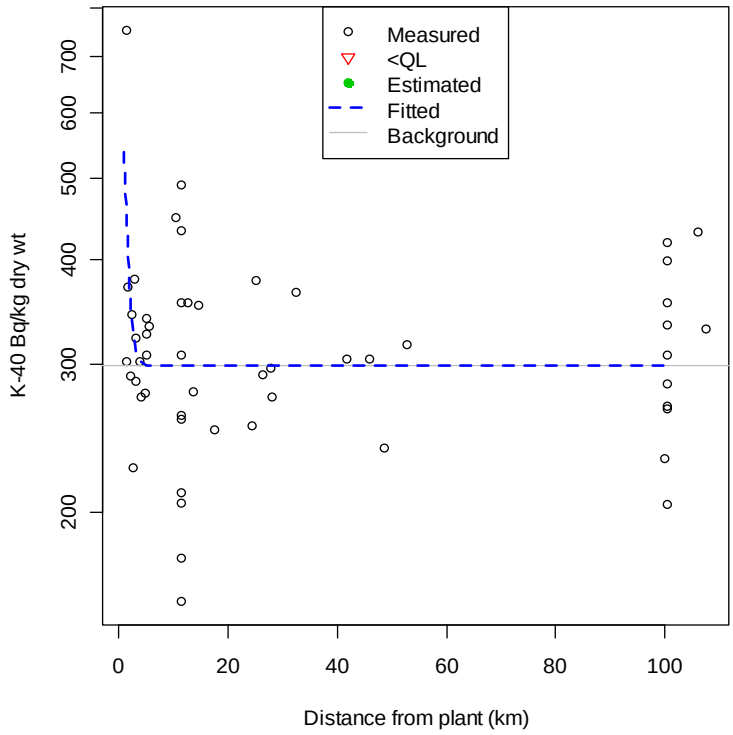

Figure 16 Effect of increasing distance from the plant on K-40 concentrations using the exponential decay model. The fitted line has been shown as broken because it did not differ significantly from the null model

Be-7

There was strong evidence of there being less Be-7 in foliage near the plant than further away (Table 1 and Table 2). There were no <QL values for Be-7, so there were less assumptions required. That makes the significance of the result very credible.

Both the inverse (Figure 18) and the exponential decay model (Figure 19) showed a significant decrease in concentration with increasing proximity to the plant.

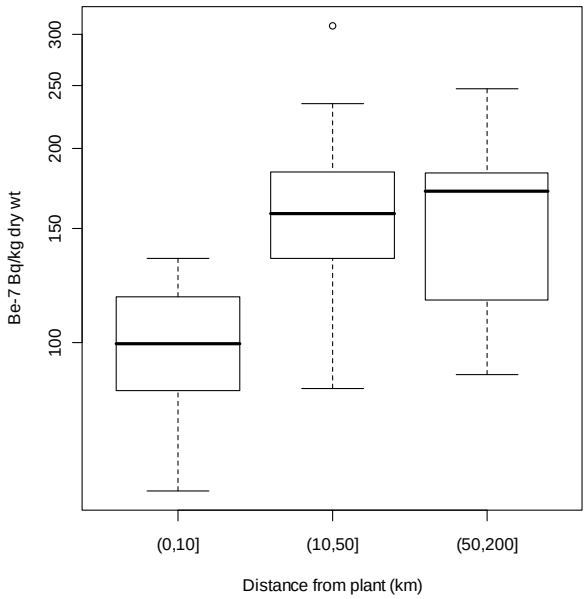

Figure 17 Boxplot of Be-7 classified into groups defined as 0 - 10 km, 10 -50 km and >50 km from the plant. The groups had 16, 24 and 13 samples respectively. The box includes the centre 50% of samples in that class, with the heavy line representing the median and the bars showing the range of values.

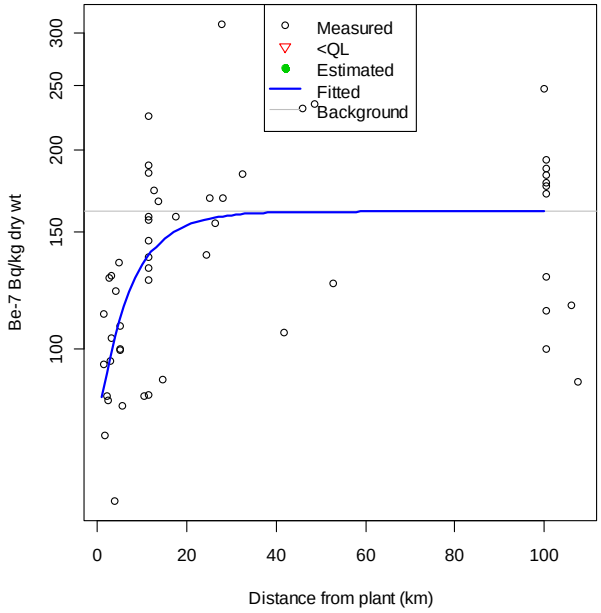

Figure 18 Effect of increasing distance from the plant on Be-7 concentrations. The fitted line is the inverse model

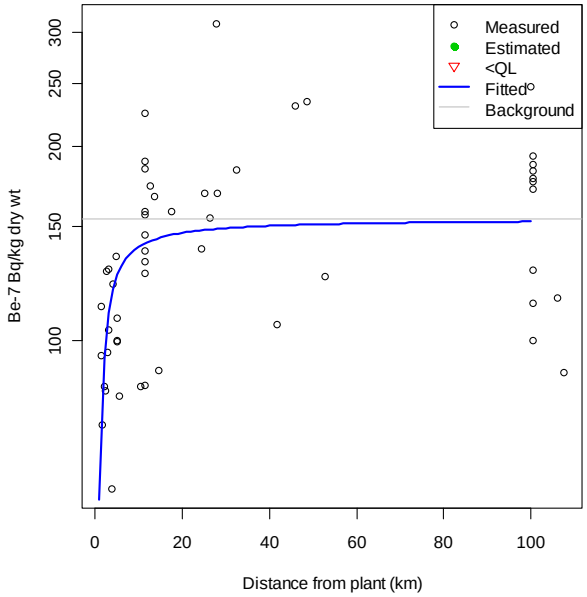

Figure 19 Effect of increasing distance from the plant on Be-7 concentrations using the exponential decay model

Predictions of radionuclide concentration levels

Inverse model

The log inverse model using the Cu surrogate approach was used to estimate levels of radionuclide concentration at given distances from the plant. The 'predict' function in R was then used to estimate log concentrations at various distances from the plant. Confidence limits were calculated on the log scale and exponentiated to give estimates in Bq/kg dry weight.

The estimates and confidence limits are shown in Table 3. The estimates at 1 km represent extrapolations as the nominal 1 km samples were all taken further than 1 km from the site for operational reasons. The 1 km predictions are unreliable and should not be used for decisions-making.

Table 3 Estimates of radionuclide concentration levels of Th-234 and Ra-226 at various distances from the centre of the plant using surrogate method for below detection and the inverse model. The units are Bq/kg dry weight.

Distance (km)	**Th-234**			**Ra-226**		
	Lower CL	Estimate	Upper CL	Lower CL	Estimate	Upper CL
1	146	240	397	46	79	134
3	7.1	8.3	9.6	4.2	4.9	5.7
5	3.8	4.2	4.7	2.5	2.8	3.1
10	2.3	2.6	2.8	1.7	1.8	2.1
25	1.7	1.9	2.1	1.3	1.4	1.6
Background	1.4	1.5	1.8	1.1	1.2	1.4

Similar confidence intervals (not shown) were also obtained for the inverse mode using bootstrapping.

Exponential model

The estimates and bootstrapped confidence limits are shown in Table 4. The estimates at 1 km represent extrapolations as the nominal 1 km samples were all taken further than 1 km from the site. The 1 km predictions are unreliable and should not be used for decisions-making.

Table 4 Estimates of radionuclide concentration level of Th-234 and Ra-226 at various distances from the centre of the plant using surrogate method for below detection and the exponential model. The units are Bq/kg dry weight.

Distance (km)	Th-234			Ra-226		
	Lower CL	Estimate	Upper CL	Lower CL	Estimate	Upper CL
1	22.9	52.4	82.3	17.1	47.7	91.5
3	8.2	11.0	14.0	4.2	6.1	13.5
5	3.5	4.9	6.5	1.95	2.9	6.2
10	1.9	2.2	2.7	1.39	1.7	2.6
25	1.5	1.8	2.0	1.30	1.6	2.0
Background	1.5	1.7	2.0	1.28	1.6	2.0

Discussion

Role of radon

Radon (Rn-222) is an inert gas that is part of the decay series shown in Table 5. A decay product of Rn-222 is Pb-210, which because of its relatively long half life, has the potential to accumulate. Elevated levels of Rn-222 could thus lead to measureable concentrations of Pb-210.

Rn-222 naturally emanates from soils and can be considered to be a transport mechanism for radionuclides which are produced from the decay of Rn-222 (such as Pb-210 and Po-210) into the environment.

Elevated concentrations of Pb-210 occur naturally and UNSCEAR (2000) reports relative concentrations of radionuclides from the U-238 decay chains as summarised in Table 6.

Table 5 Decay products and half lives of radon (Rn-222)

Nuclide	Isotope	Decay type	Half life	Decay product	
Radon	Rn-222	α	3.824 d	Polonium	Po-218
Polonium	Po-218	α	3.05 m	Lead	Pb-214
Lead	Pb-214	β	26.8 m	Bismuth	Bi-214
Bismuth	Bi-214	β	19.8 m	Polonium	Po-214
Polonium	Po-214	α	164 µs	Lead	Pb-210
Lead	Pb-210	β	22.3 y	Bismuth	Bi-210
Bismuth	Bi-210	β	5.01 d	Polonium	Po-210
Polonium	Po-210	α	138.38 d	Lead	Pb-206
Lead	Pb-206	Stable			

Table 6 Relative concentrations of 5 radionuclides from the U-238 decay chain (after (UNSCEAR 2000)

Radionuclide	Relative concentration in air
U–238	1
Th-230	0.5
Ra-226	1
Pb-210	500
Po-210	50

An elevation in Pb-210 and Po-210 concentrations could therefore be caused by an elevation of the concentration of Rn-222. Rn-222 concentrations are therefore a useful indicator of possible elevated levels of Pb-210 due to the operation.

Confounding of spatial relationships

While there have been very significant spatial relationships noted for Th-234 and Ra-226 (and also for Be-7 but in the reverse direction), the relationships cannot be directly linked to the plant operation through analysis of those data alone. Local geological anomalies are known to affect properties of vegetation (see for example Correll and Taylor 1974). It is possible that some of the observed effects could be due to local geological features.

Criteria for boundary of the footprint

Currently there would appear no objective criteria for determining the footprint of the effect of the operation using the distribution of radionuclides.

The highly variable natural levels of radionuclides make it difficult to set criteria for determining trigger levels. There may be scope for having a variable background level for radionuclides in a manner similar to that suggested by Hamon and McLaughlin (2004) for metals in soils.

Naturally high levels of radionuclides have been reported in food from Nigeria near a tin mining site (up to 83.5 Bq/kg for Ra-226 and 89.6 Bq/kg for Th-228). The authors of that work considered those dose levels as low and harmful health effects would not be expected (Jiribi *et al*. 2007). However, Ra-226 is known to bio-accumulate in

shellfish, so there is potential for bio-accumulation (Supervising Scientist 2007). This aspect needs to be considered.

Comparison of models

A comparison of the parameters resulting from using the three methods of handling the below detection values and two models is shown in Table 7. While there was general agreement between the results, there were sufficiently large differences between the methods to remind the user that the model selection can have an effect on the parameter estimates. Much of the problem is in the estimation of background where there are in fact very few data for either isotope.

The comparison of the predicted concentrations (Table 8) is more encouraging with reasonable agreement between the models at the 3 and 5 km distances. The inverse model using the Cu surrogate method typically returned a prediction near the median of the other methods. This is taken as support for choosing that combination as the preferred method of estimation. Note that the estimates given in Table 4 differ slightly as they were the median of bootstrapped samples rather than direct estimates from the model.

A further positive comparison was in the significance testing, which identified Th-234 and Ra-226 activity as declining with distance from the site and also identified Be-7 as being lower near the plant than at distance with all the methods.

No weighting of the values was used in the analysis. Changing the weights does not generally cause a bias to the data – it generally affects the efficiency of the estimates of the parameters. Weights that are not optimal yield less efficient estimates of the parameters.

In view of the differences between the models, the confidence limits quoted in Table 3 may be too narrow as they do not take into account model uncertainty. Fortunately the most consistent part of the modelling includes the distances of most concern, namely 3 -10 km.

Table 7 Comparison of parameters for 2 nuclides, using 3 methods of handling missing values (ml = maximum likelihood, sur=Cu surrogate, and 0.5 replacing below detections with half the detection limit) and two models (exponential and inverse distance)

Radionuclide	**Method**	**Exp(ml)**	**Inv(ml)**	**Exp(sur)**	**Inv(sur)**	**Exp(0.5)**	**Inv(0.5)**
Th-234	Background (Bq.kg^{-1} dry weight) *a*	0.261	0.221	0.566	0.431	0.992	0.786
	Maximum impact (Bq.kg^{-1} dry weight) *b*	5.364	5.426	4.544	5.051	5.696	4.224
	k (km^{-1})	0.322		0.300		0.514	
	Method	**Exp(ml)**	**Inv(ml)**	**Exp(sur)**	**Inv(sur)**	**Exp(0.5)**	**Inv(0.5)**
Ra-226	Background (Bq.kg^{-1} dry weight) *a*	-0.205	-0.274	0.395	0.198	-0.118	-0.103
	Maximum impact (Bq.kg^{-1} dry weight) *b*	4.457	5.26	5.54	4.17	3.338	4.931
	k (km^{-1})	0.266		0.499		0.170	

Table 8 Comparison of predictions of the concentration (Bq g^{-1} dry weight) for 2 radionuclides, using 3 methods of handling missing values (ml = maximum likelihood, sur=Cu surrogate, and 0.5 replacing below detections with half the detection limit) and two models (exponential and inverse distance)

Radio-nuclide	Distance	Exp(ml)	Inv(ml)	Exp(sur)	Inv(sur)	Exp(0.5)	Inv(0.5)
Th-234	1	4.15	5.65	3.93	5.48	4.40	5.01
	3	2.30	2.03	2.41	2.11	2.21	2.19
	5	1.33	1.31	1.58	1.44	1.43	1.63
	10	0.48	0.76	0.79	0.94	1.03	1.21
	25	0.26	0.44	0.57	0.63	0.99	0.95
	Back-ground	0.26	0.22	0.57	0.43	0.992	0.786
Ra-226	1	3.21	4.99	3.76	4.37	2.70	4.83
	3	1.80	1.48	1.63	1.59	1.89	1.54
	5	0.97	0.78	0.85	1.03	1.31	0.88
	10	0.11	0.25	0.43	0.62	0.49	0.39
	25	-0.20	-0.06	0.40	0.36	-0.07	0.09
	Back-ground	-0.20	-0.27	0.40	0.20	-0.12	-0.10

Effect of washing of foliage

The current data were based on unwashed samples of foliage on the basis that grazing animals would consume the total load including the foliage and the dust on the foliage. There is merit in this argument.

The use of unwashed foliage does pose some challenges to interpretation. For example, if there were an inert layer of dust on the foliage that could conceivably increase the dry weight and thus dilute potential concentration. That effect is likely to be small.

A more likely possibility is that there is concentration in the dust on the foliage – this could be a source of heavy metals and radionuclides. In the case of *A. aneura* this dust could be lost with a shower of rain. Conversely it could accumulate over time so that the analyte would accumulate between rain events (particularly near access tracks). If that is the case, there could be a large effect of time since the last rain on contaminant levels.

Rain in that area includes thunderstorms – these can be intense and localised. A local thunderstorm therefore has the potential to create differential reductions in surface concentration. This problem could be investigated by washing foliage from an area close to the workings (e.g. 3 km distance) and analysing unwashed foliage, washed foliage and the washing water, thus obtaining information as to whether the concentration is based on the surface or whether it has been incorporated into the foliage.

Conclusions

There is evidence that there are elevated levels Th-234 and Ra-226 close to the operation, but the exact extent of the footprint cannot be defined until a working definition of its extent has been accepted

There is a depression of Be-7 near the plant.

Estimation of background levels and the extent of the spread is very dependent on the choice of the model used and where there are values less than the quantifiable limit, the estimates are affected by how those values are handled.

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Appendix A Transformation of the data

A common assumption in statistical modelling is that the variance about the fitted model remains constant. Often the variability increases as the fitted values increase. A transformation of the data is often used to stabilise the variance. Two common transformations are the square root and logarithmic. When the variance of the data increases proportionally to the predicted mean, a square root transformation is appropriate; a classical case of this is where there are count data. When the standard deviation is proportional to the fitted mean (that is, the coefficient of variation remains constant) a logarithmic transformation can be used to stabilise the variance.

Radionuclide data are often transformed to a square root scale when they are measured by counts per unit time per kg of material. This is expressed as Bq (disintegrations) per kg of material. However, often the sampling error dominates over the counting error, and this then requires a stronger transformation. More frequently the standard deviation is approximately proportional to the mean in which case a logarithmic transformation is appropriate.

Figure 20, Figure 21 and Figure 22 show plots of the standard error (as supplied in the data) against the laboratory mean for Pb-210, K-40 and Be-7. The relationship is ambiguous for Pb-210 but for and Be-7, andK-40 the relationship appears weakly linear, implying the standard deviation is proportional to the mean (or constant coefficient of variation). A logarithmic transformation was therefore appropriate to give constant variance on the transformed scale. A logarithmic transformation is also consistent with the previous report on chemicals that may be of potential concern (Correll 2009).

Many statistical tests are robust against changes in the variance across the modelled values. This may not be so when the variance is used in modelling the maximum likelihood estimates where there are many values that are <QL.

Backtransformed means are usually biased. The correction for bias is non-trivial and dependent on assumptions such as the transformed data had normal residuals and that the variance is known. The validity of the assumptions cannot be guaranteed, and deviation from them can have a large effect. No attempt has been made to correct for the backtransformation bias.

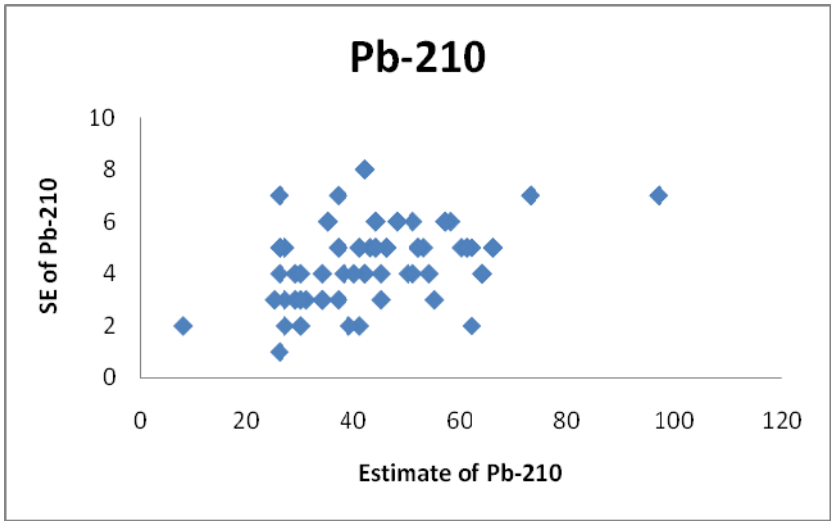

Figure 20 Relationship between reported standard error and estimate for Pb-210

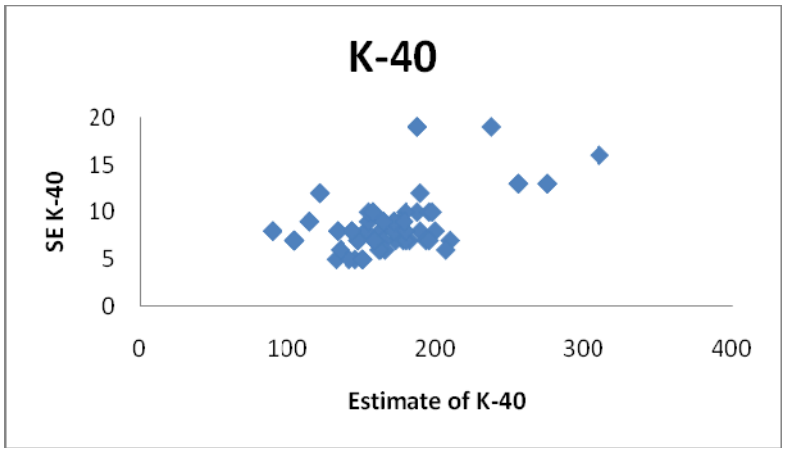

Figure 21 Relationship between reported standard error and estimate for K-40

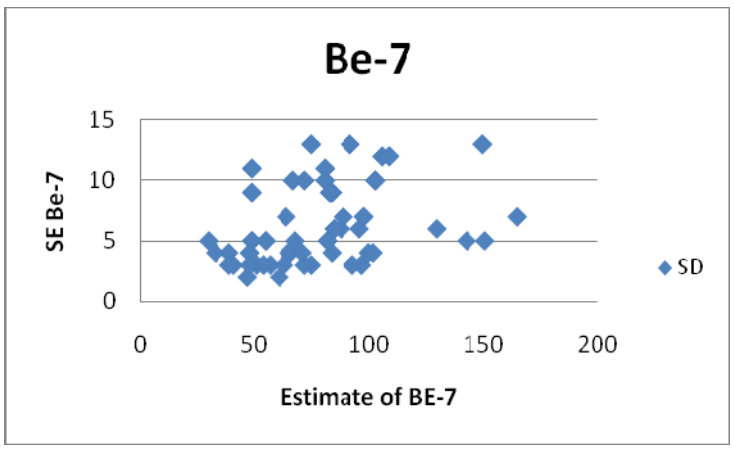

Figure 22 Relationship between reported standard error and estimate for Be-7

Appendix B Laboratory duplicates of analyses

	No.	Th-234	Ra-226	Pb-210	Ra-228	Th-228	K-40	Be-7
S10-4	1	6 ± 10	< 4	146 ± 16	< 4	5 ± 3	156 ± 17	226 ± 32
	2	2 ± 1	< 2	143 ± 13	< 4	< 3	175 ± 22	184 ± 22
W100-2	1	< 10	1.4 ± 0.8	63 ± 11	< 4	< 3	398 ± 22	115 ± 13
	2	< 10	< 2	59 ± 18	< 4	< 3	378 ± 25	130 ± 25
W10	1	< 8	< 1	21 ± 6	< 1	0.6 ± 0.4	450 ± 23	86 ± 9
	2	< 5	< 1.4	45 ± 9	< 1.8	< 1.3	524 ± 41	93 ± 14
	3	< 8	< 1.6	24 ± 7	< 1.7	0.7 ± 0.8	449 ± 23	95 ± 12

Appendix C Handling <QL values

The data contain numerous 'below detection' values and these present a challenge in the analysis. There are non-parametric methods of handling the below-detection levels (e.g. using a Kaplan-Meier approach as suggested by Lee and Helsel (2007). Other methods use a maximum likelihood approach, which assumes some underlying distribution of the data (e.g. log normal). Two alternative methods are also considered below.

Maximum likelihood approach

The maximum likelihood approach was one method used in this study to handle the <QL data.. It requires combining two components. The first part is a classical maximum likelihood of a sample of values taken from a normal distribution which can be described as $N(f(\theta;Distance), \sigma^2)$, where $f(\theta)$ is a model (either the negative exponential or the inverse distance) of the expected value of the log (base e) of the contaminant level, and θ represents the parameters; this is applied to the quantified data. In addition, the probability of obtaining an observation less than a given amount (i.e. below the quantification level) can also be estimated. The combined log of the likelihood can then be estimated. A constrained model, where some components of θ were set to zero was also estimated. Twice the difference between the log likelihoods is distributed as a χ^2. These estimates were obtained using the 'Solver' tool in Excel. No closed solution for the standard error of the parameters was available using this method.

The estimation using the maximum likelihood method makes assumptions that include that

1. The distribution is known (assumed in this case that he log of the residual has a normal distribution);
2. The observations are independent; and that
3. The variance is constant across the data set.

These assumptions are in addition to the assumption that the proposed model (exponential decay or inverse) was in fact the correct model.

There is no way, especially when a high proportion of the data are <QL that the assumptions listed above can be validated. The maximum likelihood estimate may be sensitive to departures from these assumptions. It would therefore be imprudent to rely solely on this approach.

Substitution of half QL

A simple method for handling <QL is to replace all the <QL values with half the QL. The rationale for this method is that the true value could be anywhere between QL and zero. QL/2 is then a mid-range value. The method in effect assumes a symmetric distribution bounded between zero and QL. This is a weaker assumption than the method of maximum likelihood.

The current data set has a variable QL, so different values are substituted, depending on the QL. The method has the advantage over the maximum likelihood method in that it uses this information.

Use of Cu as a surrogate

This section of the report explores the possibility of using Cu as a predictor of radionuclides. The copper data used are those analysed in Correll (2010), and came from samples of *A. aneura* collected at the same time and locations as these samples. The two nuclides that may be of potential concern are Th-234 and Ra-226.

There is some physical rationale for the use of Cu as a surrogate if it is assumed that the method of transports of all three analytes is through mineral dust.

Personal experience is that field sampling is generally the largest component to the overall variance, and that this is followed by sub-sampling in the laboratory. Generally chemical analyses are relatively precise and contribute only a small amount to the overall variance. In the current case, the same field sample would have been used for both chemical analysis and the radionuclide analysis but there would have been a sub-sampling variance component contributing to the variance of Cu estimates and to the radionuclide estimates. As well as the above there would have been a variance component from the digestion and measurement for the Cu assay on the one hand and variation and sample preparation and counting variance from the radionuclides. There was therefore no apparent reason to assume that the precision of the Cu analysis is better than that of the radionuclide data.

Because there was no reason to assume that the precision of the Cu assay and that of the radionuclides differed, they were assumed to be approximately equal. There was then no reason as to whether the Cu should be regressed against the radionuclide data, or whether the radionuclide data should be regressed against the Cu data. The compromise was to use a regression that assumed that the precision of the two assays was similar.

Simple linear regressions were fitted to both data sets – the slopes of these regressions will be biased toward zero because of the sampling error in the measurement of Cu concentration. The alternative forms of the regression that assumes equal error in both the Cu and radionuclide measures are also shown – these have steeper slopes, and are indicated by dashed lines. This approximation of equal relative precision of both variables was preferred so the dashed regression lines were used in subsequent analyses.

Diagrammatic representations of the relationships are shown in Figure 23. The correlation was strong for both radionuclides where data were available (R^2 = 0.873 and 0.733 for Th-234 and Ra-226) (Table 9). The adjusted R^2 (which takes into account the number of degrees of freedom used in the fitting process) was also high for each radionuclide. The red triangles on the graph indicate the below detection levels – as would be expected these are predominantly above the regression line. The above is conditional on the log Cu to log Th-234 line continuing to be straight. In Figure 12 we further test the assumption that the Cu surrogates are unbiased.

The graphs on the right hand side of Figure 23 include estimates of the below detection values based on half the detection limit (QL/2, where QL is the quantifiable limit). For Th-234 (upper graphs), the green dots are predominantly above the line, so had they been used as estimates of Th-234, there would have been a bias upward. This

contrasts with Ra-226 (lower graphs) where the green points are centred on the line and so using QL/2 is a fair approximation.

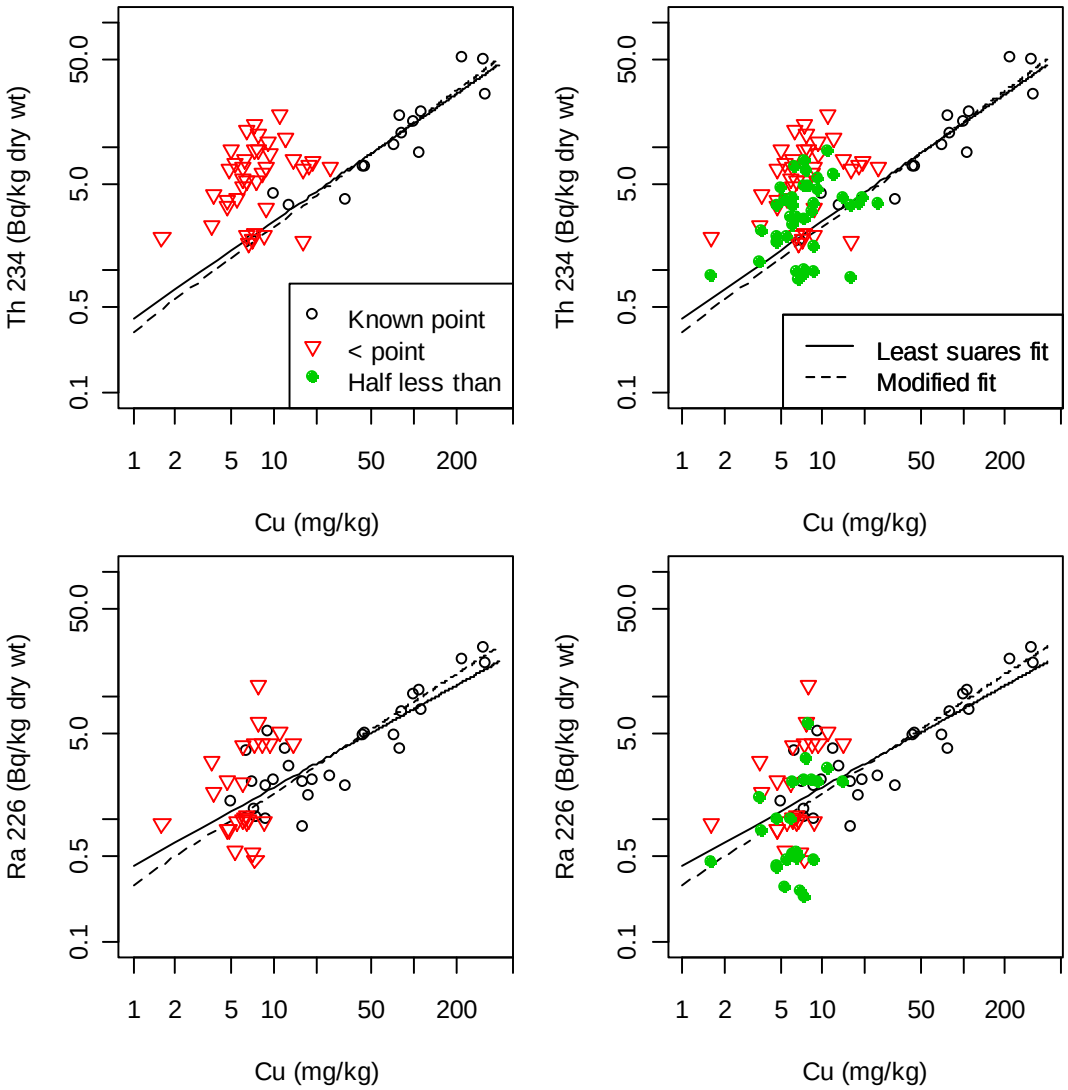

Figure 23 Relationship between Cu concentration (mg/kg) and levels of Th-234 and Ra-226

Table 9 Details of use of Cu as a surrogate for estimation of Th-234 and Ra-226

Item	Th-234	Ra-226
R^2	0.873	0.733
Adjusted R^2	0.863	0.723
Simple regression	ln(Th)=-0.922 + 0.792 ln(Cu)	ln(Ra)= -0.866 + 0.632 ln(Cu)
Modified regression	ln(Th)= -1.152 + 0.847 ln(Cu)	ln(Ra)= -1.219 + 0.739 ln(Cu)
Standard deviation about line	0.366	0. 490

Inverse model using surrogates for below detections

Summaries of the models log concentration – inverse distance models are given in Table 10 and Figure 24. The quantified samples and estimated points based on the surrogate both fit the inverse model.

Table 10 Parameter details for log-inverse distance model for Th-234and Ra-226. Units are in Be/kg dry weight.

Parameter (log base e scale)	Th-234	Ra-226
Intercept (background) *a*	0.431 ± 0.065	0.198 ± 0.069
Slope (elevation at 1km) *b*	5.051 ± 0.285	4.170 ± 0.301
s.d. about line	0.369	0.391

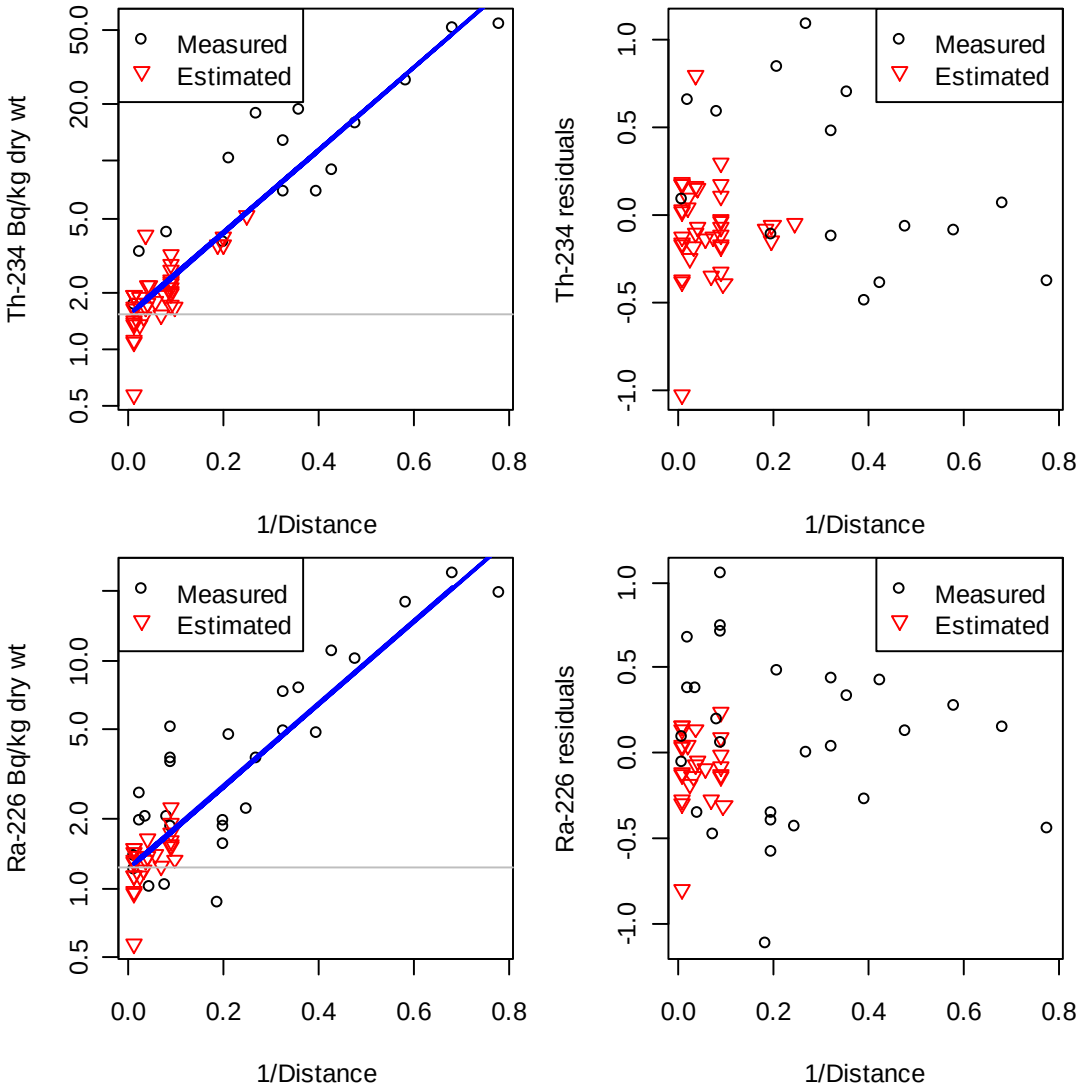

Figure 24 Inverse log model using surrogates for below detection. Plots on the left show the back-transformed model on a logged scale. Plots on the right show the residuals

Figure 24 (right side plots) shows residual plots for the model. For Th-234 the spread of the residuals of the estimates is comparable to that of the quantified samples. This infers that the variability of the quantified and estimated points are similar. There was therefore no reason to place more weight on one type of sample than the other for Th-234. For Ra-226 there was some evidence of the estimated values having less scatter than the quantified points. This might suggest that more weight should be given to the measured values than the quantified values – that would be counter-intuitive so equal weighting was again given to the estimated and quantified samples for Ra-226.

A second feature of the plots is that the residuals from the estimated points are clustered around zero. Because the means of those residuals are near zero, the contribution to the fit of those points must have been unbiased.

Taken together, the residual plots offer evidence that the estimated points from the surrogate do not contribute to heteroscedasticity (unequal variance of the residuals) nor do they bias the model. The residual plots thus validate the method of using Cu as a surrogate for Th-234 and Ra-226.

The use of Cu as a surrogate could not be used for Ra-228 and Th-228 because in each case the correlation with Cu was non-significant, and the regression slope was negative.

Conclusions about handling <QL

The method of maximum likelihood is dependent on several assumptions that cannot be validated. In contrast the QL/2 substitution, while it may not be as efficient when all the assumptions are valid, is simple and takes into account the known QL limit.

The Cu surrogate method has a direct statistical justification and it has been validated. There is also a physical rationale for its use for Th-234 and Ra-226. The surrogate method was therefore preferred for those two analytes. For Ra-228 and Th-228 the QL/2 method was used.

A further investigation of the residuals indicated that there was no strong case for applying differential weighting to the quantified and estimated values.

Appendix D Investigation of replicate data

Field replicates

Simple estimates of the variances were available for three radionuclides that had complete data (all values quantified) where 10 samples had been collected from near the same location (sites S10 and W100). Comparisons of those variances with those obtained from the exponential decay model (as defined on page 2) are given in Table 11. The variance estimates among locations S10 and W100 were comparable to those obtained from the exponential decay model.

If the modelling variance was dominated by large scale (say km) spatial variation, the within site variance would be less than the variance from the exponential decay model. Because the variances were similar, the inference from the comparisons was that the replicate estimates for S10 and W100 (or more precisely the residuals from those samples) were effectively independent. Each of those samples was therefore included in the overall modelling.

Table 11 Variance estimates for three radionuclides based on 10 samples from near the same location, modelling variance and laboratory variance. Variances for S10 and W100 were calculated from $\log_e$ transformed data of the concentration (given in Bq kg-1 dry weight).

Sample	Pb-210	K-40	Be-7
Variance in S10 samples	0.070	0.145	0.071
Variance in W100 samples	0.078	0.045	0.055
Exponential decay model variance (excluding replicates)	0.098	0.042	0.086
Laboratory variance	0.083	0.006	0.009

Laboratory replicates

A series of replicate laboratory data was supplied (see Appendix B for details). In most cases there were insufficient quantifications to enable a meaningful analysis of the laboratory precision. There were usable data for Pb-210, K-40 and Be-7, and within sample variances were computed for those samples (see Table 11). In two cases the laboratory variance was small compared to the field variance, so in those cases there is evidence that the error being introduced at the laboratory was small compared to the error associated with fitting the model. The variance between replicates for Pb-210 was higher than for the other radionuclides. Pb-210 had estimates of 21, 45 and 24 Bq kg^{-1} dry weight for sample for sample W10; this variation resulted in a high estimate of the variance for that analyte.

The high laboratory variance for Pb-210 in sample W10 is of some concern. However, where there was a quantifiable value, the error quoted for the analysis was generally less than 10% which is much smaller that the value obtained for the within laboratory error as shown in Table 11.

Overall the laboratory replicates were aligned with the estimated precision provided with the data.

Appendix E Modelling with maximum likelihood estimation of <QL values

Maximum likelihood estimates of inverse model

A summary of the results of fitting a maximum likelihood model is given in Table 12. The testing first considers a null model where there is no effect of distance (or inverse distance). In that case the model reduces to the background value *a*, which is the $\log_e$ of the background concentration in Bq kg^{-1} dry weight. In the case of the inverse model, the slope parameter *b* is taken as zero whereas in the case of the exponential model the maximum potential increase near the plant (*b*) is taken as zero. When *b* is interpreted as the increase at 1 km from the plant, the units of *b* are $\log_e$ Bq kg^{-1} dry weight. Note that in the case of the exponential model, when *b* is zero the parameter *k* is not relevant.

The results for the null model are the same as for the exponential model as given in Table 2. The conclusions from the inverse models were the same as for the exponential decay model – there was a significant effect on Th-234 and Ra-226 and a reverse effect on Be-7. It is impractical to distinguish the effectiveness of the exponential and inverse models. The exponential model is more flexible but requires an iterative fit. The simpler inverse model has therefore been used in subsequent calculations.

Table 12 Summary of modelling inverse distance using maximum likelihood. Significant increase with proximity to the plant has been indicated with red shading and significant decrease with proximity to the plant has been indicated with green shading. The parameters *a* and *b* refer to the $\log_e$ background concentration and the potential increase 1 km from the plant. Details of the models are described in the Statistical Methods section

Radionuclide	**Th-234**	**Ra-226**	**Pb-210**	**Ra-228**	**Th-228**	**K-40**	**Be-7**
***a* Background**	0.22	-0.27	4.28	-0.39	-0.10	5.7	5.04
***b* Slope**	5.43	5.26	0.23	0.06	-0.10	0.34	-1.02
s.d.	0.55	0.65	0.34	0.43	0.64	0.25	0.30
Log likelihood	-19.5	-38.5	-18.6	-15.7	-37.0	-1.9	-10.7
Null model							
***a* Background**	0.53	0.32	4.31	-0.37	-0.12	5.73	4.90
s.d.	1.62	1.36	0.34	0.43	0.65	0.26	0.35
Log likelihood	-45.1	-64.5	-18.9	-15.7	-37.0	-3.4	-19.3
χ^2 (Chi squared)	51.1	52.0	0.76	0.01	0.04	3.04	17.11
Probability	0.000	0.000	0.38	0.90	0.84	0.08	0.000

Maximum likelihood estimates of exponential model

A summary of the maximum likelihood estimates and the exponential model is given in Table 13. The parameters *a*, *b* and *k* are estimates of the parameters of the exponential decay model as described on page 2. If the parameter *b* is constrained to be zero, the model reduces to just the background parameter *a*: this model is referred

to as the null model. The log likelihood of both the exponential decay model and the null model were estimated and the change in the likelihood was calculated, Twice the change in the log likelihood is distributed approximately as χ^2 (on 2 degrees of freedom in this case as both *b* and *k* were estimated) providing a test of significance for the full model as compared to the null model.

Table 13 also includes a comparison of the standard deviation derived from the model when site replicates were included or excluded. Had the replicates from a site been similar, there would have been a reduction in the standard deviation. There was no evidence of a reduction in the standard error when the site replicates were included. It was therefore concluded that the samples from the same site were effectively independent of each other, and could therefore be included in the analysis along with all the other samples. This result is consistent with the among replicate variance being comparable to the model variance as shown in Table 11.

The estimation using the maximum likelihood method makes assumptions that include that the distribution is normal, that the observations are independent and that the variance is constant across the data set. These assumptions are in addition to the assumption that the proposed model (exponential decay or inverse) was in fact the correct model. The maximum likelihood estimate may be sensitive to departures from these assumptions.

Table 13 Parameter estimates and test of significance for exponential model for seven radionuclides. The s.d. (no dup) omitted the replicated series (S10 and W100) in the data. Significant increase with proximity to the plant has been indicated with red shading and significant decrease with proximity to the plant has been indicated with green shading. The parameters *a*, *b* and *k* refer to the $\log_e$ background concentration and the potential increase near the plant (Bq/kg dry weight) and *k* is the rate of decline per kilometre. Details of the models are described in the Statistical Methods section

	Th-234	Ra-226	Pb-210	Ra-228	Th-228	K-40	Be-7
Full Model							
a	0.26	-0.20	4.29	-0.40	-0.13	5.70	5.09
b	5.36	4.46	6.60	0.15	27.8	2.13	-0.74
k	0.32	0.27	2.04	0.37	3.57	1.32	0.14
s.d.	0.56	0.68	0.33	0.44	0.65	0.25	0.28
s.d. (no reps)	0.56	0.53	0.31	0.44	0.46	0.20	0.29
log likelihood	-18.9	-39.4	-17.5	-15.6	-36.9	-1.1	-7.5
Null model							
a	0.53	0.32	4.31	-0.37	-0.12	5.73	4.90
b	0.00	0.00	0.00	0.00	0.00	0.00	0.00
k	0.00	0.00	0.00	0.00	0.00	0.00	0.00
s.d.	1.63	1.36	0.34	0.43	0.65	0.26	0.35
log likelihood	-45.1	-64.5	-18.9	-15.7	-37.0	-3.4	-19.3
χ^2 (Chi squared)	52.5	50.2	3.0	0.10	0.21	4.6	23.5
p value	0.000	0.000	0.23	0.95	0.90	0.10	0.000

Thorium 234

The results from Th-234 data using a maximum likelihood model fitted to the data are shown in Figure 26. The data at a distance further than 10 km from the plant are dominated by samples that had levels below the quantifiable limit.

The line shown in Figure 12 fits the high values near plant very well. The shape of that curve affects the parameter *k* (the rate of decline) and also the parameter *b* – the log of the increase in concentration near the plant). The fit away from the plant is basically background estimate (parameter *a*). There would appear to be two outliers in the data. One value was 4.14 Bq/kg dry weight at 12.1 km from the centre and the other was 3.28 Bq/kg at 48.6 km distance. Both samples were on the northern transect. The other data which were < QL are consistent with the low estimate of the background.

The fit of the model is very dependent on the assumptions that the <QL data are being handled correctly and that the exponential model is appropriate.

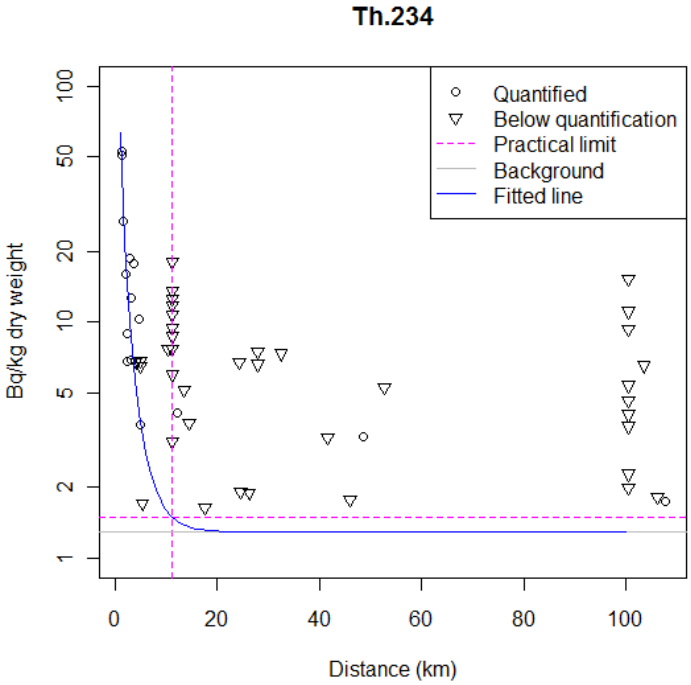

Figure 25 Maximum likelihood fit of Th-234 data using the exponential decay model

Radium 226

There was strong evidence ($p < 0.001$) of a focus of elevated Ra-226 levels near the plant with declining values with distance (Table 2) as indicated by the maximum likelihood test. A maximum likelihood fit is shown in Figure 6. Using the previous definition of a practical limit (Correll 2010), a value of 0.97 on the natural log scale is suggested as this would include 60% of samples above the background concentration. Samples beyond 12.2 km from the plant would be expected to be below that value.

There are three quantified points in the 20 – 60 km distance range that are clearly above the line. On the other hand there is a < QL point well below the line. These points show that there is considerable scatter about the line – a point reinforced by observations from field replicates taken near the 10 and 100 km distances.

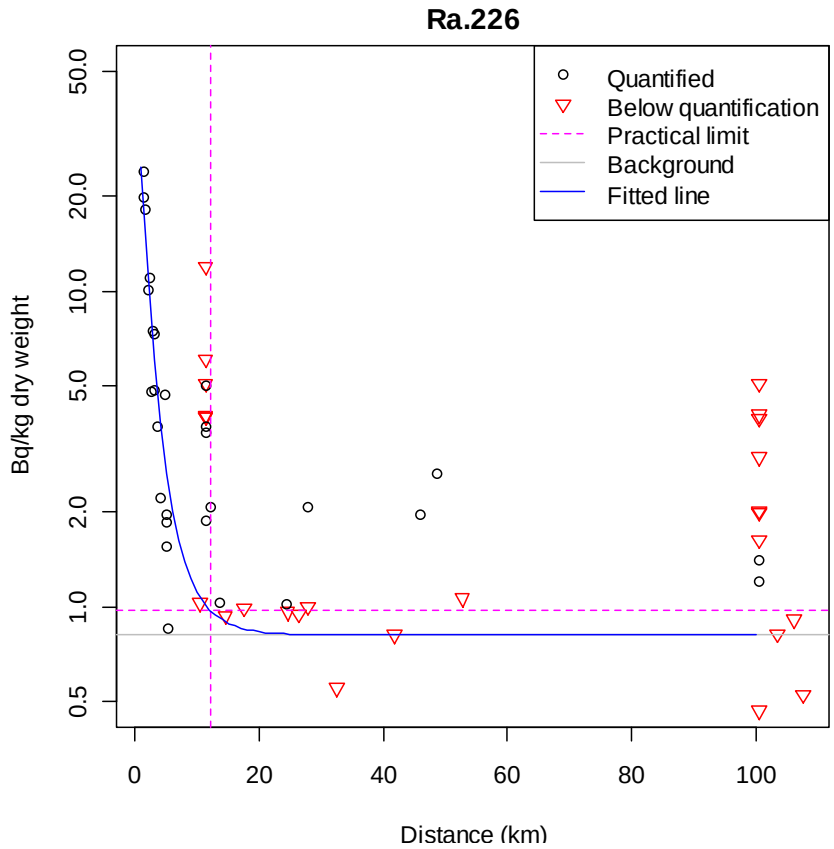

Figure 26 Effect of increasing distance from the plant on Ra-226 concentrations using the exponential decay model

APPENDIX M4

Assessment of radionuclide levels in three tissues from kangaroos in the Olympic Dam region

M4 ASSESSMENT OF RADIONUCLIDE LEVELS IN THREE TISSUES FROM KANGAROOS IN THE OLYMPIC DAM REGION

M4.1 AIMS

The aims of this radiation technical note are:

- to review data on radionuclide activities in tissues obtained during a survey of local fauna
- to provide an assessment of the potential radiological implications of ingestion of meat derived from such fauna by members of the public.

This work was initiated as part of the Olympic Dam expansion EIS process. This technical note will focus on the ingestion of kangaroo muscle from the Olympic Dam region.

M4.2 BACKGROUND

During 2005 and 2006, the Olympic Dam Environmental Department conducted radionuclide surveys in the region of the Olympic Dam mine and processing plant. Flora and fauna were sampled and analysed for radionuclide content. Fauna samples included kangaroos from inside and outside the mine lease, as well as feral cats. This technical note focuses on the results from kangaroos.

The results of the radionuclide analyses were obtained from the Olympic Dam Environmental Department for statistical analysis. The raw data are included in the Appendix and this report makes no comment on the quality of the data received.

One of the concerns noted has been that kangaroos are harvested for human consumption from the pastoral leases around Olympic Dam. If there is a significant build-up of radionuclides in tissues that are consumed, this could be transferred to consumers, resulting in a radiation dose beyond that of normal background. A potential dose, based on the measured radionuclide concentrations and Australian dietary behaviour, has been estimated in this report.

UNSCEAR (2000)[1] provides values for concentration of uranium and thorium series radionuclides in air, water and various foodstuffs.

M4.3 METHODS

M4.3.1 Sample collection

Kangaroos were shot by a professional shooter under destruction permit No. D20980.

Animals described as 'inside the mine lease' or 'mine sample animals' were collected from within the Restricted Release Zone (RRZ), in an area bounded by Eagle Way, east of the mine and south of the General Vehicles Access Road (Easting 682040, Northing 6631006). Seven kangaroos (*Macropus rufus*) were collected (four females and three males) on 28 August 2005.

Seven control animals (*Macropus rufus*, four females and three males) were collected at distances greater than 30 km from the RRZ on Purple Downs Station on 29 January 2006 and are described as 'outside the mine lease' (Easting 671660, Northing 6597445 and Easting 674826, Northing 6598397). The locations of the collection sites are shown in Figure 1.

Samples of three tissue types – muscle (upper hind leg)[2], liver and bone (lower hind leg)[3] – were collected and sent to BHP Billiton-Olympic Dam Analytical Services for analysis of five radionuclides in the U-238 decay chain (U-238, Th-230, Ra-226, Pb-210 and Po-210).

M4.3.2 Data review

This Technical Note is divided into three parts:

- statistical analysis
- review of reports and published literature
- potential human dose assessment.

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1 UNSCEAR 2000, *Report to the General Assembly. Volume 1. Sources and effects of ionizing radiation. Annex B Exposures from natural radiation sources* <www.unscear.org/docs/reports/annexb.pdf> Viewed 17.11.2009.

2 Precise nature of muscle and bone not known.

3 Daniel, DW 1984, *Biostatistics: A foundation for analysis in the health sciences*, 3rd edition. John Wiley & Sons, New York.

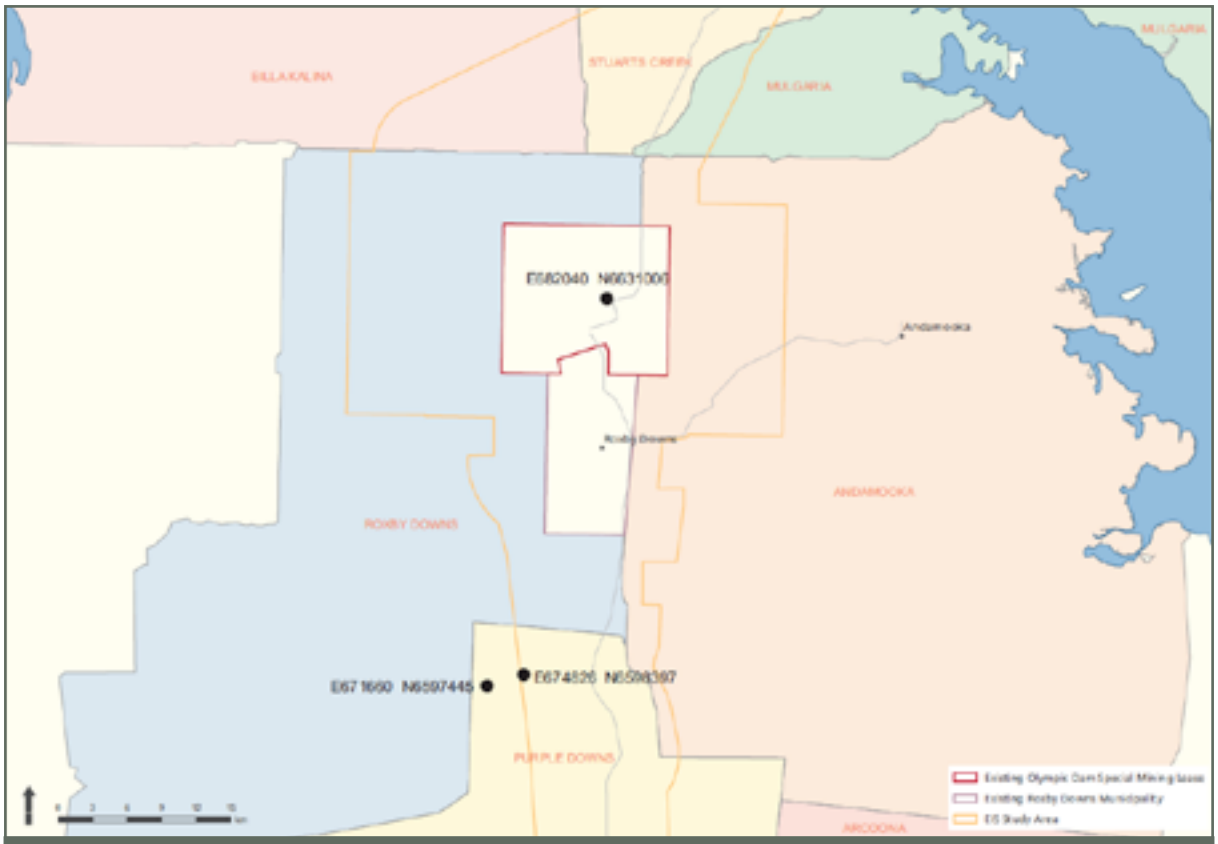

Figure 1 Location of fauna sampling sites

Statistical analysis

The first part of this Technical Note is a broad statistical analysis of the data provided (Section M4.4). This involved determining whether there were any statistically significant differences between the samples identified as being inside the mine lease and those from outside the mine lease. With only seven animals in each group it is not possible to ascertain whether the data are normally distributed. For this reason the Mann-Whitney non-parametric test of differences between median values was used.[3]

Review of reports and published literature

The second part of this report involved a search for and collation of reported radionuclide levels in fauna in the Olympic Dam region since operations began. The earliest document reviewed was the 1983 Environmental Impact Statement. Subsequent annual environmental reports were also examined. The aim of this review was to identify any changes in radionuclide concentrations that could have occurred over time due to the presence of the Olympic Dam operation. In addition, a literature search identified other similar studies carried out in Australia and overseas. The results of this review are detailed in Section M4.5.

Potential human dose assessment

The third part of this technical note (Section M4.6) surveyed practices and regulations within the kangaroo meat industry that control the quality of meat reaching the human food chain. In addition, the results of the radionuclide analyses were used to estimate a potential human dose from the ingestion of the edible portions of an animal, and this value was compared to internationally recognised dose limits. Finally, the actual amount of kangaroo meat consumed in Australia was considered.

M4.4 RESULTS

M4.4.1 Measurements at Olympic Dam

Table 1 summarises results of the statistical analyses of the activities of five radionuclides in three tissues from seven kangaroos shot on the mine lease and seven control animals shot at least 30 km from the boundary of the RRZ. Median and mean values are reported, together with significance as determined by the Mann-Whitney test (two-tailed).

Coloured cells indicate a significant difference between mine and control animals. Yellow shading indicates mine animals were higher; blue shading that the control animals were higher. Unshaded cells indicate no significant difference was identified.

Table 1: Summary of radionuclide concentrations in kangaroo tissues (median and range)

Tissue (location)	Parameter	Radionuclide (mBq/g fresh weight)				
		U-238	Th-230	Ra-226	Pb-210	Po-210
Muscle (mine)	Median (range)	**	**	0.2 (-0.2–0.9)	0.6 (0.06–2)	0.3 (0.1– 1.0)
Muscle (control)	Median (range)	**	**	0.2 (-0.6–0.8)	2.0 (0.6–3)	0.5 (-0.4–0.9)
Significance *	p			p >0.05 (NS)	p >0.05 (NS)	p >0.05 (NS)
Liver (mine)	Median (range)	**	**	0.8 (0.5–1.9)	21 (10–31)	23 (14–31)
Liver (control)	Median (range)	**	**	0.6 (0.04–1.6)	6 (3–13)	10 (4–17)
Significance *	p			p >0.2 (NS)	p <0.005	p <0.005
Bone (mine)	Median (range)	0.5 (0.1–1.5)	0.7 (0.3–1.5)	23 (6–81)	34 (23–67)	42 (19–62)
Bone (control)	Median (range)	0.4 (-0.1–1.4)	0.6 (0.2–2.0)	7 (3–60)	73 (24–278)	63 (24–146)
Significance *	p	p >0.05 (NS)	p >0.05 (NS)	p >0.05 (NS)	p <0.05	p >0.05 (NS)

* p (Mann-Whitney Test – two-tailed).
** too many values at or below the limit of detection to carry out statistical analyses.

M4.4.2 Discussion of Olympic Dam results

The activities for U-238 and Th-230 in all tissues were very low. In the case of muscle and liver, between five and 12 of the 14 values were negative, and at least one median value was negative. No statistical analyses were attempted for the data from these tissues. The raw data are presented in the appendix.

There were very low, but quantifiable, activities of U-238 and Th-230 in bone. However, there was no statistically significant difference between control and mine samples for either radionuclide.

Higher levels of Ra-226 were found in bone, but there was no significant difference between mine and control animals. Activities of Ra-226 in muscle and liver were low, and no significant difference in activities of Ra-226 was identified for either tissue.

In the case of Pb-210 and Po-210, activities in liver were significantly higher (both $p < 0.005$) in mine animals. In contrast, Pb-210 activities in bone were significantly higher in control animals ($p < 0.05$). Pb-210 and Po-210 activities were very low in muscle, and activities in mine and control animals did not differ significantly. No significant difference between mine and control animals was identified in Po-210 activity in bone.

The main conclusion from these analyses is that activities were generally low, and in most cases (12 of 15 comparisons) no significant differences were found. Where differences were observed, no consistent pattern was discernible.

Higher activities of Ra-226, Pb-210 and Po-210 were found in bone than in liver or muscle. Radium and lead are known to accumulate in bone, where they mimic calcium.[4, 5, 6, 7]

There was no consistent pattern of differences. In two cases, 'mine' animals had significantly higher activities; in one case the 'control' animals were significantly higher. In all other cases with measurable activities of radionuclides, no significant difference was identified.

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4 Agency for Toxic Substances and Disease Registry. 1990. Toxicological Profile for Radium. US Public Health Service and US EPA.

5 Harley, NH 2008. Health effects of radiation and radioactive materials. In: Klaassen, CD. *Casarett and Doull's Toxicology The Basic Science of Poisons* 7th edition. McGraw Hill, New York.

6 Rabinowitz, M.B. 1991. Toxicokinetics of bone lead. *Environmental Health Perspectives*. 91: 33 – 37.

7 Liu, J, Goyer, RA, Waalkes, MP 2008 Toxic effects of metals. In: Klaassen, CD *Casarett and Doull's Toxicology The Basic Science of Poisons* 7th edition. McGraw Hill, New York.

In summary, U-238 and Th-230 were close to or below the limit of detection in all tissues analysed, as was Ra-226 in muscle and liver, and of Pb-210 and Po-210 in muscle. Quantifiable activities of Ra-226, Pb-210 and Po-210 were found in all tissues, but were very low in muscle (all radionuclides), and liver (Ra-226). While statistically significant differences between control and mine animals were found for Pb-210 (liver and bone) and Po-210 (liver and bone), there was no consistency in the direction of the difference.

In addition, data were highly variable, as illustrated by the coefficients of variation as shown in Table 2. The coefficient of variation (ratio of the standard deviation to the mean) provides an indication of the relative variability in a data set.

Table 2: Coefficients of variation for activities of radionuclides in kangaroo tissue from Olympic Dam area (sd/mean)

Tissue (location)	Coefficient of variation (%)				
	Radionuclide				
	U-238	Th-230	Ra-226	Pb-210	Po-210
Muscle (mine)	*	*	**	83	**
Muscle (control)	*	*	**	55	**
Liver (mine)	*	*	50	33	28
Liver (control)	*	*	83	64	46
Bone (mine)	81	66	84	42	37
Bone (control)	87	97	113	89	53

* Too many values at or below the limit of detection to carry out statistical analysis.
** Mean values too close to 0 to calculate a meaningful coefficient of variation.

M4.5 EARLIER STUDIES

As part of this review, a search was conducted for data regarding radionuclide activities in kangaroo tissue in the Olympic Dam annual environmental reports published over the past 20 years.

The search identified no earlier data reporting radionuclide activity in kangaroo in the annual environmental reports. It is therefore not possible to compare these results with earlier results.

There were limited data on radionuclides in sheep and cattle in the Olympic Dam area, and these are summarised below.

In addition, a broad literature search was conducted to identify any radionuclide surveys of kangaroos or other wildlife that may have been undertaken elsewhere in Australia or overseas. Three such studies were found at Yeelirrie, WA, in south-eastern Colorado, USA, and in north-eastern Saskatchewan, Canada.

M4.5.1 Radionuclide activities in sheep and cattle from the Olympic Dam area

In the 1990–91 and 1993–94 Environmental Radiation Annual Reports[8, 9], data were presented for radionuclides in tissues (bone, muscle and liver) of sheep and cattle grazing on properties adjacent to Olympic Dam. In each case a single sheep and a single steer were slaughtered. In 1990–91 a single sample of each tissue was analysed, while in 1993–94 duplicate tissue samples were analysed.

In all samples, U-238 and Th-230 were very close to or below the detection levels available at that time. Two samples, steer bone and sheep muscle, had detectable levels of Th-230 (2 mBq/g fresh weight) in each case.

Only two tissue samples had Ra-226 activities above the detection level –steer liver (8.5 mBq/g fresh weight) and sheep bone (5 mBq/g fresh weight).

For Pb-210, activities were above the detection level in liver and bone from one sheep and one steer. In the case of the steer, activities were 8 and 23 mBq/g fresh weight for liver and bone respectively, and for the sheep activities were 12 and 57 mBq/g fresh weight for liver and bone respectively.

Po-210 was close to or below the level of detection available at that time in all muscle samples. Liver sample ranges were 7 to 12 mBq/g fresh weight (cattle) and 3 to 7 mBq/g fresh weight (sheep). In cattle bone, one sample was 16 mBq/g fresh weight, while the Po-210 level in the other bone was not able to be determined for technical reasons. The range of activities in sheep bone was 52–60 mBq/g fresh weight.

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8 Olympic Dam Operations, 1991. Environmental Radiation Annual Report – June 1990 – May 1991.

9 Olympic Dam Operations, 1994. Environmental Radiation Annual Report – June 1993 – May 1994.

Given the high variability found in radionuclides in kangaroos and other species (discussed below), it is difficult to interpret the meaning of values derived from single specimens, as described in these annual reports. However, it can be noted that the values were within the ranges reported here for kangaroo tissues.

M4.5.2 Activities in kangaroos from Yeelirrie, WA

Results for activities of Ra-226 and Pb-210 in tissues from sampling undertaken in the Yeelirrie region of Western Australia[10] in the late 1970s were obtained and are summarised in Table 3. Values were converted from pCi/g ash to Becquerels (Bq), and the denominator converted from ash to dry weight and wet weight, respectively, using the ratios provided in the report. This was done to allow comparison to be made with the current data set.

Activities of Ra-226 and Pb-210 in bone for three of the four Yeelirrie kangaroos were similar to those found at Olympic Dam. High variability was evident, with the fourth animal exhibiting Ra-226 activity in bone an order of magnitude higher than the other three.

Pb-210 activities in livers of three animals were similar to those found at Olympic Dam. One animal was an order of magnitude higher. Ra-226 in liver was similarly variable, with two animals below the level of detection, and two up to an order of magnitude higher than those found at Olympic Dam.

Ra-226 and Pb-210 in muscle were low and near the limit of detection. Both radionuclides showed high variability, with one animal exhibiting activities more than an order of magnitude higher than the others.

In all cases with high specific activities, the tissues were from the same animal (SL2).

Table 3: Summary of Yeelirrie observations of Ra-226 and Pb-210 in kangaroo tissue[10]

Activities are expressed on both a dry weight and wet weight basis calculated from figures supplied (pCi/g ash and ratio of ash to dry and wet weight respectively). Values are from individual kangaroos. BDL indicates below detection limit.

Tissue	Sample site	Ra-226 (mBq/g)		Pb-210 (mBq/g)	
		Dry weight	Fresh weight	Dry weight	Fresh weight
Muscle	SL1	0.03	0.009	3	1
	SL2	1.5	0.4	20	5
	ND1	0.05	0.01	5	1
	ND2	0.05	0.01	2	0.4
Liver	SL1	BDL	BDL	86	26
	SL2	6	2	947	218
	ND1	15	5	77	27
	ND2	BDL	BDL	55	16
Bone	SL1	63	45	22	15
	SL2	626	463	91	67
	ND1	32	25	37	26
	ND2	35	23	11	7

M4.5.3 Baseline activity in animal tissues in Colorado

Whicker carried out a baseline study at the site of a proposed uranium mill in south-western Colorado, USA. His study formed part of the permitting process, and the site had not been used previously for uranium processing. Whicker sampled tissues from three jackrabbits, three cottontail rabbits and three beef cattle. Table 4 compares Whicker's values for individual animals with the median values reported for kangaroos sampled at Olympic Dam. Not all radionuclides were reported in all tissues.

Values were generally of the same order of magnitude between species and sites. An exception was higher Po-210 (both mine and control animals) in kangaroo liver than in cattle liver.

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[10] Brownscombe AJ, Davy DR, Giles MS, Williams AR 1978, *Three baseline studies in the environment of the uranium deposit at Yeelirrie, Western Australia*. Australian Atomic Energy Commission Research Establishment, Lucas Heights, NSW.

However, given the small number of animals in the two studies, and the high variability, it appears that the radionuclide levels in kangaroo tissues (both near the mine and for controls) are very similar to those in rabbits and cattle at a site that had not then been used for uranium processing.

Table 4: Comparison of radionuclide activity in herbivore tissues

Species (Tissue)		mBq/g (wet weight)			
		Th-230	Ra-226	Pb-210	Po-210
*Rabbits[11]	(bone)	BDL	8, 22, 23, 25, 26, 30	0, 16, 20, 43, 44, 53	Not analysed
	(liver)	BDL	Not analysed	2 BDL, 4, 6, 6, 7	4 BDL, 0.9, 1
**Cow[11]	(bone)	BDL	1, 7, 12	BDL, 7, 53	Not analysed
	(liver)	BDL	Not analysed	4	BDL, 0.6, 5
	(muscle)	BDL	Not analysed	Not analysed	BDL, 1, 5
Kangaroo (Olympic Dam median values)***	(bone)	0.7m and 0.6c	23m and 7c	34m and 73c	42m and 63c
	(liver)	BDL	0.8m and 0.6c	21m and 6c	23m and 10c
	(muscle)	BDL	0.2m and 0.2c	0.6m and 2.0c	0.3m and 0.5c

* Values for three cottontail rabbits and three jackrabbits.
** Values for three cows.
*** m and c refer to animals killed in the Olympic Dam mine area and control animals respectively.

M4.5.4 Radionuclide activity in moose near Canadian uranium mines

Thomas and others[12] analysed tissues from 45 moose. Fifteen animals were collected near four uranium mines, 21 from a control area with low mineralisation, and nine (livers only) from a highly mineralised area with no uranium mines (positive control). In addition, four cattle raised in areas remote from uranium mines were sampled from a local abattoir.

Radionuclide levels were measured in bone, liver, kidney, muscle and rumen contents, and the mean activity for each collection site was calculated. The data were highly variable, and the authors concluded that elevated radionuclides found in some animals may have been due as much to natural soil types and diet as to uranium mining. They observed that:

- two moose from one mine site had low tissue concentrations
- Po-210 concentrations in kidneys were higher in cattle controls sourced from well outside the study area than in moose at all sites
- liver concentrations of Ra-226 and Po-210 from the highly mineralised control site (Hudson Bay) were as high as the highest mine site samples
- activities of Po-210 in liver and muscle of moose killed in the Wollaston Lake area declined with distance from the mining area. The authors state that natural uranium outcrops that occur in this area may have contributed to the finding[12].

Table 5 summarises their results for Ra-226, Pb-210 and Po-210 in muscle, liver and bone. These values are presented as the range of means from mine-site related samples, the mean value for the cattle controls (C), the mean value for the controls from the area with low mineralisation (ML), and the mean value for livers from the positive control area with high mineralisation (HB). Uranium values were very low and are not included in the table. These values were estimated from the graphs presented by Thomas and others.[12]

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[11] Whicker, FW 2008, *Baseline survey of radionuclides in animal tissues at the proposed Pinon Ridge Millsite*, <www.cdphe.state.co.us/hm/rad/rmlenergyfuels/dos/080827/baseline.pdf>. Viewed 16/09/2009.

[12] Thomas, P, Irvine, J, Lyster, J & Beaulieu, R 2005, *Radionuclides and trace metals in Canadian moose near uranium mines: Comparison of radiation doses and food chain transfer with cattle and caribou*. Health Physics 88: 432—438.

[13] Pople, T & Grigg, G 1999, Commercial harvesting of kangaroos in Australia. Chapter 5. The kangaroo industry past and present. <http://www.environment.gov.au/biodiversity/trade-use/wild-harvest/kangaroo/harvesting/roobg-05.html> . Viewed 24/09/2009.

All three radionuclides were low in muscle, as was Ra-226 in livers. Pb-210 was elevated in livers from animals killed near mine sites. Po-210 was similar in liver from mine site animals and from control animals. In bone, Ra-226 and Pb-210 were found at similar activities in mine site samples as in controls, while Po-210 was higher from samples taken near mine sites.

Radionuclide activities reported in moose were of the same order of magnitude as those found in kangaroos at Olympic Dam.

Table 5: Summary of data from moose collected near Canadian uranium mines (range) of means from four sites and from control animals (mean)[12]

Tissue	mBq/g (wet weight)					
	Ra-226		Pb-210		Po-210	
	Mine	Controls	Mine	Controls	Mine	Controls
Muscle	0–0.1	0.03[C] 0.04[ML]	0–3	0.9[C] 0.9[ML]	0–6	0.5[C] 0.7[ML]
Liver	0.05–0.1	0.07[C] 0.1[ML] 0.1[HB]	1–42	0.5[C] 1.0[ML]	4–10	5C 4 [ML] 10[HB]
Bone	9–30	8[C] 10[ML]	42–45	41[C] 43[ML]	15–20	7[C] 7[ML]

[C] Indicates cattle control.
[ML] Indicates moose controls from near Meadow Lake (control area with low mineralisation).
[HB] Indicates moose (positive) controls from area of high mineralisation, but no uranium mines.

M4.6 KANGAROO CONSUMPTION AND POTENTIAL DOSE ASSESSMENT

Questions have been raised regarding the likelihood of kangaroo meat harvested near Olympic Dam entering the human food chain and posing a radiological hazard to consumers. To address this issue, three topics are considered:

- the kangaroo meat industry in Australia
- studies of the extent of kangaroo home ranges
- potential for radionuclide dose from eating kangaroo meat.

M4.6.1 The kangaroo meat industry: processing and patterns of consumption in Australia

Kangaroos are harvested from the wild by licensed fauna harvesters. Handling and processing are strictly controlled to maintain meat hygiene. The animals are eviscerated in the field, and the head, limbs and, usually, the tail are discarded, but the skin is left on. The carcass is transported to a refrigerated field 'chiller', usually at a local town or depot, in the early morning before temperatures begin to rise.

Subsequently, the carcasses are collected and transported to the processing works, where the skins are removed and stored for shipment to a tannery and the carcasses are butchered in the same way as meat from domestic stock.[13] The meat is deboned and sold in Cryovac packs (wholesale) or modified atmosphere packaging (retail). Thus, only muscle meat reaches the retail market[14] and liver and bone do not enter the human food chain.

Kangaroo meat, although widely promoted as a healthy meat with fewer detrimental impacts on the environment through its production than sheep or cattle, forms only a small part of the dietary intake. Currently, only a small minority of Australians regularly consume kangaroo, as shown in Figure 2.[15] Less than 15% of respondents reported eating kangaroo meat four to six times a year or more.

Kelly[16] reports as a rough estimate that 7,000 tpa of kangaroo meat is sold on the domestic market. Kelly[17] further argues that 'Even if all the kangaroos which could be taken were taken, and all the product was sold domestically, sales would only represent approximately 4.5% of the domestic red meat market'.

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14 Macromeats website: <http://www.macromeats.com/Default.aspx>. Viewed 25/09/2009.

15 Ampt, P & Owen, K 2008, Consumer attitudes to kangaroo meat products. RIRDC Publication No 08/026. Canberra ACT.

16 Kelly, J 2009, Personal communication. 09/09/2009.

17 Kelly, J 2005, *Kangaroo industry strategic plan 2005 – 2010*. a *Report for the Rural Industries Research and Development Corporation*. RIRDC Publication No 05/105. Canberra, ACT.

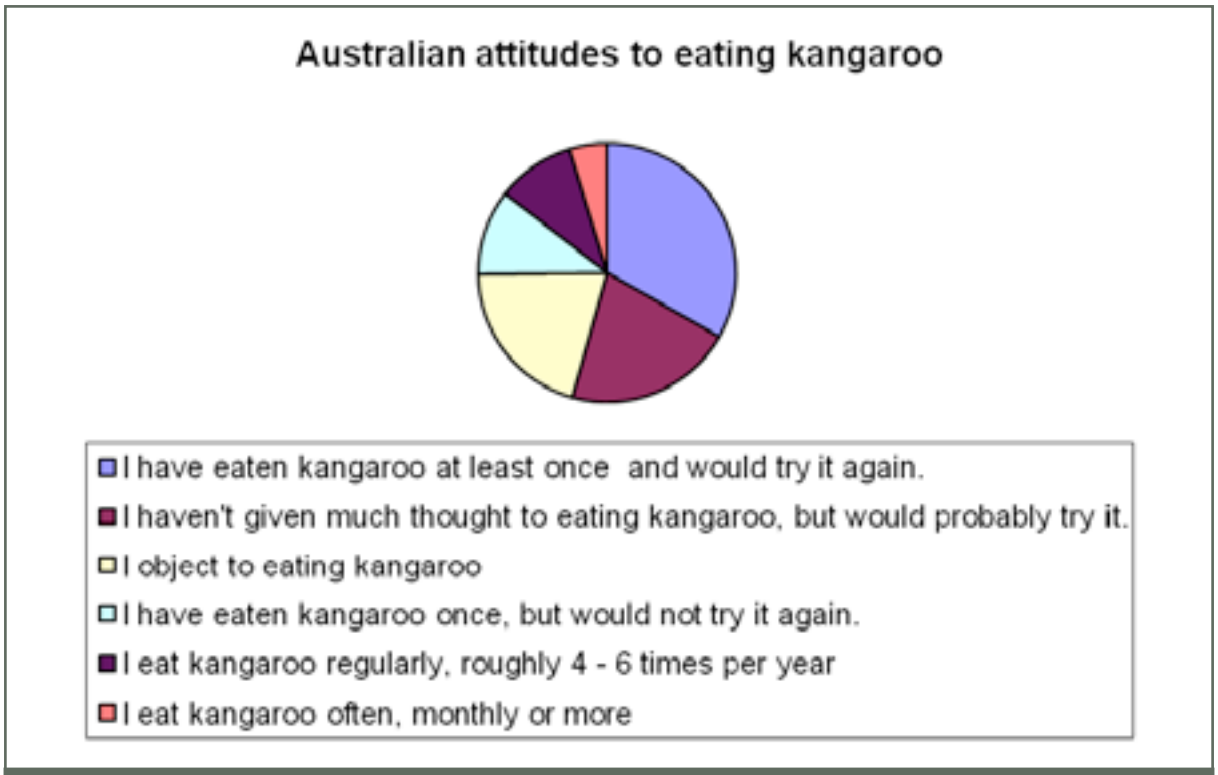

Figure 2 Responses to statements regarding consumption of kangaroo meat

In contrast to the greater than 600 kg of kangaroo muscle per annum required to achieve the member of public limit for exposure, as shown in Table 8, the average Australian eats less than 0.25kg of kangaroo meat a year. For comparison, on average, Australians consume 37 kg of beef and veal a year.[18]

In South Australia, a single processor provides kangaroo products to all supermarkets[14], thus effectively randomising the products in terms of where it was harvested. This reduces the likelihood of a consumer sourcing all his/her kangaroo meat from any one location in the state.

M4.6.2 Home ranges of kangaroos sampled

Statistically significant differences were found in only three of 15 pairs of mine/control combinations of animals collected at or near Olympic Dam. Of these, the control was statistically significantly higher in the case of Pb-210 in bone. The mine animals had higher activities in the case of Pb-210 and Po-210 in liver.

This observation led to a suggestion that the two groups may share a home range extending 30 km or more from Olympic Dam, thus casting doubt on the 'control' status of the animals harvested at distance. A literature search for information on the extent of home ranges of *Macropus rufus* was undertaken to cast some light on this question.

Kangaroo home ranges have been reported in the literature. Frith[19] studied kangaroo movements in two areas with contrasting climates. He reported that in milder climatic conditions, red kangaroos *Megaleia rufa*[20] were very sedentary. However, in the more arid area studied, the animals moved greater distances to reach green herbage. Distances observed between marking and recovery of kangaroos (and time scale) are shown in Table 6.

A number of more recent home range estimates for *M. rufus* are shown in Table 7. In some cases maximum distances travelled were indicated. Where this was not reported, an estimate of the likely distance a kangaroo would travel was calculated based on the area of the home range. This calculation assumed that the home range was roughly circular and the diameter of a circle of the stated area was taken to be an estimate of the distance a kangaroo in that home range would travel.

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[18] Mahr, K 2009, Kangaroo: It's what's for dinner. *Time Magazine.* <http://www.time.com/time/magazine article/0,9171,1894792,00.html>. Viewed 28/09/2009.

[19] Frith, HJ 1964, Mobility of the red kangaroo, *Megaleia rufa*. *CSIRO Wildlife Research* 9: 1-19.

[20] *Megaleia rufa* is a synonym for *Macropus rufus*.

Table 6: Distances travelled by *Megaleia rufa* (*Macropus rufus*)[19]

Habitat type	Number of kangaroos recovered	Range of distance from site of marking to site of recovery (km)	Elapsed time between marking and recovery for longest distance (months)
Mild	8	3–22	37
Arid	6	0–32	38

Fisher and Owens[21] estimated home ranges of 641 and 695 ha for female and male red kangaroos, respectively, which equates to distances travelled of 2.8 and 3.0 km.

Norbury and others[22] followed the movements of kangaroos in arid Western Australia over a period of drought followed by a post-drought period. They reported home ranges of 36 and 28 km^2 for males and females respectively, and noted that these were the largest home ranges reported for this species. They also reported that the longest distance travelled by males was 24 km during the drought period. This contrasts with distances of 4 km (post-drought) for males and 5 km (drought and post-drought) for females.

Priddle and others[23] observed a mean shift in the centre of the home range of 1.3 km over a three-month period for red kangaroos, and concluded that they are not nomadic. A mean home range of about 8 km^2 was observed, which gives a calculated mean distance travelled of 3.2 km. Maximum observed distance was 13 km, with the majority of animals moving less than 6 km.

In this study, the control animals were harvested about 30 km from the mine on Purple Downs Station. This distance is close to the maximum distance travelled reported by Frith, and well above the estimates of Priddle and others, Fisher and Owens and Norbury and others.

Table 7: Reported home ranges of *Macropus rufus*

Source	Home range		Estimated distance (km)		Comments
	Male	Female	Male	Female	
Fisher and Owens[21]	695 ha	641 ha	3.0 *	2.8 *	
Norbury, Norbury, and Oliver	36.1 ± 17.2 km^2	18.4 ± 5.0 km^2	6.8 *	4.8 *	Largest home ranges recorded for this species
Norbury, Norbury, and Oliver[22]	Estimate based on distances between radio collar fixes during drought (D) and post-drought (PD)		24 (D) 4 (PD)	5 (D and PD)	Distances quoted include 90% of radio fixes
Priddel, Shepherd and Wellard[23]	**Male and female**		**Maximum distance observed**		Radio collared animals tracked over 20 months
	7.74 ± 0.90 km^2		13		

* Estimated distance calculated by assuming that the home range is roughly circular and using the diameter of the circle as the distance the animals are likely to travel.

While it is not possible to conclude that the animals in this study were from separate home ranges, these observed and calculated distances suggest that kangaroos harvested 30 km or more apart are not likely to share a home range. The variability in tissue activity of radionuclides observed is likely to be natural, as reported elsewhere in kangaroos and other species.

M4.6.3 Dose assessment

Although it is unlikely that kangaroo meat from the Olympic Dam area would form a significant proportion of a person's diet, as discussed in Section M4.6.1, dose assessments were carried out to ascertain whether it were likely that such a person's exposure would exceed the member of the public dose limit.

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21 Fisher, DO & Owens, IPF 2000, *Female home range size and the evolution of social organization in macropod marsupial*, Journal of Animal Ecology 69: 1083–1098.

22 Norbury, GL, Norbury, DC & Oliver, AJ 1994, *Facultative behaviour in unpredictable environments: Mobility of red kangaroos in arid Western Australia*, Journal of Animal Ecology 63: 410–418.

23 Priddel, D, Shepherd, N & Wellard, G 1988, *Home ranges of sympatric red kangaroos* Macropus rufus*, and western grey kangaroos M. fuliginosus, in western New South Wales*, Australian Wildlife Research 15: 405–411.

The dose assessments were calculated using ICRP dose conversion factors (ICRP 60/61 and ICRP 72)[24]. The steps in the assessment are outlined below. The steps presented are for the potential doses from ingestion of kangaroo meat (muscle), since bone and viscera do not enter the human food chain.

The mean activities of radionuclides in kangaroo tissues were calculated. Negative values were taken to be 0.

Two sets of dose conversion factors were used:

- ICRP 60/61 (adult occupational exposures)
- ICRP 72 (which are age-dependent).[24]

For the purposes of perspective, quantities of material were calculated that would need to be consumed to reach the member of the public limit of 1 mSv above background.

It should be noted that in this assessment, it has been assumed that all radionuclides in the kangaroo meat are the result of the impact of the Olympic Dam project. Therefore, this is a very conservative assessment, since normal background levels have not been taken into account. Subtracting normal background levels would reduce the activity of the radionuclides in the tissues, as shown in Figure 3.

The results for each exposure scenario (ICRP 72 age-dependent factors) and (ICRP 60/61 occupational factor) are shown in Table 8. Using ICRP 72 age-dependent factors, an adult would need to ingest 648 kg of kangaroo meat a year (almost 2 kg a day) to reach the member of the public dose limit.

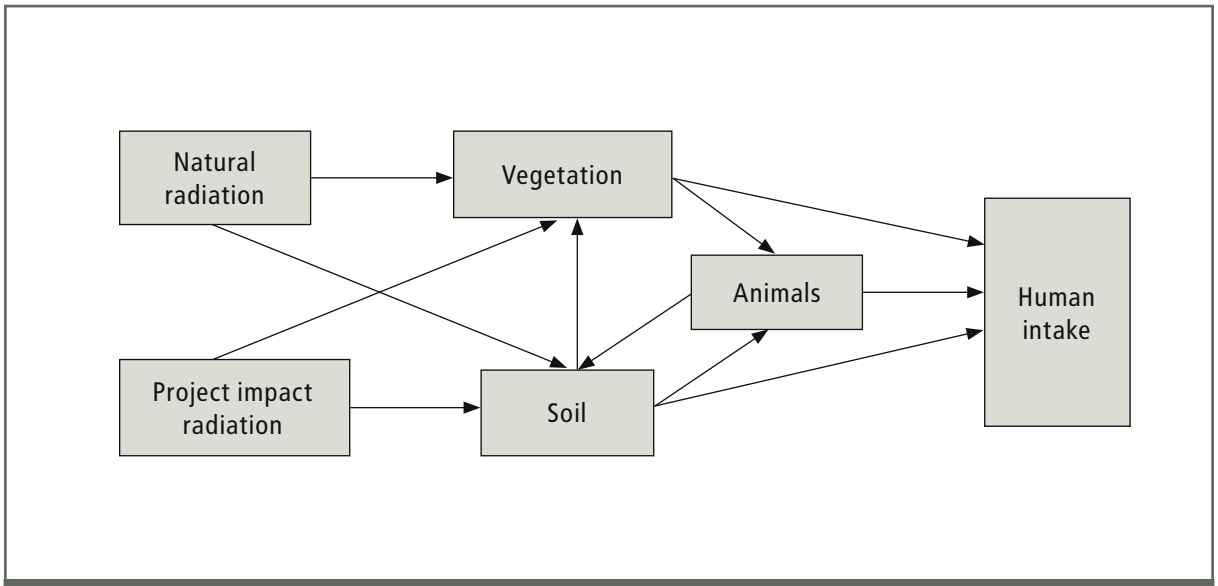

Figure 3 Diagrammatic representation of routes for ingestion of radionuclides

Table 8: Quantity of meat to be eaten based on ICRP 72 age-dependent factors or ICRP 60/61 occupational factors to receive a dose of 1 mSv in one year (calculations rounded to nearest kg)

Food Source	Kg kangaroo tissue per year						ICRP 60/61
	3 months old	1 year old	5 years old	10 years old	15 years old	Adult	Adult
Kangaroo muscle	41	113	199	256	298	648	808
Kangaroo liver	2	5	10	14	18	33	66
Kangaroo bone	1	2	3	4	4	10	13

[24] Sourced from Codell, RB & Campbell, AC 2004, *Use of ICRP-72 and age-based doses for decommissioning – a real-world example,* WM'04 Conference, February 29 – March 4, 2004, Tucson AZ. WM-4177.

Doses for infants younger than three months were calculated because dose conversion factors have been provided in ICRP 72. However, the meaning and usefulness of these estimates are questionable. Children of this age are rarely left alone and have relatively controlled diets with no solids and therefore would generally not be exposed to situations where they may eat like older children.

Although lesser amounts of kangaroo liver and bone would need to be consumed to reach the member of the public dose limit, these tissues do not enter the human food chain (see Section M4.6.3).

M4.6.4 WORST-CASE SCENARIO

Another strategy for estimating any potential impact from ingesting kangaroo meat is to calculate a worst-case scenario using food intake data from national dietary surveys.[25] Table 9 presents the mean meat intake of Australian adult males and females in grams per day (g/d), as well as the intake in g/day at the 95th percentile. The latter value represents a level of intake higher than that of 95% of the population.

For this EIS it is assumed that all meat intake is from kangaroo muscle. The total weight of kangaroo meat that would be consumed at each level has been calculated, and a factor calculated from the sum of the various radionuclides has been applied, multiplied by the relevant ICRP 72 dose conversion factor for adults[24]. In addition, background radiation has not been subtracted from the measured activities of radionuclides in tissues. Thus, the estimated exposure represents an overestimate of radiation resulting from the mine and metallurgical processing activities.

Even in the unlikely event of a man consuming meat at the 95th percentile, with all of that meat sourced from kangaroos originating near Olympic Dam, the total dose would be only 30% of the maximum allowable dose for members of the public. For those consuming lesser amounts of meat (again all of it kangaroo meat from near Olympic Dam), the doses would range from 10–20%, including background exposure.

Johnston and others[26] in the context of the clean-up of the Maralinga nuclear weapons testing site, reported that *Macropus rufus* formed a significant part of the diet of the Indigenous population of Oak Valley. An anthropological study estimated that, on average, 600 g per person per day was consumed. This is close to the 95th percentile for males, and equates to 0.34 mSv/year exposure for consuming kangaroo meat sourced from near Olympic Dam. This is a very conservative estimate, since no correction has been made for normal background radionuclide levels, and the limit of 1 mSv/year is defined as exposure above background.

Table 9: Worst-case scenario for radiation exposure from consumption of kangaroo meat*

Consumption*	Male (70 kg body weight)		Female (60 kg body weight)	
	Mean intake	95th percentile	Mean intake	95th percentile
g/day	265	552	159	354
kg meat/year	97	201	58	129
Dose conversion**	**1.54 nSv/g**			
mSv/year	0.15	0.3	0.1	0.2

* Assume all meat consumed is kangaroo muscle from the Olympic Dam area.
** Sum of measured activity of each radionuclide in kangaroo muscle x relevant dose conversion factor for adults (ICRP 72)24.

M4.7 CONCLUSION

A lack of data on background radionuclide concentration in fauna in the Olympic Dam area has made it impossible to determine whether there has been an increase in radionuclide concentrations in kangaroos over the life of the project.

Sparse data available in other reports and the literature have been used as a basis for comparison. Data from Olympic Dam and Yeelirrie, WA, suggest that radionuclide activity in kangaroo is highly variable, and that muscle has much lower activities than liver and bone. Only muscle meat enters the human food chain.

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[25] Lester, IH 1994, Food and Nutrient Intakes. Chapter 4 in Australia's Food and Nutrition, Canberra AGPS. <http://www.aihw.gov au/publications/aus/afn94/afn94.pdf>. Viewed 18/09/2009.

[26] Johnston, PN, Collett, AC, & Gara, TJ 2003, Aboriginal participation and concerns throughout the rehabilitation of Maralinga. *Proceedings of the Third International Symposium on the Protection of the Environment from Ionising Radiation (SPEIR 3) held in Darwin, Australia 22 – 26 July 2002. Protection of the environment from ionising radiation: the development and application of a system of radiation protection for the environment*. International Atomic Energy Agency.

Data from other species (cattle, jackrabbits, cottontail rabbits and moose) in North America exhibited similar variability (up to an order of magnitude) in radionuclide activity in tissues including bone, muscle and liver. Variability was observed within and between tissues, radionuclides and collection sites.

In this study, no significant differences occurred between 'control' and 'mine' kangaroo tissues in 12 out of 15 comparisons. In two cases, (Pb-210 and Po-210 in liver) the mine animals were significantly higher, and in one case (Pb-210 in bone) the control animals were significantly higher.

Radionuclide activity in kangaroo muscle is consistently lower than in liver or bone for all radionuclides measured.

UNSCEAR has published summary tables of concentrations of uranium series radionuclides in foods from various locations worldwide.[27] The median values for the five radionuclides in kangaroo muscle reported here all lie within the ranges reported by UNSCEAR.

Conservative dose assessments indicate that any potential dose from the ingestion of kangaroo meat (muscle) is low. Bone and liver, which tend to accumulate heavy metals including radionuclides, do not enter the human food chain. In addition, consumption of kangaroo meat in Australia is low, which further reduces the likelihood of harm resulting from its consumption.

The results of this assessment have been used as the basis for the dose estimates presented in the EIS.

APPENDIX: RAW DATA

Raw data for radionuclides in mine and control kangaroos.

Muscle

Location		U-238 mBq/g	Th-230 mBq/g	Ra-226 mBq/g	Pb-210 mBq/g	Po-210 mBq/g
1F Mine	Mine	-0.4	-0.04	0.7	2	0.51
2F Mine	Mine	-0.3	0.4	0.08	1	0.3
3F Mine	Mine	-0.2	0.04	0.2	2	0.1
4F Mine	Mine	0.05	0.3	0.09	0.06	0.3
5F Mine	Mine	-0.005	0.1	0.9	0.5	1.1
6F Mine	Mine	-0.15	0.5	-0.2	0.6	0.1
7F Mine	Mine	0.03	0.2	0.7	0.4	0.5
1F Control	Control	-0.2	-0.06	0.5	2	0.6
2F Control	Control	-0.2	-0.03	0.2	1	0.5
3F Control	Control	-0.1	0.2	0.6	0.6	0.7
4F Control	Control	-0.05	0.04	-0.6	0.9	0.2
5F Control	Control	-0.3	-0.4	-0.3	3	-0.4
6F Control	Control	-0.3	0.08	-0.06	3	0.9
7F Control	Control	-0.5	-0.3	0.8	2	0.1
P value (95%)		0.022	0.022	0.022	0	0.022

[27] UNSCEAR 2000, *Report – annex B: Exposures from natural radiation sources*, <www.unscear.org/unscear/en/publications/2000-1.html>. Viewed 1/09/2009.

Liver

Location		U-238 mBq/g	Th-230 mBq/g	Ra-226 mBq/g	Pb-210 mBq/g	Po-210 mBq/g
1L Mine	Mine	0.6	0.3	0.5	10	17
2L Mine	Mine	0.03	0.19	0.6	21	18
3L Mine	Mine	-0.1	0.01	1.4	17	25
4L Mine	Mine	0.2	0.3	1.0	31	31
5L Mine	Mine	-0.1	-0.1	0.6	29	29
6L Mine	Mine	-0.2	-0.2	1.9	21	23
7L Mine	Mine	-0.08	0.3	0.8	20	14
1L Control	Control	-0.5	-0.2	0.17	3	8
2L Control	Control	-0.2	0.06	0.5	12	17
3L Control	Control	-0.4	-0.2	1.2	7	10
4L Control	Control	0.8	0.3	0.04	3	14
5L Control	Control	0.6	-0.2	0.6	6	7
6L Control	Control	-0.5	0.3	1.2	13	17
7L Control	Control	0.1	-0.2	1.6	3	4
P value		0.022	0.022	0.0	0.022	0.022

Bone

Location		U-238 Concentration mBq/g	Th-230 Concentration mBq/g	Ra-226 Concentration mBq/g	Pb-210 Concentration mBq/g	Po-21 Concentration mBq/g
K - 1b Mine	Mine	0.3	2	14	60	59
K - 2b Mine	Mine	0.1	0.5	12	67	62
K - 3b Mine	Mine	0.5	0.7	45	28	31
K - 4b Mine	Mine	1.1	0.3	38	23	19
K - 5b Mine	Mine	1.5	0.5	81	44	42
K - 6b Mine	Mine	0.2	1.1	23	34	43
K - 7b Mine	Mine	0.7	1.5	6	28	35
K - 1b Control	Control	0.3	0.6	8	80	64
K - 2b Control	Control	-0.1	0.4	3	174	146
K - 3b Control	Control	0.4	0.2	6	73	63
K - 4b Control	Control	1.0	2	15	71	54
K - 5b Control	Control	1.4	0.4	4	63	60
K - 6b Control	Control	0.3	0.2	7	278	87
K - 7b Control	Control	0.9	0.8	60	24	24
P value		0.036	0.022	0.022	0.022	0.022
Calibration		0.1600				

APPENDIX M5

Briefing paper on effects of radiation on the non-human environment

URANIUM EQUITIES

Uranium Services

Olympic Dam Expansion EIS

Technical Note

Radiation exposure to Non-Human Biota

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August 2007

The information prepared here is on the basis of Good Faith only and is based on information provided by the client and other sources at the date of production.

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CONTENTS

1 Introduction

Historically the risk assessment and management of radionuclides entering or present in the environment is generally based on human health considerations alone. This has been underpinned by the ICRP 1990 recommendations:

> "The Commission believes that the standards of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species. At the present time, the Commission concerns itself with mankind's environment only with regard to the transfer of radionuclides through the environment, since this directly affects the radiological protection of man."

Recently there has been increasing awareness of the vulnerability of the environment and of the need to be able to demonstrate that it is protected against the effects of industrial pollutants, including radionuclides. The ICRP, in its recently approved (March 2007) new fundamental Recommendations on the protection of man and the environment against ionising radiation, has addressed this by giving more emphasis to the protection of the environment.

To facilitate the better understanding of the radiation effect to non-human biota, the ICRP has also formed a Committee 5. The aim of this committee is to ensure that the development and application of approaches to environmental protection are compatible with those for radiological protection of man, and with those for protection of the environment from other potential hazards.

ICRP publication 91, entitled "A framework for assessing the impact of ionising radiation on non-human species" was published in October 2003. This document provided a review of the various methods that have been developed for the assessment of radiological impacts with the objective of identifying and suggesting the best framework. One alternative recommended was to make an initial assessment using primary (generic) reference organisms for flora and fauna to give an order of magnitude assessment of the probability and severity of the likely and different effects of radiation exposure on the population. The majority of organisms could be eliminated as negligible or low risk at this point leaving room for a more detailed assessment of any high risks areas identified. This approach was adopted by the European Union as part of their ERICA project, see section 2.

In early 2005, the ICRP circulated a draft document for discussion entitled "The concept and Use of Reference Animals and Plants for the purposes of Environmental Protection". This included a list of the Reference Animals and Plants (RAP) that would be adopted by the ICRP and outlined the framework for a common methodology of assessment. There is no indication from the commission as to the proposed publication date.

UNSCEAR are also in the process of reviewing all available data on this subject and will be assessing a second draft of this work during 2007, with an expected publication date some time during 2008.

Currently there is a significant amount of research occurring into the topic of radiation effects to Non-Human Biota (NHB) and a number of different assessment methodologies have become available. At the completion of the ICRP Committee 5 initial charter, a standard framework for assessment should be

completed. Until this time assessments will need to be undertaken using the best available and applicable methodology and data.

2 The ERICA Tool

The European Commission funded a project in 2004 (completed in February 2007) called ERICA (Environmental Risk from Ionising Contaminants: Assessment and Management). This project produced an assessment tool that predicts the risks to reference animals and plants from ionising radiation. The assessment tool is structured around a three tiered approach, this is explained in detail in the ERICA final report given in Appendix A and summarised to follow.

The Tier 1 assessment is designed to be simple and conservative, requiring a minimum of input data and enabling the user to quickly eliminate a number of scenarios as negligible risk and identify those requiring the next tier of assessment. The tool uses a dose rate screening value to determine Environmental Media Concentration Limits (EMCL) for all reference organism/radionuclide combinations. It then compares the input media concentrations with the most restrictive EMCL for each radionuclide and determines a risk quotient (RQ). If the RQ is less than one, then risk is negligible and the tool suggests that the user should exit the assessment process. If the RQ is greater than one, the user is advised to continue with a tier 2 assessment.

The default screening value in the ERICA tool is an incremental dose rate of 10 µGy/h. This value was derived from a species sensitivity distribution analysis performed on chronic exposure data and is supported by other methods for determining predicted no effect values, see appendix A.

Tier 2 is a more detailed risk assessment, using best estimate values, that yields a low, moderate or high risk return. Any high risk areas identified will require a detailed, site specific, radiological assessment for identified radionuclides and organisms to be undertaken, this is termed a tier 3 assessment.

The ERICA tool was utilised to undertake an assessment of the potential risk to the terrestrial environment (non-human biota) surrounding the proposed Olympic Dam expansion project.

3 ERICA Assessment Results

The critical pathway for transport of radionuclides into the environment will be airborne. Unlike the assessment of dose to humans, where radon inhalation is significant, the major pathway for exposure results from long term dust deposition.

The Olympic Dam (OD) Environmental Impact Statement (EIS) air dispersion model was utilised to determine the dust deposition immediately outside operational areas. This showed a maximum of approximately 25g/m^2/y inside the lease boundary, falling to below 6g/m^2/y outside the lease boundaries.

Over the 40 year project life there would be mixing of the deposited dust with the underlying soil resulting in an increase in the radionuclide content in that surface layer. No references could be located indicating the exact amount this mixing, so for the purposes of this assessment it was assumed that deposited dust will mix into the top 10 mm of the soil. Using a uranium concentration in dust of 92ppm (average ore grade for entire pit as calculated for the EIS radiological assessment) with highest deposition rate for a 40 year mine life and assuming a 10mm mixing zone gives a maximum predicted uranium concentration in surface soil of 6 ppm U (c.f. natural soil is typically 3ppm).

This uranium concentration was used for the tier 1 assessment. The default screening value was selected and all other uranium decay products were assumed to be in equilibrium. The assessment yielded a risk quotient of <1 (negligible risk) for all but two radionuclides, Ra-226 in Detritivorous invertebrates and Po-210 in mammals (Rat & Deer). For the purposes of this assessment “Rat” was taken to mean small mammals and “Deer” was taken to be kangaroos.

A tier 2 assessment was then conducted for these radionuclides and reference animals. Once again the maximum predicted soil concentration was input into the tool and throughout the assessment the most conservative position was always selected. The resultant risk quotient for this assessment was <1 for each radionuclide and reference animal. Thus indicating that there is negligible risk and that a detailed tier 3 assessment will not be required.

The results of this assessment indicate that the radiological risk to non-human biota as a result of dust deposition from the Olympic Dam Expansion operation, for a 40 year mine life, can be considered negligible.

APPENDIX

ERICA Assessment Tool Report for Olympic Dam Expansion

ERICA Assessment Report

August 2007

Project: ODX EIS Assessment
Assessment name: ODX EIS
Author: Sharon Paulka

Purpose of the assessment
Assessment for the Olympic Dam Expansion EIS. To determine if there is any risk to the environment (animals and plants) as a result of long term dust deposition and the subsequent accumulation of uranium in soils.

Performed tiers
Tier 1, Tier 2

Tier 1

Stakeholder Involvement

This will go in a report to stakeholders (EIS)

Problem Formulation

Description

Assessment of effects of increase in Uranium Concentration in soils from dust deposition Maximum concentration values were obtained form atmospheric modeling that showed a maximum possible deposition rate, close to the project in one small location of 25g/m2/year. Assuming this mixes with 10 mm of top soil over a 40 year period and the dust has a U concentration of 92 ppm gives an 6 ppm or 75 Bq/kg of U-238. Also assume that all daughter products are in equilibrium and stay that way.

Pathways and endpoints

Pathway: Uranium Contamination of Soil from Dust deposition up to 10 ppm increase.

End Point: Contained within in ERICA program and consistent with the new ICRP recommendations. Early mortality, morbidity or reduced reproductive success such that they have a negligible impact on the maintenance of biological diversity, the conservation of species or the health and status of natural habitats, communities and ecosystems.

Specifically looking to see if there is anything greater than a negligible effect. If so further studies will be conducted.

Assessment Details

Ecosystem	Terrestrial
Transport model	None

Isotopes

U-238
U-234
Th-234
Th-230
Ra-226
Pb-210
Po-210

Inputs

Media Activity Concentration in soil or air [Bq kg-1 d.w. or Bq m-3]

Isotope	Value
U-238	7.50E1
U-234	7.50E1
Th-234	7.50E1
Th-230	7.50E1
Ra-226	7.50E1
Pb-210	7.50E1
Po-210	7.50E1

Parameters

Terrestrial Environmental Media Concentration Limit (ERICA) [Bq kg -1 or Bq m-3]

Isotope	Value
U-238	1.97E3
U-234	1.68E3
Th-234	1.78E5
Th-230	1.58E3
Ra-226	4.16E0
Pb-210	5.02E3
Po-210	7.31E0

Outputs

Risk Quotient [unitless]

Isotope	Value	Limiting Reference Organism
U-238	3.82E-2	Lichen & bryophytes
U-234	4.46E-2	Lichen & bryophytes
Th-234	4.22E-4	Grasses & Herbs
Th-230	4.76E-2	Lichen & bryophytes
Ra-226	1.80E1	Detritivorous invertebrate
Pb-210	1.49E-2	Lichen & bryophytes
Po-210	1.03E1	Mammal (Deer), Mammal (Rat)

Sum of Risk Quotients [unitless]

Value
2.84E1

Decision

Justification for the decision

Effect to Terrestrial Environment is negligible for all but Ra-226 and Po-210

A tier 2 assessment required for these parameters

Tier 2

Stakeholder Involvement

This will go in a report to stakeholders (EIS)

Problem Formulation

Description

A more detailed tier 2 assessment of effects of increase in Ra-226 and Po-210 on the most sensitive fauna (Detritivorous invertebrate and Mammal (Deer and Rat)). Once again taking the absolute maximum of 6 ppm U or 75 Bq/kg which is an extremely conservative estimate.

Pathways and endpoints

Ra-226 at 75 Bq/kg to Detritivorous invertebrate Po-210 at 75 Bq/kg to Mammal (Deer and Rat).
Endpoints same as for Tier 1 Assessment.

Assessment Details

Ecosystem	Terrestrial
Transport model	None
Dose rate screening value	ERICA
Uncertainty Factor	3.0

Organisms

Detritivorous invertebrate
Mammal (Deer)
Mammal (Rat)

Isotopes

Ra-226
Po-210

Inputs

Percentage d.w. [%]

Value

1.00E2

Media Activity Concentration in soil or air [Bq kg-1 d.w. or Bq m-3]

Isotope	Value
Ra-226	7.50E1
Po-210	7.50E1

Parameters used in assessment

Weighting Factor of internal alpha [unitless]

Value

1.00E1

Weighting Factor of internal beta gamma [unitless]

Value

1.00E0

Weighting Factor of internal low beta [unitless]

Value

3.00E0

Dose Conversion Coefficients used by ERICA tool

Dose Conversion Coefficient of external alpha radiation in air [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coefficient external beta gamma radiation in air [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0

Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coefficient of external low beta radiation in air [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coefficient of external alpha radiation in soil [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coeff. external beta gamma radiation in soil [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	9.10E-4
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	8.50E-4
Po-210	Detritivorous invertebrate	4.60E-9
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	4.30E-9

Dose Conversion Coefficient of external low beta radiation in soil [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0

Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coefficient of external alpha radiation on soil [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coeff. external beta gamma radiation on soil [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	3.50E-4
Ra-226	Mammal (Deer)	1.80E-4
Ra-226	Mammal (Rat)	3.40E-4
Po-210	Detritivorous invertebrate	1.70E-9
Po-210	Mammal (Deer)	8.60E-10
Po-210	Mammal (Rat)	1.70E-9

Dose Conversion Coeff. of external low beta radiation on soil [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coefficient of internal alpha radiation [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	1.36E-2
Ra-226	Mammal (Deer)	1.38E-2
Ra-226	Mammal (Rat)	1.34E-2
Po-210	Detritivorous invertebrate	3.10E-3
Po-210	Mammal (Deer)	3.10E-3
Po-210	Mammal (Rat)	3.10E-3

Dose Conversion Coefficient of internal beta gamma radiation [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	4.20E-4
Ra-226	Mammal (Deer)	1.20E-3
Ra-226	Mammal (Rat)	7.00E-4
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Dose Conversion Coefficient of internal low beta radiation [µGy/h / Bq/m3]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	0.00E0
Ra-226	Mammal (Deer)	0.00E0
Ra-226	Mammal (Rat)	0.00E0
Po-210	Detritivorous invertebrate	0.00E0
Po-210	Mammal (Deer)	0.00E0
Po-210	Mammal (Rat)	0.00E0

Occupancy Factor [unitless] (default values)

Habitat	Organism	Value
On-soil	Detritivorous invertebrate	0.00E0
On-soil	Mammal (Deer)	1.00E0
On-soil	Mammal (Rat)	0.00E0
In-soil	Detritivorous invertebrate	1.00E0
In-soil	Mammal (Deer)	0.00E0
In-soil	Mammal (Rat)	1.00E0
In-air	Detritivorous invertebrate	0.00E0
In-air	Mammal (Deer)	0.00E0
In-air	Mammal (Rat)	0.00E0

Concentration Ratio (CR) [Bq/kg (f.w.) per Bq/kg soil (d.w.) (default values)

Nuclide	Organism	Value
Po	Detritivorous invertebrate	1.00E-1
Po	Mammal (Deer)	2.78E-3
Po	Mammal (Rat)	2.78E-3
Ra	Detritivorous invertebrate	9.00E-2
Ra	Mammal (Deer)	2.65E-2

Ra	Mammal (Rat)	2.65E-2

Dose Rate Screening Value (ERICA) [µGy/h] (default)

Organism	Value
Detritivorous invertebrate	1.00E1
Mammal (Deer)	1.00E1
Mammal (Rat)	1.00E1

Uncertainty Factor [unitless]

Value
3.00E0

Outputs

For all reference organisms the probability of exceeding the selected screening dose rate is below the probability selected.

Risk

Organism	Total Dose Rate per organism [µGy h-1]	Screening Value [µGy h-1]	Risk Quotient (expected value) [unitless]	Risk Quotient (conservative value) [unitless]
Detritivorous invertebrate	1.22E0	1.00E1	1.22E-1	3.66E-1
Mammal (Deer)	2.97E-1	1.00E1	2.97E-2	8.90E-2
Mammal (Rat)	3.39E-1	1.00E1	3.39E-2	1.02E-1

External Dose Rate [µGy/h]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	6.82E-2
Po-210	Detritivorous invertebrate	3.45E-7
Ra-226	Mammal (Deer)	1.35E-2
Po-210	Mammal (Deer)	6.45E-8
Ra-226	Mammal (Rat)	6.38E-2
Po-210	Mammal (Rat)	3.22E-7

Internal Dose Rate [µGy/h]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	9.19E-1
Po-210	Detritivorous invertebrate	2.32E-1
Ra-226	Mammal (Deer)	2.77E-1

Po-210	Mammal (Deer)	6.46E-3
Ra-226	Mammal (Rat)	2.69E-1
Po-210	Mammal (Rat)	6.46E-3

Total Dose Rate [µGy/h]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	9.88E-1
Po-210	Detritivorous invertebrate	2.33E-1
Ra-226	Mammal (Deer)	2.90E-1
Po-210	Mammal (Deer)	6.46E-3
Ra-226	Mammal (Rat)	3.32E-1
Po-210	Mammal (Rat)	6.46E-3

Activity Concentration in organism [Bq/kg f.w.]

Isotope	Organism	Value
Ra-226	Detritivorous invertebrate	6.75E0
Po-210	Detritivorous invertebrate	7.50E0
Ra-226	Mammal (Deer)	1.99E0
Po-210	Mammal (Deer)	2.08E-1
Ra-226	Mammal (Rat)	1.99E0
Po-210	Mammal (Rat)	2.08E-1

Media Activity Concentration in soil [Bq/kg]

Isotope	Value
Ra-226	7.50E1
Po-210	7.50E1

Risk Quotient [unitless]

Organism	Value
Detritivorous invertebrate	1.22E-1
Mammal (Deer)	2.97E-2
Mammal (Rat)	3.39E-2

Total Dose Rate per organism [µGy/h]

Organism	Value
Detritivorous invertebrate	1.22E0
Mammal (Deer)	2.97E-1
Mammal (Rat)	3.39E-1

Decision

Justification for the decision

The tier 2 assessment for Ra-226 and Po-210 and the most sensitive organisms (identified in Tier 1) has shown the risk as negligible

END OF REPORT