3 ENVIRONMENTAL RESEARCH AND MONITORING

The *Environment Protection (Alligator Rivers Region) Act 1978* established the Alligator Rivers Region Research Institute (ARRRI) to undertake research into the environmental effects of uranium mining in the Alligator Rivers Region (see Map 1). The scope of the research program was widened in 1994 following amendments to the Act. The Alligator Rivers Region Research Institute was subsequently renamed the Environmental Research Institute of the Supervising Scientist (***eriss***).

The core work of ***eriss*** comprises ongoing monitoring and conduct of research to develop and refine leading practice monitoring procedures and standards for the protection of people and the environment, focusing on the effects of uranium mining in the Alligator Rivers Region (ARR). The expertise of the Institute is also applied to conducting research on the sustainable use and environmental protection of tropical rivers and their associated wetlands.

The content and outcomes of the ***eriss*** research program are assessed annually by the Alligator Rivers Region Technical Committee (ARRTC) using identified Key Knowledge Needs (KKN). These KKNs define the key research topics within each of the geographic domains in the ARR relating to monitoring, closure and rehabilitation for current (Ranger and Jabiluka), rehabilitated (Nabarlek) and legacy (South Alligator River Valley) sites. The charter and activities of ARRTC are described in chapter 4 of this Annual Report and the current list of KKNs is provided for reference in Appendix 1.

eriss contributes to the addressing of each of the Key Knowledge Needs by applying a broad range of scientific expertise across the research fields of:

- Ecotoxicology
- Environmental radioactivity
- Hydrological and geomorphic processes
- Monitoring and ecosystem protection
- Spatial sciences and remote sensing

Highlights from the 2009–10 research program are presented in this report, with an overview introduction to these topics below.

Ongoing enhancement of monitoring methods is one of the key processes followed by SSD to ensure that leading practice continues to be employed for detection of possible impacts arising from the Ranger mining operation.

SSD has been undertaking an intensive evaluation since 2005–06 (see previous Annual Reports for details) of the use of continuous monitoring to provide essentially real time coverage of changes in water quality upstream and downstream of the Ranger minesite, in both Magela and Gulungul Creeks. This effort represents a major investment of Divisional resources and will result in substantially improved surveillance capacity compared with the historical weekly grab sampling approach to monitoring water quality. For wet seasons up to

2009–10 the grab sample and continuous monitoring programs were run in parallel. It is now planned that, starting with the 2010–11 wet season, the continuous monitoring system with associated event-based automatic sampling will become SSD's primary water quality monitoring platform.

To meet this goal has meant that FY 2009–10 has been a year of consolidation for all components of the research program that underpin the acquisition and interpretation of the continuous monitoring data. This has included enhancing the capability of the deployed instrumentation, improving the capacity for data transmission and analysis in Darwin, and continuing to completion the extensive ecotoxicological testwork, involving exposure of a suite of five aquatic test organisms to pulses of magnesium over periods of 4, 8, and 24 h (see 2008–09 Annual report for details), required to derive appropriate trigger values spanning this range of exposure conditions. Since this development work is incremental and will not be completed until the third quarter of 2010, it was decided to defer further reporting until the next Annual Report when the monitoring system will have been fully implemented and the interpretation framework developed and in place.

In the last Annual Report the major program of works being done by ***eriss*** to instrument four erosion plots on an eight hectare trial landform constructed during late 2008 and early 2009 by Energy Resources of Australia Ltd (ERA) was described. Data required to derive sediment and solute export concentrations and loads were collected through the 2009–10 wet season. An initial assessment is presented here of the very large amount of information obtained during the first wet season following construction. Updates of the findings from this multi-year project will be presented in future Annual Reports.

Gulungul Creek (see Map 2), a tributary of Magela Creek, is assuming increasing importance in the context of potential for runoff from the recently lifted tailings dam walls and the prospect of future mine-site infrastructure that may be located in the catchment of this creek. Accordingly, biological monitoring using the aquatic snail in situ method – described for Magela Creek in previous Annual Reports and in Chapter 2 of this report – was deployed in Gulungul Creek for the first time during the 2009–10 wet season. This first year was a pilot to establish the deployment logistics and investigate snail survivability in a new catchment regime. The snail data provided by this and subsequent wet seasons will provide a response baseline against which to assess the effects any future increases in mine-related activity on the aquatic ecosystem health of Gulungul Creek.

Research on the effect of dissolved organic carbon (DOC) on modulating metal toxicity was extended to aluminium during 2009–10. Aluminium is a potentially important component of the early first flush waters in the ARR given the acidic pH of the rain at the start of the wet season. It is also contained in acidic seepage and runoff waters from many operating and legacy minesites (including Rum Jungle) in the northern tropics. The results from this work have shown that the DOC naturally present in water in the Magela Creek catchment can substantively ameliorate the toxicity of Al.

Commissioning of the process water treatment plant at Ranger was completed in October 2009. As part of this process ***eriss*** undertook toxicity testing of the final treated water stream (reverse osmosis permeate) to confirm that there was no unanticipated toxicity, and that the toxicity of water produced by the full scale operating plant was comparable with that of the treated water originally produced by the pilot plant in 2001.

Accurate characterisation of the pre-mining radiological baseline is an essential precursor to being able to quantify rehabilitation success for a uranium minesite and to provide assurance that the requisite international standards for protection of members of the general public post rehabilitation are being met. In the case of Ranger mine, there was insufficient on-ground pre-mining survey work done to provide this baseline assessment. Consequently substantial research effort is being devoted to inferring this baseline using a combination of aerial radiometrics acquired for the lease area before mining started and contemporary intensive ground characterisation of undisturbed radiological anomalies (radiological analogues).

Since 1985, pond water stored in Retention Pond 2 and in the mine pits (firstly Pit 1 and then Pit 3) during the wet season has been disposed of on site using land application methods. This water contains uranium and other radionuclides (such as radium 226) that become bound to the near surface horizon of the soil. Over time the radiological load of the land application areas has increased. A comprehensive collaborative study is currently underway to definitively characterise the radiological status of these areas in the context of determining the extent of rehabilitation that will be needed for them. The contribution that ***eriss*** is making to the characterisation of the land application areas is described in this chapter.

In this report the first stage of developing a remote sensing monitoring framework for the ARR is described. The framework will provide the basis for efficiently and cost-effectively acquiring the spatial data needed to be able to place the land surface status of operating and rehabilitated minesites into a regional context.

Measurement of the radionuclide content of traditional bushfoods or ‘bushtucker’ obtained from many locations has been made by ***eriss*** over the past three decades. This unique resource continues to be updated on an annual basis. Emerging technologies such as Google Earth, Arc Explorer and ArcGlobe are being used to develop a user-friendly system to store and retrieve the data, and to present it in an understandable way to members of the local community.

During 2009–10 a project was commissioned to integrate the large volumes of knowledge acquired by ***eriss*** across its research program areas into a series of conceptual models describing potential contaminant transport pathways associated with uranium mining in the Alligator Rivers Region. This is being done as part of the evolving ecological risk assessment framework being developed by the Supervising Scientist for the operating phase of the mine. One of the key objectives of the project is to determine if there are any significant gaps in our scientific knowledge about the pathways that could potentially adversely impact on the health of the environment outside of the mine lease.

More comprehensive descriptions of ***eriss*** research are published in journal and conference papers and in the Supervising Scientist and Internal Report series. Publications by Supervising Scientist Division staff in 2009–10 are listed in Appendix 2. Presentations given during the year are listed in Appendix 3. More information on the Division’s publications, including the full list of staff publications from 1978 to the end of June 2010, is available on the SSD web site at www.environment.gov.au/ssd/publications.

3.1 Monitoring of erosion and solute loads from the Ranger trial landform

3.1.1 Introduction

A trial landform of approximately 200 m x 400 m (8 ha) was constructed during late 2008 and early 2009 by Energy Resources of Australia Ltd (ERA) adjacent to the north-western wall of the tailings storage facility (TSF) at Ranger mine (Map 2). The trial landform will be used to test landform design and revegetation strategies to assist ERA develop a robust rehabilitation strategy for deployment once mining and milling have finished.

The landform was designed to test two types of potential final cover layers:

1 Waste rock alone

2 Waste rock blended with approximately 30% v/v fine-grained weathered horizon material (laterite).

The landform is divided into six treatment areas (Figure 3.1).

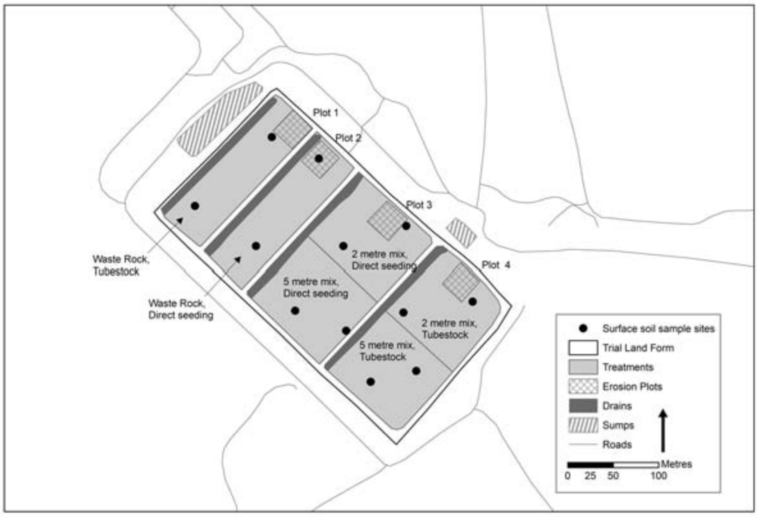

Figure 3.1 Layout of the plots on the trial landform

Each treatment was designed to test different planting methods and substrate types as follows:

1 Tube stock planted in waste rock material

2 Direct seeded in waste rock material

3 Direct seeded in waste rock mixed with laterite to a depth of 2 m

4 Direct seeded in waste rock mixed with laterite to a depth of 5 m

5 Tube stock planted in waste rock mixed with laterite material to a depth of 2 m

6 Tube stock planted in waste rock mixed with laterite material to a depth of 5 m

During the 2009 dry season, surface samples were collected by spade to a maximum depth of 10 cm from 12 randomly chosen locations over the trial landform surface to characterise the particle size distribution. Eight mixed and four waste rock only samples made up the total of 12. For all 12 samples, more than half of each sample (by weight) (53–78%) was larger than 2.0 mm in diameter showing the influence of the waste rock on the composition of the cover treatments. The fraction greater than 2.0 mm from surface soil on the natural surrounding Koolpinyah surface is always less than 50% and is generally no greater then 10%.

Four erosion plots (30 m x 30 m) (location marked by cross hatched small squares on Figure 3.1) were constructed on the landform surface and physically isolated by engineered borders from runoff from the rest of the area. Half-section 300 mm diameter U-PVC stormwater pipes were placed at the down slope ends of the plots to catch runoff and channel it through rectangular broad-crested (RBC) flumes (Figure 3.2) where rainfall event triggered discharge is measured. A reservoir (stilling basin) is located upstream of the inlet to each flume to trap coarser material eroded from the plot. The outlet of each erosion plot was instrumented with the following sensors:

- pressure transducer and shaft encoder to measure stage height
- a turbidity probe
- electrical conductivity probes located at the inlet to the stilling well and in the entry to the flume to provide a measure of the concentrations of dissolved salts in the runoff
- an automatic water sampler to collect event based samples
- a data logger with mobile phone telemetry connection

Figure 3.2 Runoff through flume on trial landform erosion plot 3 during a storm event

A rain gauge was also installed at the downstream end of each plot near the instrument shelter. Data acquired during the 2009–10 wet season were downloaded daily by mobile phone access and then stored in the hydrological database Hydstra.

During the 2009–10 wet season runoff, turbidity (surrogate of fine suspended sediment), bedload (coarser material deposited in the stilling basin) and EC (surrogate of water quality) were measured. The first rainfall event of 26 mm occurred on 23/9/09 and the last significant rainfall event of 17 mm occurred on 17/4/10. The total rainfall for the 2009–10 wet season (averaged across the four plots) was 1491 mm.

During rainfall induced runoff events water samples were collected by automatic water samplers triggered by pre-programmed increases in stage height, turbidity and EC. The trial landform was visited once a week to collect the water samples and the bedload. This task was shared between staff from SSD and ERA, with the allocation of staff resources and workplan defined in a formal memorandum of understanding between SSD and ERA. SSD was responsible for processing and analysis of all of the samples collected for the sediment transport component of the project; ERA was responsible for chemical analysis of the water samples.

3.1.2 Topographic surveys

Two topographic surveys of the trial landform were completed during 2009–2010. The first manual survey using a total station was undertaken in December 2009 prior to the onset of heavy rains. A total of 1737 points were collected across the surface at approximately 5-metre intervals and used to generate a medium resolution (5 metre) digital elevation model (DEM) (Figure 3.3).

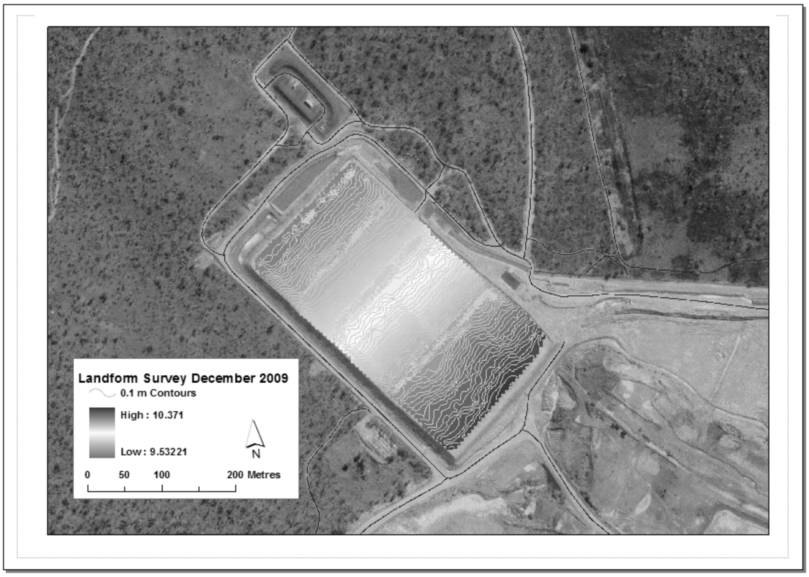

Figure 3.3 Digital Elevation Model of trial landform (with overlaid 0.1 m contour lines) produced from the December 2009 survey

During the course of this survey it was noted that the vegetation growth that will occur over the next few years will progressively compromise line-of-sight or optical surveying methods. Consequently, it will be necessary in the future to employ survey technologies (for example, LIDAR – Light Detection and Ranging) capable of penetrating through vegetation cover to measure ground level.

A second survey was undertaken in June 2010 during the early dry season using a Leica ScanStation2 laser scanning instrument and differential GPS. In contrast to the earlier manual point survey, the use of the laser scanner enabled both surface elevation data as well as surface features (such as the current status of vegetation communities) to be captured.

Twenty-five scans were made across the landform (Figure 3.4). Three scans were undertaken within each of the erosion plots, at a scan resolution of 2 cm. A further 13 scans were made across the landform at a coarser resolution of 20 cm.

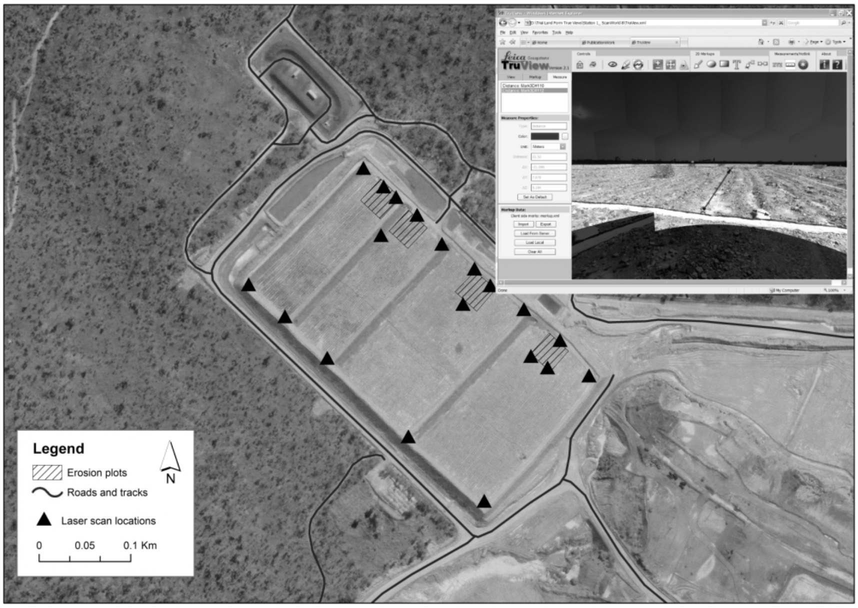

Figure 3.4 Locations (marked by triangles) of scanning laser instrument. Inset shows an example of the type of composite digital image synthesised from multiple images captured at each scan location.

The data collected from the second survey are currently being processed to provide a very high spatial resolution DEM of the surface. To date, data have been extracted to generate a DEM for Erosion Plot 2 with a horizontal resolution of 20 cm (Figure 3.5). The DEM spans an elevation range of 1.24 m between the highest and lowest points in the plot. At this resolution, the rip lines, boulders and pits in the plot are clearly visible

The very high resolution digital elevation model was acquired to underpin several components of ***eriss***’s minesite rehabilitation research. In particular, it will provide the input data needed for the CAESAR and Siberia landform evolution models that are being used to test the long-term stability of the trial landform against the erosive effects of high intensity rainfall events.

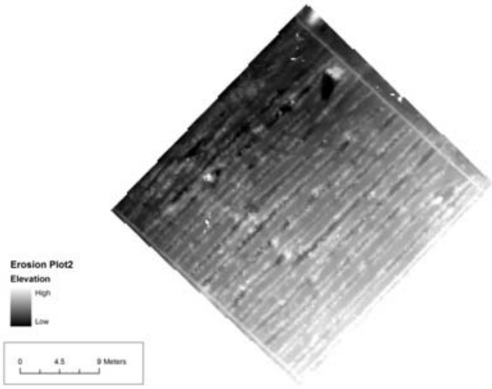

Figure 3.5 High resolution digital elevation model of erosion plot 2. Lighter colours represent areas of greater elevation. Riplines, boulders (light) and pits (dark) visible on surface.

3.1.3 Sediment transport

Fine suspended sediment

Turbidity sensors were installed at the exit to each of the settling basins on each of the erosion plots. Turbidity provides a measure of the concentration of fine suspended sediment. It is this fine material that is of most immediate relevance from the perspective of the potential for downstream environmental impact of material eroded from a newly constructed mine landform.

An example of concurrent typical turbidity events occurring across each of the four erosion plots in response to a rainfall event is shown in Figure 3.6 below.

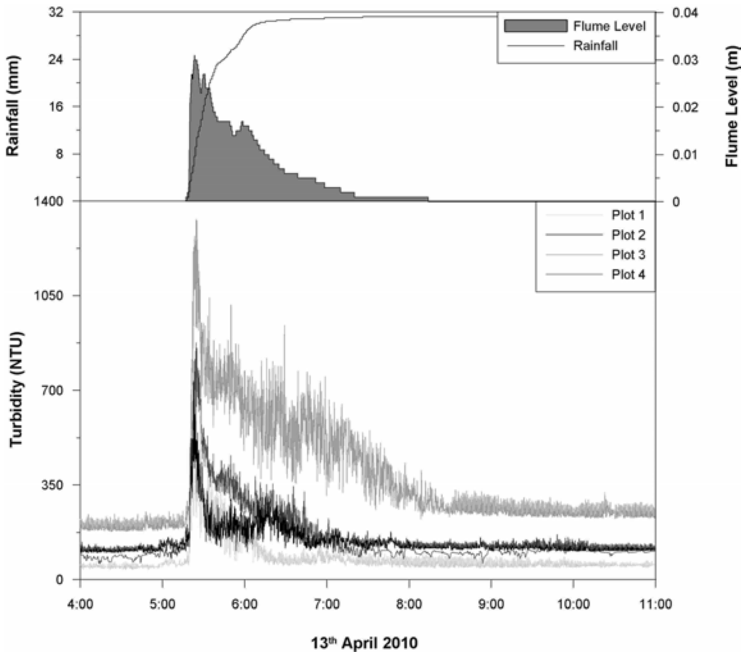

Figure 3.6 Rainfall induced turbidity events occurring between 4 and 11am on 13 April 2010. Top panel shows the cumulative rainfall and flume water level (surrogate of flow) for plot 2. The bottom panel displays the continuous turbidity data from each of the four erosion plots.

The magnitude of the pulses for the waste rock plots (plots 1 and 2) are generally similar to one another and lower than the pulses observed for the mixed waste rock and laterite plots (plots 3 & 4). Throughout the season, the turbidity measured at plot 3 was consistently higher than that measured at plot 4.

Water samples were collected during rainfall events using autosamplers activated using a combination of pre-programmed stage height, EC and turbidity values. All samples triggered by turbidity were analysed for total suspended sediment (TSS) concentration (sediment fraction between 63 µm and 0.45 µm). The TSS concentration was determined by firstly passing the water sample through a 63 µm sieve and then filtering a standard volume through a 0.45 µm filter. The weight of the dried residue on the filter paper was then measured. The TSS data will be used to define the relationship between TSS and turbidity measured in situ, allowing estimation of continuous TSS concentration from the continuous turbidity data.

The TSS is the most readily transportable fraction of sediment and is a key indicator of landform surface erosion rates. Selected TSS samples will be analysed for associated trace metal concentrations (including uranium) to derive the loads of sediment-associated contaminants transported from each of the erosion plots during the 2009–10 wet season.

Bedload

The coarser bedload material is deposited in both the half pipe defining the downslope boundary of the plot and in the stilling basin upstream of the flume. The total amount of bedload collected from each plot over the wet season is shown in Table 3.1. Similar amounts of bedload material were washed from each of the plots, with no systematic difference between the two surface treatments.

TABLE 3.1 TOTAL BEDLOAD COLLECTED FOR 2009–10 WET SEASON

Erosion plot	Basin (kg)	Half-pipe (kg)	Total (kg)
EP1	24.2	71.7	95.9
EP2	9.6	117.6	127.2
EP3	15.9	86.3	102.3
EP4	64.9	57.4	122.3

The particle size distributions measured for bedload samples collected on 17/03/2010 and 15/04/2010 are provided in Table 3.2 to illustrate the different behaviours between the plots, and the influence of rainfall event magnitude. Sieving was used for size classification above 63 µm. The hydrometer (gravity settling) method was used for more detailed classification (not shown here) of the less than 63 µm fraction.

TABLE 3.2 BEDLOAD PARTICLE SIZE DISTRIBUTION DATA (DRY WEIGHT BASIS) FOR SAMPLES COLLECTED ON 17 MARCH 2010 AND 15 APRIL 2010

Sample erosion plot	Sample date	Sample mass (kg)	% > 2.00 mm	% < 2.00 mm	
				% > 0.0063 mm	% < 0.0063 mm
EP1	17/03/2010	1.5	18.7	73.6	7.7
EP2	17/03/2010	1.9	17.9	59.7	22.4
EP3	17/03/2010	1.3	28.2	61.0	10.8
EP4	17/03/2010	1.5	15.0	75.1	9.9
EP1	15/04/2010	14.4	33.3	61.7	5.0
EP2	15/04/2010	15.2	24.6	63.7	11.7
EP3	15/04/2010	12.9	53.6	44.5	1.9
EP4	15/04/2010	12.4	45.2	52.2	2.6

The rainfall events that produced the amounts of bedload reported in Table 3.2 are shown in Table 3.3. The bedload collected on 17/3/10 resulted from 49 mm of rainfall over 4 events and was correspondingly much lower in mass than the bedload collected from 15/4/10 which was the result of 254 mm of rainfall over 8 events.

TABLE 3.3 RAINFALL EVENTS DURING THE WEEK PRIOR TO BEDLOAD COLLECTION

Sample date	Total rain (mm)	No of events	Event 1 (mm)	Event 2 (mm)	Event 3 (mm)	Event 4 (mm)	Event 5 (mm)	Event 6 (mm)	Event 7 (mm)	Event 8 (mm)
17/3/10	49	4	5	16	9	15				
15/4/10	254	8	58	5	11	47	30	41	25	26

3.1.4 Solute transport

EC sensors were installed at the entrance and the exit of the sediment settling basin at each of the erosion plots. The information from both of the sensors was used to derive event-based EC data for each site over the 2009–10 wet season. The behaviour of EC observed over an event will be determined by the condition of the basin preceding the rainfall. Two possible conditions apply for this system:

1. The basin was empty and clean prior to rainfall, in which case the EC is indicating the composition of surface runoff throughout the event.

2. The basin was full prior to rainfall, in which case the EC trace measured at the exit to the basin could be impacted by ‘stale’ water that has remained in the basin between rainfall events.

Condition 1 events give a clear indication of the surface runoff water quality. Condition 2 events are confounded due to the mixing of the surface runoff with ‘stale’ water in the basin that has accumulated from a varying number of antecedent events. While the majority of events occurring throughout the wet season occurred under condition 2, the potential confounding caused by the ‘stale’ water can be removed by comparison of the EC values measured at the entrance and the exit of the basin. The time at which the two EC readings converge will indicate when complete flushing of the ‘stale’ water has occurred. Detailed analysis of the time series EC data for the condition 2 events is still in progress. Consequently the results reported here will focus on condition 1 events.

Thirteen condition 1 events occurred during the 2009–10 wet season. However, the intensity of the rainfall and associated runoff volume for the majority of these events was low, with only five of the 13 events falling in the upper 50^{th} percentile of rainfall volume and intensity for the season. Figure 3.7 shows summary statistics describing the peak (maximum) EC values recorded for each of the 13 events for plots 2 and 4, representing the waste rock and waste rock mixed with laterite, respectively. The box and whisker plot shows that the medians and general distribution of the peak EC values for each plot are similar. The scatter plot shows that the distribution of peak EC values as a function of total rainfall for each event are similar for both plots, indicating that the total amount of solutes derived from both treatments are similar (for condition 1 events less than 35 mm).

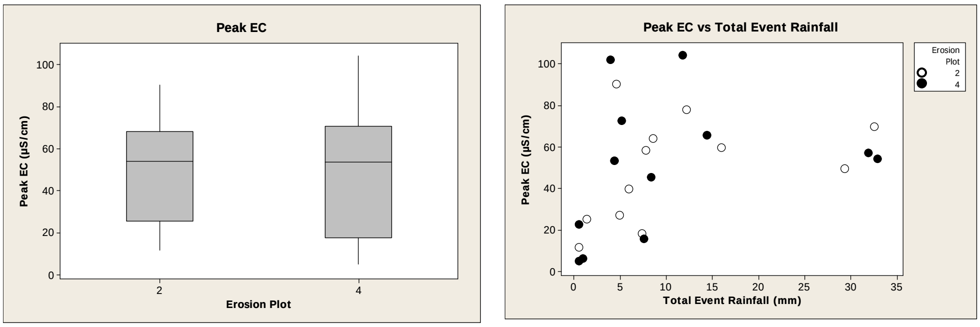

Figure 3.7 Box plot summarising the mean, maximum, minimum, third quartile and first quartiles of the maximum first flush EC values; and scatter plot of the maximum first flush EC values and total event rainfall

Water samples were collected for chemical analysis from each of the erosion plots using autosamplers which were activated using a combination of stage height and EC triggers. The EC-triggered samples were analysed by ERA in its on-site laboratory for a suite of trace elements and major ions. The results obtained for Mg, SO_4 and U only are presented here (Figure 3.8) since these solutes are the most relevant for potential environmental impact from the site. The box plots in Figure 3.8 show the concentration means and ranges measured for each of the three solutes in the water from each of the four plots.

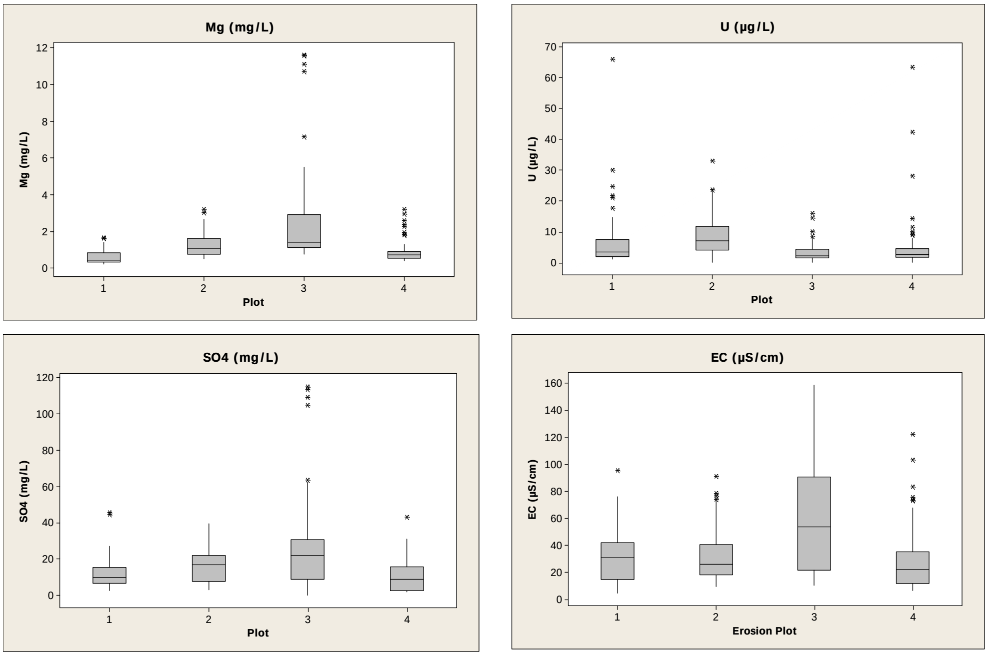

Figure 3.8 Box plots showing the mean, maximum, minimum, third quartile and first quartile values of Mg, SO4, U and EC measured in the water samples collected from each plot over the 2009–10 wet season. The number of data points (n) for erosion lots 1, 2, 3 and 4 are 54, 76, 67 and 86, respectively.

The summary statistics provided show that:

- Mg and SO_4 exhibit similar behaviour over the four erosion plots
- EC exhibits similar behaviour to Mg and SO_4, indicating that these ions are major contributors to the EC of the surface runoff from each plot
- Plots one, two and four all have similar concentration ranges for both Mg and SO_4 (and hence EC). However, plot three has a broader range and maximum values at least double that of each of the other plots
- The highest concentrations of U were measured for plots one and four, noting that the majority of U concentrations were less than 30 µg/L and that the means, except for plot two, were all less than 6 µg/L, which is the current ecotoxicologically derived limit for U in Magela Creek.

Apart from some individual higher U concentrations measured for plots one and four, each of the plots show a similar distribution of U concentrations. While plot three stands out from the others due to the generally higher solute concentrations, plots one, two and four exhibit similar surface runoff water quality.

Being a composite of all of the data, the box plot summaries do not demonstrate the dynamic range of concentrations that occur through a rainfall event. To do this, individual events need to be analysed. Figures 3.9 and 3.10 show examples of the time series concentrations of Mg and U measured through two rainfall events that produced sustained flow through the installed flumes.

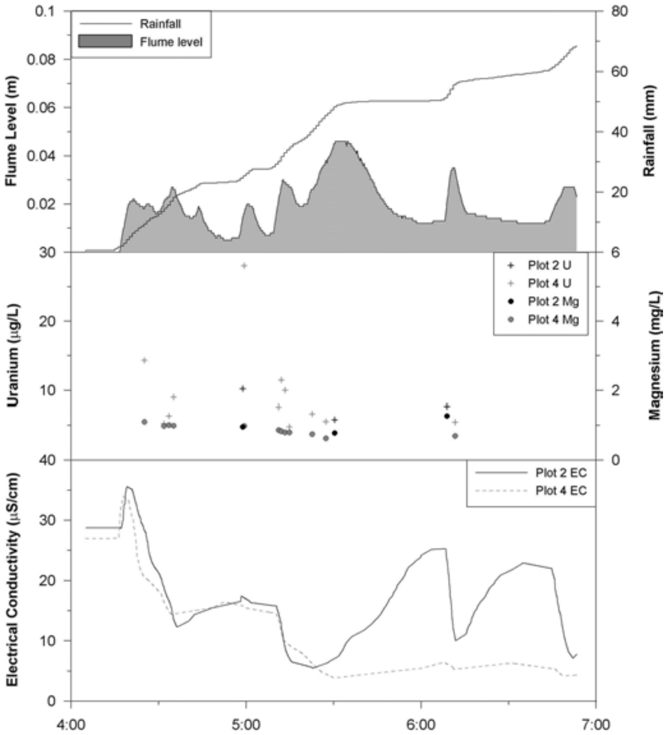

Figure 3.9 Rainfall event occurring between 4 and 7 am on 1 February 2010. Rainfall, EC, Mg, U and flume level from plots two and four are shown for comparison between wasterock and laterite treatments.

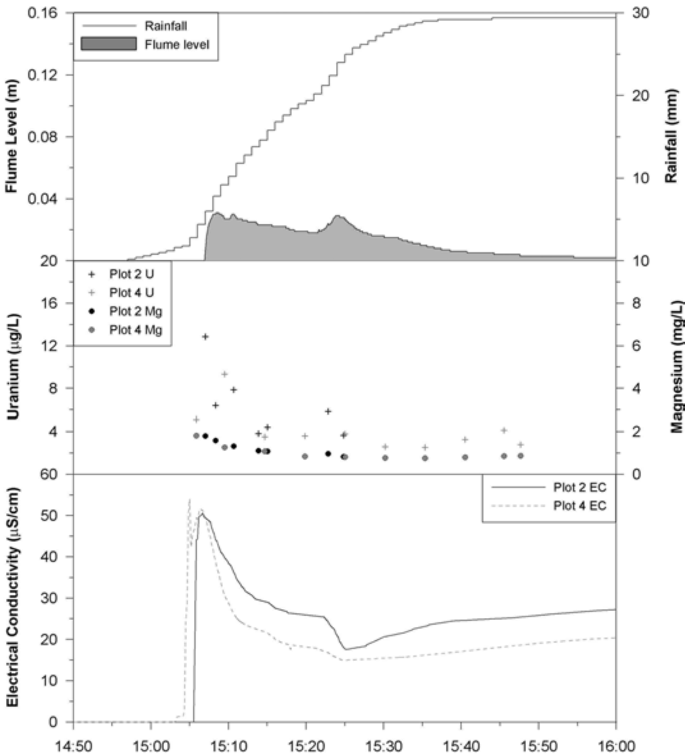

Figure 3.10 Rainfall event occurring between 3 and 4 pm on 23 March 2010. Rainfall, EC, Mg, U and flume level from plots two and four are shown for comparison between waste rock and laterite treatments.

The concentrations of Mg and U are very similar between the two plots for these events. There is a difference in EC between 0530 and 0700h in Figure 3.9. However, this particular event represents the low end of the EC range (0–700 µS/cm) measured over the wet season so the effect on solute load of the differences observed between the plots for this event is low. Further data analysis is required to statistically define the significance of such variation.

3.1.5 Future work

Considerable resources are being devoted to processing, collating and analysing the large amounts of data produced from the trial landform during the 2009–10 wet season. Examples have been provided in this report of the wide range of information that is being produced by the project. The findings will be used to inform analysis of the suitability of options for the design and revegetation of the final rehabilitated Ranger site. During Q1 and Q2 of the 2010–11 financial year it is anticipated that loads of solutes, suspended sediment and bedload material will be derived for each of the plots, enabling quantitative comparison of the behaviours of the two types of surface treatments. These results will be documented in the next Annual Report.

The scope of the trial landform monitoring program for the 2010–11 wet season will be refined using the findings from the 2009–10 season, with more selective sampling and analysis of the runoff streams.

3.2 In situ biological monitoring in Gulungul Creek

In recognition of the increasing importance of Gulungul Creek in the context of runoff from the recently lifted tailings dam walls and the prospect of future mine-site infrastructure that may be constructed in the catchment due to proposed expansion of mining and milling at Ranger, SSD has increased its environmental monitoring effort in this creek. In addition to upgrades of the continuous monitoring equipment in the creek, biological (toxicity) monitoring also commenced in the 2009–10 wet season with the trial in situ deployment of the freshwater snail reproduction technique. This method of biological monitoring has been routinely deployed in Magela Creek over many years with the results documented in previous Annual Reports. As with toxicity monitoring in Magela Creek (section 2.2.3.2), it is intended that in situ biological monitoring will be used in Gulungul Creek as an early detection method for identifying changes in water quality.

The trial deployment was conducted firstly to establish the logistics of reliably conducting toxicity monitoring procedures in the creek and secondly to start acquiring biological response data to develop a baseline prior to any significant future disturbance in the catchment. The test design was the same as that used for the routine monitoring of Magela Creek (see section 2.2.3) with upstream ‘control’ and downstream ‘exposed’ sites co-located with water quality monitoring (Gulungul u/s and Gulungul d/s on Map 2). While the control and exposed sites in Magela Creek are accessible by boat throughout the wet season, the upstream control site on Gulungul Creek is not accessible by boat at any time, nor by road for the majority of the wet season. Hence it is necessary to access this site by helicopter.

Five tests were conducted through the 2009–10 wet season, over a range of flow conditions, and in alternate weeks to the routine Magela Creek testing. Tests were conducted in the periods 25–29 January, 22–26 February, 22–26 March, 9–13 April and 19–23 April 2010. The results, together with comparative results from Magela Creek, are shown in Figure 3.11. The range in egg number observed in Gulungul Creek was similar to that recorded in Magela Creek (Figure 3.11).

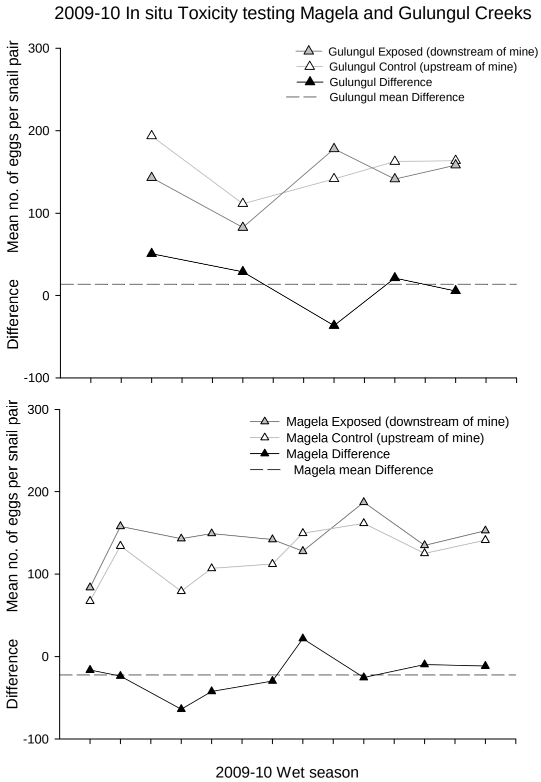

Figure 3.11 In situ toxicity monitoring results for freshwater snail egg production for Gulungul Creek compared with results from Magela Creek, 2009–10 wet season

Four out of the five tests resulted in positive difference values, ie egg production was higher upstream than downstream. This pattern was opposite to that observed in Magela Creek during the same period, where eight of the nine tests resulted in a negative difference value (Figure 3.11). High statistical power in this toxicity monitoring technique is potentially available when, in the absence of human-related disturbance downstream of potential sources of impact, the responses measured at upstream and downstream sites are very similar in magnitude to one another over time. This concordance (or 'tracking') in egg number between upstream and downstream sites is the typical pattern in Magela Creek (Figure 3.11), and also appears to be the pattern in Gulungul Creek.

It is anticipated that fortnightly in situ toxicity testing will be implemented in Gulungul Creek during the 2010–11 wet season.

3.3 Developing sediment quality criteria for uranium

Research and monitoring of the impacts of mining at Ranger have historically focused on the water column, as this environmental compartment is the primary transport vector for solutes released from the minesite. However, since uranium (U, present in water as the uranyl ion) has a high affinity for sediments, sediment quality assessment and the derivation of protective trigger values for sediments are aspects of aquatic ecosystem protection that also need to be considered. Such trigger values will have application both for operational water management and for the development of sediment quality closure criteria for the Ranger site.

There has been little work conducted on the toxicity of U in sediments to aquatic biota, and the toxicity estimates produced by the few international studies that have been published have varied by at least three orders of magnitude (from 5.3 to >5000 mg U/kg dry weight). The lack of a robust toxicity guideline for U is of concern – not only for the local situation but also nationally given the projected expansion of the uranium mining industry.

On the Ranger lease, concentrations of U in the sediments of mine-influenced waterbodies such as Georgetown Billabong (GTB, up to 45 mg/kg) are higher than reference waterbodies (1.2–4.3 mg/kg). While U concentrations in the sediments of GTB have been higher than in other billabongs of the region since before the start of mining, there appears to have been a further increase in GTB since about 2002. Additionally, the communities of benthic macroinvertebrates in the mine-influenced waterbodies, Georgetown and Coonjimba Billabongs, currently exhibit lower diversity than reference billabongs.

Concurrent investigations are presently underway to determine whether the observed impoverishment of benthic macroinvertebrates is due to the presence of higher concentrations of U, or other mining or non-mining (for example, differences in natural habitat such as bed sediment type) related factors. This report documents the progress that has been made over the past year to develop U sediment quality criteria for the protection of sediment-dwelling biota. These criteria are required for both the operational life of the mine and for its successful closure. In particular, sediment quality criteria will be required for on-site sentinel wetlands, which will serve to capture and 'polish' seepage and runoff waters from the rehabilitated mine site, as well as for downstream receiving waterbodies in the rehabilitation phase.

A pilot field sediment U toxicity study was undertaken, in collaboration with the CSIRO Centre for Environmental Contaminants Research and Charles Darwin University, during the 2009–10 wet season. Field studies have several benefits over laboratory assessments. In particular, a field experiment can be more time and cost-effective than a laboratory approach (not requiring selection and culturing of numerous suitable local test species, nor development of test protocols); and will be able to assess responses of whole communities of organisms with the results more likely to be directly applicable to managing the natural environment.

The experimental approach involved the deployment of U-spiked sediments (in retrievable containers) in (unimpacted) Gulungul Billabong over the duration of a wet season. At the end of the exposure period, the extent of colonisation of macroinvertebrate, microinvertebrate and microbial communities was measured in the control and U-treated replicates. Research activity during 2009–10 focused on analysis of data collected during a site characterisation field trip, and undertaking of a pilot experiment during the 2009–10 wet season.

3.3.1 Site characterisation

Eighteen sediment samples (~20 cm × 20 cm × 10 cm; ~4000 cm^3) were collected from Gulungul Billabong in April 2009 to determine the baseline physico-chemical and biological conditions of the study site. Baseline whole sediment concentrations for key metals were: Al – 49 000 mg/kg (dry weight); As – 2 mg/kg; Cu – 30 mg/kg; Fe – 11 700 mg/kg; Mn – 61 mg/kg; Pb – 12 mg/kg; U – 6 mg/kg; and Zn – 13 mg/kg. The sediment is classified as a granular medium sand, with approximately equal proportions of sand (<2 mm – >0.063 mm), silt (<0.063 mm – >0.0039 mm) and clay (<0.0039 mm). There was zero gravel (>2 mm) present (Wentworth grain size scale).

Taxa numbers and abundances of macroinvertebrates in the sediments (>500 µm fraction) were low (mean ± standard deviation of 8.4 ± 2 taxa and 335 ± 183 organisms per sample respectively; n = 18), possibly reflecting the fine-grained sediment particles (restricting habitat availability) and low dissolved oxygen environment characteristic of billabong waters at depth during the late wet season. Microinvertebrate taxa numbers in the sediments were higher than for macroinvertebrates (mean ± standard deviation of 18 ± 5 and 12 ± 3 taxa per sample for 63–125 µm and 125–500 µm fractions respectively; n = 3) while microinvertebrate abundances were orders of magnitude higher (mean ± standard deviation of 180 000 ± 37 000 and 18 000 ± 9 000 organisms per sample for 63–125 µm and 125–500 µm fractions respectively; n = 3).

Sample processing for microinvertebrates in particular, is a very laborious process, requiring meticulous separation of (often) cryptic (ie concealed or camouflaged) organisms from the fine-grained sediment particles using fine tungsten needles. Consequently, only three of the original 18 samples were processed for microinvertebrates. Among the samples processed, the fauna were dominated by protists (in particular, Rhizopoda, Difflugiidae – amoeboids inhabiting a test or shell) and rotifers (in particular, the Lecanidae).

Characterisation of the microbial assemblage in the sediment was done by extracting DNA and using the technique of terminal restriction fragment length polymorphism (TRFLP). This revealed over 130 ‘operational taxonomic units’ (OTUs; bacterial species and/or strains of species) with a mean ± SD of 38 ± 25 OTUs per sample (n = 18). A metagenomic analysis of the sediments was also conducted, using pyroseqencing techniques. Most of the bacteria identified in the TRFLP data also appear in the metagenomic analysis, although numerous other bacteria were identified with pyrosequencing. Analysis at the phylum level revealed that the community composition along the transect was relatively uniform, with the soil bacteria *Acidobacteria*, *Proteobacteria* and *Actinobacteria* well represented. Of the total number of OTUs revealed, 40–50% from each site represented yet to be identified bacteria.

3.3.2 Pilot study

Preparations for the pilot study over the 2009–10 wet season, commenced in August 2009. A bulk sample of moist sediment (~150 kg) was collected from the exposed littoral zone at the study site in Gulungul Billabong. The sediment was frozen for 1 week (to kill the majority of the biota resident in the samples) then wet sieved through a 2 mm mesh size with deionised water. This created a slurry (1:1.4 sediment to water ratio) suitable for spiking with U. The slurry was split into four 30 kg batches for the following treatments: zero addition control; 5400 mg/kg (sodium) sulfate control; 400 mg/kg U; and 4000 mg/kg U (U was added as uranyl sulfate). The batches were mixed in a cement mixer for 1 hour once every 2 days for 14 days.

Following mixing, the sediments for each treatment were placed in the dark at 4°C for 21 days to allow time for adsorption of spiked metal (or ion) to the sediment. Following a 10-d period of drying at ambient temperature (24–35°C), the sediments were transferred to the experimental containers. There were nine containers (~20 × 20 × 15 cm plastic containers with ~5 mm mesh size sides and base) for each treatment, with each container holding ~2 L (or 2000 cm^3 – 20 × 20 × 5 cm) of sediment. The test containers were then placed in holding containers, covered, and left in the dark at 4°C for approximately 10 weeks prior to their deployment in the field. Sub-samples for sediment and porewater chemistry were collected at regular intervals throughout the equilibration periods.

The sediments were deployed at pre-determined locations at the study site on 9 December 2009 (Figure 3.12), approximately 2 weeks prior to the onset of the first monsoonal rains of the wet season. At this time, replicates of an additional control treatment (Gulungul Control; GC) were included in the design, namely natural surface sediment from the study site that had not been pre-treated in the same manner as the other four treatments. For this GC treatment, natural sediments were excavated and placed in test containers of the same type used for the other four treatments. Because the GC sediments were essentially dry, they were broken down by hand and moistened using deionised water so that they filled the containers with no significant air spaces remaining.

Equivalent volumes of natural sediments at the site were excavated at the designated locations, to create cavities for the field placement of the test containers. The containers were set such that the surface of the sediment in the container was flush with that of the surrounding natural sediment. The field placement for the 5 × 9 (treatment × replicates) containers used a statistical design that ensured the elimination of biases in potential environmental gradients at the site that could otherwise potentially confound results.

By early January 2010, the site was inundated with water (Figure 3.12), and remained so for the rest of the wet season The containers of test sediment were retrieved on 30 March 2010, after being submerged for 3 months. Prior to processing, cores of sediment (30–50 mm depth × 15 mm diameter) were obtained from each container for detailed chemical and microbial analysis. The contents of each replicate container was then elutriated through stacked sieves of 8 mm, and 500, 125 and 63 μm mesh, with the > 500 μm fractions retained for macroinvertebrate characterisation and each of the smaller mesh fractions preserved separately in 90% ethanol for microinvertebrate (125 and 63 μm) characterisation.

Water and sediment material left over from the processing of the two uranium treatments were combined in a bulk container and retained for later safe disposal using an approved protocol.

At the time of collection of the retrieved samples it was observed that the method of preparation in the laboratory followed by exposure to hot and dry conditions in the field for about three weeks prior to inundation, had profoundly altered their physical condition compared with the undisturbed in situ sediment. In particular, the samples had a compacted and hardened ('baked') appearance, unlike that of the naturally occurring mostly softer and yielding sediments at the site, and that of the GC controls that had been prepared at the time of deployment of the extensively pre-treated material. At this time it was suspected that the greatly changed physical nature of the pre-treated sediment could have inhibited penetration and colonisation by organisms present in the surrounding natural sediments and surface waters, thereby creating an experimental artefact.

Initial results appear to confirm this expectation, with densities of macroinvertebrates appreciably lower in the extensively manipulated treatments compared with the GC controls. Processing of additional replicates will be done to determine the extent to which the method of preparation compromised the validity of the pilot trial. The work program for 2010–11 will focus on developing methods for sediment preparation and spiking that do not so greatly disturb the physical characteristics of the sediment. Pending the outcome of these investigations, the conduct of a comprehensive uranium sediment toxicity field trial has been deferred until 2011–12.

Figure 3.12 Top: deployment of uranium-spiked sediments for pilot experiment at Gulungul Billabong study site, 9 December 2009. Bottom: Gulungul Billabong study site, 7 January 2010.

3.4 Toxicity testing of Ranger process water permeate

Active treatment of process water at Ranger was implemented in late 2009 to accelerate reduction of the process water inventory. Untreated process water typically has a pH of ~4, an electrical conductivity > 25 000 µS/cm and contains highly elevated concentrations of sulfate – >30 000 mg/L, magnesium – >5000 mg/L, total ammonium ~900 mg/L, uranium (U) – >25 mg/L, aluminium – >400 mg/L and manganese – >2000 mg/L).

The treatment of process water comprises lime and carbon dioxide softening, followed by microfiltration/ultrafiltration, and finally reverse osmosis. The water treatment plant was designed to produce water to a standard such that the treated water, after an additional passive wetland polishing treatment, would be suitable for release to the off-site aquatic environment with no measurable biological impact. The final wetland step was specifically intended to remove residual ammonia (present in solution as ammonium ion) given that it was anticipated that the reverse osmosis treated water (permeate) from the water treatment plant could contain up to 20 mg/L of this species. Ammonia is both a toxicant and a nutrient so it is important that its concentration is reduced to environmentally acceptable levels prior to release of the final treated water.

A key question to be addressed from both an operational and environmental perspective, notwithstanding the wetland biopolishing step, was the extent to which the permeate contained residual toxicity, and whether this toxicity could be accounted for by the ammonium present. Toxicity testing in 2001 of the permeate produced from a pilot water treatment plant indicated low toxicity to three aquatic species, with IC/LC50 ratios ranging from 44% to >100% permeate. The aims of the present study were to (i) assess the toxicity of permeate from the full scale treatment plant commissioned at Ranger mine and, if residual effects were observed, to (ii) identify the cause/s of the effects.

Commissioning of the process water treatment plant at Ranger was completed in October 2009. On 26 October 2009, following advice from ERA that the permeate being produced was representative of typical outputs, SSD staff collected a sample for toxicity testing in the SSD Darwin laboratories. Separate samples of the permeate were collected for analysis of chemical constituents.

The chemical composition of permeate is compared with process water and Magela Creek water in Table 3.4. The treatment process was highly effective in removing major ions and metals from process water, including U. Analytes present in the permeate at concentrations substantially above those of natural Magela Creek water included ammonia (6.7 mg/L, as total ammonia-N), boron (236 µg/L), bromine (49 µg/L), rubidium (4 µg/L) and rhenium (10 µg/L). The ammonia concentration, although greatly reduced by the treatment process , was still at least seven times higher than the Australian and New Zealand water quality trigger value of 0.9 mg/L applying at the pH of the permeate (pH 8). Existing toxicity data suggest that the other analytes listed above were unlikely to be a concern.

Although the concentration of U (0.07 µg/L) in the permeate was an order of magnitude greater than background concentrations in Magela Creek water (0.005 µg/L), it was two orders of magnitude lower than the derived site-specific Limit for uranium in Magela Creek of 6 µg/L. Hence U is not a toxicant of concern in the permeate sample submitted for testing.

TABLE 3.4 WATER QUALITY OF MAGELA CREEK WATER AND UNTREATED AND TREATED PROCESS WATER FROM RANGER URANIUM MINE

Variable	Magela Creek water	Process water	
		Untreated [a]	Treated)
pH	6.2	3.9	8.3
Electrical conductivity (μS/cm)	18	28 200	91
Dissolved organic carbon (mg/L)	2.6	NM[b]	1.5
NO_3_N (mg/L)	<0.005	1.77[c]	0.005
NH_3_N (mg/L)	<0.005	1040[c]	6.8
Ca (mg/L)	0.2	602[c]	<0.1
Mg (mg/L)	1.1	6390[c]	<0.1
Na (mg/L)	1.3	97[c]	4.9
SO_4 (mg/L)	0.3	38 600[c]	2.4
Al (μg/L)	5.5	491 000	3.1
B (μg/L)	12	NM	236
Br (μg/L)	10	NM	49
Cu (μg/L)	0.23	12 600	0.2
Fe (μg/L)	40	10 300	<20
Mn (μg/L)	2	2 520 000	<0.01
Pb (μg/L)	0.22	4480	0.05
Rb (μg/L)	0.5	NM	4
Re (μg/L)	<0.05	NM	10
U (μg/L)	0.005	32 300	0.074
Zn (μg/L)	0.5	6130	<0.1

a Unless otherwise stated, values for untreated process water represent measurements from a sample collected on 2 November 2009, one week after the permeate sample for toxicity testing was collected. Data supplied by Energy Resources of Australia Ltd (ERA).

b NM: Not measured

c Values supplied by ERA from a sample collected on 30 November 2009

The toxicity of permeate was assessed using five local freshwater organisms – a unicellular alga (*Chlorella* sp), macrophyte (duckweed; *Lemna aequinoctialis*), cnidarian (*Hydra viridissima*), crustacean (*Moinodaphnia macleayi*) and a fish species (northern trout gudgeon; *Mogurnda mogurnda*). The test organisms were exposed to concentrations of 6.25%, 12.5%, 25%, 50% and 100% permeate, as well as a Magela Creek water control. The permeate used for testing was diluted with fresh Magela Creek water.

Significant effects of permeate were observed for all species, at concentrations above 12.5% permeate, with the responses ranging from growth stimulation to moderate toxicity (Figure 3.13, Table 3.5).

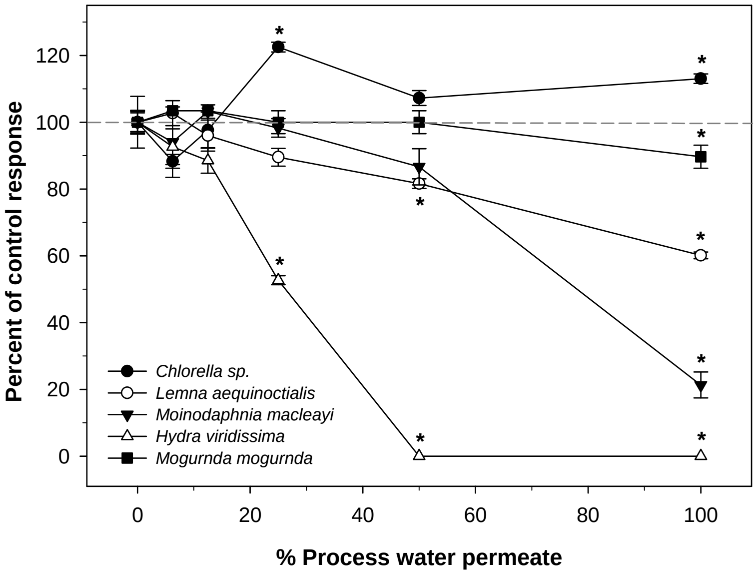

Figure 3.13 Responses of five tropical freshwater species to treated process water from Ranger Uranium Mine, expressed as percentages of the control response (see Table 3.5 for control response data). Data points represent the mean ± standard error of three replicates (10 replicates for *Moinodaphnia macleayi*). Asterisks denote treatments that are significantly different ($P \leq 0.05$) from the control response.

TABLE 3.5 TOXICITY ESTIMATES FOR TREATED PROCESS WATER FROM RANGER URANIUM MINE

Species	Control response (mean ± standard error)	Toxicity (% process water permeate)	
		IC10[a] (95% CL)[b]	IC50[a] (95% CL)
Chlorella sp	Doublings per day = 1.6±0.04	NC[c]	NC
Lemna aequinoctialis	Growth rate = 0.43±0.01	22 (0–45)	NC
Moinodaphnia macleayi	Offspring per adult = 35.2±2.7	43 (5–54)	78 (69–83)
Hydra viridissima	Growth rate = 0.31±0.01	10 (0–18)	26 (23–29)
Mogurnda mogurnda	Percent survival = 97±3	67[d] (0–100)	NC

a IC10 and IC50: concentrations that result in a 10% and 50% inhibition of response compared to the control (ie unexposed) response, respectively. Estimates were derived using linear interpolation (ToxCalc V5.0.23).

b 95% CL: 95% confidence limits

c NC: Not able to be calculated since there was insufficient response across the dilution gradient

d Value represents an LC05 (ie concentration resulting in 5% mortality of larval *M. mogurnda*; derived using non-linear interpolation; ToxCalc V5.0.23). A lower effect level than 10% was selected given the test is an acute test.

Chlorella sp growth rate was significantly enhanced at permeate concentrations of 25% (22% enhancement compared with the control) and 100% (13% enhancement). Exposure of *L. aequinoctialis*, *M. macleayi*, *H. viridissima* and *M. mogurnda* to 100% permeate resulted in significant reductions in responses of 40%, 80%, 100% and 10%, respectively. *Hydra viridissima* exhibited the strongest response of all the species, with a full response at 50% permeate and 47% reduction in growth rate at 25%. Based on the extent of response (negative or positive) at 100% permeate, the order of sensitivity of the species (from highest to lowest) was: *H. viridissima* > *M. macleayi* > *L. aequinoctialis* > *Chlorella* sp ≈ *M. mogurnda*.

The process water treatment process is clearly effective at removing the majority of contaminants and hence reducing or eliminating toxicity, compared with the composition of the untreated process water.

The effects of the reverse osmosis permeate, including the stimulatory response by *Chlorella* sp, are hypothesised to be primarily due to residual ammonia (present largely as ammonium ion). Alternatively, or in addition, the adverse responses of some of the species could be due to the very low concentrations of nutrients (other than N) or essential trace elements in permeate preventing normal growth, development and/or survival. This was previously shown to be the case for treated pond water permeate from Ranger (see 2007–08 ***eriss*** Research Summary). Additional work is being undertaken to confirm if the effects of permeate are largely caused by the residual concentration of ammonium ion. This will involve the selective removal of ammonia (as ammonium) from the permeate followed by toxicity testing of the residual solution.

3.5 Influence of dissolved organic carbon on the toxicity of aluminium to tropical freshwater biota

This work is part of a PhD project studying the influence of dissolved organic carbon (DOC) on metal toxicity to freshwater organisms. The first part of the project assessed the effects of DOC on uranium toxicity and the results were presented in the 2008–09 Annual Report.

Aluminium (Al) is a metal of general ecotoxicological concern for the mining industry. Inputs of Al to surface waters can occur through acidic seepage or discharge of acidic mine waters from legacy, closed and operating mine sites. Examples of such sites in the Northern Territory include the legacy Rum Jungle and Rockhole Creek uranium mines and metal mine sites throughout the Pine Creek Geosyncline metal province. The outcomes of the assessment done by the Supervising Scientist Division for the Rockhole Mine Creek site located in the Alligator Rivers region were documented in the 2008–09 Annual Report.

The classic acid drainage conditions exhibited at these sites provide an environment in which the bioavailability and toxicity of Al to biota are potentially much increased. In the case of fish , Al binds to the gills where it leads to respiratory dysfunction. Al has also been found to bioaccumulate in filter feeding invertebrates, in particular those feeding on benthic detritus. There are few toxicity data for Al in freshwater, particularly at acidic pH. The only water quality guideline available for Al in freshwater at low pH is a *low reliability* trigger value of 0.8 µg/L Al.[5] This guideline also does not incorporate the influence of DOC, which can form strong complexes with Al and potentially influence its bioavailability and toxicity.

The objective of this study was to quantify the influence of DOC on the toxicity of Al to three tropical freshwater species at low pH (5.0) and alkalinity (2–14 mg/L as $CaCO_3$). The selected tropical species, green hydra (*Hydra viridissima*), green alga (*Chlorella* sp), and the cladoceran (*Moinodaphnia macleayi*) were chosen to cover a range of trophic levels.

The influence of DOC was assessed using two sources of DOC: (i) the international standard Suwannee River fulvic acid (SRFA) and (ii) a local DOC present in water sourced from Sandy Billabong located adjacent to Magela Creek upstream of Ranger mine in Kakadu National Park. Four concentrations – 1, 2, 5 and 10 mg/L – of SRFA and local DOC were used and test species were exposed to up to 5 mg/L total Al. For the SRFA, toxicity testing was conducted using diluted (25% dilution with Milli-Q water) Magela Creek water (DMCW), containing a natural DOC concentration of <1 mg/L, as the test medium. DMCW, rather than synthetic Magela Creek water (SMCW), was used as the diluent because its low concentrations of background DOC (~1 mg/L) and alkalinity were required to provide buffering capacity to maintain the low test pH of 5.0 (SMCW, which lacks DOC, was not able to hold pH at a pH lower than pH 6).

For the local DOC, Sandy Billabong water (SBW), naturally containing 10 mg/L DOC, was diluted to the required DOC concentrations (1, 2, 5, 10 mg/L) using SMCW containing a similar inorganic composition to SBW but lacking in DOC. For the *Chlorella* test, nitrate and phosphate were added as nutrients (3.28 mg/L nitrogen and 0.046 mg/L phosphorus).

Test systems were static, with 24 h renewal of test solutions for *H. viridissima* only (there was no renewal for the *Chlorella* sp or *M. macleayi* tests). Test temperatures were maintained at 27 ± 1ºC for *M. macleayi* and *H. viridissima* and 28 ± 1ºC for *Chlorella* sp. For each species, four tests were conducted for SRFA and three tests for the SBW DOC, in order to fully characterise the concentration-response relationships.

Test durations and endpoints were as follows: *H. viridissima* – 96-h population growth rate; *Chlorella* sp – 72-h growth rate; *M. macleayi* – 24-h neonate survival. For all tests, general water parameters (pH, DO and EC) were monitored daily. At the beginning of each test, water samples were taken for analyses of DOC, alkalinity, hardness and a standard suite of metals and major ions. For each species, response data from the tests were pooled, and concentration-response relationships were determined using non-linear regression analyses.

Concentration-response relationships and associated linear regressions of toxicity (expressed as IC_{50} – the concentration that results in a 50% inhibition of the test response relative to the control response) against fulvic acid concentration are shown in Figures 3.14 and 3.15, respectively, while the toxicity summary data are shown in Table 3.6. Al toxicity was reduced in the presence of both DOC sources. For *H. viridissima*, SRFA was ~5 times more effective (based on the increased slope of the IC_{50} versus DOC plot) at reducing Al toxicity than the local SBW DOC. For *Chlorella* sp, SRFA was only ~2 times more effective at reducing Al toxicity than the local DOC. For *M. macleayi*, Al toxicity was reduced by a similar factor in the presence of both DOC sources.

5 ANZECC/ARMCANZ 2000. Australian and New Zealand guidelines for fresh and marine water quality. National Water Quality Management Strategy Paper No 4. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

TABLE 3.6 EFFECT OF TWO DIFFERENT FORMS OF DISSOLVED ORGANIC CARBON (DOC), (I) SUWANNEE RIVER FULVIC ACID STANDARD I, AND, (II) DOC IN SANDY BILLABONG WATER, ON THE TOXICITY OF ALUMINIUM TO THREE LOCAL FRESHWATER SPECIES

Species	DOC[a] (mg/L)	IC_{50}[b] (95%CL)[c]		Extent of amelioration of Al toxicity (µg Al mg/L DOC^{-1})[d]	
		DMCW+SRFA[e]	SBW diluted with SMCW[f]	DMCW +SRFA	SBW
Hydra viridissima	1	35 (29–39)	49 (NC–149)	21	4.0
(green hydra)	2	59 (40–71)	61 (48–72)		
	5	119 (91–138)	69 (54–81)		
	10	226 (204–242)	87 (65–101)		
Chlorella sp	1	275 (189–384)	437 (315–679)	225	115
(unicellular alga)	2	805 (560–1032)	801 (560–1134)		
	5	1427 (1242–1582)	1251 (870–1724)		
	10	2260 (1830–2867)	1635 (1410–1895)		
Moinodaphnia macleayi	1	164 (123–206)	950 (939–983)	147	141
(cladoceran)[g]	2	691 (610–767)	905 (608–1293)		
	5	1162 (972–1390)	1214 (868–1510)		
	10	1584 (1277–1930)	2113 (2083–2140)		

a DOC: dissolved organic carbon, b IC_{50}: the concentration that results in a 50% inhibition of the test response relative to the control response; c 95% confidence limits; d extent of amelioration is the slope of the regression between IC50 and the concentration of DOC (Figure 3.15). e SRFA made up in dilute Magela Creek water (25%) ; f SBW diluted with SMCWr; g For *M.macleayi*, toxicity estimates relate to concentrations that affect percentage survival (as a % of control survival), compared to sub-lethal endpoints, such as growth and reproduction, for the other species.

Physicochemical variables were input into the WHAM (Windermere Humic Aqueous Model) chemical speciation computer model to estimate the effect of DOC on Al speciation, which was related back to Al toxicity. For both DOC sources, the decrease in Al toxicity with increasing DOC can be attributed to a reduction in the free (Al^{3+}) and monomeric hydroxy ($Al(OH)_2^{+}$) ion concentrations (the two most toxic species), due to Al being bound by DOC. These results and those of additional speciation modelling used to investigate finer aspects of the observed responses to Al, will be presented in more detail in subsequent publications.

Based on the responses of the three test species to Al in the presence of 1 mg/L DOC (IC_{50}s ranging from 50–950 µg/L Al), it appears that the current *low reliability* trigger value of

0.8 µg/L Al, which does not account for the influence of DOC, is likely to be overly protective for natural waters containing this level, or greater, of DOC.

Extending the number of species tested to 5 or 6 would enable a high reliability trigger value to be derived for Magela Creek (and similar composition) waters. However, to do this would be technically very challenging. For a species to be suitable for this testing it would need to be able to tolerate water at pH 5 and exhibit effects within the solubility limits of Al (which for water at pH 5 is around 400–500 µg/L).

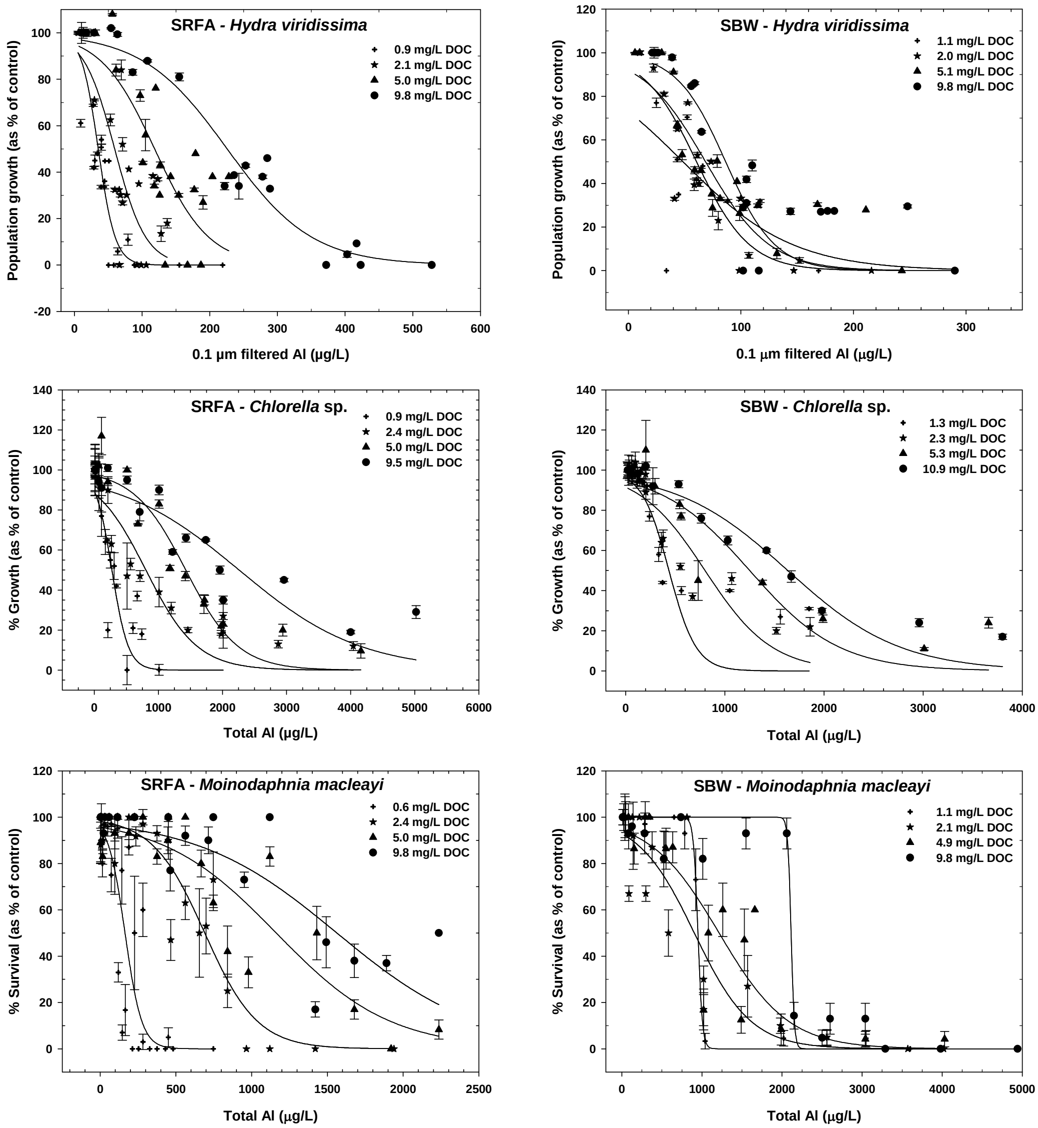

Figure 3.14 Concentration-response plots for Al exposures. Left: using Suwannee River fulvic acid (SRFA) in dilute Magela Creek water, 4 pooled tests for each species. Right: Sandy Billabong Water (SBW) diluted in synthetic Magela Creek water, 3 pooled tests for each species. Data points represent the mean of 3 replicates ± SE for *Chlorella* sp and *M. macleayi*, and 2 replicates ± SE for *H. viridissima*.

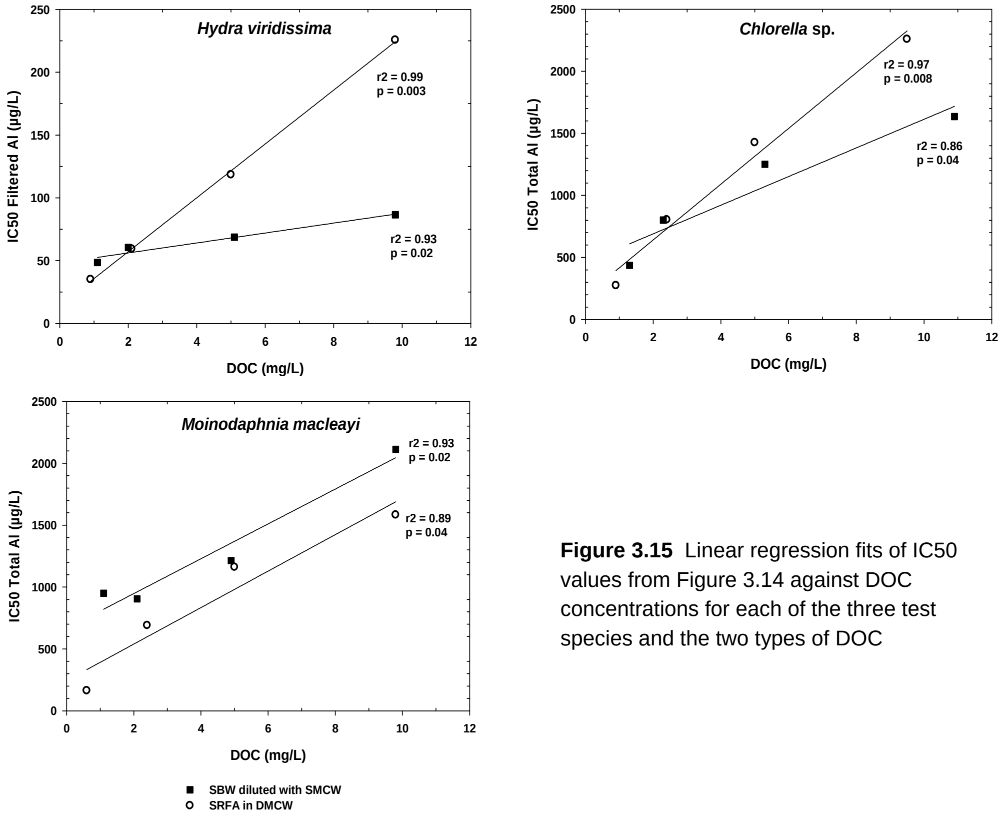

Figure 3.15 Linear regression fits of IC50 values from Figure 3.14 against DOC concentrations for each of the three test species and the two types of DOC

3.6 Characterisation of the pre-mining radiological footprint at Ranger

The ICRP recommends that the total annual effective radiation dose to a member of the public from practices such as uranium mining should not exceed 1 millisievert (mSv). This dose is on top of the natural pre-mining background dose and includes the external gamma, inhalation and ingestion pathways. In a high natural background area such as the area around Ranger mine, determining an additional dose due to mining activities presents a challenge, especially when pre-mining data are scarce and focus on delineating the extent and location of an orebody, rather than determining area wide radiological conditions.

Pre-mining radiological conditions need to be quantified so that post-mining changes can be assessed in the context of the success of rehabilitation from a radiological perspective. Historical airborne gamma surveys (AGS), coupled with ground truthing surveys, have the potential to provide a powerful tool for an area wide assessment of pre-mining terrestrial gamma dose rates. AGS and ground truthing surveys have been commissioned and used for regional assessments of radiological conditions at rehabilitated and historic mine sites elsewhere in the Alligator Rivers Region. Whilst a pre-mining AGS was flown over the Alligator Rivers Region including the Ranger site in 1976, no ground radiological data of the resolution and spatial coverage needed to calibrate the AGS data are available from that time. The novelty of this project is to use recently measured high resolution ground data from an appropriate undisturbed radiologically anomalous area to calibrate the AGS survey data for this anomaly, and then to use the calibrated 1976 AGS to infer pre-mining radiological conditions over the whole Ranger lease.

3.6.1 Methods

1976 AGS data were acquired from Rio Tinto by the NT Government and are available in the public domain (the *Alligator River Geophysical Survey*). Data were re-processed in 2000 by the Northern Territory Geological Survey (NTGS) and then resampled by NTGS at a pixel size of 70 m in 2003. The line spacing of the survey was 300 m, however, the flying height is unknown. The 1976 AGS has been used to identify undeveloped radiological analogues in the vicinity of the Ranger lease as potential candidates for ground truthing. A comparison of signal intensity with known uranium occurrences in the MODAT database suggested that Anomaly 2 to the south of the Ranger lease may be a suitable analogue site for Ranger pre-mining radiological conditions as it exhibits a strong airborne gamma signal in the data, has not been mined, nor is it influenced by operations associated with the Ranger mineral lease.

In addition, Energy Resources of Australia (ERA) has made data available to SSD from an AGS that was flown in 1997 at a low flying height (50 m) and a higher spatial resolution (200 m line spacing) than the 1976 survey. This dataset was used to further refine extensive groundtruthing fieldwork conducted in the dry seasons 2007 to 2009 to establish the exact location and intensities of the Anomalies immediately south of the Ranger lease. To date approximately 2000 external gamma dose rate measurements have been conducted using environmental dose rate meters, in addition to the determination of soil uranium, thorium and potassium activity concentrations via gamma spectrometry at selected sites.

Dry season radon exhalation was measured using conventional charcoal cups, with 3 charcoal cups deployed at each of the 25 sites for a period of three days. The charcoal cups were then analysed using the SSD NaI gamma detector. In addition, external gamma dose rates were measured and soil scrape samples were taken at the 25 sites for high resolution gamma spectrometry analyses. Track etch detectors were also deployed for three months at these sites to measure dry season airborne radon concentration and to establish whether there is a correlation between airborne radon concentration and radon exhalation flux or soil ^{226}Ra activity concentrations. At some of the sites, track etch detectors were deployed at various heights to represent the breathing zones of a person lying down with the head slightly raised, sitting and standing, to investigate changes in radon concentration with distance from the ground.

3.6.2 Results

Groundtruthing of the airborne gamma survey

Figure 3.16 shows the results from the 1997 ERA AGS data (total counts) compared with external gamma dose rate measurements ($\mu Gy{\cdot}hr^{-1}$) from SSD's groundtruthing. It is apparent that the groundtruthing survey has clearly distinguished Anomalies 2A (in the middle) and 2B (to the northeast), and a third Anomaly further to the southwest. Maximum uranium concentrations at the surface of Anomaly 2A are greater than 6000 mg/kg and maximum gamma dose rates measured at 1m height exceed 20 $\mu Gy \cdot hr^{-1}$. Typical environmental background uranium concentrations in the vicinity are 4–6.5 mg/kg and background gamma dose rates are approximately 0.16 $\mu Gy \cdot hr^{-1}$.

To groundtruth an AGS, the data acquired in the field (gamma dose rates, uranium, thorium, and/or potassium concentrations) are plotted against the count rates from the respective channels in the AGS. As the groundtruthed data at Anomaly 2 have been acquired at a much higher resolution than both the 1997 and 1976 AGS data, the image is much 'sharper', and it is thus essential to determine appropriate 2-dimensional smoothing algorithms which allow a comparison to be made between the groundtruthed and the AGS data. Ground-based data are typically smoothed by averaging such that the resolution is similar to that of the AGS.

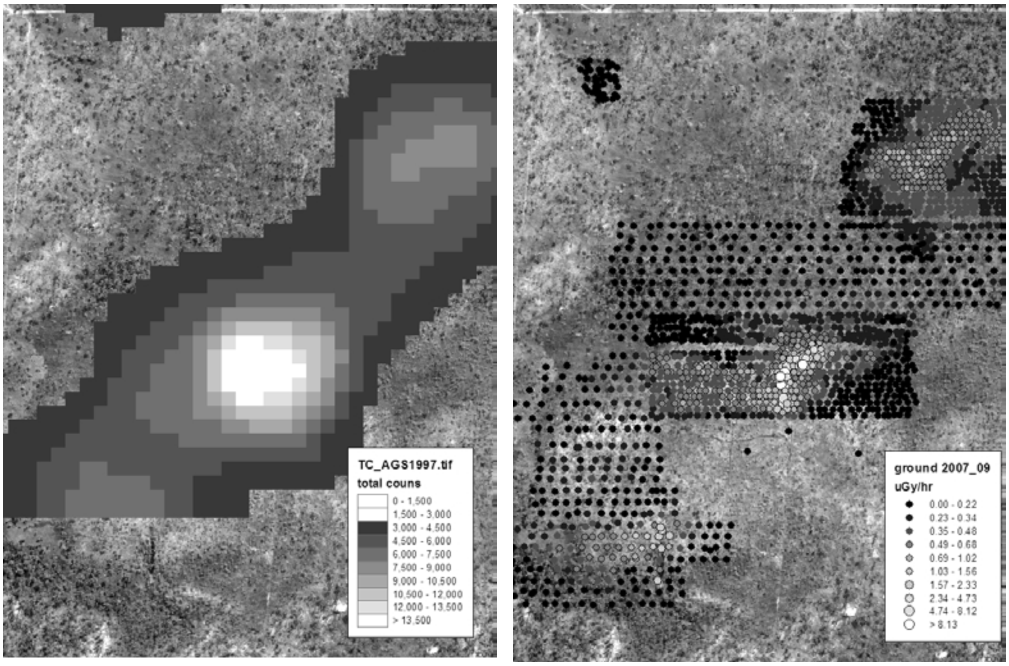

Figure 3.16 1997 AGS data (courtesy of ERA, left) and the results of the on ground gamma dose rate measurements (right) performed from 2007 to 2009, overlaid on a 2006 Quickbird image of the area immediately south of the Ranger lease

The best correlation between the 1997 AGS and the ground based dataset using a circular footprint is achieved after applying a small spatial shift and using a smoothing radius of ~80 m for the ground data. To take into account the fact that the plane is in motion as data is being acquired, more work is currently underway to investigate the effects of using an ellipsoidal footprint to smooth the ground data.

Radon

Radon (^{222}Rn) is a radioactive noble gas and part of the ^{238}U decay series. It is exhaled from soils and rocks, and exhalation is generally higher for fine grained soils rich in its parent, ^{226}Ra. Once airborne, the shortlived radon decay products (^{218}Po, ^{214}Pb, ^{214}Bi) are produced by the decay of radon and it is these decay products that deliver a radiation dose following inhalation, rather than the radon gas.

To determine the source strength, or radon flux, expected for an undisturbed uranium anomaly, radon flux densities have been measured across the Anomaly 2 area. In addition, gamma dose rates and soil ^{226}Ra activity concentrations were measured at these sites to investigate whether they can be used as a proxy to predict radon flux from the area. Figure 3.17 shows the geometric means of the radon flux densities versus the soil ^{226}Ra activity concentrations measured at the sampling sites, both plotted on a logarithmic scale.

In Figure 3.17 the sampling sites have been divided according to soil type (identified by visual inspection in the field) and sampling location, and results are plotted for fine gravel, loamy sand and coarse gravel/rocks on top of the anomalies. It appears that radon exhalation does not change significantly with increasing ^{226}Ra activity concentration of the soil directly above the outcropping anomaly, where typical radon flux densities (geometric mean) are 5.6 ± 2.4 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$, similar to values measured above the Ranger #1 and #3 orebodies before mining started (2.5–5.5 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$). For soil ^{226}Ra activity concentrations in the range of 10–2500 $Bq{\cdot}kg^{-1}$, radon flux densities can be predicted by multiplying the measured soil ^{226}Ra activity concentrations by 2.2 $g{\cdot}m^{-2}{\cdot}s^{-1}$. This value is similar to those reported earlier for non-compacted fine grains in the region (2.7 ± 0.4 $g{\cdot}m^{-2}{\cdot}s^{-1}$).

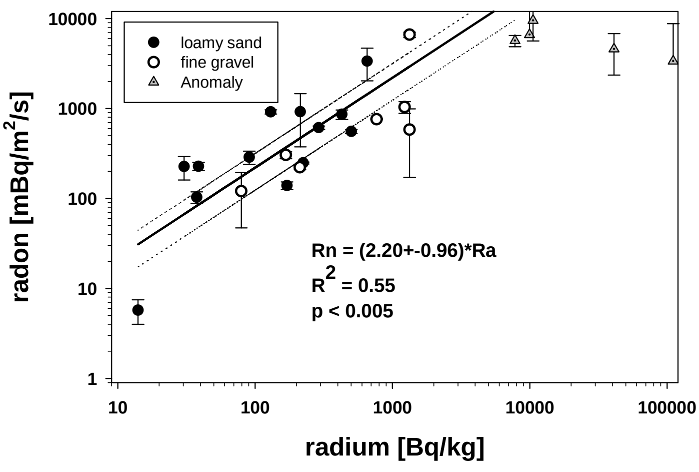

Figure 3.17 Radon flux densities plotted versus soil radium activity concentrations measured at Anomaly 2. The solid line is a linear fit to the data, the dotted line represents the 95% confidence interval.

Whereas radon flux density from the soil into air varies by three orders of magnitude, the radon activities measured in air ($Bq{\cdot}m^{-3}$) at 1.5 m height vary much less, indicating good lateral mixing. However, there is still a positive correlation ($p < 0.005$; $R^2 = 0.4$) with radon exhalation flux densities from the soil underneath. The typical dry season radon concentration (geometric mean) 1.5 m above Anomaly 2 is ~150 Bq·m^3, which is about 5 times higher than typical dry season radon concentration measured at Jabiru, but lower than the Australian indoor reference level for existing and new dwellings of 200 Bq·m^3. The radon concentration increases by ~1 Bq·m^{-3} for every 370 Bq·kg^{-1} increase in soil ^{226}Ra activity concentration. Wet season radon concentrations in air are generally lower than the values given above as previously determined at other areas in the Alligator Rivers Region.

Figure 3.18 shows the radon concentration measured at three different heights at various sites across the area surveyed, and the corresponding soil ^{226}Ra activity concentrations. The figure illustrates that at areas away from 'hot spots' radon concentration is relatively uniform vertically, but concentrations, and thus inhalation doses, are significantly higher when sitting or lying in close vicinity to the outcropping uranium anomalies with high ^{226}Ra activity concentrations. This potential exposure route and its dependence on height needs to be taken into consideration, in addition to land use, including diets, of indigenous people in the area, when assessing potential doses to humans in the region before mining started.

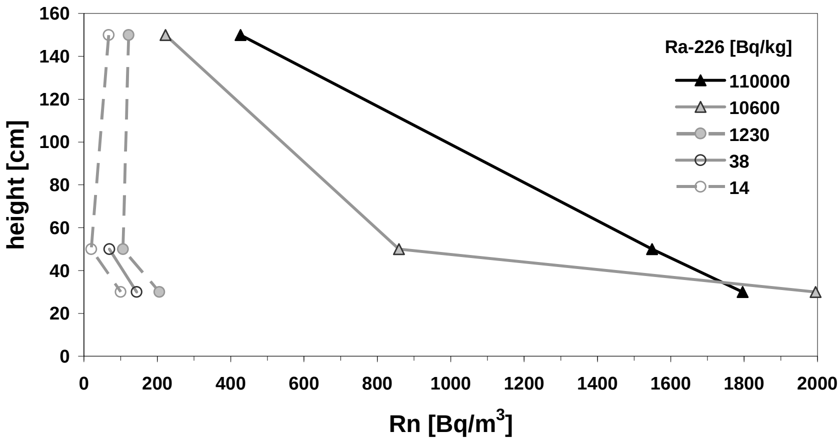

Figure 3.18 Radon concentration in air for various heights (30 cm, 50 cm, 150 cm) above the ground

3.6.3 Conclusion and future work

The correlation of historical AGS data from Anomaly 2 with recent ground truthed data from the area will allow determination of average external gamma dose rates across the Ranger lease area before mining started. The spatial resolution of the extrapolated dose rates is limited by the resolution of the AGS, which is at least 1ha for the 1976 AGS, but this resolution will suffice to determine pre-mining averages across orebodies #1 and #3 and other areas on site. The behaviour of radon in the vicinity of Anomaly 2 has also been studied, and the results will allow the determination of doses from the inhalation of radon progeny above the pre-mining footprint at Ranger, using appropriate equilibrium and dose conversion factors, respectively.

The potential contribution from the dust inhalation pathway still needs to be established and a separate study is currently underway to quantify the resuspension of dust in the area. Published resuspension factors for the region are comparatively high and need to be verified before radionuclide activity volume concentrations in air ($Bq \cdot m^{-3}$) are inferred from soil radionuclide activity concentrations extrapolated from the AGS survey data.

Further work is required on algorithms to upscale the results from the groundtruthing, in particular taking into account that the aircraft is in motion as data are being acquired, so that a similar comparison can be made with the 1976 AGS. Once data analysis is complete, the radiological conditions on ground around Anomalies 2A and 2B will be correlated to the pre-mining 1976 airborne signal to extrapolate to the area wide radiological conditions at the Ranger lease area before mining commenced. The results will be reported in a subsequent Annual Report.

3.7 Radiological characterisation of Ranger mine land application areas

3.7.1 Background

Water management is a major issue at Ranger uranium mine, given its location in the wet-dry tropics where up to 2 m of rainfall can occur within a single wet season. Release of water from the site into the downstream environment is minimised by the use of a series of retention ponds (RP1–RP3) (see Map 2). RP1 water is of relatively good quality, and free release into Magela Creek occurs routinely during most wet seasons. Since 1985 water stored in RP2 during the wet season has been disposed of on site using land application methods. RP2 receives runoff and seepage from the low grade ore and waste stockpiles and other areas on the minesite.

The history of development of the land application areas (LAAs) on the Ranger site is summarised in Table 3.7. The Magela Land Application Area (MLAA) was the first to be established using the spray irrigation method. Additional LAAs were developed as the amount of water to be disposed of rose through time as a result of the increasing area occupied by waste and low grade ore stockpiles. Starting in 1995, the RP1 and Djalkmara wetland filters were used to polish RP2 water before it was applied to the RP1 and Djalkmara East and West LAAs. In this context, and in contrast to the other LAAs, it should be noted that the MLAA has received untreated RP2 water throughout its entire operational life. It is therefore likely to contain the highest concentrations of metals and radionuclides.

From 2006 onwards increasing volumes of pond water have been treated by microfiltration/reverse osmosis (MF/RO) water treatment during the wet season, with the RO permeate being discharged along the Corridor Creek catchment line. The introduction of active pond water treatment during the wet season has progressively reduced the volume needed to be disposed of by land application during the dry season.

TABLE 3.7 SOURCES OF WATER FOR LAND APPLICATION AREAS AT RANGER URANIUM MINE

Land Application Area	Source of applied water	Total area (ha)	Year commissioned
Magela –Area A (MALAA)	RP2 water	33	1985
Magela –Area B (MBLAA)	RP2 water	20	1994
RP1	polished RP2 water	46	1995
Djalkmara East (E. Dj)	polished RP2 water	18	1997
Djalkmara West (W. Dj)	polished RP2 water	20	1999
Jabiru East (JELAA)	RP2 water	52	2006
RP1 Extension (RP1 ext)	RP2 water	8	2006
Corridor Creek (CCLAA)	RP2 water	141	2007

The use of land application as a water treatment method relies on the fact that radionuclides and most heavy metals have a tendency to bind to the organic rich surface horizons of soil profiles. These bound metals and radionuclides have a low leachability and will therefore be unlikely to impact the aquatic environment downstream of Ranger. However, there has been ongoing stakeholder concern about the radiological status of the Ranger LAAs, in particular with regards to the Magela LAAs and their capacity to continue to adsorb radionuclides at the current rate of application. The concentration of radionuclides adsorbed in the soil could potentially require the area to be rehabilitated at closure, based on current ICRP recommendations.

The Environmental Strategy Department within ERA, in collaboration with *SafeRadiation*, Brisbane and the Environmental Research Institute of the Supervising Scientist (***eriss***), has initiated a project to identify and quantify current radiological issues associated with the LAAs. The aims of this project are to characterise the magnitude and extent of radiological contamination at each of the Ranger LAAs and to suggest options for their rehabilitation. The nature of these options will strongly depend on the estimated post rehabilitation radiation doses to people from data produced by this current project.

3.7.2 Methods

Soil samples were collected at various distances (0–15 m) from the sprinkler heads at all LAAs and also included samples not influenced by irrigation. Soil samples were taken to a depth of 10 cm. In addition, ten soil cores were collected and sampled at a resolution of 5 cm down to 20 cm depth. Whole soil samples were dried and crushed, and prepared for radionuclide analysis via gamma spectrometry at ***eriss***. Leaf litter samples were also taken at various distances from the sprinklers. This material was ashed and homogenised and analysed by gamma spectrometry. The radionuclide activity concentration results were used to determine vertical and horizontal depositional patterns and to calculate the total load of radionuclides retained in LAA soils. These loads (in $kBq \cdot m^{-2}$) were then compared with loads calculated from the known volumes and water quality data provided by ERA for the water applied at the various LAAs over the years.

Radon (^{222}Rn) exhalation flux density was also determined at various distances from the sprinkler heads using conventional charcoal cups. There was no irrigation of mine waters during and immediately prior to charcoal cup exposure. The charcoal cups were analysed using the ***eriss*** NaI gamma detector. Measurements were made in the dry season 2008 and in March 2009 (wet season) to quantify the effect of season.

Passive dust collection stations were established along transects that intersect the boundaries of the Magela A and Magela B land application areas (see Figure 3.19 for transect locations). The stations are triangular in shape and approximately 2 m high. Each face of the stations has four collector panels made of sticky vinyl, centred at 0.3, 0.7, 1.2 and 1.5 m above ground, representing the breathing zones of a person lying down, sitting, a juvenile standing and an adult standing, respectively. The stations were deployed in the dry season of 2008 and remained in place until the end of the dry season 2009. The sticky vinyl panels were changed every three months so that the deposition rates were measured quarterly over a seasonal cycle.

3.7.3 Results

Soil and leaf litter radionuclide activity concentration

The maximum ^{238}U soil activity concentration measured was 28 000 $Bq \cdot kg^{-1}$ (2270 $mg \cdot kg^{-1}$ uranium) and the average was ~1700 $Bq \cdot kg^{-1}$ (137 $mg \cdot kg^{-1}$). In contrast, the maximum measured ^{226}Ra soil activity concentration was only a little above 1000 $Bq \cdot kg^{-1}$, with an average of ~190 $Bq \cdot kg^{-1}$. A large number of ^{226}Ra activity concentration values are in the range 100–500 $Bq \cdot kg^{-1}$. Most samples exhibit an activity concentration trend of $^{238}U >> {}^{226}Ra > {}^{210}Pb$, which reflects the signature of RP2 water applied to the soils. This is important for the external gamma pathway, as uranium is only a weak gamma emitter. The majority of the terrestrial gamma dose rate measured in air originates from ^{226}Ra decay products (^{214}Bi and ^{214}Pb) rather than uranium.

Although the activity concentration in surface leaf litter ($Bq \cdot kg^{-1}$ dry weight) is ~10 times higher than that measured in the underlying soil, only a small fraction of the total load of applied radionuclides appears in the leaf litter. It was found that approximately 90% of the applied radionuclides have been retained in the top 10 cm of the soils, in agreement with earlier studies conducted in the MLAA.

To put the radiation source term of the MLAA into context it should be noted that the concentration of uranium in waste rock can be up to 200 $mg \cdot kg^{-1}$, which translates to ~2100 $Bq \cdot kg^{-1}$ of ^{226}Ra in radioactive equilibrium with ^{238}U. The combined exposure to the external gamma radiation and radon progeny inhalation pathways is a function of both the magnitude of ^{226}Ra activity concentration in the soil and its depth of occurrence. The typical diffusion path length for radon in soil is 1–2 m. Thus the 10 cm effective depth of elevated ^{226}Ra (maximum value of 1000 $Bq \cdot kg^{-1}$) in the soil of the LAA needs to be compared with the potentially many metres of depth of waste rock containing up to 2100 Bq·kg^{-1}. Consequently annual doses via those two pathways will be less significant over the footprints of the LAAs compared to areas that will contain substantial depths of waste rock after remediation of the site.

The ^{238}U and ^{226}Ra soil activity concentrations in the top 10 cm decrease with distance from the sprinkler heads. This decrease can be approximated mathematically using an exponential equation, which has been used to estimate radionuclide activity loads deposited within the sprinkler wetting zone. The results derived from the direct measurement of soil activities, and subsequent integration over the LAA areas, compare well with the applied loads calculated from historical radionuclide inventories in RP2 water and irrigation rates provided by ERA.

The activity ratio of ^{226}Ra/^{210}Pb has been used to distinguish areas affected by application of mine waters from areas that may have naturally higher soil radionuclide activity concentrations. For environmental background soils, ^{226}Ra and ^{210}Pb are in radioactive equilibrium within the soil grains but deposition of ^{210}Pb from the atmosphere (which is produced by ^{222}Rn decay in air) shifts the ^{226}Ra/^{210}Pb activity ratio to values less than one. For natural uranium mineralised areas, the ^{226}Ra/^{210}Pb activity ratio is close to one and the effect of ^{210}Pb deposited from the atmosphere on the ^{226}Ra/^{210}Pb activity ratio is negligible due to the much higher concentrations of ^{226}Ra and ^{210}Pb arising from the uranium mineralisation. For soils subject to land application of pond water, the ratio should be greater than one as RP2 water contains significantly higher amounts of ^{226}Ra compared with ^{210}Pb.

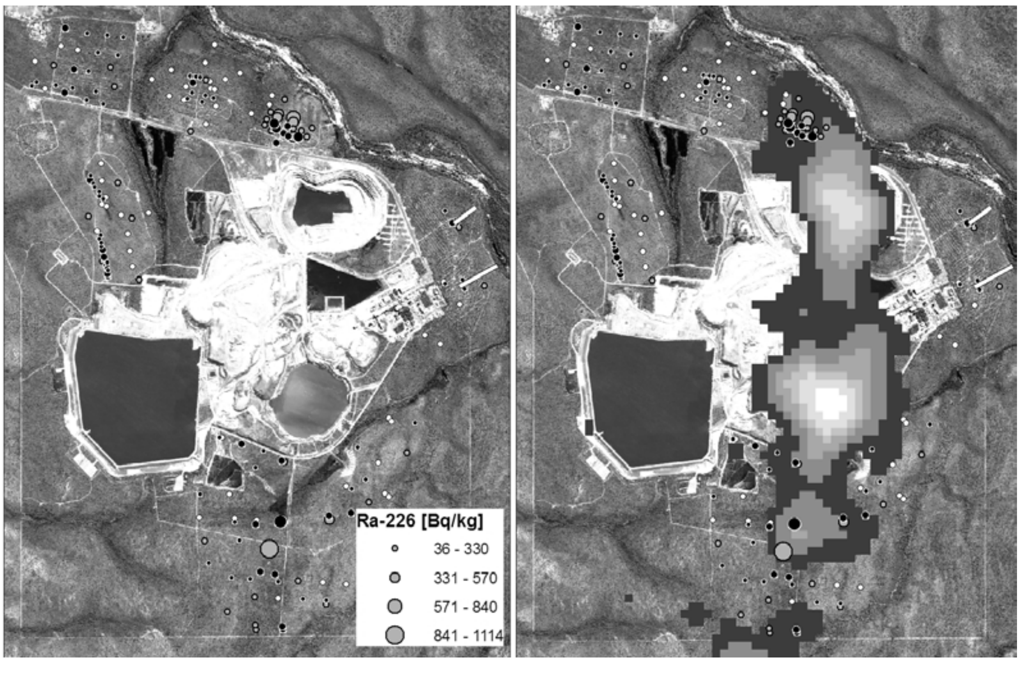

Figure 3.19 (a) ^{226}Ra/^{210}Pb activity ratios (white: ^{226}Ra/^{210}Pb < 0.9; grey: 0.9 < ^{226}Ra/^{210}Pb < 1.1; black: ^{226}Ra/^{210}Pb > 1.1) and ^{226}Ra activity concentrations of the soils collected. (b) Data overlaid on results from a 1976 airborne gamma survey (courtesy of the Northern Territory Geological Survey). Indicated are areas exhibiting counts per seconds in the airborne gamma survey significantly above background, black is lowest white is highest. White lines to the east show the locations of the dust transects at the MLAA.

Most samples measured exhibit a $^{226}Ra/^{210}Pb$ activity ratio of ≥ 1, whereas most of the lower activity soils have a $^{226}Ra/^{210}Pb$ activity ratio < 1. However, there are some areas of relatively high ^{226}Ra and ^{210}Pb activity concentrations with $^{226}Ra/^{210}Pb$ activity ratios close to radioactive equilibrium, indicating that in some areas within the LAAs naturally elevated ^{226}Ra activity concentration exists that is not attributed to irrigation.

Figure 3.19a (left) shows the location of the soil samples collected and a classification with regards to their ^{226}Ra activity concentration (indicated by the size of the circles) and their $^{226}Ra/^{210}Pb$ activity ratio (indicated by their colour). The white circles ($^{226}Ra/^{210}Pb < 0.9$) are generally small in size and some of the samples are outside the zone of influence from the sprinklers. Figure 3.19b (right) shows the same data overlaid on results from an airborne gamma survey conducted in 1976. Soils with high ^{226}Ra activity concentration that exhibit a $^{226}Ra/^{210}Pb$ activity ratio of approximately 1 (big grey circles) are located within areas that exhibited higher natural backgrounds before mining started. This is particularly obvious in samples from the Djalkmara East LAA, to the northwest of Pit 3. This finding is important in the context of post irrigation dose assessment, as a proportion of the determined radiation doses will be due to existing natural radiation anomalies at these areas. Estimation of pre-mining doses at Ranger are subject of a separate project being conducted by ***eriss*** (see Section 3.6 – Characterisation of the pre-mining radiological footprint at Ranger in this report).

Radon exhalation

Dry and wet season measurements of radon flux densities were conducted in 2008–09 and a summary of the results is shown in Figure 3.20. In this figure average radon flux densities measured in the dry (August 08) and wet season (March 09) are plotted versus distance from the sprinklers at the various LAAs.

The decrease of radon flux densities with increasing distance from the sprinklers is more pronounced during the wet season compared with the dry. The Jabiru East, Corridor Creek and Djalkmara LAAs show on average higher radon flux densities during the dry season as compared with the wet, most likely due to lower soil moisture during the dry season. However, the trend appears to be opposite in the Magela and RP1 LAAs. This could potentially be an effect of the higher radium loads in the top few centimetres of the soils in these areas, in particular at the MLAA, that dry out more quickly compared with the deeper sections of the soil profile, and thus contribute relatively more to the radon flux than at the other areas.

Typical environmental (ie background) radon flux densities measured in the region are approximately 40–70 $mBq \cdot m^{-2} \cdot s^{-1}$. The Magela, Corridor Creek and East Djalkmara LAAs exhibit geometric means that are higher, whereas the remaining LAAs exhibit no noticeable increase above background. It is likely that the higher average radon flux densities at the Corridor Creek and East Djalkmara LAAs are caused by the presence of natural radiogenic anomalies (see Figure 1), whereas the increase at the MLAA (Area A) is largely due to the application of mine waters. However, it is known that natural anomalies are also present underneath and in the vicinity of the MLAA (Area A).

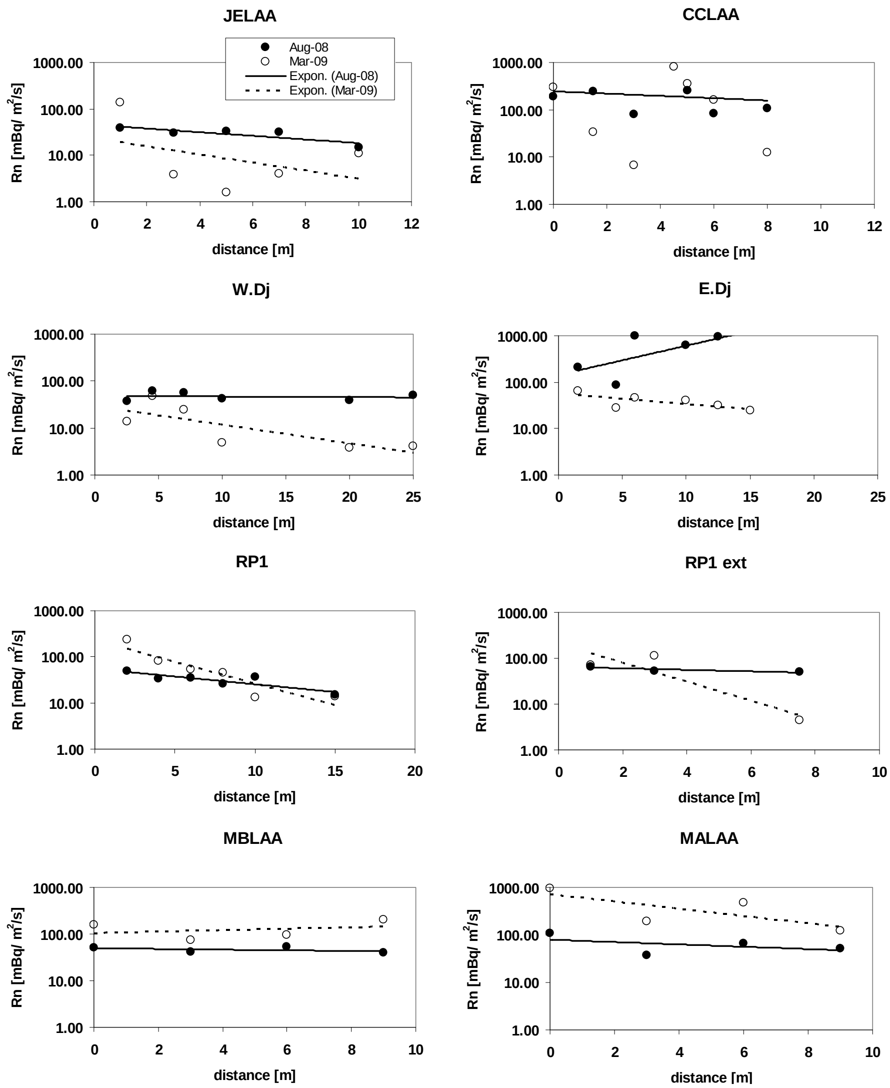

Figure 3.20 Radon flux densities measured in the dry and wet season, respectively, at various distances from the sprinklers at LAAs on the Ranger lease. The lines are exponential fits to the data.

Dust

Although it has been shown that the inhalation of dust contributes little to the radiological dose to the public in the off minesite areas of Jabiru East or Jabiru, it is possible that people accessing the LAAs may receive a higher dose from the inhalation of radionuclides in dust resuspended from the top few centimetres of LAA soils. This is an important factor to quantify in the context of assessing the rehabilitation requirements for the LAAs.

Dust samples were collected on sticky vinyl panels mounted on dust collector stations located along transects in the Magela LAAs (Figure 3.20). The panels were analysed for total alpha activity (Figure 3.20). The analyses showed that alpha activity is generally higher in samples closer to the ground indicating that a person sleeping will receive a higher dose from inhalation of dust than a person standing up. There is also a sharp drop of more than one order of magnitude in total alpha activity within the first 70 m outside the LAA boundary.

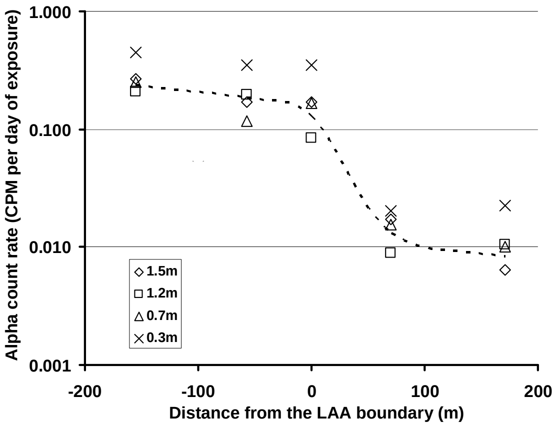

Figure 3.21 Total alpha activity (logarithmic scale) collected on sticky vinyl at various heights above ground in the dry season 2008 along a transect in the Magela B land application area. Positive distances shown on the x axis are outside the boundary of the LAA, negative distances are within.

Total alpha activity drops to about 0.01 cpm (counts per minute) per day between 100–200 m, a value similar to values measured at the ***eriss*** field station, about 4 km northwest of the transect. This indicates that while there is only limited transport of dust away from the LAAs, it may be a significant contributor to dose in the event of accessing or camping on the LAAs for an extended period of time. Further work is being done to verify published dust re-suspension factors and to quantify the dust inhalation pathway.

3.7.4 Conclusions and future work

This investigation has shown a substantial increase of radionuclide activity concentration in soils at the Magela, Djalkmara and RP1 LAAs due to irrigation of polished and unpolished RP2 water. However, this accumulation of radionuclides is restricted to the top 10 cm of the soil profile where most of the applied load is captured. There is very good agreement between measured radionuclide loads in the LAAs, and loads inferred from water quality data and irrigation rates over the past 25 years.

It can be expected that doses received via the external gamma and radon progeny inhalation pathways will only be little above background in the LAAs, in agreement with predictions from earlier studies. The dust inhalation pathway in the LAAs may become increasingly important and efforts currently focus on determining the resuspension factors for the area to quantify this pathway.

There are several rehabilitation options that could be used to reduce exposure of people potentially accessing the footprint of the LAAs, in the event that it was determined that such a reduction was needed. These options include removal of the top 10 cm of contaminated soil and placing it into the pit, tilling of the soil, or a mixture of both.

The extent of above background doses at Ranger post-remediation depends on pre-mining radiological conditions and future use of the area by indigenous people. The status of determining pre mining radiological conditions using Ranger Anomaly 2 as an analogue is addressed in the preceding section of this chapter. An agreed position by stakeholders on future land use and occupancy of the area is required as a pre-requisite to being able to predict applicable doses to humans post-remediation, and the possible need to carry out specific rehabilitation of the LAAs.

3.8 Remote sensing framework for environmental monitoring within the Alligator Rivers Region

Broad scale characterisation of landscapes in the Alligator Rivers Region (ARR) is required to be able to place the land surface status of operating and rehabilitated minesites into a regional context. To date there is little information on landscape ecology variables (in the context of rehabilitation, close out and known risks and threats) and their scale of interaction. Application of remote sensing technologies to address this knowledge gap requires the development of a remote sensing monitoring framework. The framework will provide the basis for most efficiently and cost effectively acquiring the required data by direct investment.

In May 2010 a systematic remote sensing data capture, incorporating full ground control and coincident spectral data collection, was done of the Magela floodplain and Ranger uranium mine. The data capture was undertaken in collaboration with the Tropical Rivers and Coastal Knowledge research hub's Theme 5.3 project (Food webs and biodiversity: river–floodplain food web studies). Three World-View 2 images covering 730 km^2 of the Magela Creek catchment were acquired. Table 3.8 shows the spectral bandwidth resolution of the satellite's sensor. The spatial resolution supplied is 0.5 m for the panchromatic band and 2.0 m for the multispectral bands.

The following scene parameters and data format were requested from the supplier: nadir angle less than 20º; cloud cover threshold 0–15%; and 16 bit data format. The potential capture dates for the imagery provided were May 6, 11, 14, 22 and 25. In order to ensure all required field and calibration data were available at the time of image capture, locational positioning and spectral calibration needed to be collected. Therefore, ground targets with accurate known locations had to be deployed prior to 6 May and suitable spectral calibration targets had to be in position, with spectral characteristics measured as close as possible to the time of image acquisition.

TABLE 3.8 SPECTRAL BANDS FOR THE WORLD-VIEW 2 SENSOR

Sensor band	Wavelength
Panchromatic	450–800 nm
Coastal	400–450 nm
Blue	450–510 nm
Green	510–580 nm
Yellow	585–625 nm
Red	630–690 nm
Red Edge	705–745 nm
Near-IR$_1$	770–895 nm
Near-IR$_2$	860–1040 nm

Historically there has been poor ground control for acquiring remote sensing data for the Magela Creek floodplain. High accuracy ground control is especially important in this case given the very low topographic relief of the area. This was achieved for the current capture by collecting 33 ground control points across the image acquisition area (Figure 3.22a).

Twenty-seven 3.5 x 3.5 m square silver-coloured tarpaulins were positioned on the ground prior to the image acquisition window and six image objects (features such as cross roads evident in previous image data) were selected for measurement. The centre of each of the tarpaulins and image objects were measured with a dGPS (Figure 3.22c) to within 12 mm x,y accuracy. These ground control points enable accurate orthorectification of the imagery.

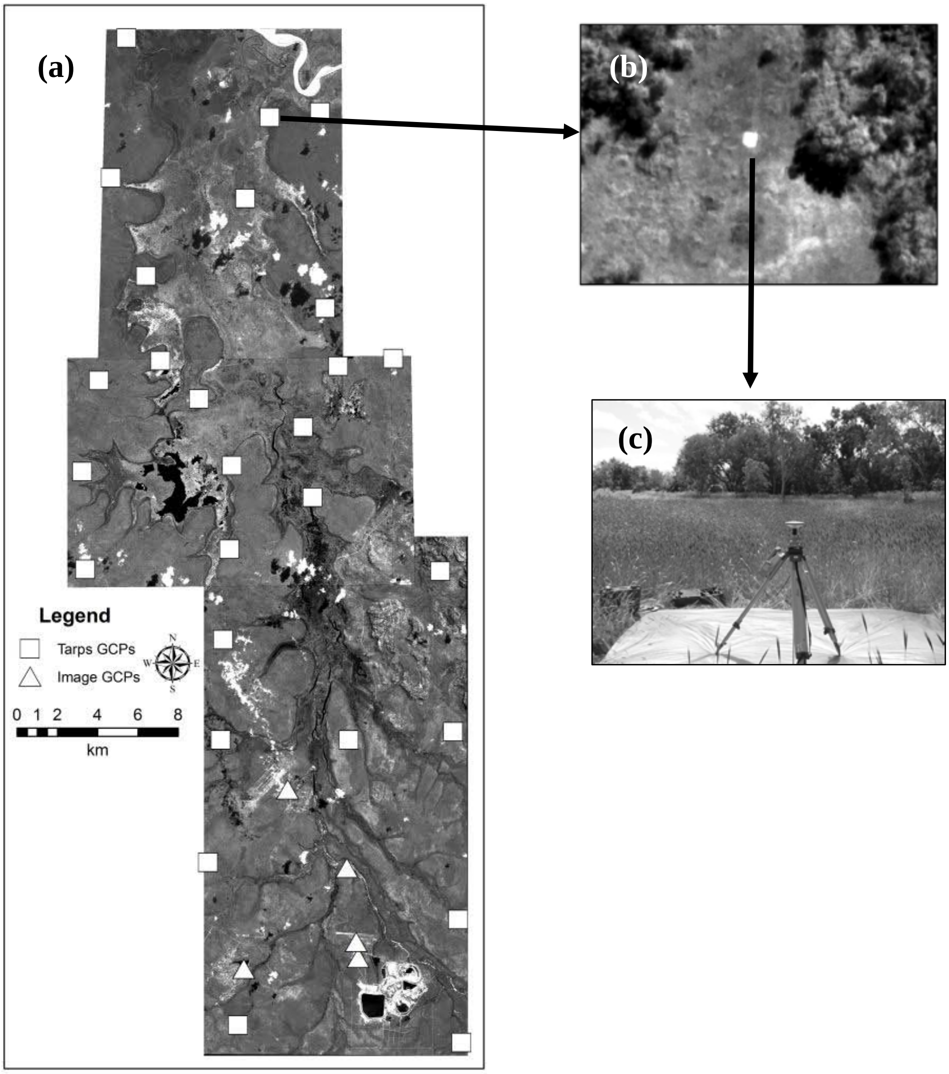

Figure 3.22 (a) Distribution of the Ground Control Points (GCPs) across the WV-2 imagery, (b) example of a tarpaulin GCP (site 2) captured by the WV-2 satellite in the panchromatic band, (c) collection of the dGPS data from the site 2 GCP

Atmospheric correction of satellite imagery using an empirical line method requires that high quality spectral measurements of suitable ground targets are acquired as close as possible to the time of image acquisition (in this case 10:30 am). After testing, using laboratory measurements of reflectance spectra, the suitability of various industrial products as ground targets, four materials were chosen to represent dark and bright targets. These were: black synthetic upholstery material (2% reflectance); silver plastic weave tarpaulin (23% reflectance); white plastic weave tarpaulin (67% reflectance); and Tyvec, a building insulation product (95% reflectance). The targets needed to be sufficiently large enough to be detected in the satellite imagery (Figure 3.23).

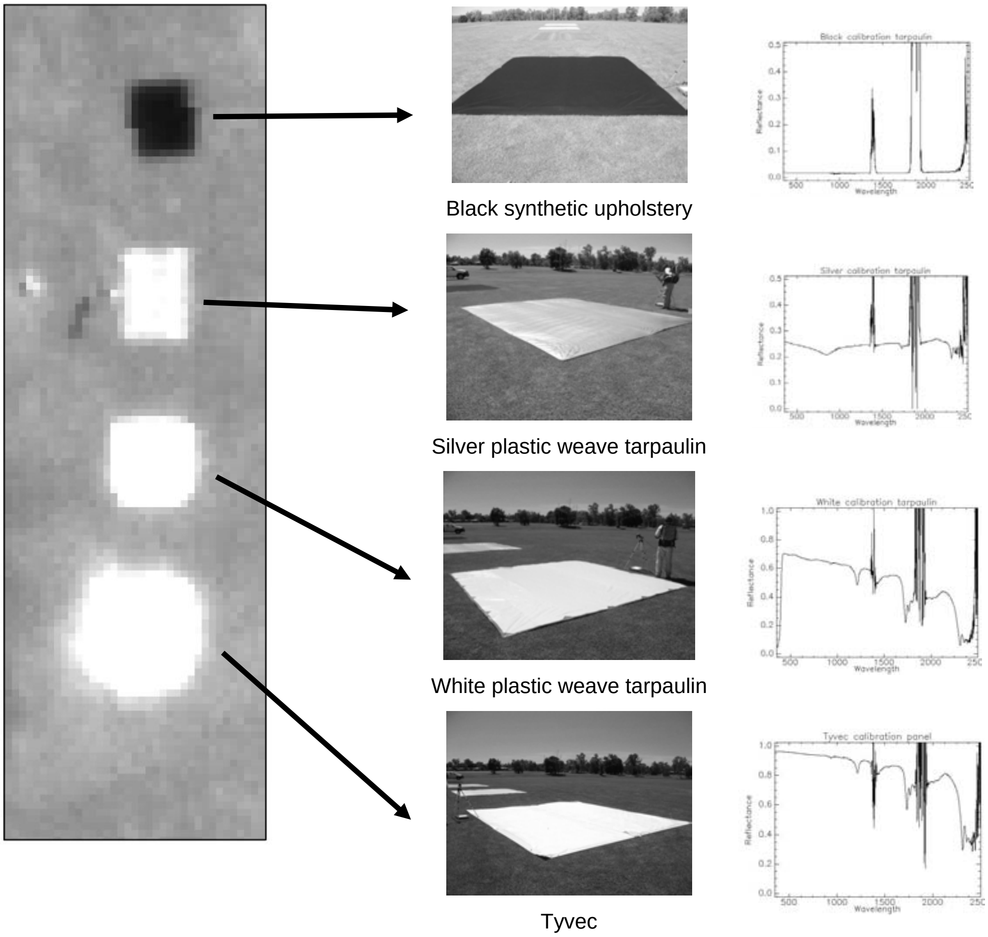

Figure 3.23 Calibration tarpaulins as captured by the WV-2 satellite (panchromatic band), photographs of target set up on Jabiru Oval, and the spectral signal of each tarpaulin as measured with the ASD SpecPro spectrometer

Prior to the 10:30 am satellite overpass, the four targets were deployed on Jabiru Oval. During the satellite overpass atmospheric solar irradiance data were collected using the ASD FieldSpecPro Spectrometer. Following the measurement of the solar irradiance data, multiple measurements of each of the four ground targets were collected. To assess the accuracy of the empirical line method for calibration of the WV-2 data, spectra of various invariant targets such as deep water, bare earth and well maintained golf green were also collected. These spectra were measured on the day of the overpass.

The majority (95%) of the areas of the three requested scenes were captured with the specified scene parameters (nadir angle of 13.8º and total cloud cover <2%) on 11 May. The remaining 5% of one of the scenes was captured on 22 May (nadir angle of 11.6º). All 27 ground control tarpaulins were visible in the imagery. Figure 3.22b shows an example of how ground targets appear in the imagery.

To produce a quality final product suitable for high resolution mapping of vegetation and habitat types, orthorectification of the imagery, atmospheric correction and the development of mapping applications will be required. This detailed work will be done over the next year.

3.9 The Bushtucker database

3.9.1 Background

For the past 30 years, information on the bioaccumulation of radionuclides in traditional bushfoods or 'bushtucker' has been gathered by SSD from many locations for a wide range of species. The database continues to be updated on an annual basis with data from routine monitoring programs and sometimes more opportunistically, for example in relation to the rehabilitation of the old mine workings in the South Alligator River Valley. Although the methodology and findings have been the subject of several journal and conference papers, as well as previous SSD Annual Reports and Research Summaries, there has been no prior integration of this material in an easily accessible format.

Newly available spatial technologies such as the 3-Dimensional virtual Earth/Globe viewing programs (hereafter referred to as virtual globe) such as Google Earth, Arc Explorer and Arc Globe offer a means to integrate and display this complex information in a format that is available to a wide range of potential users.

It is intended to develop a user friendly system to store and retrieve the data and present it to the local people of the area and to the wider public. The virtual globe environment will allow the user to navigate around the Alligator Rivers Region using high resolution satellite imagery and 'fly' to sampling sites to view available information. This gives the user a unique perspective of the terrain and appreciation of where the sampling sites are located relative to uranium mines, populated places and favoured bushtucker hunting and gathering sites. The virtual globe software is free for non-commercial applications and is easily downloaded from the internet, making it generally available to the community.

3.9.2 Scope of work

It is anticipated that three platforms will be developed – two for public viewing and a third for internal research use only. The first public viewing product is a Keyhole Markup Language file (KML) that contains all of the virtual globe features such as callout boxes, terrain flyovers and internet page hyperlinks. This compressed file will be available as a download from the internet or on a CD if necessary. It is simply loaded into the virtual globe program and the information tour starts automatically. KML files can also be loaded into web-based viewers such as Google Maps, Bing Maps, Yahoo Maps and Whereis.com to name a few.

The second public viewing product for non-internet users is a movie that has been created in a virtual globe environment and copied to a DVD. The movie will take the viewer on a tour of the bushtucker sampling sites with ‘pop-up’ information appearing along the way.

There is a need for caution when presenting data of this nature to the public because interpretation of the results is usually complex and there is potential for confusing or misleading interpretations to be made from individual numbers. In the event of data having been published previously in reports or papers, links will be provided to this reference source since they typically contain more detailed explanations/interpretations of the data.

The third ‘internal use only’ product would be similar to the first product but would contain all of the detailed radionuclide information. This product will enable SSD research staff to readily locate all of the available information in a spatial context and facilitate the use of the data across the Division.

3.9.3 Status

Figure 3.24 shows a sample Google Earth image with a callout box containing a graphic, text and links to, in this case, the catchments where bushtucker has been sampled.

The aims of this phase of the project are to make the information more accessible and understandable. In particular, to make the exploration of data more entertaining using the power of the virtual globe and the ability to ‘fly’ around the region and zoom in on features such as the escarpment, floodplains, billabongs and mining infrastructure. Included are many high quality photographs of the bushtucker fauna and flora species.

In Figure 3.25, the dialog box contains icons that provide a clickable link to web pages with information on radionuclides, aspects of the species’ biology and available information on how local Aboriginal people may catch and prepare the food. Both English (upper) and Gundjeihmi (lower) language names for the species are provided. Gundjeihmi is the major local indigenous language group.

Work on development of the primary KML files and construction of the internet pages is largely complete. These components provide the basis for all three outputs from this phase of the project. The KML files and internet pages can be readily updated to incorporate new data that become available from the Environmental Radiation group at ***eriss***. This approach to data presentation could be followed for other spatially-based datasets (for example, soils and vegetation) acquired by ***eriss*** over the years.

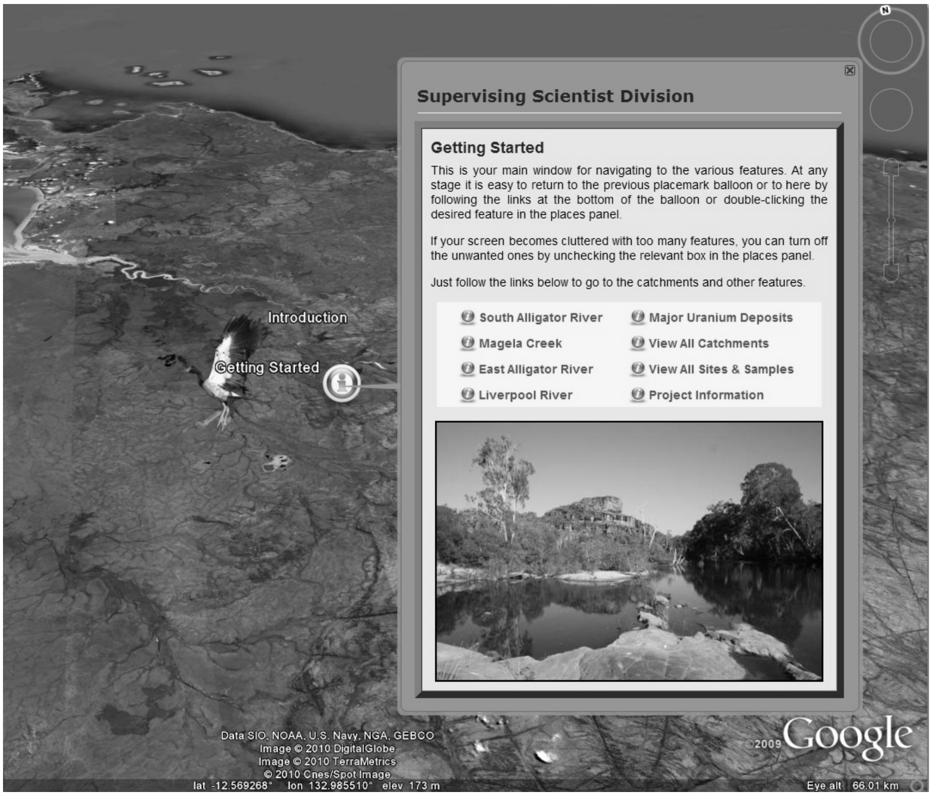

Figure 3.24 Google Earth snapshot with an information callout box and customised icons

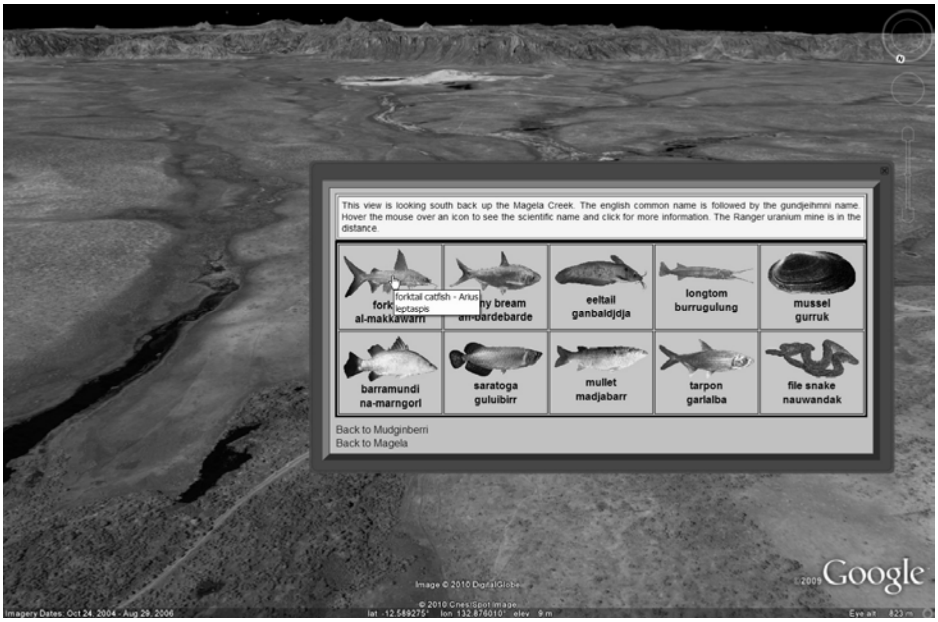

Figure 3.25 Another Google Earth snapshot showing a callout box with icons for the bushtucker species sampled at that site. In this Figure, Ranger uranium mine and Magela Creek are in the top centre.

3.10 Conceptual models of contaminant pathways for operational phase of Ranger uranium mine

Conceptual models of contaminant pathways associated with uranium mining in the Alligator Rivers Region (ARR) have been developed as part of the evolving ecological risk assessment framework being developed by the Supervising Scientist since the early 1980s. In response to recommendations by the World Heritage Commission Independent Scientific Panel and the Alligator Rivers Region Technical Committee (ARRTC), a specific project was initiated to produce an up-to-date comprehensive conceptual model of contaminant pathways associated with the operational phase of the Ranger uranium mine (RUM).

The new conceptual model framework was developed using an internal scientific expert panel approach involving senior ***eriss*** scientific staff to identify the main chemical, physico-chemical, biological and radiological contaminant types (stressors – see below) that could be potentially transported from the Ranger mine lease into the surrounding environment. For each contaminant class the source/s, potential transport mechanisms off-site, affected environmental compartments, receptor organisms, routes of exposure, types of effect (where known) and measures of effect (where available) were detailed. The conceptual model identified six main types of stressors and nine transport mechanisms associated with the operational phase of mining at Ranger (Table 3.9).

A diagram of the conceptual model elements was completed and validated by workshopping with external technical stakeholders in 2006. A sub-model diagram for the transport of inorganic toxicants via the surface water to surface water pathway was also completed to demonstrate the methodology that was being used. However, sub-model diagrams and narratives for the other potential contaminant pathways (up to 30) identified in the conceptual model were not developed at this time. Finalisation of the remaining contaminant pathway sub-models was identified as a priority by ARRTC during its most recent revision of the Key Knowledge Needs (KKN). Resources were allocated to progress the project in the second half of 2009.

A comprehensive review of the status of scientific knowledge regarding the various contaminants and pathways was undertaken and the content and structure of the conceptual model elements were revised as required. Draft sub-model diagrams for each of the potential contaminant pathways showing linkages between various model pathway elements (source, transport mechanisms, environmental compartments) and relevant measurement and assessment endpoints were also developed. The draft sub-models were revised following a technical workshop involving ***eriss*** Program Leaders and other senior scientific staff in September 2009. A report on progress was provided to ARRTC in November 2009. An example of the structure and content of the revised sub-models can be seen in the sub-model for the transport of inorganic toxicants via the surface water to surface water contaminant pathway (Figure 3.26). Supporting narratives for each of the sub-models were also drafted to provide explanatory information on the various pathway components, including spatial or temporal characteristics, the level of scientific knowledge and scientific certainty and any knowledge gaps. The narratives were refined with input from senior ***eriss*** scientific staff in early 2010.

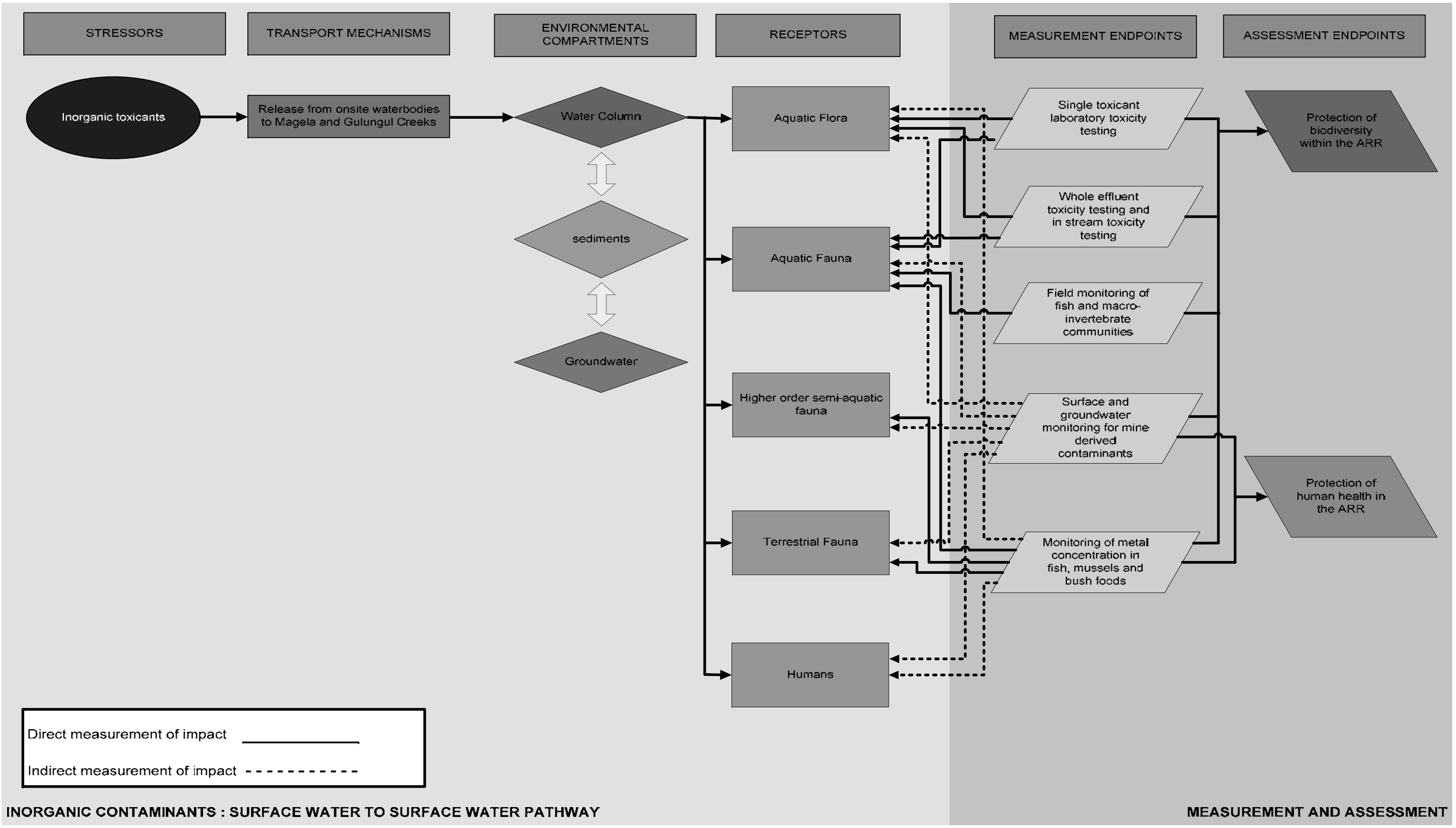

Figure 3.26 Conceptual model diagram for transport of inorganic toxicants from Ranger uranium mine via surface water to surface water pathway

TABLE 3.9 POTENTIAL STRESSORS AND TRANSPORT MECHANISMS ASSOCIATED WITH RANGER URANIUM MINE OPERATIONAL PHASE[2]

Potential stressors	**Inorganic toxicants** (eg uranium; magnesium; sulfate; manganese; ammonia) **Organic toxicants** (eg chlorinated aliphatic hydrocarbons, monocyclic aromatic hydrocarbons, polycyclic aromatic hydrocarbons, total petroleum hydrocarbons, organic sulfur compounds, volatile organic compounds) **Radionuclides** (eg Uranium – 238, 234, 235; Thorium-230; Radium-226; Lead-210; Polonium-210) **Radon-222 and its progeny** (eg Polonium-218, Lead-214, Bismuth-214, Polonium-214) **Weed propagules** (terrestrial and aquatic) **Suspended sediments** (<63 µm diameter)
Transport mechanisms	Release from mine site waterbodies direct to Magela and Gulungul Creeks Seepage from minesite waterbodies to groundwater and possible discharge to surface water systems Land application of mine water followed by (i) infiltration to groundwater and discharge to surface water and/or (ii) direct runoff to surface water Stormwater runoff from non-mine areas of lease Airborne dust and other particulates from mine site Airborne emissions from mill stacks and vehicles from mine lease Exhalation from mine lease Bioaccumulation and trophic transfer to mobile species visiting mine site waterbodies Human and non-human vectors (including vehicles)

1 Not all transport mechanisms are relevant to all stressors

2 from van Dam et al (2004)

The overall project approach and draft outputs were considered and endorsed by ARRTC in April 2010. Following this, it was decided that the importance of the contaminant pathways should be assessed in terms of their inherent potential to adversely impact on the environment within the ARR. In this context it should be noted that inherent potential does not equate to actual potential in the event of various management strategies (eg impounding of runoff followed by water treatment) being in place to provide mitigation.

A technical workshop involving senior ***eriss*** scientific staff was held in June 2010 in which each of the contaminant pathways were assessed based on the nature and size or generating capacity of the contaminant source, and the volume (and rate) of contaminants able to be transported off the mine lease via the pathway transport mechanisms. Project outcomes will be made available as a SSD Internal Report, in the Annual Research Summary and, eventually, a Supervising Scientist Report.

The content, design and functionality of various communication products arising from the project will be determined based on consultation with ARRTC members, traditional owners and other relevant stakeholders. This project will also contribute towards the future development of a risk-based framework, identified as a knowledge need by ARRTC, to support ***eriss*** research activities and scientific knowledge management.