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The sediment concentration - turbidity relation: Its value in monitoring at Ranger Uranium Mine, Northern Territory, Australia

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THE SEDIMENT CONCENTRATION - TURBIDITY RELATION: ITS VALUE IN MONITORING AT RANGER URANIUM MINE, NORTHERN TERRITORY, AUSTRALIA

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1. ABSTRACT

Some earlier studies of erosion of the schist-dominated waste rock dump of Ranger Uranium Mine, Northern Territory, Australia, used turbidity as a surrogate for sediment concentration. Subsequent detailed studies of the turbidity-sediment concentration relation for a number of sites on the waste rock dump, reported in this paper, demonstrate that turbidity cannot be used to accurately predict sediment concentration. The results of this experiment suggest that environmental monitoring of erosion at mines in the Kakadu region of Northern Australia should use direct methods of measurement of the sediment concentration in runoff and that turbidity measurements will not provide sufficient accuracy to satisfy audits of environmental compliance to regulatory guidelines.

Keywords: turbidity, erosion, monitoring, uranium mining

2. INTRODUCTION

The sometimes poor and often complex relation between sediment concentration and discharge (Olive and Rieger, 1985) makes the proposition of continuously recording sediment concentration highly attractive. An indirect and cost-effective measure of sediment concentration, particularly where a large number of water samples have to be analysed, has many potential benefits over the time-consuming standard filtration and gravimetric methods. Turbidity, as a surrogate for sediment concentration, offers an attractive solution, and turbidity meters have been used in laboratory and field situations for the measurement of sediment concentration (Gippel, 1989, 1995; Lawler and Brown, 1992). This paper discusses the value of using turbidity measurement as a surrogate for direct measurement of sediment concentration in runoff from the Ranger Uranium Mine, Northern Territory, Australia.

The quality of the relationship between turbidity and sediment concentration is crucial in determining the value of the turbidity surrogate. Problems have been noted with this relation, arising from variations in particle size, shape and composition, instrument stability, lighting conditions, organic load and biological activity on the probe (Burz, 1970; Gippel, 1995). Recent research suggests that infrared turbidity probes may overcome some of these problems, although coefficients of variation of 5% or more in the predictability of the concentrations were encountered by Clifford *et al.* (1995) and careful calibration of the relation was required. Hence, the purpose of the study described in this paper was to:

- determine the reliability of the turbidity-sediment concentration relationship at Ranger Uranium Mine,
- outline the limitation of using turbidity-based sediment concentration measurements in erosion research at Ranger, and

- examine the implications of turbidity as a monitoring tool in a regulatory environment.

3. EROSION ASSESSMENT AT RANGER URANIUM MINE

Ranger Uranium Mine is located in Kakadu National Park, Northern Territory, Australia (Fig 1) and the waste rock dump represents a potential environmental risk to surrounding wetlands. Details of the issues involved in the management of waste rock and uranium mill tailings are discussed by Riley (1995a, b). A critical aspect of planning the rehabilitation of Ranger is a detailed understanding of erosion processes. The need to develop process-response hydrogeomorphic models is detailed by Riley and East (1991) and aspects of the research program into erosion are discussed by Riley (1994).

Assurance of the effectiveness of erosion control at Ranger is undertaken through a combination of monitoring and modelling. Monitoring is necessary for assurance of erosion control in the operational phase of mining, while modelling is necessary to predict the long-term compliance of erosion control with whatever conditions are imposed by regulatory authorities. The need for reliable data on the sediment concentration in runoff is obvious, and in the World Heritage Listed area of Kakadu, the need to provide assurance of erosion control has international implications.

Turbidity has value in itself as a measure of water quality, particularly in the context of ecosystem viability, as stream waters in the region have very low sediment concentrations and associated turbidities (Riley, 1994). The turbidity surrogate has great attraction because of the ability to use it in continuous monitoring at automated stations, as well as the costsavings outlined above. The turbidity-sediment concentration relation has been used in some erosion studies at Ranger Uranium Mine (Curley, 1991; East *et al.*, 1994).

Early erosion studies at Ranger Uranium Mine used turbidity measurements as a surrogate for sediment concentration of runoff from the waste rock dump (Curley, 1991). These data were then used in estimates of erosion (East et al., 1994). Concern about the value of the relation as a means of measuring sediment concentration in monitoring and rainfall simulation studies led to its re-evaluation. The relation developed by Curley (1991) was

$$\text{Non-filtrable residue} = -30.97 + 1.65\ \text{Turbidity (NTU)} \qquad (1)$$

$$r^2 = 0.87$$

The error associated with the relation is large and for a turbidity less than 20 NTU (approximately) it is possible to predict a negative sediment concentration. The relative residuals of the predicted values of the original data set from which the regression equation was developed show that the equation is a poor predictor. Errors in the estimate of sediment concentration are of the order of 200% or more.

The turbidity study reported in this paper was part of a larger study of erosion at Ranger which concentrated on the waste rock dump. Details of the study are reported in Riley and East (1991) and Riley (1994). The waste rock dump is important in erosion studies at Ranger; it will occupy the largest area of the site (3 to 4 km^2) at the conclusion of mining and covers for tailings containment structures and pits will be constructed out of waste rock. The schist-dominated waste rock weathers rapidly in the seasonally wet Tropical environment, which has an average annual rainfall of 1500mm (Riley, 1994).

4. METHOD

A small area of the waste rock dump was monitored in 1989 for the purpose of developing and calibrating hydrogeomorphic models (Riley and Gardiner, 1992). The experimental site was within 100 metres of the batter plot site examined by East et al. (1994) in their 1986-1989 monitoring program. A number of small catchments on the flat cap and steeper batter sites of the waste rock were instrumented for rill and wash processes (Fig 1, Table 1). These plots/catchcments were established at each of the Cap and Batter sites in order to assess the significance of length of plot and to account for the effect of variability in the waste rock material on hydrology and erosion in the plots.

During the 1989-90 wet season these plots/catchments were monitored and samples of runoff collected either by automatic pump samplers or manually in 600 ml bottles at the discharge measuring flumes located at the outlets. The number of samples was not great because of instrument errors, an unusually dry wet season, and the usual teething problems in establishing a complex monitoring program. However, sufficient evidence was collected to throw considerable doubt on the quality of the turbidity-sediment concentration relation (Fig 2, see subsequent discussion). Confirmation of the unsatisfactory nature of the turbidity-sediment concentration relation was sought from the water quality sampling conducted during the monitoring program of the 1990-91 wet season. During the 1990-91 wet season 1119 samples of water from runoff in rills, small gullies and rainwash monitoring sites were collected.

In the laboratory the sealed bottles were washed and dried on the outside and then weighed. The samples were agitated and a small aliquot removed, placed in a turbidity meter cell and a turbidity reading taken, using a Hach Turbidimeter (model 439000). The turbidity was reported as NTU, as per the instrumentation. Turbidity measurements using this method conformed to the previous study (East *et al.*, 1994) for the purpose of comparison. At the same time, the method is in conformance with the recommendations outlined in Standard Methods 17th Edition (1989, confirmed in the 19th edition, 1995) for low turbidities. Tests of the repeatability of the turbidity readings were undertaken in the early phase of the program by taking several successive aliquots and recording the values. No significant differences were noted between the readings. Turbidity measurements were occasionally repeated as a quality assurance of the reliability of the operator, instrument and readings.

In the 1989-90 wet season monitoring program each sample in its 600ml bottle, including the aliquot used for the turbidity measurement, was then washed, with distilled water, though a 64μm sieve and filtered. For the 1989-90 and 1990-91 wet seasons the samples were filtered through a 0.45μm millipore filter, which had been previously dried in a desiccator for 24 hours prior to weighing. The filter paper with its sediment was oven dried, placed in a desiccator for 24 hrs and weighed on a balance with a precision of 100 μg. The sediment in the sieve (1989-90 wet season) was washed onto a sample dish, dried and weighed.

A test of the repeatability of the filtering technique was undertaken by filtering approximately 400 ml of distilled water and calculating the apparent sediment concentration from the change in weight of the filter paper. Six replicates gave an average sediment concentration of 8.2 mg/l with a standard deviation of 2.6mg/l. The gain in weight may result from a number of factors, including hydroscopic effects, balance errors, and contamination of filter papers. However, within the scope of the present study it was considered that an error of +/- 10 mg/l was not an issue when compared with the concentrations measured in the discharge from the waste rock dump.

5. RESULTS

There were never any readings with zero turbidity (NTU=0) but there were a number of sediment concentration measurements with zero or near-zero values. As there is a 10 mg/l error in the measurement of sediment concentration all estimates of sediment concentration less than this value were corrected to it. The effect on regression analyses and interpretation of results was modelled by examining some of the relations with and without the corrections and the results indicated that the regression relations were improved with this correction to the low sediment concentration values.

1989-90 wet season

Because there were insufficient data in the 1989-90 sampling program to examine the turbidity-sediment concentration relation for each individual sampling site, data from all the sites were included in a single analysis. A number of statistical tests were undertaken to confirm the best form of the regression relation. The linear equation used by Curley (1991) was a poorer predictor (lower correlation coefficient) than the log-log relation used in the following.

The regression relation between sediment concentration and turbidity (equation 2) is statistically significant, but the relation only explains 38% of the variance in the sediment concentration. Thus, while the correlation is statistically significant there is considerable uncertainty in the reliability of the predicted value of sediment concentration.

$$\text{Sed Conc} = 0.009\ \text{Turb}^{0.691} \qquad (2)$$

$$r^2 = 0.38,\ p<0.001,\ n=155$$

The relation between sediment concentration and turbidity (Fig 2), for each of the sediment fractions (equations 3 and 4), is significant but the correlation is weak, explaining less than 30% of the variance. Turbidity is the value for the whole sample (coarse and fines).

$$\text{Sed conc - coarse} = 0.00007\ \text{Turb}^{1.14}; \qquad (3)$$

$r^2 = 0.20$, $p<0.001$, $n=92$

$$\text{Sed conc - fines} = 0.0152\ \text{Turb}^{0.56}; \qquad (4)$$

$r^2 = 0.30$, $p<0.001$, $n=155$

Turbidity is obviously associated with the fine fraction, as can be seen in Figure 2 from the high turbidity readings associated with near-zero concentrations of coarse sediment. However, removing the coarse fraction does not improve the correlation, in fact it decreases the explained variance in sediment concentration (equation 4).

Splitting the data and examining the regression relation for turbidities greater than and less than 300 NTU did not reveal any significant improvements in the correlation coefficients except for equation 5, the coarse fraction of samples with NTU greater than 300. Comparison of equations 3 and 5 shows that the constant of the equation has not changed but the exponent has increased for the second equation. The improved correlation in equation 5 is not considered significant in predicting the total sediment concentration using turbidity because the majority of the sediment in the runoff sample is in the fine fraction.

>300NTU

$$\text{Sed conc - coarse} = 0.00007\ \text{Turb}^{1.55} \qquad (5)$$

$r^2 = 0.54$

$$\text{Sed conc - fines} = 0.0024\ \text{Turb}^{0.93} \quad (6)$$

$$r^2 = 0.34$$

<300NTU

$$\text{Sed conc - coarse} = 0.0203\ \text{Turb}^{0.59} \quad (7)$$

$$r^2 = 0.16$$

$$\text{Sed conc - fines} = 0.015\ \text{Turb}^{0.62} \quad (8)$$

$$r^2 = 0.17$$

The lower turbidity values (<300 NTU) have the poorer correlations and less than 17% of the variance in sediment concentration is explained by turbidity readings.

One means of assessing the value of the regression equations in predicting the sediment concentrations is to examine the relative residuals, the absolute values of the ratio between the difference between the predicted and actual value of sediment concentration and the actual sediment concentration ($= \frac{|(predicted - observed)|}{observed}$). Plots of the relative residuals of the correlations for the total data sets (both coarse and fine sediments, Fig 2) show that the errors in the predicted concentrations are up to 6 times as large as the measured sediment concentrations for the lower values. It is only for the higher sediment concentrations that the errors of the prediction equations are low. Predicted sediment concentrations are often two to three times different from the actual values (Fig 2).

1990-91 wet season

The far more detailed sampling of the 1990-91 wet season allowed an examination of the turbidity relation between the sites (Table 2). The separation of coarse and fine fractions was abandoned because the 1989-90 Wet season program showed that many samples had no coarse sediment and the proportion of coarse sediment in a sample was not large for almost all samples.

On the whole the power relation is much better than the linear relation, as was shown for the 1989-90 wet season data. The log-log regression relation is used in all the subsequent analyses (Fig 3). Examples of the relationships are presented in Figure 3.

There is an improvement in the correlation coefficients of many of the regression relations, compared with the relations derived from the 1989-90 wet season data (Table 2, Fig 3). Eight of the 14 sites had regression relations that explained more than 50 percent of the variance in sediment concentration. Two of the seven batter sites had r^2 values greater than 0.5 while five of the seven cap sites had r^2 values higher than this. A possible reason for the difference is the higher proportion of coarse sediment in the runoff samples collected from the Batter site plots/catchments. The Batter sites have high slope gradients and thus have higher stream powers per unit catchment area per unit runoff to erode sediment.

The variance explained in the regression relations (Table 2) appears to increase with catchment area. There is a high r^2 for the regression relation between catchment area and variance explained for the Batter site turbidity-sediment concentration relations (equation 9). The regression relation is not significant for the Cap site, even when the large catchment (COUT) is left out of the regression (equation 10).

Batter site Variance Explained (r^2) = 0.003 Catchment area (m^2) + 0.08 (9)

$r2 = 0.63$

Cap site Variance Explained (r^2) = 0.0003Catchment Area (m^2) + 0.47 (10)

$r2 = 0.26$

The most important result of the study is that the relations between sediment concentration and turbidity are significantly different for many of the sites. The sites are less than 200 metres from each other (Fig 1) and developed on the same waste rock material, which is all treated in the same way when deposited in the waste rock dump (Riley, 1994). In order for the reliability of the turbidity surrogate to be confirmed it would be necessary to calibrate the turbidity-sediment concentration relation for each site at which it was decided to use a turbidity meter. As only three out of the 14 monitoring sites used in this experiment produced regression relations that explained more than 75% of the variance, it is obvious that the sediment concentration will have to be measured directly.

The value of a common regression relation between turbidity and sediment concentration was assessed using the combined sets of data from the 14 sites. The variance explained when the data are combined for the 1990-91 wet season is improved compared to the regression relation obtained from the 1989-90 wet season results. The relation is statistically significant (Fig 4). There is some justification for using a common relation, but the value of the relation must be judged against the likelihood of erroneous estimates of sediment concentration. The deviation between the predicted and actual values shows that the majority of predictions deviate by more than 30% and a large number differ by more than 100% from the actual values. Some samples have relative errors in excess of 1000% and one sample (not shown on Figure 4) had measured sediment concentration that was 67 times larger than that predicted by the regression relation. In a regulatory environment it has to be asked whether a 100% or greater error in an estimate of sediment concentration is acceptable.

6. DISCUSSION

The wide range of sediment concentrations sampled at each of the sites and the range of hydrological conditions that were sampled suggest that the data set for the 1990-91 Wet season is sufficient to adequately judge the quality of the turbidity-sediment concentration relation.

Even though some of the regressions explain more than 70% of the variance a number are very weak. This means that the value of even a site-specific specific regression relation has to be questioned. Two issues have to be considered. Firstly, the effort in calibrating the regression relation is not much less than undertaking direct measurements of sediment concentration. Secondly, there is no guarantee that the regression relation is going to be reliable or sufficiently accurate to satisfy operational and regulatory needs. Efforts to correct the relations for catchment characteristics proved unsatisfactory, using the obvious differences of site and catchment area that exist between the Batter and Cap sites used in this study. Using regression equations specific to individual sites is not warranted and would result in erroneous estimates of sediment concentrations.

The high degree of variability between the sites suggests that the use of a reliable and accurate single regression equation for the waste rock dump, let alone the rest of the mine site, is not possible. An example of the significance of the differences among the regression equations for the different sites is demonstrated for four sites (Fig 5). The data are presented in the order in which the samples were collected at the site (but not referenced to time of sampling). The predicted values of sediment concentration, using regression equations from different sites, are estimated from the measured values of turbidity (Fig 5). Almost all the predicted values of sediment concentration are higher than the actual values, particularly for the higher concentrations, and the difference between the actual and the predicted values of sediment concentration are significant. Comparisons of actual sediment concentrations with estimated values for other sites and other storms show the same patterns of deviation between the actual and estimated values.

Observations of erosion during the sampling suggest some reasons for the poor quality of the turbidity-sediment concentration relation. There are obvious differences in the mixtures of fines and coarse materials entrained during runoff. These differences arise from factors like first-flush, sediment exhaustion, and variations in contributions from different sections of the plots during storms. Clearly, the same sediment concentrations can have different grain size mixes. Also, there appears to be some differences in the shape and colour of the sediment that moves through the catchment to the outlet. Organic particles tend to move quickly, while the micaceous sands are moved more easily than the rounder quartz sands. These particles would have different effects on the turbidity - sediment concentration relation (eg the shiny surfaces of the micas).

The value of turbidity measurements, apart from their ecological implications, lays in the continuous nature of the sampling, It is not economically feasible, over a long period, to collect water samples to estimate sediment concentration every minute. This means that complex relationships have to be developed between the sediment concentrations measured on samples taken by the pump or manual sampler and the more continuous samples taken by the turbidity meter. These relationships are not demonstrated here but this study demonstrates that the algorithms for correcting turbidity measurement will have to be developed for each site.

The errors arising from turbidity estimates of sediment concentration would be compounded in complex process-response models of erosion when these models were optimised. The poor estimates of sediment concentration would significantly effect the quality of calibrated erosion models. Underestimates of sediment concentration of the order of half of the actual values would result in predictions of the longevity of containment and rehabilitation structures nearly twice as long as they would be in reality. Estimates of loads exported to adjacent billabongs and streams over the 1000 year structural life of a rehabilitated uranium mine would be less than half of the actual.

Finally, there are implications for regulatory monitoring of erosion on the mine site. It has to be asked whether any regulatory organisation would accept estimates of readings for sediment concentration that, for 30% of the samples, stated the sediment concentration at less than a half or more than twice of their actual values. The estimate of mean sediment concentration of Magela Creek, which runs pass the Ranger Mine, is less than 20mg/l. The errors that are identified in this study greatly exceed this value.

7. CONCLUSION

While there are statistically significant relations between the turbidity of water samples and their sediment concentration the error in predicting the sediment concentration using turbidity measurements is site specific and, for some uses, too high. This study has shown that the differences in the relations vary significantly between sites that are less than 200m apart on waste rock dumps. There appear to be no simple means for correcting the relations based on the site characteristics of position in the landscape and catchment area.

The errors inherent in the relations suggest that it is not possible to accept turbidity as a surrogate for sediment concentration. Water samples have to be collected in order to measure sediment concentration. Even if a turbidity probe is used and a calibration equation is derived, it is obvious that a continuous program of water sampling has to be undertaken to improve the reliability of each measurement and to assess for 'drift' in the calibrated relation.

8. ACKNOWLEDGEMENTS

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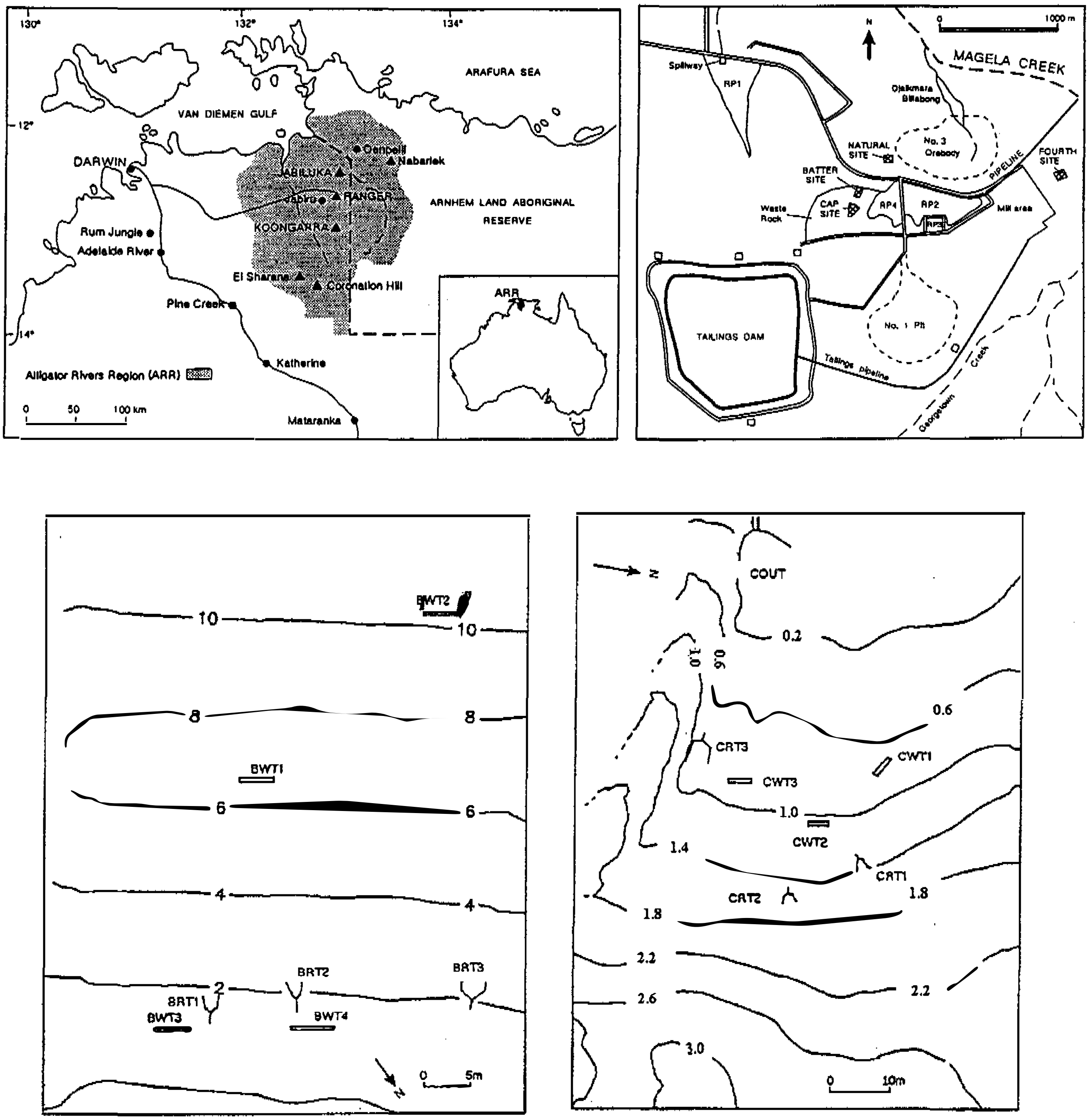
10. LIST OF FIGURES

Figure 1. Location of sampling sites, Ranger Uranium Mine, showing the position of the rill and wash traps on the Cap ('C' sites) and Batter ('B' sites) monitoring sites. Contour lines (metres) on the monitoring site maps are to relative datums.



Batter site

Cap site

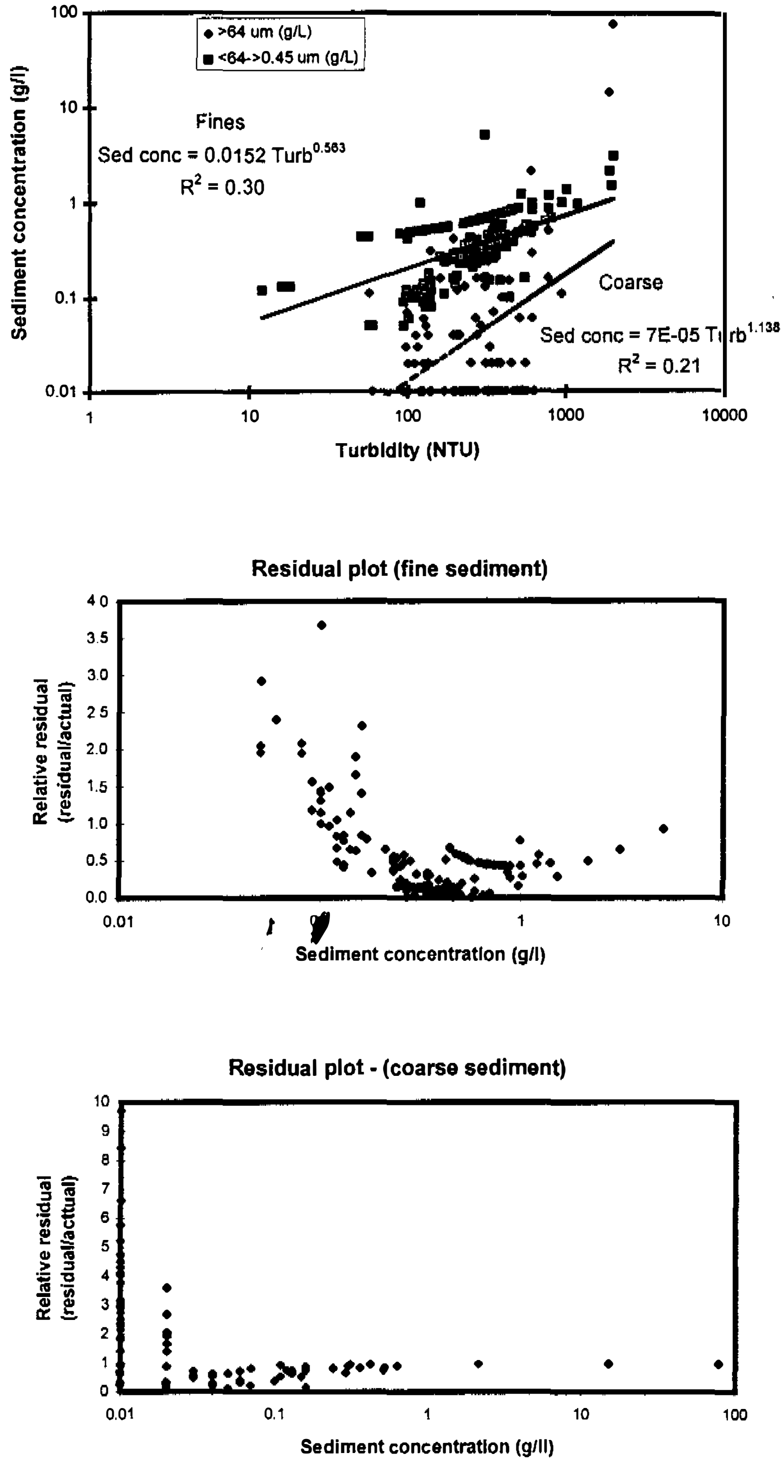


Figure 2. Sediment concentration - turbidity relations for the coarse and fine fraction of samples collected during the 1989-90 wet season. Relative residuals for each relation are presented in the lower diagrams.

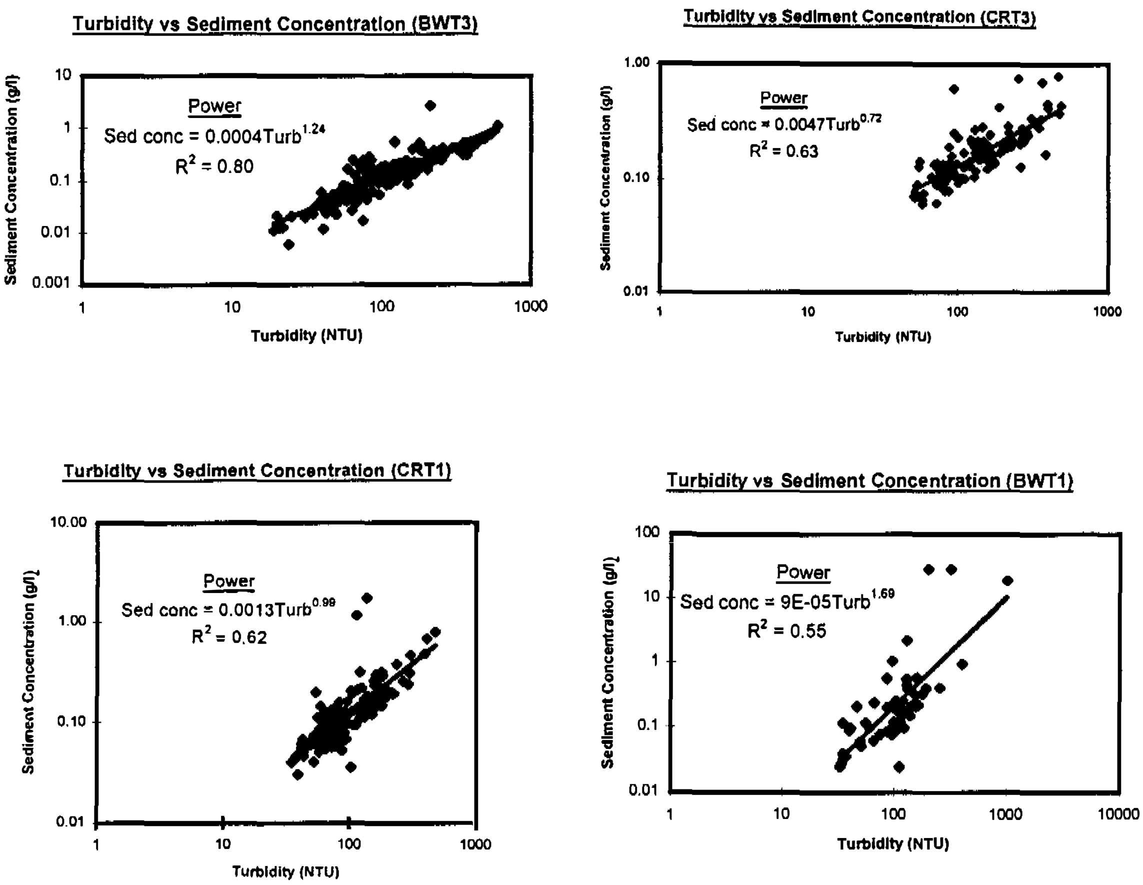
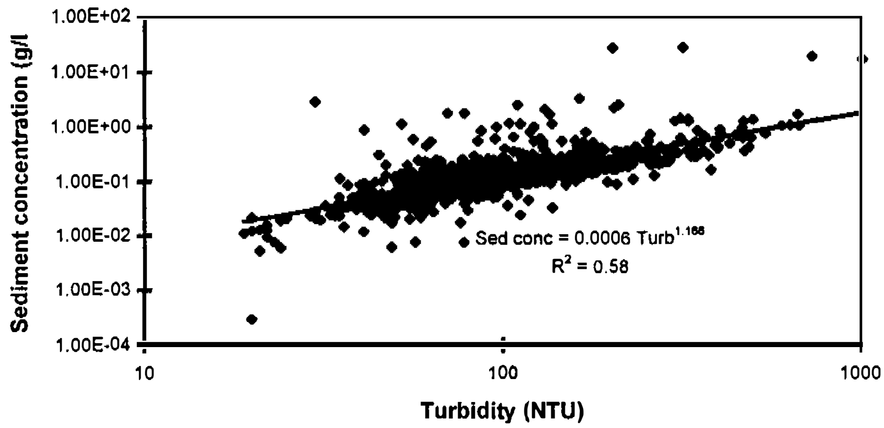


Figure 3. Examples of the relations between sediment concentration and turbidity for the 1990-91 wet season sampling program

SEDIMENT CONCENTRATION-TURBIDITY RELATION
(FOR ALL SAMPLES)



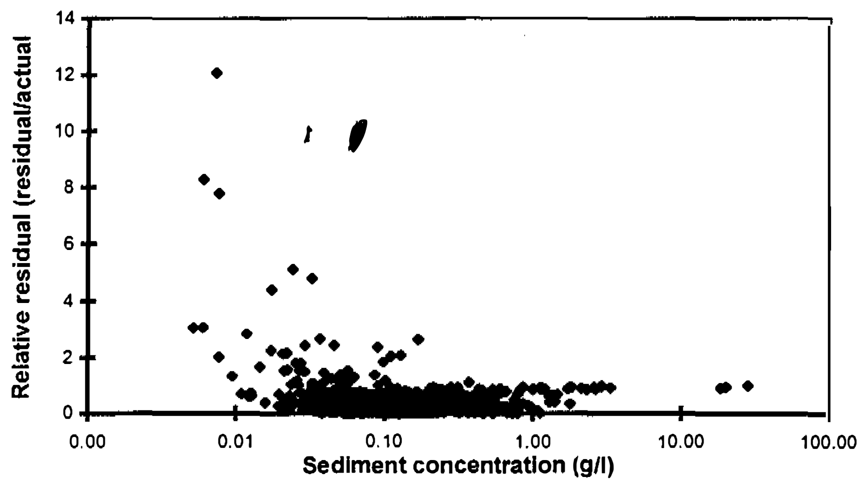


Figure 4. Sediment concentration - turbidity relation for all samples collected during the 1990-91 wet season, Ranger Uranium Mine. One value with a relative residual value of 67 has been omitted from the graph to improve presentation of the data.

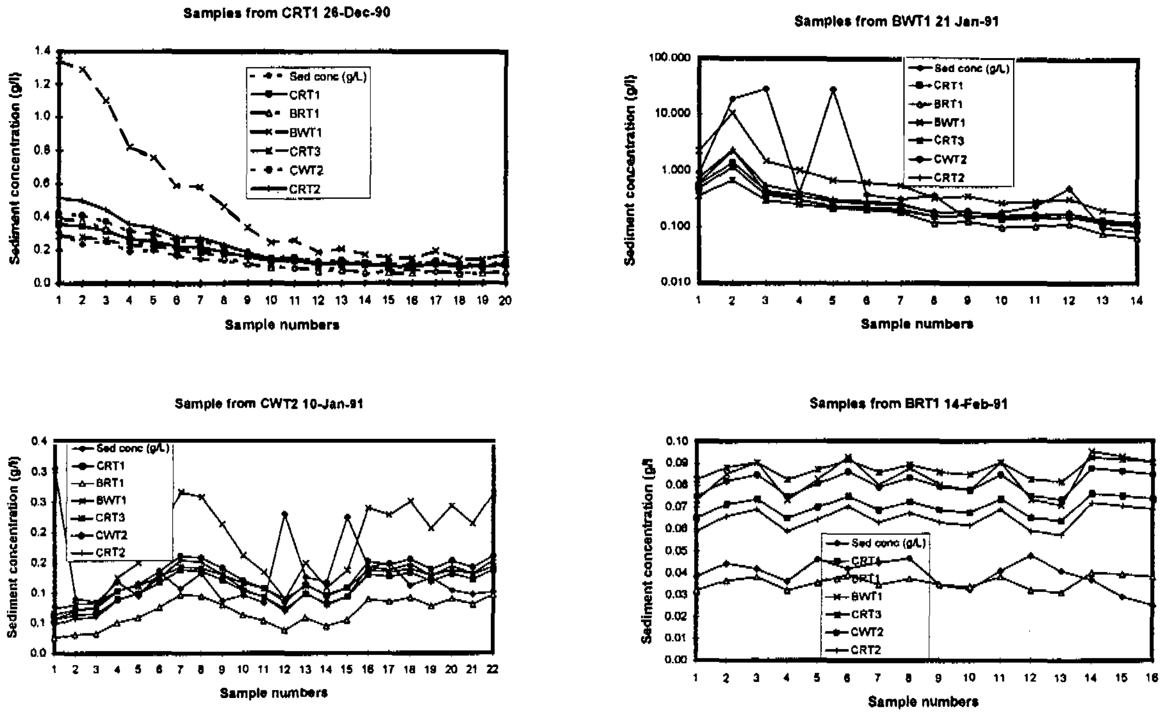


Figure 5. Examples of predicted and measured sediment concentrations for samples taken from Rill Trap 1 and Wash Trap 2 on the Cap site (CRT1 and CWT2) and Rill Trap 1 and Wash Trap 1 on the Batter site (BRT1 and BWT1). Horizontal axis shows sample numbers and are not spaced according to time of sampling. Predictions are based on regression equations developed for each sampling site.

Table 1

Cap site catchment characteristics

CAP SITE (C)			BATTER SITE (B)		
SITE	CATCHMENT AREA (m^2)	MEAN SLOPE (m/m)	SITE	CATCHMENT AREA (m^2)	MEAN SLOPE(m/m)
COUT	2182	0.03	BRT1	103	0.18
CRT1	461	0.029	BRT2	94	0.19
CRT2	330	0.039	BRT3	134	0.18
CRT3	731	0.034	BWT1	105	0.21
CWT1	149	0.04	BWT2	32	0.20
CWT2	102	0.035	BWT3	234	0.19
CWT3	91	0.036	BWT4	204	0.19

RT = rill trap

WT = wash trap

OUT = catchment outlet

Table 2

Turbidity - sediment concentration relations for each sampling site (power relations)

Site No.	Equation	R^2	p	n
BRT1	Sed Conc. = $0.0001(\text{Turb.})^{1.4536}$	0.32	0.000	63
BRT2	Sed Conc. = $0.0043(\text{Turb.})^{0.7413}$	0.17	0.001	57
BRT3	Sed Conc. = $0.0004(\text{Turb.})^{1.2654}$	0.32	0.000	67
BWT1	Sed Conc. = $9E\text{-}5(\text{Turb.})^{1.6881}$	0.55	0.000	55
BWT2	Sed Conc. = $1E\text{-}5(\text{Turb.})^{1.9447}$	0.35	0.013	17
BWT3	Sed Conc. = $0.0003(\text{Turb.})^{1.2909}$	0.79	0.000	60
BWT4	Sed Conc. = $0.0003(\text{Turb.})^{1.2462}$	0.87	0.000	37
COUT	Sed Conc. = $0.0012(\text{Turb.})^{1.0371}$	0.54	0.000	116
CRT1	Sed Conc. = $0.0013(\text{Turb.})^{0.9863}$	0.62	0.000	142
CRT2	Sed Conc. = $0.0004(\text{Turb.})^{1.258}$	0.88	0.000	81
CRT3	Sed Conc. = $0.0047(\text{Turb.})^{0.7223}$	0.63	0.000	122
CWT1	Sed Conc. = $0.0009(\text{Turb})^{1.0719}$	0.50	0.000	104
CWT2	Sed Conc. = $0.0014(\text{Turb.})^{1.0027}$	0.42	0.000	88
CWT3	Sed Conc. = $0.0007(\text{Turb.})^{1.213}$	0.44	0.000	99

Risk assessment: a component of the design of containment structures for uranium mill tailings, Ranger Uranium Mine, Australia

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ABSTRACT: Predictive assessment of the environmental risk of the release of uranium mill tailings and the other products of erosion of containment structures is an important aspect of rehabilitation design and assessment. Any meaningful assessment should include a measure of the environmental change arising from dispersed tailings and erosion products. This paper discusses issues in assessing the non-radiological ecological risk of a tailings spill using toxicity testing protocols, with an example from the Ranger Uranium Mine, Northern Territory, Australia. A conceptual model is presented of the relation between environmental risk, probability of failure of the containment structure, design life and the community expectation of acceptable risk.

1. INTRODUCTION

Mines have to be rehabilitated because mining disturbs the immediate environs of the mine site and has the potential for off-site environmental impacts (Birrell *et al.*, 1982; Hannan, 1981; Ollier, 1984). Large scale uranium mines produce radioactive mill tailings that require containment and isolation for thousands of years (Commonwealth of Australia, 1987). During these long periods, erosion and weathering have significant effects on cover materials and site stability and the products of erosion may impact on surrounding areas (Schumm et al., 1981, 1982; Pidgeon, 1982). Thus, the geomorphic stability of containment structures and associated landforms is important in the rehabilitation of uranium mines, as it is for other mines (Toy et al., 1987; Cecille, 1991; Watts et al., 1993; Riley, 1994). This paper discusses some of the issues related to environmental risk assessment of the rehabilitation of Ranger Uranium Mine (RUM), Northern Territory, Australia.

Ranger Uranium Mine (RUM) is located in the world heritage listed area of Kakadu National Park (Fig 1) and is within the Magela Creek valley. The valley is occupied by its traditional aboriginal owners, and downstream of the mine is an area of substantial wetlands. One proposed rehabilitation strategy at Ranger (Unger et al., 1996), the above-grade option, incorporates the existing tailings dam ($1 km^2$ in area) in the waste rock dump and low-grade ore stockpile and uses the waste rock as a cover material. The final landform will be more than $4 km^2$ in area and rise more than 17m above the surrounding lowlands. The waste rock is dominated by a highly chloritised schist, which weathers rapidly in the seasonally wet tropical climate.

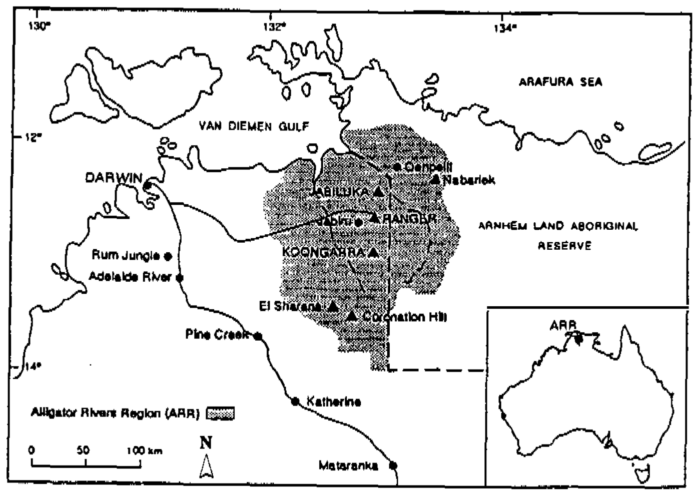

Figure 1. Location of Ranger Uranium Mine, Northern Territory, Australia

2 REHABILITATION ISSUES

It is necessary to estimate the risk to the environment of failure of the containment (Waggitt and Riley, 1994) for this will affect the design standards in terms of acceptable levels of failure and desired design and structural life (Fig 2a). The higher the level of risk the greater must be the security of the containment system for the design period (Fig 2b). Furthermore, the longer the period of risk (the period of effective risk) the longer the design life of a containment structure.

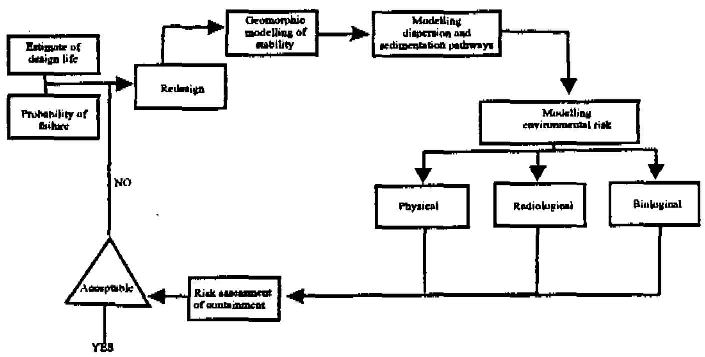

Figure 2a. Procedures in analysis of environmental risk of release of uranium mill tailings

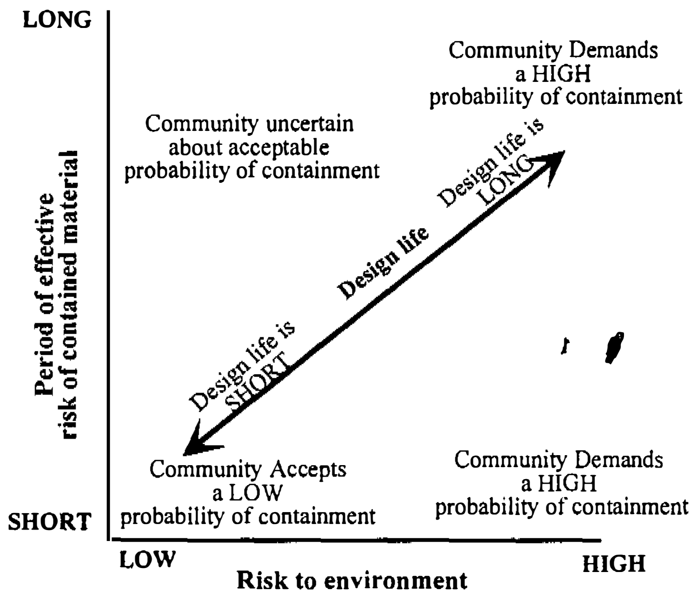

Figure 2b Conceptual model of the relation between Risk to the environment, period of effective risk, acceptable probability of failure and design life of the structure. Community Expectation of the the security of the structure (=probability of containment) ranges from lower values to the left of the diagram to higher values to the right. The design life increases from the origin (of low risk and short period of effective risk) outwards.

Details of the geomorphic issues of stability are given in Riley (1995a, b). Geomorphic modelling of a proposed rehabilitation structure at Ranger shows that there is substantial erosion in its central area and on the margins of the steeper batter slopes. In some areas the predicted erosion exceeds 7m in depth after 1000 years. Details of the modelling are presented in Willgoose and Riley (1993) and the role of vegetation and surface management in erosion and hydrologic processes is discussed by Riley et al. (1995).

There are both radiological and non-radiological aspects of the hazard to the environment of a release of uranium mill tailings. The tailings may be toxic to the environment irrespective of their radiological hazard. Most uranium mill tailings containment structures are constructed from natural materials, usually the waste rock produced during the mining operation. Erosion of these cover materials poses an environmental hazard, irrespective of the exposure and release of the contained tailings.

Just as it is necessary to undertake geomorphic modelling to assess the stability of the containment structure so it is necessary to undertake modelling to assess the environmental impacts of the release of tailings under different failure scenarios. The environmental assessment can be divided into physical, radiological and biological components (Fig 2a). The physical impacts can be modelled by examining the dispersion pathways and related processes (Riley and Waggitt, 1992; Wasson, 1992) and radiological models for hazard and risk assessment are available (AECL, 1992; Akber et al., 1992; Moroney, 1992; Carter, 1992). In order to model the potential biological impacts it is necessary, at the very least, to have information on the toxicity of the tailings and the eroded products of waste rock. For the Alligator Rivers Region, as for many other ecosystems throughout the world, there is a paucity of data for the impact of constituents of released water and sediments on relevant species. While there is a substantial body of literature on risk of failure of containment structures there is less concerning toxicity assessment of the contained materials (Deason and Bunch, 1990; Bedinger and Stevens, 1990; Brown and Lemons, 1991; Cecille, 1991; IAEA, 1992) on which an environmental risk assessment can be based. Yet a holistic approach to risk assessment requires that estimates of probabilities of failure of containment structures must be placed in the context of the impact on the environment (natural and human) of the contained material.

Toxicity testing involves the exposure of an organism, or part of an organism or tissue culture, to a concentration series of test chemical or chemicals and has the advantage of examining the impact of the suite of chemicals that contribute to the environmental risk. This holistic approach, together with obtaining a relevant biological response, has distinct advantages in establishing the hazard of test substances (Rippon and Chapman, 1994). However, the results of toxicity tests have to be considered in the context of the selective nature of the species testing. There can be no absolute assurance of ecological security because it is not possible to test the impact of potential contaminants on all aspects of the environment, including the complex interactions among species.

2.1 *Environmental hazard at Ranger*

The likely deposition sites for tailings and eroded waste rock have been described by Riley and Waggitt (1992) and Wasson (1992) and include the major stream courses, billabongs and a $30km^2$ section of the backwater plain of Magela Creek below Mudginberri. It is likely that the areas first affected by eroded material would be the streams and associated billabongs that drain the mine site, where the coarser materials would be deposited. Changes are likely to be morphometric, eg the billabongs would infill and the sediments of the sand bed streams would become finer. The fines (silts and clays) would be carried further, to the outer extremities of the flood plains and the downstream sections of Magela Creek backwater plain (Wasson, 1992).

The likely rates of discharge of sediment would be a function of the type of erosion and whether the containment structure fails. The fine grained nature of the tailings would facilitate rapid erosion, but the rate of erosion would clearly depend on their exposure. Wasson (1992) estimated that if the erosion rate of the tailings was four times that of the natural erosion rate then the proportion of tailings on the backwater plain would constitute 40% of the total sediment deposit. Riley and Waggitt (1992) estimated that a loss of 5000 tonnes per annum of tailings or eroded material from the containment would double the sedimentation rate of the backwater plain immediately below Mudginberri Billabong.

Once deposited the tailings fines or material from the waste rock cover may be subjected to cycling between extremes of some water quality parameters, such as redox potential or pH, especially if seasonal conditions are extreme. Any metals released in this cycling will be affected by other processes, such as the development of thermoclines, flood plain and billabong hydrology, microbial activity and the amount of organic material in the associated natural sediment matrix. Thus, a number of potential scenarios exist for considering the fate of tailings and other eroded materials in the environment.

The initial non-radiological environmental impact of uranium mill tailings would be chemical or physical. As discussed, the physical impacts are related to changes in the texture of surface materials or in the landform geometry. Other transient, but detrimental, effects might occur *in stream* due to an increase in turbidity. The physical impacts are clearly related to changes in habitat quality, and assessment of the impact can be judged by the proportion of habitat that could be affected. The ways in which tailings physically impact on the environment may be subtle, eg. fine grained tailings may alter the infiltration characteristics of soils which may impact on water availability to plants and soil fauna, which in term may influence plant growth and reproduction. Chemical changes are related to the complex geochemical processes within each ecosystem and are less easily assessed because of the potential for chemicals to disrupt ecosystems through their effects on any number of species or species interactions, hence the value of toxicity testing as an aid in assessing environmental impact (Brown, 1986; Cairns and Mount, 1990).

Toxicity testing of the tailings and runoff from the waste rock has been conducted (Rippon et al., 1994; Rippon and Riley, 1996). The most significant result of the tests was that the tailings neutralised flood plain sediments and did not appear to represent a hazard to the test species. Some uncertainty in interpreting this result in terms of the ecosystem is to be expected because it is unlikely that a complete understanding of the flood plain ecosystem will ever be achieved. However, this uncertainty should not preclude us from attempting the interpretation. Any fundamental change in sediment quality could affect the nature and abundance of the sediment fauna and microflora. It could, for instance, disrupt processes such as nutrient cycling mediated by sediment-dependent organisms and possibly reduce the amount of food available to bottom-feeding fishes and other animals. The measurement of fluxes in ecosystems, the principal functional groups, and abiotic factors of the ecosystem, could be used to give a measure of ecosystem function (or integrity). The cautious approach to the results of the toxicity testing of the tailings would suggest that the non-radiological environmental risk of the tailings as a chemical entity is probably low but not yet well defined.

This analysis of environmental hazard has not considered atmospheric or groundwater issues as it focuses on the tailings containment. Atmospheric issues are not considered important and groundwater hazard is related to potential long-term use of groundwater resources and the complex aspects of dilution as groundwater returns to the surface.

3 IMPLICATIONS FOR REHABILITATION

Risk assessment of the above-grade rehabilitation proposal suggests that there will be a high risk of failure of the containment structure within a 1000 year period. Furthermore, the quantities of sediment eroded from the containment structure will be in excess of 1 million cubic metres. This sediment loss equates to an annual erosion rate of approximately 2 to 4 tonnes/ha/yr (depending on the density estimate) and a denudation rate greater than 0.3mm/yr (0.3m/1000yr). The natural denudation rate for the lowlands on which the mine is located is less than 0.02mm/yr (Riley, 1995c). These estimates of the stability of Ranger rehabilitated landforms are approximate, based on the results of modelling and monitoring, and include factors for a revegetated containment structure.

Some of the eroded material will be in soluble form, a result of weathering, and will be transported through the fluvial system. The majority will be particulate and the volumes released will far exceed the volumes that would have been eroded from the area now occupied by the rehabilitation structure. Thus the physical impact of the eroding rehabilitation structure on the surrounding area is likely to be high, leading to loss of habitat and change in the spatial distribution of ecosystems. However, it is unlikely that the physical impacts will lead to species extinction in the area and the rates of change caused by the sedimentation, while much higher than the background (natural) rates, are unlikely to be perceived as significant by the casual observer operating in a secular timeframe. Infilling of artificial wetlands and billabongs may be noticeable.

The bio-chemical impact of the tailings does not seem to be significant but the toxicity of eroded waste rock material may be much higher. This paper is concerned with the tailings, tests into the chemistry and toxicity of runoff from the waste rock are another issue.

Radiological aspects of the environmental risk may be significant if tailings are released but will be insignificant if only waste rock is released into the fluvial system of the Magela floodplain, which is the case in the early period after completion of rehabilitation works.

The acceptable probability of failure of a structure is a socio-economic decision in the political arena. Differences in culture, economics and environmental perceptions can result in considerable variation in acceptable levels of failure even within communities, let alone between different sites. Traditional Owners appear to be reluctant to permit the opening of a second mine in the Magela Valley, at Jabiluka, because their perception of the risk is such that mining is unacceptable to them. The perception of the level of risk is dominated by issues that are not directly related to the physical impact of the mining, and include social impact and religious values. Thus, as has been noted in other contexts (Douglas and Wildavsky, 1982), there is a complex relation between risk, the period of risk, the acceptable probability of failure and the design life accepted for the containment structure (Fig 2b). Sandman (1993) has indicated that the perceived risk is a combination of hazard plus "outrage",

(RISK=HAZARD+OUTRAGE)

and perusal of letters to national papers will show that outrage about uranium mining is high for some sections of the Australian community.

4 CONCLUSION

Previous studies suggest that the risk of failure of the proposed rehabilitation structure at Ranger Uranium Mine over a 1000 year period is high but that the direct environmental bio-chemical hazard of released tailings is low. It appears that the greatest environmental risk to the environment, and of immediate concern in the design of rehabilitation structures, relates to the physical impact of the eroded containment material. Erosion products have the potential to infill wetlands, designed to remediate water quality of runoff, and physically impact on the aquatic and riparian ecosystem.

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