internal report **365**

Model predictions of sulfate import and export in the RP1 catchment: Revision of a 1997 estimate of temporal solute flux, incorporating additional data

leGras C & Klessa D

May 2001

supervising scientist

MODEL PREDICTIONS OF SULFATE IMPORT AND EXPORT IN THE RP1 CATCHMENT: revision of a 1997 estimate of temporal solute flux, incorporating additional data[1]

Christopher leGras and David Klessa

Abstract

Keywords: waste rock, tailings seepage, magnesium sulfate, temporal flux, Ranger, RP1.

A model is proposed to enable calculation of the temporal flux of magnesium sulfate in the catchment of Ranger Retention Pond 1. The main reason for developing this model is to assess the long-term potential for solute generation within the catchment, and subsequent release to the environment. Therefore, the time frame of interest is principally the rehabilitation and post-rehabilitation phases of mining. For this reason, operational releases of solutes to the catchment, for example by flood irrigation of wetland-modified Retention Pond 2 water, have not been considered in great detail.

The main medium to long term sources of magnesium sulfate within the catchment are:

- new placements of waste rock, particularly from the #3 orebody and,
- surface expression of seepage from the tailings dam (whether tailings are rehabilitated *in situ* or not).

To assess the rate of production of soluble erosion products from waste rock, the 'northern' waste rock dump (in the catchment of Retention Pond 4) was used as a case study. Calculations based on historical solute fluxes from this disused dump predict that soluble effluent will decline to low levels within about 20 years from last placement. Solute evolution was relative slow from this dump when operational, possibly because continual occlusion of previous rock placements impeded the initiation and propagation of weathering processes.

The volume of seepage from the tailings dam has been estimated using: the dimensions of the 'shallow' (that is, C_2-horizon) aquifer; the velocity of the seepage plume, as measured by progressive water quality in observation bores; and the porosity of the aquifer material. Contaminated groundwater enters the catchment principally by surface expression during the Wet Season. The volume reaching the surface has been estimated using discharge, water quality and isotope-composition data. Direct intersection at ground level has not been demonstrated for this aquifer. The model predicts that solute evolution from this source is likely to continue for several decades after tailings seepage is terminated, either by removal of the tails or by impermeable capping.

Modelling of seepage via the C_2-horizon aquifer using additional data since 1997 (leGras & Klessa 1997b) has returned substantially different findings in detail, particularly regarding aquifer flow velocity, and dispersion of the plume as water moves down gradient. However, the recalculated estimate of surface-expressed solute is not greatly different from the original value (attenuation being somewhat more rapid). As the seepage component of the overall model is relatively small, the effect on overall solute flux is calculated to be minimal.

---

[1] Revisions to leGras & Klessa 1997b (including diagrams) are printed in blue

Introduction

Background

Water entering the catchment of Ranger Retention Pond 1 (RP1) has free egress to the environment, via a spillway constructed in the pond wall. Since its construction RP1 has contained elevated concentrations of solutes, dominated by magnesium sulfate ($MgSO_4$). Until recently these solutes have come from two sources:

- degradation of waste rock that forms the external mantle of the tailings dam. This rock contains minor sulfide and magnesite mineralisation which forms $MgSO_4$ on oxidative erosion.
- surface expression of tailings dam seepage, which is also dominated by $MgSO_4$.

The first of these sources is rapidly declining in importance, as the sulfide content of the waste rock becomes exhausted. The second source will remain important for many years after rehabilitation, whether or not tailings are removed from the current tailings impoundment.

At present, some Retention Pond 2 (RP2) water is being ameliorated during the Dry Season by passage through a constructed wetland filter in the catchment of RP1. After removal of a substantial fraction of Mn and U values (see, for example, leGras and Klessa 1997a) the water is flood irrigated elsewhere in the RP1 catchment. The quantity of water disposed of in this way, and its quality, are functions of previous Wet Season conditions and operational requirements of the mine. In the years of operation of the Constructed Wetland Filter, approximately 100-300 tonnes of $MgSO_4$ have been irrigated to the RP1 catchment per year (leGras and Klessa 1997a and ERA internal data). However, although wetlands may be a feature of the post-rehabilitation landscape, the active disposal of large volumes of water with moderate solute concentrations will probably be restricted to the operational phase of mining. It is therefore considered in our temporal solute flux model mainly for comparison purposes.

ERA has designed its plans for waste rock placement with a view to final rehabilitation. This has resulted in progressive encroachment of material into the RP1 catchment. The rock is derived from Pit 3, and for the purposes of our model is assumed to be mineralogically equivalent to that from Pit 1. ERA intends to use substantial volumes of Pit 3 waste rock to backfill the Jabiluka void, should the latter project proceed (ERA 1996a). This eventuality would alter the volume of waste rock placed in the RP1 catchment. Nevertheless, new waste rock in the catchment will be the dominant source of solutes in the medium to long term, and its contribution has therefore received most attention for the work described in this report.

Modelling approach

The temporal flux of two sources of $MgSO_4$ in the RP1 catchment have been modelled:

- solutes derived from the oxidative degradation of sulfide minerals in new waste rock.
- solutes derived from surface expression of tailings seepage.

To estimate the potential solute load in waste rock, documentary sources were consulted to determine the sulfur content of fresh waste rock. Other sources were used to discover the history of waste rock in the dump in the catchment of RP4. This location was used as a case study for three reasons.

1. The dump is no longer used for new placements of waste rock, which simplifies the mathematical treatment of solute loss data.
2. A reasonable documentary record exists of placements in this catchment, and of the final stored mass.
3. A very good documentary record exists of solute fluxes in the RP4 catchment because of regulatory requirements.

For tailings dam seepage, the main components for the model are:

- The dimensions of the aquifer that conducts the seepage.
- The porosity of the aquifer.
- The flow velocity of seepage in the aquifer.
- The mass and proportion of aquifer solute load that surface expresses per year.

The first three components allow calculation of the annual solute load conducted by the aquifer. This is the load potentially available to the RP1 catchment. The fourth component permits the derivation of the actual load discharged. Calculation of the *proportion* of solute load discharged via surface expression allows an estimate of the dilution factor of the aquifer load to be made. This estimate is required to calculate the rate of diminution of solute expression should tailings seepage be terminated or greatly reduced. This situation may arise either because of impermeable capping, or relocation, of the tailings mass.

We note that most of the numerical values acquired from documentary sources and experimental measurements have moderate to large uncertainties. These compromise the robustness of the model to some extent. An indication of the approximate magnitude of the uncertainties is found in the degree to which much of the documentary evidence is either inconsistent or contradictory. In these cases judgement has been used to assess the relative integrity of the data. Therefore, the model predictions must be regarded as semiquantitative.

Modelling Results

The 'Northern' (RP4) Waste Rock Dump

According to ERA internal data (ERA 1996b), the northern waste rock dump occupies an area of 260 000 m^2 with a mean vertical height of 15 m. This gives a volume near 3 900 000 m^3 if the dump is assumed to be a rectangular prism. In fact, the dump more closely approximates a rectangular pyramid with sides sloping at about 36° (angle of repose) and the apex removed to 15 m from ground level. However, the calculated difference in volume is slight, and well with the uncertainties imposed by other data. The mean density of rock on the dump is stated to be 2.06 t/m^3 (ERA 1996b). This yields a total rock mass of about 8 000 000 tonnes in 1996.

Placement of waste rock in the northern dump was scheduled to cease in 1989 (ERA 1988) and this seems to have occurred. The same document quotes a rock inventory, excluding 'construction material' at June 1988 of 9 943 000 tonnes. Using these figures, the mass of rock stored in this dump when placement concluded should have been more than 10 000 000 tonnes. Other documentary sources yield presumptive inventories greater than this, but these evidently include 'construction material' and/or very low grade ore in their estimates. Negligible quantities of these latter two categories of material remain in the northern dump.

There is obvious difficulty in reconciling the 1988 and 1996 estimates of the mass of waste rock in the northern dump. Both purport to measure the same material, and the data sets seem to be internally consistent, as far as this can be assessed. The number of significant figures included in the estimates suggests that those who performed the calculations regarded them has having a reasonable degree of accuracy.

Some erosion losses have occured since the dump was decommissioned, but these are unlikely to have been more than about 5 000 tonnes (Evans and Loch 1996), which is an insignificant fraction of the difference. Apparently waste rock, in addition to construction material, is used for road building and maintenance, and possibly for miscellaneous purposes such as bunding. However, these uses cannot plausibly account for the discrepancy (for example, a road or bund 10 metres high by 10 metres wide by 2 000 metres long would use only about 400 000 tonnes of rock). Most likely the estimates contain fairly large unaccounted uncertainties. We have used the average of the 1988 and 1996 values for our model (9 000 000 tonnes).

We next addressed the question of the sulfur content of waste rock. A small number of studies have reported determinations of this element in rocks of the Alligator Rivers Region (Ferguson and Winer 1980; Frishmann, Nutt and Grauch 1986; and Milnes and Fazey 1988). The former documents record values of sulfur in a wide range of rocks, though with some emphasis on ore samples from the Jabiluka and Ranger deposits. The mean sulfur contents of the rocks determined were 0.12% and 0.11% respectively. These values, though in close agreement, are not very relevant to the present study, because waste rock contains much less sulfur on average than those examined by these workers. Milnes and Fazey measured the sulfur contents of 36 samples of Ranger waste rock, and obtained a mean value of 0.006% with a standard deviation of 0.007%. Although specific rock types and their sulfur contents were identified in this report, the relative abundance of each type was not estimated. However, this uncertainty is unlikely to result in gross error because of the approximate similarity of sulfur contents between rock types. The mean value equates to 60 mg S/kg of rock, which would produce about 0.18 kg/tonne of SO_4^{2-} after oxidative degradation of the rock. Therefore, the waste rock in the northern dump should have initially contained the potential to produce about 1 650 tonnes of SO_4^{2-}. We assume that all sulfur is initially present as sulfide and that this is quantitatively converted to SO_4^{2-} on oxidation. We account for solute fluxes in terms of SO_4^{2-} rather than $MgSO_4$. This is mainly a matter of convenience, and is approximate because a strict stoichiometric equivalence does not exist between Mg and SO_4^{2-} (some Ca being present). The total mass of solutes can be approximated by multiplying the value for SO_4^{2-} by 1.3.

The northern waste rock dump was designed so that runoff from the site was directed to RP4. For regulatory reasons, stored volume and water quality parameters are measured frequently in RP4. Where water release is permitted, discharge volumes and flow-weighted concentrations are determined. These procedures result in a detailed and reasonably accurate database from which solute mass flux can be calculated. These are presented in Table 1#.

Season	Final mass SO_4^{2-} in RP4 tonnes	Mass released from RP4 tonnes	Net SO_4^{2-} inflow to RP4 tonnes	Cumulative solute loss %
1982-83	1.0	0	1.0	**0**
1983-84	4.0	2.0	5.0	**0**
1984-85	4.0	4.3	4.3	**1**

Season	Final mass SO_4^{2-} in RP4	Mass released from RP4	Net SO_4^{2-} inflow to RP4	Cumulative solute loss
	tonnes	tonnes	tonnes	%
1985-86	7.2	4.2	7.4	**1**
1986-87	7.4	34	34	**3**
1987-88	15	0	7.6	**4**
1988-89	46	22	53	**7**
1989-90	50	3.3	7.3	**7**
1990-91	49	18	17	**8**
1991-92	110	14	65	**12**
1992-93	150	34	74	**17**
1993-94	56	240	150	**26**
1994-95	35	220	200	**38**
1995-96	20	150	130	**46**
1996-97	17	120	120	**53**
1997-98	44	38	66	**57**
1998-99	18	140	120	**64**

Table 1. Sulfate flux in the RP4 catchment

#: Derived from ERA Environmental Annual Reports (and equivalents) and the Ranger minewater Access database

The total net inflow of SO_4^{2-} from 1982 to 1999 was approximately 1070 tonnes, which accounts for about 64% solute mass predicted to be potentially available. Sulfur-isotope measurements of RP4 water (leGras, unpublished data) suggest that some sulfur may be fixed as sulfide in the sediments of RP4, as a result of bacterial reduction of SO_4^{2-} . However, this does not appear to be cumulative as extensive reoxidation apparently takes place with the onset of each Wet Season. The contribution of sediment-fixed sulfur to the overall budget is therefore likely to be small.

Three observations can be made from these data:

1. In the active phase of the dump, that is, while new rock was being placed there, solute production was relatively low, even allowing for the smaller total rock mass in early years. In the period until the end of the 1989-90 Wet Season, approximately 120 tonnes of SO_4^{2-} were released, about 7% of that predicted to be available. This may be at least partly due to continual placement of new rock over old before significant degradation of the latter had occurred. Covering waste rock evidently slows down erosion of the material and concomitant solute evolution, presumably because of restricted access of water and oxygen.
2. A surge in solute evolution is observed, commencing 2-3 years after cessation of new rock placement. Maximum solute evolution occurred in the Wet Season 5-6 years after decommissioning. This period apparently corresponds to an induction phase whereby oxidative erosion processes are initiated and then propagate.
3. A slow decline in solute evolution is observed thereafter. This probably reflects progressive exhaustion of solutes as surface rock weathers completely, to be replaced by much slower degradation of deeper material.

We attempted to model the decline in solute evolution observed since 1994-95 by assuming that these data represented a sustainable exponential decrease in available solute. The results are presented in Figure 1, where experimentally derived, and calculated, data points are overlain by the curve of best fit. Year 0 corresponds to 1982-83, and year 40 to 2023-24. Figure 1 includes data for 1997-98 and 1998-99, which were not available for the original report (leGras & Klessa 1997a). The two additional points fit adequately on the original curve, though the 1998-99 inflow is larger than expected. This may be due to that Wet Season having the highest precipitation (1891 mm) of the period under review. This could have led to abnormal mobilisation of solutes, including flushing of material usually entrained within the rock mass during less severe seasonal conditions.

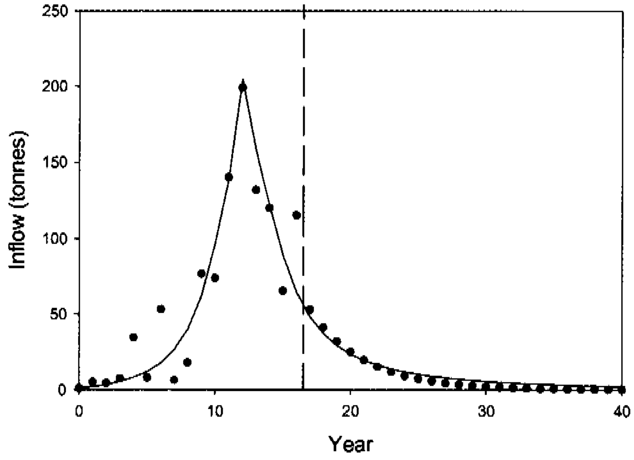

Figure 1. Actual (years 0-16) and predicted (years 17-40) solute loss from the RP4 WRD

Figure 2 shows the same data presented as a cumulative percentage of the SO_4^{2-} potentially available for release. This was calculated to be 1 650 tonnes, as described above.

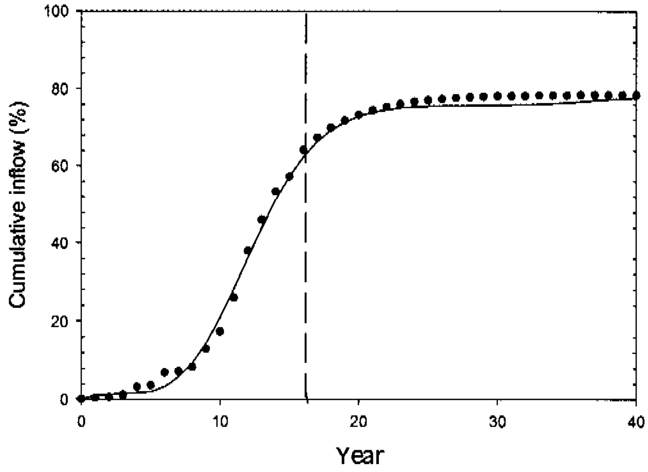

Figure 2. Actual (years 0-16) and predicted (years 17-40) cumulative loss of SO_4^{2-} from the RP4 WRD

It is evident from Figure 2 that, if the present trend for loss of solutes from waste rock in the catchment of RP4 continues, only about 80% of the sulfur presumed to be originally present will be converted to SO_4^{2-}. There are at least five reasons why this may be the case. The first two relate to possible inadequacies in the model, while the last three provide possible explanations where the model is assumed to be substantially correct.

- *The mass of rock in the dump and/or its average sulfur contents may have been overestimated.*

This explanation cannot be excluded. However, we cannot assess the likelihood of this possibility because we have used the best available information for our calculations.

- *The projected rate of loss of sulfur from the waste rock may not be correct.*

The rate constant that specifies the exponential loss of solute from waste rock, shown in Figure 1, was derived from only three data points. Significant uncertainty must therefore attach to extrapolated mass fluxes that are calculated using this number. Monitoring of solute evolution from the catchment of RP4 should continue until a more confident estimate can be made of the rate of solute production. This will require data for at least a further three years.

- *Some sulfur may be retained in the sediments of RP4 as sulfide minerals, as a result of bacterial reduction of SO_4^{2-}.*

Our measurements of the sulfur-isotope composition of RP4 water (leGras, unpublished data) show a seasonal variation in $\delta^{34}S$ of about 2‰, with the highest values being observed at the end of the Dry Season, together with a diminution of the ratio $[SO_4^{2-}]/([Mg]+[Ca])$. These observations are consistent with the bacterial reduction of about 10% of the ponded SO_4^{2-} (leGras *et al* 1991). However, the evidence also suggests that the phenomenon is cyclic, with sediment-fixed sulfides oxidising at the onset of the following Wet Season, when turbulent, well-oxygenated water enters the pond. Although there may be a slow accumulation of sulfides in the sediments of RP4, this explanation can only explain a small fraction of the projected SO_4^{2-} deficiency.

- *Dissolved SO_4^{2-} may have percolated downwards and be pooled beneath the rock mass.*

Richards (1987) has proposed that soil loading by waste rock may cause a depression under the dump where solute-laden water may accumulate. According to his calculations, the maximum depth of a depression may be 2-3 metres. This would translate to a rock-filled void of about 500 000 m^3. If the porosity of this mass were 30%, about 70 tonnes of SO_4^{2-} could be dissolved in the interstitial water. This water would presumably be exchanged on a seasonal basis, and would provide a gradually attenuating pool of solute as the supply of $MgSO_4$ entering the interstitial water progressively diminished. Because there is no direct evidence for the existence of a depression under the waste rock dump, the applicability of this explanation cannot be evaluated.

- *The discrepancy may reflect the relative inaccessibility of some waste rock to weathering processes.*

Garvie and Gibson (1994) have studied oxidative degradation of a waste-rock dump at Mt Lyell, Tasmania. They conclude that SO_4^{2-} release may proceed by two mechanisms: the rapid oxidation of sulfur-enriched ‘pods’ within the dump, which should be complete on a time scale of 20-60 years; and a slower process whereby low-sulfur bulk material weathers at a rate controlled primarily by oxygen diffusion. This latter process may proceed over several centuries. If a similar process occurred in the RP4 waste rock dump, the projected SO_4^{2-} deficiency may eventually be released, but at a rate up to 100 times slower than that observed at present.

Seepage from the tailings dam

The amount of tailings seepage reaching the surface over time has been modelled using three parameters.

1. *The total solute (that is, SO_4^{2-}) flux in the catchment of RP1.* This is calculated from the annual release volume from RP1, the volume of standing water in the pond and water quality data. All these values are known with a moderate to high degree of confidence.

2. *The mass of solute passing Upper Coonjimba Creek near OB44.* This was estimated from creek discharge and water quality. This calculated value is identified with the total mass of solute attributable to waste rock (dam wall) runoff and tailings seepage. The 'near OB44' figure was then compared with total solute flux for the catchment, adjusted for flood irrigation contributions from the RP1 constructed wetland filter. The proportion of 'near OB44' solute attributable to tailings seepage alone was estimated using the sulfur-isotope composition of the surface water, together with a knowledge of the isotope signatures of the contributing sources (leGras *et al* 1993). Isotope data are available for only one year, and hence the confidence with which this parameter can be evaluated is correspondingly less.

3. *The volume of water held by the 'shallow' (C_2-horizon) aquifer, and from observation bore water quality data, the mass of solute.* Using an estimate of annual solute release from groundwater through surface expression, combined with aquifer capacity parameters, the rate at which reserves of solute in the aquifer are depleted by displacement once seepage stops can be calculated. It must be noted that the fate of the seepage plume downgradient from OB44 is unknown. It evidently does not discharge to RP1 directly, because its measured rate of progress should have ensured that observation bores on the margins of RP1 currently show degraded water quality. The most recent data shows that this is not the case. This accords with the view of ERA (R. Salama, private communication) that a subterranean ridge exists between OB44 and RP1 which renders the C_2-horizon aquifer discontinuous at that point. Whether affected groundwater surface expresses there, pools, or is diverted is not known.

The catchment of RP1 is approximately 236 hectares, excluding the area of the Constructed Wetland Filter (ERA 1995a). From the net holding capacity of RP1, evaporation data, and the spillover volume from the pond (ERA 1996c), the total influent volume to RP1 for the 1995-96 Wet Season can be calculated. This totalled approximately 1 400 000 m^3. An additional 1 700 000 m^3 was retained by the catchment as infiltration, and lost by evapotranspiration, as calculated using rainfall data (ERA 1996c). The mean [SO_4^{2-}] in RP1 water during 1995-96 was approximately 40 mg/L, giving a SO_4^{2-} inflow of about 60 tonnes.

The volume of rainfall falling in the RP1 catchment upstream of OB44 (~70 hectares) was about 940 000 m^3. Assuming that infiltration and evapotranspiration were similar for this sub-catchment as for the catchment as a whole, the total discharge of Coonjimba Creek near OB44 for 1995-96 was about 380 000 m^3. The mean [SO_4^{2-}] was about 80 mg/L (leGras, unpublished data), although this value is only roughly flow-weighted because of the absence of accurate temporal discharge data at this point. This gives a total mass contribution of about 30 tonnes for SO_4^{2-} passing this point. The discrepancy between this value and the estimate of total SO_4^{2-} inflow to RP1 (60 tonnes) can be reconciled by considering the mass of SO_4^{2-} applied to the RP1 catchment from the RP1 Constructed Wetland Filter during the

1995 Dry Season. Approximately 284 000 m^3 of water, with a mean [SO_4^{2-}] of 500 mg/L was applied (ERA 1995b), yielding an irrigated mass of 140 tonnes. The movement of about 20% of this load to RP1 during the subsequent Wet Season would account for the calculated shortfall. There is therefore no need to invoke a large component of SO_4^{2-} ingress to the RP1 catchment via Coonjimba Creek downstream from OB44 (for example by surface expression of seepage water). Conversely, there is no evidence that this does not occur to some extent.

It is evident that SO_4^{2-} is not quantitatively transported from the sites of flood irrigation to RP1 in the Wet Season immediately following. This suggests that SO_4^{2-} will accumulate in the catchment soils during the years of application. However, if 20% of the residual is lost from the catchment each year, the annual yield will decline to about 10% of the original value in 10 years, once further application ceases.

Most of the SO_4^{2-} in discharge passing OB44 derives from oxidation of sulfide minerals in the construction rock used on the outer surface of the tailings dam. However, a proportion originates from surface expression of tailings dam seepage, with the C_2-horizon aquifer the established conduit for seepage water (leGras et al 1993). The sulfur-isotope composition of the SO_4^{2-} in Coonjimba Creek can be used to determine the proportion from each source, because the alternatives have very different sulfur isotope ratios ($\delta^{34}S \sim 5‰$ for waste-rock derived SO_4^{2-} and ~25‰ for observation bore water, with a measurement precision of about 0.2‰ (leGras et al 1991 and 1993)). Figure 3 depicts an almost uninterrupted increase in the values of $\delta^{34}S$ determined during the 1995-96 Wet Season. This trend strongly suggests that runoff sources of SO_4^{2-} are progressively supplanted by groundwater sources (identified with tailings seepage) as the season proceeds. From the measured [SO_4^{2-}] associated with these $\delta^{34}S$ values, a putative contribution of about 30% (10 tonnes) from tailings sources is calculated.

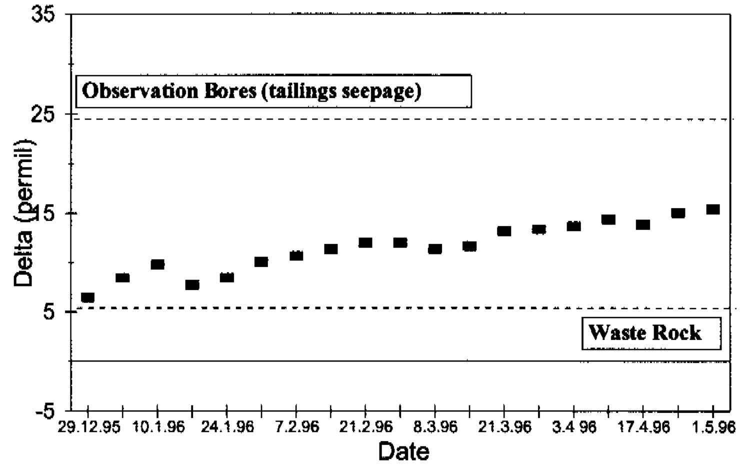

Figure 3. Progressive increase in tailings contribution to runoff in the RP1 catchment during 1995-96

The hydraulic properties of the C_2-horizon aquifer

The C_2-horizon aquifer is semi-confined, and located in slightly to moderately weathered metamorphic rock, mostly of gneiss grade. It forms part of the Cahill formation, with mineralogy dominated by quartz, biotite, 'white mica' and feldspar (Coffey and Partners Pty Ltd, 1982). The aquifer is confined at depth (below ~20 metres) by unweathered basement rocks, and semi-confined above ~10 metres by relatively impermeable highly weathered rock. Coffey and Partners remark that the zone of high (> 10^{-4} cm/sec) permeability is restricted to the vicinity of Coonjimba Creek, but its lateral extent has not been unequivocally established. There is some diagrammatic suggestion (Figure A4 in Coffey) that the zone of high transmissivity may be about 300 metres wide, centred on Coonjimba Creek. This estimate is consistent with the observation of slightly elevated solute concentrations in OB24, which is about 400 metres west from the creek. OB13A and OB16 have a lateral separation of about 150 metres, and both are greatly plume affected. This provides a minimum lateral extent for the plume-affected aquifer, and hence the zone of increased permeability. The distance from the dam wall to RP1 is about 800 metres, and as remarked above, the aquifer is apparently interrupted between OB44 (~500 metres from the dam wall) and RP1.

For the purposes of this report, we have assumed that the dimensions of the C_2-horizon aquifer are 650 metres long, 300 metres wide and 10 metres deep. We have also assumed a mean porosity of 0.17 (the *Zone 3* of Lowson and Jeffrey, 1988). This yields a saturation capacity for the C_2-horizon aquifer of about 330 000 m^3.

The temporal [SO_4^{2-}] profiles for three representative observation bores that intersect the C_2-horizon aquifer, OB16, OB15 and OB44 are presented in Figures 4-6. In each case the data can be fitted satisfactorily to a sigmoidal curve, which is superimposed on the data points. The distances of these bores from the tailings dam wall are respectively 70, 340 and 500 metres.

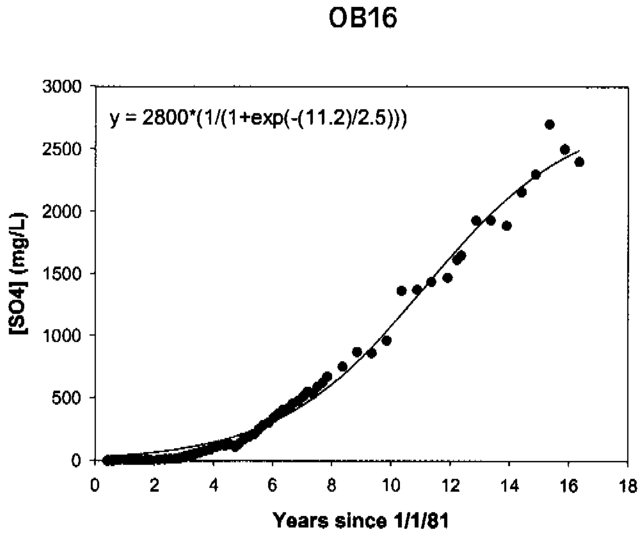

Figure 4. Concentration profile of SO_4^{2-} in OB16

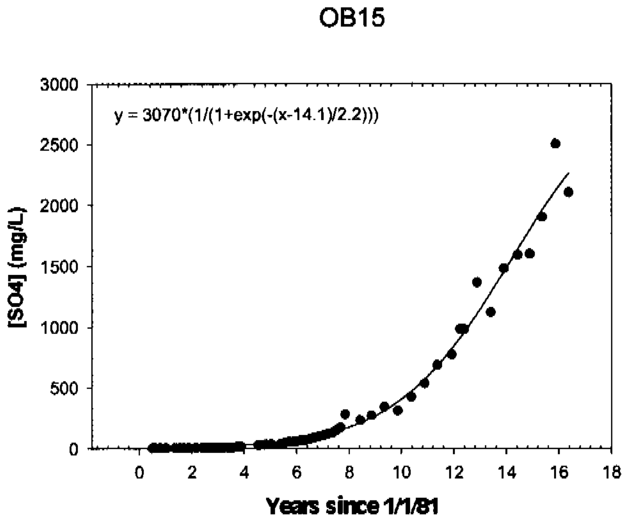

Figure 5. Concentration profile of SO_4^{2-} in OB15

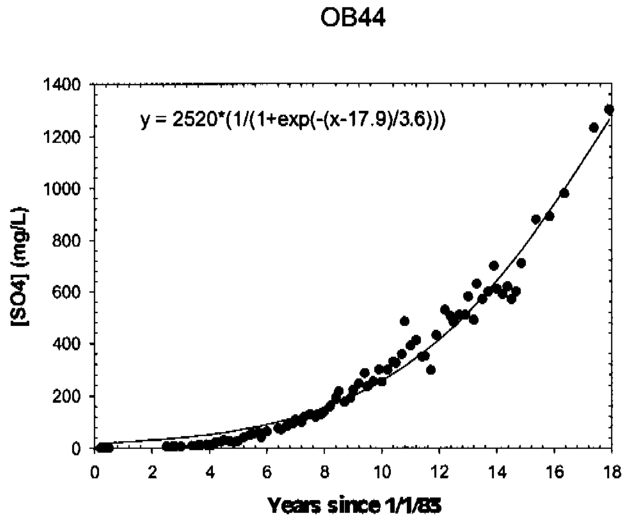

Figure 6. Concentration profile of SO_4^{2-} in OB44

It is evident from these data that, for the bores more distant from the dam wall, not only is the arrival of elevated solute concentrations delayed, but the extrapolated final concentration of solutes is attenuated, presumably by dilution with unaffected groundwater. The data are summarised in Table 2.

Bore	Distance from tailings dam (metres)	Time to arrival of plume# (years)	Final $[SO_4^{2-}]$§ (mg/L)
OB16	70	11	2800
OB15	340	14	3100
OB44	500	17	2500

#: assessed as the inflection point of the breakthrough curve from a base date of 1 January 1981

§: extrapolated from the curve of best fit

Table 2. Hydrological data for seepage plume in the C_2-horizon aquifer

These data imply a permeability between OB16 and OB15 of about **90** m/y (**2.8** x 10^{-4} cm/s), and between OB15 and OB44 of **50** m/y (**1.6** x 10^{-4} cm/s). These figures are in good agreement with the permeability estimates made by Coffey and Partners Pty Ltd (1982) from bore-pumping experiments. However, the mean permeability between the dam wall and OB16 is much smaller, presumably because of the far less permeable lining of the tailings dam, and the lower hydraulic head in earlier years.

The values reported in Table 2 for plume velocity and ultimate $[SO_4^{2-}]$ are quite different from the estimates made for the 1997 report, especially regarding OB44. It is now evident that solute attenuation is much smaller between OB15 and OB44 than previously calculated. This implies that diluted tailings dam seepage is minimally supplemented by unaffected groundwater between these bores. Indeed, it is likely that the attenuation observed can be completely accounted by loss of solute from surface expression. There is no evidence for a widening or deepening of the aquifer between OB15 and OB44 that could account for plume attenuation. The aquifer geometry proposed in 1997 is retained as a model for aquifer discharge. However, we have revised the flow rate in line with the new estimates:

$$\mathbf{150\ m}\ \text{(width)} \times \mathbf{10\ m}\ \text{(depth)} \times \mathbf{90\ m/y}\ \text{(flow rate)} \times \mathbf{0.17}\ \text{(porosity)} = \mathbf{23\,000\ m^3/y}$$

A revised minimum estimate for SO_4^{2-} flux past OB15 is ~60 tonnes/year, based on a new estimate for $[SO_4^{2-}]$ of 2 500 mg/L. This compares with an estimate for surface expression upstream from OB44 during 1995-96 of about 10 tonnes.

We propose a model for the aquifer, after equilibrium solute concentration has been achieved, as approximating a rectangular prism with dimensions:

$$\mathbf{650\ m}\ \text{(length)} \times \mathbf{300\ m}\ \text{(width)} \times \mathbf{10\ m}\ \text{(depth)} = \mathbf{2\,000\,000\ m^3}$$

With a porosity of 0.17 and a mean $[SO_4^{2-}]$ of 2 500 mg/L, the final presumptive aquifer load of SO_4^{2-} is ~830 tonnes. As stated previously, the fate of this solute is unknown once it meets the discontinuity before RP1. Presumably it must either be forced to the surface or be diverted elsewhere.

Modelling conclusions

Evolution of solutes from waste rock

In modelling the predicted loss from future placements of waste rock, we have made the following assumptions.

- Waste rock from Pit 3 has the same composition and mineral distribution as that from Pit 1.
- Rock is placed in such a way that solute loss is small until the dumping site is decommissioned.
- Hydrological and infiltration properties of the stockpiled rock are similar to that occurring at the dump in the RP4 catchment.

In this way we developed the temporal model summarised in Figure 7. It predicts a rapid increase in solute evolution after the cessation of rock placement, peaking in year six. Thereafter, the mass of solute produced declines rapidly, yielding very small quantities after about 20 years. These model conclusions are not altered by the incorporation of an additional two years data for RP4 catchment flux. As RP4 is now decommissioned, no further data will be available.

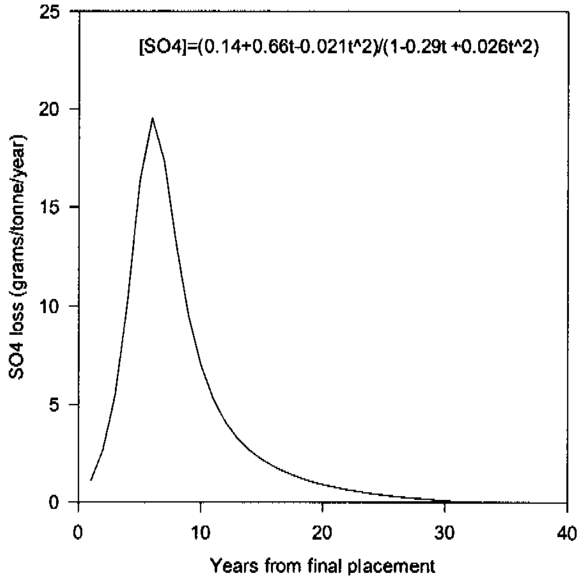

Figure 7. Predicted solute evolution (in grams SO_4^{2-} /tonne of rock/year) from stockpiled waste rock

The total mass of evolved SO_4^{2-} is calculated from the amount of rock deposited. For example, for a total mass of 20 000 000 tonnes, the maximum SO_4^{2-} flux is predicted to be ~400 tonnes during year six after final placement. For a Wet Season with rainfall of 1 600 mm (inflow of ~2 000 000 m^3 to RP1), the resulting contribution to the concentration of SO_4^{2-} in RP1 would be ~200 mg/L.

Evolution of solutes from seepage expression

We have modelled a scenario where additional seepage from the tailings dam is stopped, either by removal of the tailings or by impermeable capping. The aquifer will at that point contain ~830 tonnes of SO_4^{2-}. We assume that the only mechanism for removal of seepage solutes from the C_2-horizon aquifer is by surface expression, at a rate of 20 tonnes/year SO_4^{2-}. This mass is arrived at by factoring in an increased surface-expression contribution as $[SO_4^{2-}]$ approaches steady state, compared with measured $[SO_4^{2-}]$ in 1995-96 when an estimate of surface expression was made. It also accounts for the probability that some surface expression occurs downstream from OB44. A refinement to the 1997 model involves the assumption that the aquifer flow rate declines from 50 m/year at OB44 to 20 m/year, thus providing an additional loss mechanism from the C_2-horizon aquifer apart from surface expression. This is assumed to approximate the effect of diminution in hydrostatic head once the tailings dam is emptied, based on the calculated steady-state ratio of seepage to unaffected groundwater (which would continue to exert hydrostatic pressure).. The loss from the catchment, net of surface expression between the tailings dam and RP1, is therefore estimated initially at 13 tonnes/year. The results, as tonnes of SO_4^{2-} released to the catchment of RP1 annually, are shown in Figure 8. The equation describing this relationship is:

$$SO_4^{2-}{}_{\text{Expressed}} = e^{(3.0-0.04t)}*0.6$$

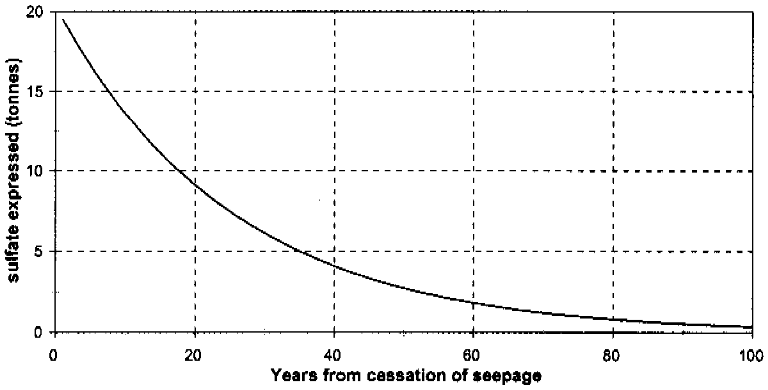

Figure 8. Contribution of tailings-sourced SO_4^{2-} to the RP1 catchment after cessation of seepage

This modelling outcome is similar to that previously described (leGras & Klessa 1997). Attenuation of SO_4^{2-} is slightly more rapid under the revised model, but this does not alter the overall model projections in a material way.

Evolution of solutes from flood irrigation

Solute contribution from flood irrigation of wetland-ameliorated RP2 water is likely to be a transient phenomenon. There is evidence from mass-balance calculations that about 20% of the SO_4^{2-} mass irrigated is lost from the catchment each year. This yields the diminution curve presented in Figure 9, once irrigation ceased. The figure assumes an ultimate total SO_4^{2-} load of 1 000 tonnes from this source. The equation describing the annual loss of solute is:

$$SO_4^{2-}{}_{Removed} = e^{(5.4-0.2t)}$$

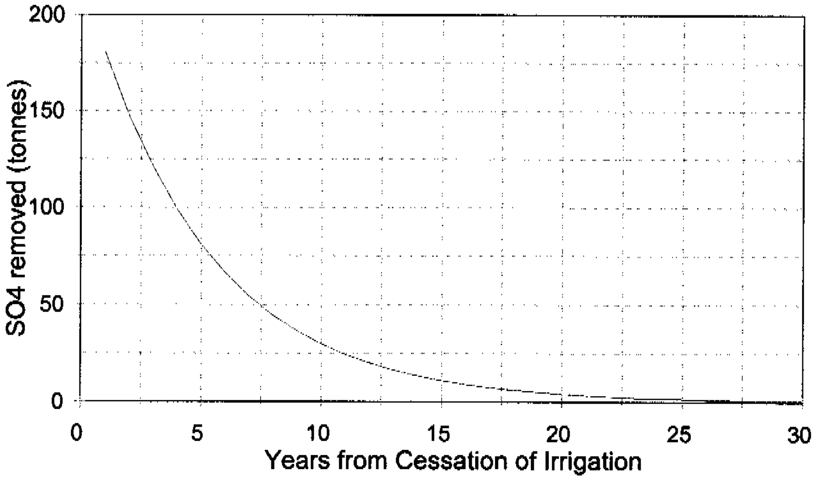

Figure 9. Contribution of flood-irrigation sourced SO_4^{2-} to the RP1 catchment

Total evolution of solutes from all sources

We have combined the modelled SO_4^{2-} contributions from each of the sources and derived the projected SO_4^{2-} flux for the hundred years (Figure 11) following cessation of the activities giving rise to solute evolution. The mass of waste rock deposited in the RP1 catchment is assumed to be 20 000 000 tonnes, yielding a peak SO_4^{2-} mass of 400 tonnes/year and an average $[SO_4^{2-}]$ for that year of 200 mg/L. For simplicity we assume that the relevant operations stop at the same time. The mean concentration of SO_4^{2-} in RP1 is calculated assuming Wet Season rainfall of 1 600 mm, which, based on observed runoff volumes from the RP1 catchment since 1982-83 (Figure 10, taken from leGras, unpublished data) should yield ~2 000 000 m³.

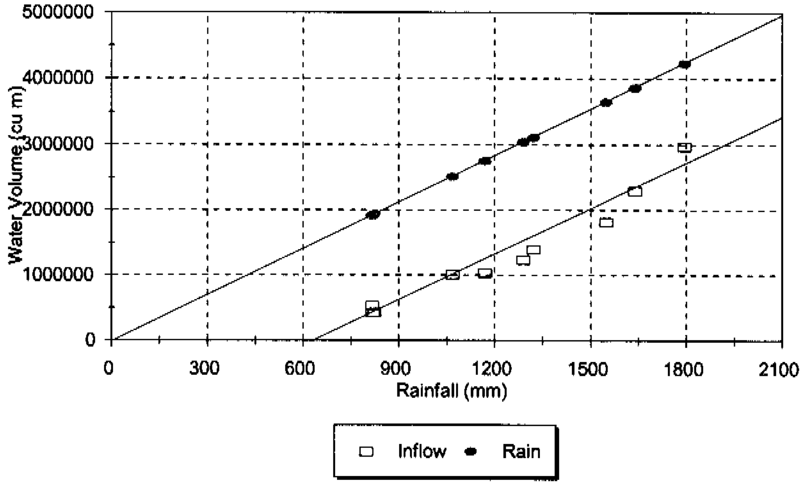

Figure 10. Rainfall volume and pond inflow in the RP1 catchment, 1982-1997

The equation of best fit that describes the total annual solute flux in the RP1 catchment is:

$$SO_4^{2-}{}_{\mathrm{Removed}} = (131-2.8\times t+0.096\times t^2)/(1-0.24\times t+0.020\times t^2)$$

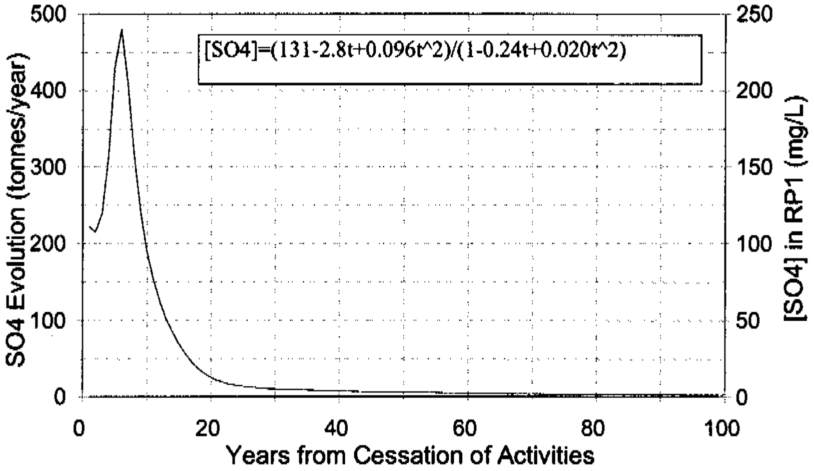

Figure 11. Combined contributions of SO_4^{2-} to the RP1 catchment

Concluding remarks

Solute flux in the catchment of RP1, dominated by $MgSO_4$, originates primarily from three sources:

- oxidation of sulfide minerals in waste rock;
- seepage from the tailings dam that is subsequently surface expressed;
- mobilisation solutes originally present in wetland effluent flood irrigated in the catchment.

The first and third or these sources will produce the greatest mass of solute, but are likely to be less persistent than the second, in spite of new calculations that suggest that the attenuation of surface-expressed seepage is likely to be more rapid than previously estimated. Solutes derived from flood irrigation should be quantitatively removed from the catchment a few years after application stops. There is a strong possibility that this would be before the commencement of rehabilitation, or at least in the early stages. We therefore believe it unlikely that solutes from this source will make a meaningful contribution to solute flux in the post-rehabilitation phase of mining. The fate of seepage-derived solute cannot be predicted with a high degree of accuracy because of uncertainty about the location of the aquifer containing it downstream from OB44. However, this source provides the smallest contribution to overall flux, so the predictive model is not very sensitive to the inherent imprecision in estimating its value.

A temporal model for solute evolution is proposed for each of the three sources, based on mass balance observations to date. One significant finding , in the case of waste-rock degradation, is the apparent existence of an 'initiation' period. Release of solute is relatively minor while additional rock is placed in dumps. We attribute this to continual covering of existing rock with new placements, effectively retarding the weathering process. Maximum solute release occurs about six years after final decommissioning of the dump. The models for each source of solute were combined, based on the assumptions described in the text, to give an overall temporal flux model for the RP1 catchment for one hundred years after operational activities have ceased.

References

Coffey and Partners Pty Ltd 1982. *Groundwater investigation and monitoring system, Ranger on-site area.* RAN 138.

ERA 1988. *Ranger Uranium Mines Pty Ltd five year rehabilitation plan, August 1988.* RAN 183.

ERA 1995a. *Ranger mine water management system operation manual, revision 17.* RAN 46.

ERA 1995b. *Wetland filter performance report 1995.* RAN 300.

ERA 1996a. *The Jabiluka project: draft environmental impact statement*, Kinhill Engineers Pty Ltd and ERA Environmental Services, p 4-54.

ERA 1996b. *Ranger Uranium Mines Pty Ltd five year rehabilitation plan, August 1996.*

ERA 1996c. ERA-Ranger mine Environmental Annual Report. RAN 22.

Evans KG & Loch RJ 1996. Using the RUSLE to identify factors controlling erosion rates of mine soils. *Land Degradation & Development* 7, 267-277.

Ferguson J & Winer P 1980. *Pine Creek Geosyncline: statistical treatment of whole rock chemical data*, in J Ferguson and AB Goleby (eds), Uranium in the Pine Creek Geosyncline, IAEA, Vienna, pp 191-208.

Frishmann D, Nutt CJ & Grauch RI 1986. *Geochemical data for unaltered rocks and uranium ores from the Cahill Formation, Northern Territory, Australia.* US Geological Survey Open File Report 86-454.

Garvie AM & Gibson DK 1994. *Monitoring hole installation and quantification of oxidation in a waste rock dump at the Mt Lyell mine.* Report to Renison Goldfields Consolidated Ltd by ANSTO Environmental Science Program.

leGras C, Akber RA & Andrew A 1991. *The sulphur-isotope composition of mineral, surface- and groundwater samples from the Ranger Uranium Mine, Northern Territory.* Supervising Scientist Internal Report 44.

leGras C, Akber RA & Andrew A 1993. *The sulfur-isotope composition of pore water, seepage and infiltration samples from the tailings dam, Ranger Uranium Mine, Northern Territory.* Supervising Scientist Internal Report 126.

leGras C & Klessa D 1997a. *An assessment of the performance of the Ranger RP1 constructed wetland filter during releases in 1995 and 1996.* Supervising Scientist Internal Report 255.

leGras C & Klessa D 1997b. *An estimate of temporal solute flux in the catchment of Ranger Retention Pond 1.* Supervising Scientist Internal Report 267.

Lowson RT & Jeffrey J 1988. *The attenuation of solutes (chemical and radioactive species) in tailings water by adsorption on soil and rock materials down gradient of the Ranger tailings dam.* Report to ERA by ANSTO Environmental Science Program. RAN 234.

Milnes AR & Fazey PG 1988. *Acid leaching from ore stockpiles and waste dumps in the Ranger project area, East Jabiru.* RUM Technical Paper 2. RAN 197 #2.

Richards BG (1987). *Settlement under rock dumps, focus report #1 to Ranger Uranium Mines.* RAN 196.