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TABLE OF CONTENTS

Page

KEYNOTE PRESENTATIONS

Detailed Characterisation of the Waste Rock Dumps at the Kennecott Utah Copper Bingham Canyon Mine – Optionality for Closure ARD	<i>P.L. Brown, M.J. Logsdon, B. Vinton, I. Schofield, and K. Payne.....</i>	1
INAP Updates in a Challenging Environment and Outcomes of the 9th International Congress on Acid Rock Drainage 2012, Ottawa, Canada	<i>T. Chatwin.....</i>	13
Savage River Rehabilitation Project – 2002 to 2013	<i>D.J. Williams, G.W. Wilson, P.A. Scott, B. Hutcheson, S. Kent, and A. Hughes.....</i>	21

FROM BOARDROOMS TO MINE SITES – THE BUSINESS CASE FOR IMPROVED AMD MANAGEMENT

Commercial Implementation of Mine Closure Research Bringing Tools to Practice for Management Decisions	<i>M. O’Kane, G. McKenna, and C. Qualizza.....</i>	37
Leading Practice Solutions for Acid Rock Drainage Prevention and Control: A Key To Achieving a Sustainable Future for Mineral Resource Development	<i>S. Miller.....</i>	51

FUTURE IMPLICATIONS FOR AMD MANAGEMENT IN AUSTRALASIA

Management of Pyrrhotite Tailings: A Ten Year Case Study on Continuous Learning and Improvement	<i>A.M. Robertson, N. Kawashima, R.St C. Smart, and R. Schumann.....</i>	67
The Mt Gordon Challenge: Impacts of Historic Legacies, Highly Variable Climatic Conditions and Tough Economic Conditions on Management of Acid Mine Drainage	<i>K. Sell and D. Allen.....</i>	79
Beyond the PAF Cell	<i>S.R. Pearce.....</i>	97
Overall Acid Production Rates in Mine Wastes	<i>A. Garvie, C. Linklater, J. Chapman, and A. Watson.....</i>	111
The Rum Jungle National Partnership Agreement	<i>T. Laurencont and M. Rider.....</i>	125

Rehabilitation Planning at the Former Rum Jungle Mine Site, Northern Australia	<i>P. Ferguson, C. Wels, and T. Laurencont.....</i>	131
The Influence of Biotite and Phlogopite Dissolution on AMD Neutralisation at Brukunga Pyrite Mine, South Australia	<i>F. Ali, J. Webb, and J. Taylor.....</i>	141
What do Elevated Background Contaminant Concentrations Mean for AMD Risk Assessment and Management in Western Australia?	<i>C.D. McCullough and J.I. Pearce.....</i>	147
Field and Desktop Waste Rock Classification Guide for a Metalliferous Mine in the Northern Territory, Australia	<i>M. Landers, B. Usher, D. Faulkner, P. Marinelli, and K. Masterman.....</i>	159

A GLOBAL VIEW ON ACID AND METALLIFEROUS DRAINAGE

Designer Tailings: Improving the Management of Tailings through Collaborative Research	<i>M. Edraki, T. Huynh, T. Baumgartl, L. Huang, M. Andrusiewicz, K. Tungpalan, M. Tayebi-Khorami, E. Wightman, S. Palaniandy, E. Manlapig, C. Evans, S. Farrokhpay, D. Bradshaw, and S. Vink.....</i>	173
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MANAGING AMD – WHAT’S WORKING, WHAT’S NOT, AND WHAT COULD BE

Cover Systems.... Are We There Yet?	<i>M. O’Kane.....</i>	183
-------------------------------------	-----------------------	-----

WASTE ROCK LANDFORMS AND COVER DESIGN

Characterisation of 26 Year Old Waste Rock Stockpiles at The Detour Lake Project	<i>A. Cash, G.W. Wilson, D.W. Blowes, R.T. Amos, J. Robertson, and M.H. Turgeon.....</i>	197
Landform Design for Geita Mine: Comparison of Siberia and Caesar-Lisflood	<i>B. Dobchuk, I. Taylor, G. Hancock, Cooper, H., T. Coulthard, and R. Stephen.....</i>	211
Defining Effective Closure and Reclamation Measures for Tailings and Waste Rock Storages: An African Case Study	<i>P.A. Scott, R. Stephen, B. Dobchuk, M. Phillip, E. Mihayo, and I. Taylor.....</i>	225
Evaluation of Cover Systems Utilising Geosynthetic Layers Constructed in a Seasonally Humid Geographic Location for the Closure of Coal Waste Rock Piles	<i>G.P. Meiers, M. O’Kane, D. Mayich, and M. Barteaux.....</i>	239
Alkalinity Producing Covers for Minimisation of Acid Mine Drainage Generation in Waste Rock Dumps	<i>W. Olds, P. Weber, and M. Pizey.....</i>	253
Alkalinity Addition for Pyrite Passivation of Waste Rocks from Mt Lyell, Western Tasmania	<i>M.G. Sephton, J.A. Webb, and G. Cordery.....</i>	263

Performance of a Cover on a Bulk Sample Tailings Dam at Century Mine	<i>P.L. Defferrard, T.K. Rohde, and B.J. Milsom</i>269
---	--

EFFECTIVE SAMPLING AND TESTING – IN PREDICTING DRAINAGE CHEMISTRY

Using Kinetic Geochemical Testwork to Assist with Mine Planning, Operations and Post Closure	<i>B. Davis, N. Bourgeot, and J. Taylor</i>281
Heterogeneity Profiling: A Technique to Improve Geochemical Sampling and Analysis for AMD Assessments	<i>S.R. Pearce, V. Aktas, and R. Naithani</i>295
Mineralogy of Mine Site Neutralising Materials: A Missing Link in AMD Control Planning	<i>A.R. Gerson, R. Smart, J. Li, N. Kawashima, R. Fan, S. Zeng, R. Schumann, G. Levay, P. Dielemans, P. Mc Latchie, B. Huys, A. Hughes, S. Kent, and B. Hutchison</i>313
Assessing Metal Leachability from Low Sulfur Wastes - Sequential Extraction Methods	<i>C. Linklater, J. Chapman, P. Brown, R. Green, and S. Leake</i>325
Oxidation and Solute Accumulation in Pit Wall Rock: Limiting Changes to Pit Lake Water Quality	<i>A. Garvie, C. Linklater, R. Staines, J. Chapman, and R. Green</i>339
An Acid and Metalliferous Pit Lake Risk Assessment Tool	<i>S. Hannam and R. Green</i>351

PASSIVE AND ACTIVE WATER TREATMENT OPTIONS IN MANAGING AMD

Passive Treatment of Antimony in New Zealand, by Two Methods: Formation of Stibnite in a Sulphate-Reducing Bioreactor and Adsorption Onto AMD Precipitates	<i>D. Trumm and J. Pope</i>363
Instrumentation in Waste Rock Dumps: Going Deeper	<i>S. Pearce and M. Barteaux</i>371
The Dynamic Neutralising Behaviour of Alkaline Waste Rocks (Chlorite Calcite Schist) In Flow- Through Treatment of AMD Waters	<i>J. Li, A.R. Gerson, R.St.C. Smart, K. Kaplun, N. Kawashima, R. Fan, R.C. Schumann, A. Hughes, B. Hutchison, and S. Kent</i>387
Nanofiltration for Resource Recovery from Mine Influenced Waters	<i>M. Mullett and R. Fornarelli</i>401

EMERGING TECHNOLOGIES AND RESEARCH

Integration of Water Chemistry and Acidity Components: A New AMD Assessment Model	<i>P.H. Pham, B. Noller, S. Golding, and M. Edraki</i>411
Gypsum Sludge and Tailings at Brukunga Mine Used as Partial Clay Substitute for Making Construction Materials: A Lab Scale Study	<i>Y. Tian, R. Mollehuara, I. Gajjar, and P. Baker</i>423

Tools for Assisting with the Assessment of Acid and Metalliferous Drainage (AMD)	<i>J. Waters, S. Pape, and J. Taylor.....</i>	435
Neutralisation of Aluminium-Rich Acid Mine Drainage to Form a Magnetic Sludge	<i>W. Stanford and J. Webb.....</i>	449
Geochemical and Geotechnical Investigations at the Reddale Coal Mine, Reefton, New Zealand	<i>P. Weber, W. Olds, and M. Pizey.....</i>	455
The Inert Gas Mixture (IGM) Technology to Prevent Acid Mine Drainage	<i>L. Ameglio and H. Barrie.....</i>	467
Domaining Acid Rock Drainage Risks Using Geometallurgical Data	<i>A. Parbhakar-Fox and B.G. Lottermoser.....</i>	483
Low Cost Acid Mine Drainage Management Using a Modified Biological Reduction Process Producing Waste Heap Armouring Components	<i>S.P. Costin.....</i>	495

DETAILED CHARACTERISATION OF THE WASTE ROCK DUMPS AT THE KENNECOTT UTAH COPPER BINGHAM CANYON MINE—OPTIONALITY FOR CLOSURE

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ABSTRACT

A detailed characterisation of the waste rock dumps at the Kennecott Utah Copper (Rio Tinto) Bingham Canyon Mine has been acquired from the placement and instrumentation of four paired borings in the dumps over their full depth and analysis of the drill core. This has been coupled with data from a substantial number of historical studies to develop a conceptual model of the geochemical and physical behaviour of the dumps. The data show conclusively that pyrite contained in the dumps is oxidised, with air being supplied by both diffusion and convection. The dominant mechanism is oxygen supply by convection.

The generation of acidic effluent from the dumps could be reduced if the air supply from convection could be controlled. Placement of new waste rock into the dumps could be used to reduce convection by ensuring that minimal segregation of waste rock occurs. Historically, segregation has occurred due to the dumping practices used. Reduced convection will result in reduced acidic effluent generation which will have closure benefits.

1.0 INTRODUCTION

The existing waste rock dumps at the Kennecott Utah Copper (Rio Tinto) Bingham Canyon Mine (BCM) occupy a footprint of 2157 ha containing 4.43 billion tonnes of material. Leaching of the dumps for copper recovery occurred from the 1930s until leaching was terminated in 2000. An upgraded collection system using cut-off walls keyed into bedrock down hydraulic gradient of the waste rock dumps along their east side was installed between 1992 and 1995 to capture acidic water flowing from the waste rock. Acidity arises from the oxidation of sulfide minerals, principally pyrite, present in the waste rock, which is moderated by the precipitation of secondary minerals, principally jarosite, and dissolution of gangue aluminosilicate minerals. In some parts of the waste rock, there is also sufficient calcite present that in situ acid neutralisation may be quantitatively important. While the mine is operating, the final fate of the captured acidic water is neutralisation and metals precipitation in the tailings pipeline.

The aim of the present study was to develop an understanding of the mechanisms and rates of geochemical and hydrogeological processes responsible for acid generation and release from the BCM waste rock. In formulating the understanding, the study relied primarily on quantitative results obtained from investigations of the waste rock dumps that occurred during 2011 and 2012. In addition, the very extensive suite of prior studies on the dumps have been reviewed and utilised, including the studies from the 1970s research and development program that was aimed at evaluating the commercial leaching of copper from the dumps. Where considered appropriate, key results from these earlier studies have been used to support the acquired conceptual understanding. The understanding has been used

to assess management strategies (e.g. dumping strategies, covers) that could be used in mine and closure planning aimed at reducing acid release from the dumps post-closure.

2.0 CHARACTERISATION METHODOLOGY

During 2012, eight borings were placed in the waste rock dumps in four pairs. The pairs consisted of one shallow boring to a depth of about 33 m and one deep boring, targeted to terminate when bedrock was encountered. Three of the four deep borings encountered bedrock, but the fourth had reached a depth of about 274 m when it was terminated without contacting bedrock, as this was considered to be the depth limitation of the drill rig. Table 1 provides a summary of the borings that were placed in the waste rock dumps. Figure 1 illustrates the location of the borings in the waste rock dumps and the catchments relevant for the east side cut-off wall collection system.

Table 1: Summary data for borings placed in BCM waste rock dumps

Well	DH12-01		DH12-02		DH12-03		DH12-04	
	S	D	S	D	S	D	S	D
Waste Rock Dump	Code 51		Keystone		N. Keystone		Yosemite	
Catchment	Bluewater 2		Keystone		Keystone		Yosemite	
Total Depth (m)	32.6	182.9	32.6	274.3	32.6	216.4	32.6	216.4
Bedrock Depth (m)	NA	176.6	NA	Deeper	NA	207.3	NA	208.0

Note: S signifies the shallow and D the deep completions in the paired borings at each location

The drill holes were established using a sonic drilling method (with multiple telescoped casings) to enable the collection of core from each of the borings that was assayed for its chemical composition as a function of depth. A mineralogical analysis of composite samples from each hole was obtained using quantitative scanning electron microscopy (QEMSCAN). The moisture content as a function of depth was also determined. Some field observations and measurements were also collected during construction of the drill holes. These included material classification, lithology and oxidation descriptions, waste rock density, occurrence of elevated moisture or steam vents, and temperature.

A 50 mm fiberglass support pipe with a bottom end cap was inserted into each drill hole to its full depth. Instrumentation was placed within, or connected to the outside of the support pipe, to enable collection of in situ data. This instrumentation included a thermistor string, tubes for collection of gas (oxygen) samples, suction lysimeters, a vibrating wire piezometer and a time-domain reflectometer cable. Following installation of the support pipe and instrumentation, the entire annulus was filled with either bentonite or sand/filter mediums. Subsequent to placement of the drill holes in the waste rock dumps, oxygen flux measurements were taken on the dump surfaces near to the drill hole locations.

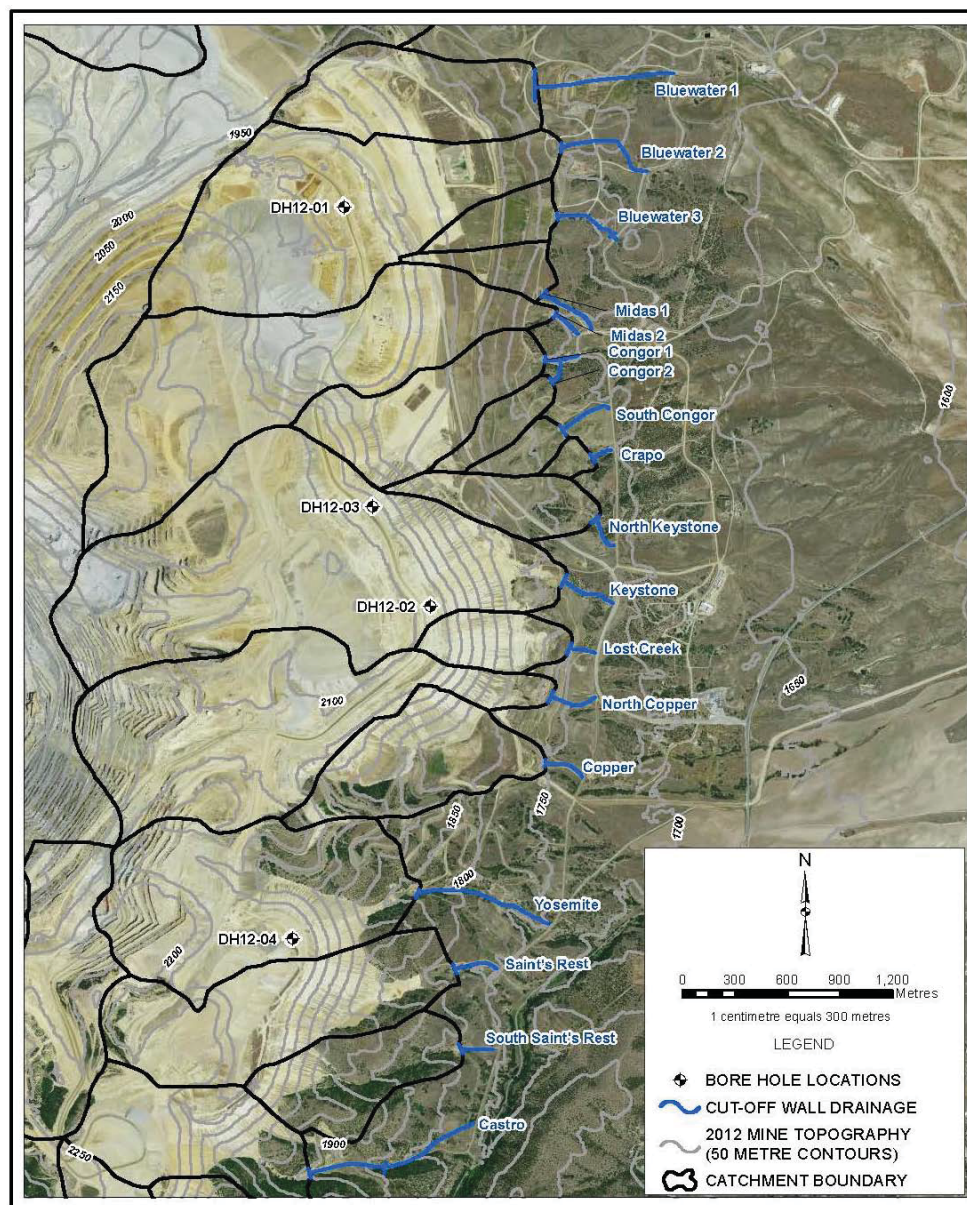


Fig. 1. BCM waste rock dumps and catchments for east side cut-off wall collection system and location of bore holes.

3.0 OXYGEN AND TEMPERATURE

As an example of the temperature and oxygen concentration profiles measured in the dumps, data for two of the pairs of borings are illustrated in Figure 2. Temperature was measured using the thermistor strings internal to the fiberglass pipe installed within the shallow and deep bores. The oxygen concentration was measured at a number of depths in both the shallow and deep borings, and a measurement was made at a depth of about 30 m in both borings.

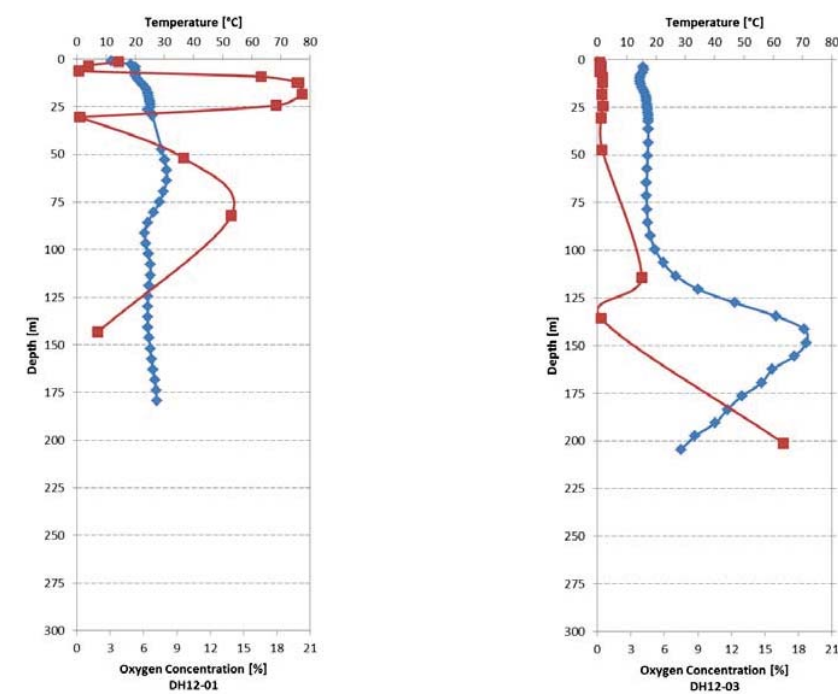


Fig. 2. Temperature (blue) and oxygen concentration (red) measured as a function of depth in borings DH12-01 (left) and DH12-03 (right).

DH12-01 exhibits much lower temperatures throughout its profile than the other borings. It is believed that these relatively low temperatures are a direct result of the extensive leaching that occurred in the region where the boring is located, the leaching removing heat from the dumps. However, it should be noted that the average ambient temperature at the mine site is about 10 °C, so the temperatures of between 20 and 30 °C observed at this location still indicate a substantial amount of in situ heat generation. In addition, the change in the temperature profile at a depth of about 90 m may be indicative of the change in dumping practices that occurred at this depth. The oxygen concentration profile shows very high concentrations (up to almost 21% between 6 and 30 m and about 14% at about 85 m). The very low oxygen concentration at about 30 m is believed accurate as both the shallow and deep bore exhibited a low concentration at this depth.

The high oxygen concentrations are clear evidence of convective air transport into the dumps. If only diffusive air transport occurred, the oxygen concentration would decrease from ambient (21%) at the surface of the dump to near zero at some depth below the surface (in this particular boring, this depth is 6 m). Below this depth, the oxygen concentration would remain low because no oxygen is transported further into the dump. High concentrations can only occur at depth if there is another transport mechanism, which in this case is convective supply due to density (temperature and/or pressure) differences in the dump drawing additional air (containing oxygen) into the dump.

Beneath the zone of diffusive oxygen consumption (depth of 6 m), the oxygen concentration then increases to near ambient followed by a very steep drop in concentration to near zero. This drop is associated with a further increase in temperature as a result of the oxidation of pyrite, with oxygen supplied by convective transport. On the basis of the temperature and

oxygen concentration profiles, the thickness of the oxidising zone in this region is estimated to be about 12 m. Beneath this zone, the oxygen concentration again increases and there is an associated decrease in temperature, beneath which the oxygen concentration again decreases which is coupled with an increase in temperature. In this zone, again it is believed that there is oxidation of pyrite fuelled by convective air supply. The thickness of this zone is estimated to be about 22 m. Thus, the total thickness of oxidising material at this location is calculated to be about $(6 + 12 + 22 \text{ m}) = 40 \text{ m}$.

The oxygen concentration and temperature profiles in DH12-03 are very different from those obtained from any of the other borings (Figure 2). The temperature measured in this boring is below 20 °C for the top 100 m of the profile, but for at least the first 50 m, the oxygen concentration is near zero. The low temperatures and oxygen concentrations in the top 50 m at this location suggests that the air permeability in this region is quite low, limiting the ability of air to diffuse into the dump from the surface or to migrate into the top portion of the dump by convection from depth. On the basis of the small temperature increase that occurs to a depth of about 36 m, this depth has been derived for diffusive oxidation at this location (and confirmed by surface oxygen flux measurements).

Beneath a depth of 100 m, there is a rapid increase in temperature. This increase is coupled with a decrease in the oxygen concentration. In this zone there is oxidation of pyrite occurring due to convective air supply. The thickness of oxidising zone is of the order of 21 m. Near the base of the dump, at this location, there is a large increase in the oxygen concentration, again confirming the supply of air into the dump via convective mechanisms. The total thickness of oxidising zones at this location is about $(36 + 21 \text{ m}) = 57 \text{ m}$.

In 1975, the Wyoming Minerals Corporation drilled 48 holes in the BCM waste rock dumps, using a Becker hammer drill. Twenty-four of the total of 48 holes did not collapse after drilling and were completed in plastic pipe to a range of depths from a few to 81 m. In addition, the perforated plastic pipe was tested for total oxygen content (over the whole depth), and temperature was measured with thermistors over 3 m intervals ranging from 12 to 58 m. Although the drilling did not reach the depths achieved in the 2012 drilling, and the techniques used for measurements, especially for oxygen, were not as detailed or robust as is currently possible, the large number of measurements and the care with which they were taken and assessed was far ahead of its time, and provides valuable data to the current study. The 1975 holes provided a larger distribution of data over the surface of the dumps, particularly away from the dump crests, than was possible with only the four drill holes installed in 2012.

In the 1975 measurements [KUC, 1976], the range of observed temperatures was from 19 to 73 °C, which is entirely comparable with the range observed in the 2012 testing. Fifteen of the wells obtained average oxygen measurements. The range of those values was from 0 to 20% (by volume), again, entirely comparable to the 2012 range. The range of both oxygen concentration and temperature within the dumps in 1975 provided clear evidence that there was active thermal convection occurring, particularly in the dumps that were under (or had recently ceased) leaching. There was a tendency for the wells that were within 30 m of the crest to have higher oxygen concentrations, typically greater than 15%, whereas there was a tendency for wells completed more than 90 m from the crest to have relatively low oxygen concentrations (i.e. less than 5%). However, the low oxygen concentration wells far from the crest tend to have high temperatures, and it was inferred that this signified that the low oxygen concentrations were a consequence of active oxidation in the region of the low oxygen concentrations. In addition, there were two wells at distances greater than 90 m from the crest that had oxygen concentrations greater than 14%.

From the 1975 data, it is inferred that the convection observed today is a physical process that has been characteristic of the entire history of the waste rock dumps. In addition, the data show that there has been active oxidation occurring in the deeper part of the dumps for many decades.

3.1 Oxygen Consumption

On the basis of the oxygen concentration and temperature measurements, the average thickness of oxidation in the dumps is 59 m, whereas the average thickness of oxidation due to diffusive oxygen supply is about 10% of this. These measurements, when coupled with the oxygen flux measurements, lead to an average oxidation rate of about $2.7 \times 10^{-8} \text{ kg(O}_2\text{)/m}^3\text{/s}$. The flux of oxygen consumption in the dumps is thus $2.7 \times 10^{-8} \text{ kg(O}_2\text{)/m}^3\text{/s} \times 59 \text{ m} = 1.6 \times 10^{-6} \text{ kg(O}_2\text{)/m}^2\text{/s}$.

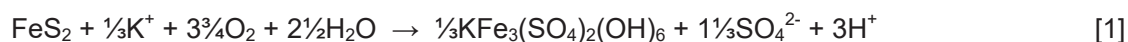
The flux of sulfate collected in the east side drainages can be utilised to assess the fraction of sulfate being released from the dumps of the total amount of sulfate that would be produced from the oxidation of pyrite. These calculations indicate that the flux of sulfate collected in the east side drainages is less than the oxygen consumed within the heap, indicating that a large amount of the sulfur is remaining within the heap in secondary minerals, such as jarosite.

4.0 INFILTRATION INTO AND EXFILTRATION FROM THE WASTE ROCK DUMPS

O'Kane Consultants (OKC) have undertaken several studies examining the infiltration into a number of different scenarios (test plots) relating to BCM waste rock [OKC, 2005, 2009, 2011]. In their 2009 study, OKC determined the net infiltration into both weathered and fresh waste rock. The range of net infiltration into weathered waste rock was from 13 to 32% of incident precipitation whereas that into fresh waste rock was somewhat higher, ranging from 17 to 42%. In a subsequent study [OKC, 2011] they indicated that during wet years the net infiltration could increase to over 50% of incident precipitation. The moisture content of the dumps following such high precipitation seasons would likely increase and then drain down more slowly over time during drier periods. OKC indicated that the most likely net infiltration into the BCM waste rock dumps would be similar to what they measured for weathered waste rock, which averages 23% of incident precipitation. The range of net infiltration defined by OKC [2009] was almost identical to that calculated in their previous study [OKC, 2005] which varied between 15 and 30%, which also has an average value of 23%.

The quantity of water exfiltrating the waste rock dumps has been determined from water balance calculations on two occasions [KUC, 2001, 2006]. Both of these studies indicated that the quantity of water emanating from the waste rock dumps was approximately 20% of incident rainfall. On average this value is about 3% less than that entering the dumps, and the difference can be readily accounted for by water consumed in chemical reactions within the dumps for which there is strong mineralogical and geochemical support that cannot be neglected.

The dominant geochemical processes occurring in the dumps are the oxidation of pyrite and the subsequent precipitation of jarosite. The combination of these processes can be represented by the following chemical reaction:



It is evident from this reaction that $3\frac{3}{4}$ moles (120 g) of oxygen react in the oxidation of pyrite at the same time as $2\frac{1}{2}$ moles (45 g) of water is consumed. The load of water entering the

waste rock dumps from meteoric precipitation is $138 \text{ kg(H}_2\text{O)/m}^2\text{/year}$. At the average oxidation rate measured in the dumps ($2.7 \times 10^{-8} \text{ kg(O}_2\text{)/m}^3\text{/s}$), the infiltrating water would be completely consumed in the dumps (i.e. no exfiltration) over a distance of 432 m, assuming that all of the waste rock was oxidising at the same rate. However, it was shown that the thickness of the oxidation zones beneath any point on the dumps has an average thickness of 59 m. Therefore, the amount of water consumed in the dumps is equal to $23\% \times (59 \text{ m} / 432 \text{ m}) = 3.1\%$. This latter value is in excellent agreement with the amount of water consumed that was determined from difference between the average net infiltration entering weathered dumps (23%) and the measured exfiltration (20%) from the dumps. The potassium in reaction [1] arises from the dissolution of muscovite and/or biotite [Jambor, 2005]. It is believed that the impact of other reactions that consume or produce water have a negligible effect on the water balance.

Figure 3 illustrates that reaction [1] does occur in the waste rock dumps. The figure shows a sample taken from the dumps where a pyrite crystal is rimmed by jarosite precipitates. It should be noted that in Figure 3 jarosite precipitation does not occur around all sulfide containing grains, even in close proximity to grains in which jarosite-pyrite contact is extensive. This is very often seen in waste rock systems, and is thought to be associated with the details of microscopic conditions of pH and fluid hydrochemistry at mineral-grain contacts. In the detailed mineralogical analysis of samples from the 2012 drilling, the results show that a textural association of jarosite and pyrite is present in only about 30% of the grains examined. These statistical results are considered consistent with the optical record [Jambor, 2005], as illustrated in Figure 3.



Fig. 3. Sample obtained from the Keystone dump at BCM showing a plain reflected light image (left) and an internal reflected image (right) of the same field (1.3 mm in width). On the left of the two images is a pyrite clast that is contained within a jarositic cement (greenish colouration on the right image).

5.0 DRILL LOGS, ASSAY DATA AND MINERALOGY

The colouration and lithologies logged as a function of depth in the drill core obtained from DH12-02 are shown in Figure 4. The grey colour is indicative of predominantly unoxidised waste rock, with the red and orange colours representing oxidised waste rock. The orange colour is indicative of the presence of jarosite and the red colour either ferrihydrite and/or goethite.

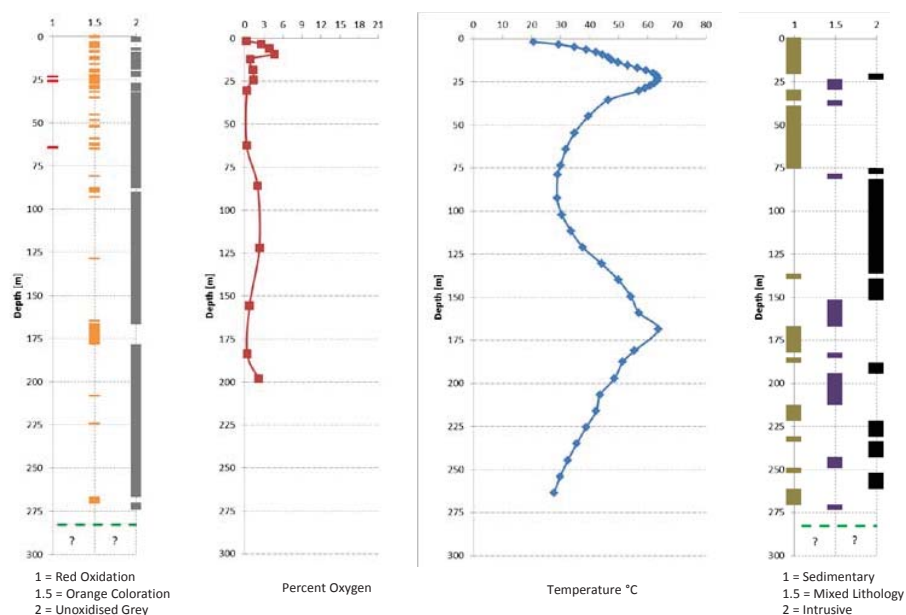


Fig. 4. Colouration and lithology of drill core as a function of depth compared to oxygen concentration and temperature measured in the DH12-02 bore.

There is much less red and orange colouration noted in this core than was the case for DH12-01 (due to the substantial leaching that took place at the latter location there is much more secondary mineralisation), there being only three small areas where red colouration was noted. It is also worth noting that the main areas of orange colouration coincide with the highest temperatures, consistent with the endothermic nature of jarosite precipitation. That is, the precipitation of jarosite is favoured at higher temperatures. The higher oxygen concentrations in the centre of the profile coincide with an area that is unoxidised that consists of a large band of intrusive rocks. In these regions, it is believed that air transport occurs mainly through relatively coarse zones where the surrounding fine material has a relatively low permeability. Thus, some oxidation can occur but at a reduced rate. Neither the basement (bedrock contact) nor the overlying alluvium was intersected in this bore; pre-mine topography data indicate the bedrock occurs at a depth of 283m, about 9 m deeper than the hole could be drilled.

The most important assays reported on the cores taken during drilling relate to those of iron and sulfur. These two elements are predominantly contained in pyrite and jarosite. The molar ratio of the two elements in pyrite and jarosite is quite different; in pyrite it is 0.5 whilst in jarosite it is 1.5. Thus, determining the molar iron to sulfur ratio from the assay data as a function of depth allows the relative proportions of pyrite and jarosite in the core to be ascertained. Ratios close to 0.5 will contain predominantly pyrite whereas those close to 1.5 will be predominantly jarosite; a ratio of 1.0 will be indicative of equal molar proportions of the two minerals. The molar ratio of iron to sulfur from all four bore holes clearly showed that the ratio only ranged between 0.5 and 1.5; only in basement rocks does the ratio go above 1.5, where the iron content remains relatively high but the sulfur content is quite low. As shown by the colour in the drilling logs (Figure 4), the precipitation of jarosite is generally high near the surface of the dumps and increases at depth in the regions of high temperature.

Table 2 provides an example of the mineralogical results provided on the drill core obtained from DH12-04. The data were obtained using two techniques; quantitative scanning electron microscopy was used to determine the +20 μm fraction and X-ray diffraction was used to determine the -20 μm fraction. The combined results from both fractions are presented in Table 2.

Table 2: Mineralogy of drill core samples from well DH12-04. Minerals are reported in %.

Monitoring Hole	DH12-04						
Depth (ft)	10-20	30-40	50-70	90-110	190-210	390-410	590-610
Chalcopyrite	0.04	0.14	0.43	0.05	0.09	0.12	0.18
Bornite	0	0	0.01	0.02	0	0.01	0
Covellite	0	0	0	0	0	0.02	0
Chalcocite / Digenite	0	0.03	0.06	0	0.01	0.06	0.01
Other Cu minerals	0.06	0.78	0.34	0.07	0.13	0.22	0.42
Pyrite	0.98	6.29	5.56	5.18	5.21	4.82	3.51
Jarosite	2.16	0.48	0.40	0.57	0.43	0.86	2.36
Quartz	54.6	52.0	61.8	58.6	54.6	48.2	29.3
Plagioclase	3.64	5.29	3.15	4.65	4.93	7.18	6.28
K-feldspar	9.75	11.6	6.20	8.77	11.9	12.5	13.2
Muscovite	3.74	4.05	2.80	8.06	3.68	7.62	9.15
Biotite / Phlogopite	1.09	0.59	0.86	1.03	0.94	0.90	1.14
Chlorite	4.25	3.01	2.68	2.74	2.67	2.01	2.23
Kaolinite	0.39	0.87	0.33	0.24	0.29	0.63	0.78
Pyroxene / Amphibole	4.93	2.28	5.90	2.71	4.69	5.59	6.16
Talc	4.59	1.18	1.52	1.29	1.90	1.67	2.30
Calcite	0.15	2.62	2.40	0.39	1.96	0.70	4.22
Iron oxides	2.97	1.13	0.40	0.43	1.13	1.65	2.91
Gypsum	1.33	2.04	0.89	0.57	0.17	0.81	2.73

An important difference in the mineralogy is found for DH12-01, the boring located where extensive leaching had taken place, as compared to the mineralogical results from DH12-02, DH12-03 and DH12-04 (where no or limited leaching occurred). For DH12-01, the mineralogical results show negligible calcite content, relatively low pyrite content and high jarosite content. Conversely, the mineralogy results from the other three sonic drill cores show a significant calcite content in most samples, much higher pyrite content and lower jarosite content. Thus, it would appear that there has been an insufficient flux of acid generated in these locations to access and dissolve all of the calcite that is present. Sufficient acid may have been produced in the vicinity of these locations to dissolve the amount of calcite that is present, but it must be able to diffuse into the waste material to be able to first access and then dissolve the carbonate minerals. The presence of calcite, however, does not imply that drainage from these areas will be near neutral because, as indicated, the acid may not easily access the locations where calcite is present and so fluid with low pH and high acidity may bypass some calcite-containing zones. It would also appear that in the zones that have not been leached the amount of oxidation is also less, as evidenced by the higher pyrite content and lower jarosite content. The secondary minerals that precipitate within the dumps are largely associated with the -20 μm fraction.

The mineralogical assessment confirms the molar ratio analysis carried out on the assay data, in that there are substantial quantities of both pyrite that is yet to be oxidised and jarosite formed from the oxidation of pyrite present throughout the drill core profiles. The upper profile in the core from DH12-01 shows a predominance of jarosite over pyrite as was indicated by the iron to sulfur ratio. In other locations, there are smaller quantities of jarosite relative to pyrite, where in some locations pyrite predominates. This also supports the observed range of the iron to sulfur ratio data that was shown to vary between 0.5 and 1.5.

6.0 CONCEPTUAL MODEL

A conceptual model of the geochemical and physical processes occurring in the waste rock dumps is illustrated in Figure 5. The model has been derived from the detailed evaluation of data described in the previous sections. The figure shows that air ingress into the waste rock dumps occurs by both diffusive and convective mechanisms. The convection of air into the dumps occurs at the toe of the dumps through coarse zones created because of material segregation by mass that occurred during the top-dumping process (often from heights of 150 to 300 m). Water ingress into the dumps now only occurs from incident precipitation (snow melt and rainfall), whereas historically the majority of the water ingress for several of the dumps was as a result of irrigation for leaching purposes.

The diffusion of air into the dumps occurs in a narrow thickness at the top of the dumps that is of the order of 2 to 6 m, based on the measured rates of oxygen consumption and the observed oxygen concentrations in the pore space of the dumps. As indicated in Figure 5, convective air supply occurs primarily from the base, although there may also be a smaller supply that occurs through the slopes of the dumps (as seen in DH12-02). The ingress of air into the dump results in the oxidation of sulfide minerals, principally pyrite, within the dumps. Oxidation in the diffusive zone occurs throughout the whole thickness of the diffusion and also in deeper zones due to convective air supply. These layers may be immediately beneath the diffusive ingress zone or may occur deeper within the dump (i.e. away from the batters). The thickness of the convective oxidising layers is of the order of 9 to 20 m for the BCM waste rock. Outside of these oxidising zones, oxidation may still occur but at a much lower rate. The flux from such zones is unlikely to be greater than 10% of the flux from the oxidising zones which, considering the thickness of the non-oxidising zones, suggests an oxidation rate about two orders of magnitude lower than in the oxidising zones. Convective air supply into the dumps arises due to differences in density (caused by temperature and/or pressure differences) within the dump as compared to the ambient conditions (i.e. hotter temperatures inside the dump resulting from heat generated from the oxidation of pyrite and/or lower pressures resulting from the consumption of oxygen in air through reaction with pyrite).

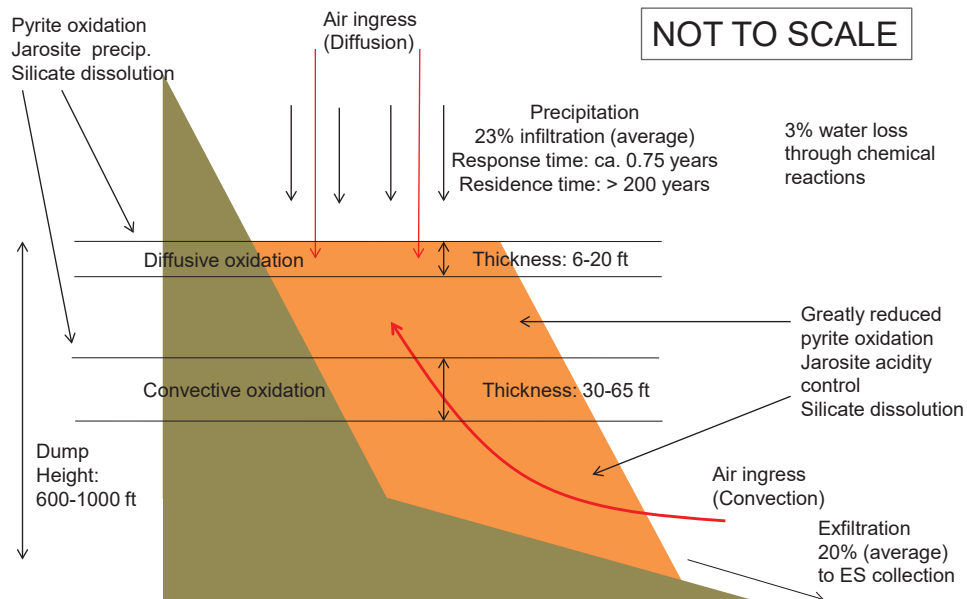


Fig. 5. Conceptual model of processes occurring in the BCM waste rock dumps.

7.0 OPTIONS FOR CLOSURE

The acid generation from the BCM waste rock dumps could be substantially and sustainably reduced if convective air flow into the dumps could be controlled. Oxidation from convective air supply accounts for the majority (90%) of the acid generation in the dumps. Convective air supply occurs largely from the base of the dumps, due to segregation of the waste rock during truck dumping. The air passes into the dumps via “chimneys” and the rate of oxygen ingress increases significantly due to the differences in temperature and/or pressure within the dumps.

BCM is planning an expansion of the mine that would lead to the generation of an additional 1.3 billion tonnes of waste rock. Placement of this additional waste rock in an engineered dump in front of the existing dump would have the ability to prevent convective air ingress into the existing waste material as well as the new dump material. The essential conceptual element of the design would be to construct the dump extension in such a manner that high convective fluxes of air cannot move into the dumps. This can be accomplished, whilst maintaining the necessary drainage required for physical stability, by using construction techniques that will prevent the formation of high gas permeability zones that are characteristic of traditional top-dumping methods in which coarse boulders accumulate near the base of the dumps (i.e. shorter dump lifts, blading fine-grained material from dump crests to dump toes, different dumping directions). This would have a substantial impact on the amount of oxidation that occurs in the dumps, potentially reducing it to about one third of the current amount. Some convective air transport may still occur in the slopes of the dumps, but at depth within the dumps convection would largely be eliminated. This would therefore allow neutralisation of acidity, due to dissolution of gangue minerals, to occur over a substantially greater depth, leading to a reduction in the acidity emanating from the dumps. Placement of covers on the outer slopes of the dumps (obtained from existing fine-grained soils and sub-soils recovered from the new dump footprint prior to construction) would lead to a significant reduction in net infiltration into the dumps.

8.0 CONCLUSIONS

Four paired and instrumented borings were placed in the waste rock dumps at BCM that extended the full depth of the dumps. Data acquired from the borings and dump surface were utilised to characterise the geochemical mechanisms occurring in the dumps that lead to generation of acidic effluent. The data have been combined with an extensive amount of historical information on the dumps to show conclusively that oxidation of pyrite within the dumps occurs as a result of both diffusive and convective air supply into the dumps, with the convective supply being dominant.

The generation of acidic effluent in the dumps could be reduced by decreasing the convective supply of air (containing oxygen) into the dumps. Controlling the convective supply of air into the dumps could potentially be achieved by placing new waste rock in front of the existing dumps that would also provide closure benefits via decreasing the acidity of exfiltrating water.

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INAP UPDATE: CHALLENGING ENVIRONMENT AND OUTCOMES OF THE 9TH INTERNATIONAL CONGRESS ON DRAINAGE 2012, OTTAWA, CANADA

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ABSTRACT

The recent contraction of the mining industry has been difficult for all elements of this sector. INAP has also experienced some of these challenges, but we are continuing to grow in the breadth and depth of our activities. This paper presents these activities and recognizes the support that we have received, not only from our member companies, but also from the numerous world-wide volunteers from the Global Alliance. A summary of the very successful 9th ICARD in Ottawa, Canada is also presented with the plans for the 10th ICARD in Santiago, Chile on 20-25 April 2015.

1.0 INTRODUCTION

I have been asked to present an update on INAP and the 9th ICARD, but before I do that I would like to outline some of the background for the INAP activities, and why they have such strong momentum during this difficult time in the mining industry.

This story begins at the 7th Australian Workshop in Darwin that occurred in April 2011. Many of you were there and perhaps you are part of this story. At that workshop, Dr. Bruce Kelley, the Rio Tinto Global Practice Leader for Environment at that time presented an excellent paper on the management of mining waste. He focused his talk on the substantial effort that is placed on stripping the overburden and mining waste to recover a very small volume of ore. In fact, he suggested that mining companies might more accurately be called mine-waste management companies, except there is little value derived from this considerable effort of mine waste management. Historically, the corporate focus has been on the ore, while the significant volume of mine waste was considered a corporate orphan, with its major concern being how to remove and dispose of it with a minimum of cost. Dr. Kelley used the phrase, “the elephant in the room” to describe this dichotomy of the potential importance of waste management and its apparent lack of concern. Dr. Kelley went on to suggest that this topic was worthy of further consideration.

“The Elephant in the Room” captured the interest of good portion of the attendees at that workshop, and an ad hoc group was formed to discuss this topic in greater detail. I sat in on this discussion, and I was pleased to see about 40 experienced practitioners engaged in a spirited discourse that ran until dinner. Early the next morning, they were back again developing a path forward and a plan to engage SMIKT and INAP in its plan. I was struck by the focus and energy of the group and by all those that attended the workshop.

I would like to make a quick side comment at this point. The enthusiasm and commitment of the SMIKT and the AMD practitioners in Australia reflects a common characteristic of many of the Global Alliance (GA) members. Once they have the critical size of membership in their region, they can become a fantastic source of ideas and energy and are able to implement amazing activities. Examples of this trait, which are described in this paper, are shown by

the quality of the 9th ICARD as organized by MEND and the support of ADTI for the Sulfate Workshop recently held in Salt Lake City, UT, and the publication of a 6 volume set of workbooks on Mine-Influenced Waters prepared by ADTI and published by the Society of Mining, Metallurgy and Exploration.

At the next INAP Operating Committee (OpCom) meeting in November 2011 following the Darwin AMD workshop, a presentation was made to the INAP OpCom by members of the Australian group assisted by SMIKT. Resulting from that presentation, the OpCom was in favor of looking further at mine-waste management, but they were concerned with limitations of the INAP budget. They also knew that the 9th ICARD was fast approaching where other options might be presented, and they postponed a decision on this proposal.

2.0 THE PATH FORWARD SYMPOSIUM

To engage a broader discussion of mine-waste management and other topics of interest, INAP and the GA organized the “Path Forward Symposium” that was held the day preceding the ICARD in May of 2012. Global Alliance members (ADTI, MEND, INAD and SMIKT) participated in this symposium. The format of this symposium was unusual, in that the initial session was planned for half a day. Hence, our time for presentations was limited, as there were 18 presentations scheduled. To enable this tight schedule, the speakers were allowed only 5 minutes per presentation followed by 2 minutes for a brief question and answer period. Thus we had 18 “Elevator Speeches” or “Lightening Presentations” that were designed to catch the attention of the audience of 70 ARD and mine-waste-management experts. These 18 presentations were grouped into 4 general topics, which included 1) Mine-waste management; 2) Biogeochemistry; 3) Innovative technology and 4) Stakeholder engagement.

Following the opening session, the audience was allowed to separate into one of the four topic groups for further discussion. The break-out sessions were designed as “free-form” to follow the general interests of those in the break-out group. There was a concern that this approach could create a situation where a strong leader would control the direction of the group. However with the breadth of leaders in the groups, this possibility did not occur, and most participants felt that they had ample opportunities to express their thoughts.

Following the break-out session, the participants reconvened and presented a draft summary of their findings. The organizers felt that by having this symposium prior to the ICARD, it would capture the imagination of the participants and that they would think about these waste management and discharge issues during the ICARD proceedings. Hence, it was proposed to the assembly that they meet for a second summary meeting at the end of the ICARD. This suggestion was met with enthusiasm, and a room was reserved for this final meeting. Some of the groups met independently and formalized their thoughts and concepts in more detail.

We met early on Friday before the ICARD tour and each group made a final presentation. The discussion was then opened to the audience. There were some frank comments expressing the need for the mining industry to take a stronger position relating to some of these topics. A list of recommendations was compiled and this list was utilized at the INAP Strategic Planning Meeting that was held in the fall of 2012 in Vancouver, BC. This list is presented below.

Findings from the Path Forward Symposium May 2012 - Ottawa, Canada

- Mine Waste Management
 - Volume of Mine Wastes
 - Potential Impacts
 - Soil Covers
 - Land Form Design
 - Recovery of Low-grade Ore
- Biogeochemistry
 - Advisory Panel
 - Application of Biogeochemical Tools
 - Stakeholder Issues
- Develop Stakeholder Capacity
 - Expand Stakeholder Dialogue
 - Expand use of Global Alliance
 - More Focused Workshops
- Innovative Technologies
 - INAP be more supportive
- Case Studies
 - Linked to GARD Guide
 - Great Learning Tools
 - “What the mining industry is doing right”
 - Lessons Learned

3.0 INAP STRATEGIC PLANNING MEETING

The 2nd INAP Strategic Planning Meeting was held in Vancouver, BC in conjunction with the INAP OpCom and Board Meeting in November 2012. This meeting was facilitated by Dr. Dirk van Zyl, and it was preceded by a number of development conference calls and planning sessions. Much of the preparatory input was based on recommendations from the 2008 INAP Strategic Planning Meeting, prior OpCom discussions, as well as the inputs from the “Path forward Symposium”. We also had a set of Global Alliance conference calls to obtain input from these partners as well.

From these inputs, it was determined that many INAP members, partners and stakeholders thought the mining industry would be facing significant changes in the future. It was also believed that these changes would come from many sources affecting the mining industry including: 1) Environmental, 2) Regulatory, 3) Societal, 4) Water issues, 5) Mine-waste management, and 6) Increasing size of mines.

3.1 GARD Guide

As expect, one of the major topics discussed was the GARD Guide and how to enhance its acceptance by the mining community. It is recognized that there is strong support for use of the GARD Guide in the consulting and regulatory community. But often exploration and mine operations are concerned that designing and operating for closure increases their present costs and with little recognition for their effort to minimize future savings. Hence, there was much discussion as to developing a holistic mine management approach that would not only view mine planning and operations on a “time-value” approach, but would also consider “sustainable development” where the future use of land, water and resources are respected. With this vision in mind, it was proposed to prepare focused versions of the GARD Guide for exploration geologists, mine planners, and mining financial accountants.

There was also much discussion about preparing less technical and illustrated versions for non-technical stakeholders and community leaders, who need to understand the basis of ARD/AMD mitigation and prevention actions without being inundated with the technical details of the process.

3.2 Transformative Leadership

We discussed transformative leadership, and how it applied to the mining industry and its commitment to preventing ARD/AMD. It was recognized as a long and involved process that needs the commitment and support of the entire mining industry from the truck driver who places the potentially acid generating (PAG) waste to the financial planners, who prepare the detailed budgets to the upper management, who must understand and articulate the value of mitigating ARD/AMD to both stakeholders and stockholders.

Because of the vast framework of this overarching process of transforming the thinking of the mining industry, it is imperative that we partner with all groups and organizations of similar interest including ICMM, IFC, World Bank, AMC, MAC, and NMA. This need for partnering is also being recognized by these other organizations. At the recent annual meeting of the Society for Mining, Metallurgy and Exploration in February 2014, I met with Dr. Anthony Hodge, the president of ICMM, who stated that it has been decided that ICMM needs to team with INAP to distribute and promote the GARD Guide and other INAP technical endeavors that are relevant to the mission of ICMM. INAP was also requested to review the upcoming International Finance Corporation's (IFC) mining guidance document for issues relating to mine waste management and ARD, at the recent Sulfate Workshop. I believe that these partnerships can be a significant stepping stone in the implementation of transformative leadership within the mining industry.

3.3 INAP Technical Sharing

Another key topic discussed during the strategic planning meeting was the continued commitment to the ideal of sharing of information, experience and ideas between INAP members. The members recognize that the expectations of their lenders, shareholders and stakeholders, require companies to develop parallel expertise. But these companies also know that by sharing this environmental information the mining industry as a whole prospers with minimal loss of internal advantage resulting from being the "first user". In fact, I believe that recognition of being innovative by mining industry stakeholders often times outstrips the potential advantage of being a "first user".

Since the strategic planning meeting, I have seen a strong growth of internal INAP information sharing. Rio Tinto has offered to share its 10-years of experience in minimizing environmental risk including their audit process and its internal mechanisms. At recent, OpCom meetings we have one to two presentations by member companies on topics relating to INAP focuses including: Anglo American - the "True Value of Water", Barrick - "Soil Cover Design for the Gold Strike Operations" and Freeport McMoRan - "Pilot Plant Design for Mining Discharges". At the recent INAP/ADTI Sulfate Treatment Workshop, we had presentations by five INAP member companies. At the most recent OpCom meeting, INAP toured the Freeport McMoRan Copper and Gold's Bisbee waste-rock management and tailings closure facilities and the Sierrita mine-water discharge pilot facility, which includes 3 separate pilot operations treating separate groundwater and process-waste-water streams.

3.4 Capacity Building

The importance of INAP supporting the capacity development of mining industry stakeholders across the globe was discussed at the strategic planning meeting. This is of particular importance in many of the developing regions. It has been observed that as regulatory personnel become trained and experienced, they are often hired away by service and consulting firms resulting in regulatory agencies continually training new personnel.

INAP has made a commitment to support the promotion of best practices in all regions of the globe. Obviously, this was one of key justifications for the development of the GARD Guide. Portions of the GARD Guide have been translated to Spanish and French; we understand that the GARD Guide Executive Summary is being translated into Turkish, by local ARD practitioners.

INAP believe that this is an excellent way for the GARD Guide translation to proceed. INAP will continue to upgrade the English version of the GARD Guide on a regular basis, and then global practitioners and service providers can translate the relevant sections as needed.

The GARD Guide Executive Summary was also translated to Spanish by the U.S. EPA and appended to the Central American Mining Industry Guidance that was prepared by U.S. EPA for use by countries in Central America. This recognition of the value of the GARD Guide and of INAP's efforts is very satisfying.

3.5 Case Studies

INAP believes that case studies are a critical part of the capacity development of stakeholders. Not only are case studies a great learning tool, they can be an effect method for the mining industry to illustrate their commitment preventing and mitigating ARD/AMD.

Over the past several years, some members of the Global Alliance have been pressing for a case study companion volume to the GARD Guide. This volume would be composed of brief case studies that illustrate the successful application of best practices put forth in the GARD Guide as well as lessons learned. The recent downturn in the mining industry has slowed the publication of relevant case studies and delayed the development of this case study compendium.

During a recent Global Alliance conference call, GA members suggested the idea that INAP should sponsor a contest or recognition of brief 4 to 5 page case studies that would capture successful operations or specific ARD migrating techniques that could be compiled and used as a companion volume to the GARD Guide or used separately as an illustration of best practice or as a learning tool. This idea was forwarded to other GA members and received strong support.

To expand interest in ARD prevention, mitigation, modeling and treatment case studies and also in the GARD Guide, it is proposed that INAP and the Global Alliance sponsor a challenge to recognize the best brief of a published or non-published ARD case study that reflects ARD best practices of principles described or reflected in the GARD Guide. Potential sources of published articles could be the proceedings of ICARD, IMWA, BC/MEND ARD Workshop, and the Australian AMD Workshop, also the following journals Mine Water and the Environment, Environmental Science and Technology and Mining Engineering. We believe that this activity would be of particular interest to academics, since this would be an excellent exercise for students in geochemistry, geology, environmental, mining and civil engineering as well as related fields of study.

The competition would be held on two levels. The first selection would be done on a regional basis and sponsored by the Global Alliance member (ADTI, CNAMD, INAD, MEND, PADRE, SANAP, SMIKT, and WRC) with the support of INAP. Each of these winners would then be submitted to the judges to select a grand winner, in which additional awards would be added. All regional and subsequent awards would be recognized at the 10th ICARD in Santiago, Chile in April 20 – 25, 2015.

The case study brief should be in English and contain about 1000-1500 words. The length of the brief should be 4 to 5 pages including figures or pictures. The regional award would be \$500 USD for the editor and \$500 USD for the original authors if the case study is based on a published article offered by INAP. Additional awards may be granted by Global Alliance members. It was also felt that obtaining written approval of the original authors or owners of copyright. The challenge would be open to all. Since some published case studies have particular merit, multiple briefs of the same paper will be accepted. All of the best ARD briefs as selected by a panel of judges would be published in the INAP ARD Case Study Compendium.

4.0 MINE WASTE MANAGEMENT

As INAP pursued options relating to mine-waste management, we encountered a MEND report entitled, "Cold Regions Cover System Design Technical Guidance Document," by O'Kane Consultants (OKC). This report caught the attention of a number of INAP members. While this document was focused on cold regions, it included many of the mine-waste characteristics that INAP was interested in investigating a broader range of climatic conditions. These characteristics including cover design theory, objective and application. It also included a discussion of failure analysis and utilized case studies to good advantage.

INAP approached OKC with the intent of using this report as a model for a guidance document with a broader climatic range of cover design and application. INAP also wanted a broad range of international experts to participate in this activity. As part of the proposed agreement OKC proposed to have a meeting in Adelaide at the time of the 8th Australian AMD Workshop to engage Australian practitioners into the review process. In September 2013, INAP entered into an agreement with OKC to product a guidance document for soil covers addressing a broad range of climatic conditions. Michael O'Kane will be presenting on the investigation at this conference.

INAP is still pursuing other activities in the area of mine waste management and source control for acid rock drainage to sponsor.

5.0 INAP WORKSHOPS

On February 27 and 28, 2014, INAP organized a Sulfate Treatment Workshop in Salt Lake City, Utah with the help of an outstanding committee composed of Scott Benowitz, Lisa Kirk, Charles Bucknam, Carol Russell, Linda Figueroa, Chris Howell and Jim Fricke. Many of these committee personnel are members of ADTI (Acid Drainage Technology Initiative). Over 110 participants attended the 1 and a half day workshop. INAP was also supported by three sponsors: Veolia Water, Golder Associates and Arcadis in funding the workshop.

The workshop began with an up-date of the Lorax Environmental's investigation of Sulfate Treatment in Water, which was developed for INAP in 2003. The update was presented by Bruce Mattson of Lorax. This presentation was followed by a panel discussion with members from industry, academia, consulting, state and Federal governmental agencies.

These two preliminary activities set the stage for the remaining 20 presentations and case studies that discussed methods of treating sulfate in water, ranging from source control, to active and passive treatment. These treatment technologies also cover all the current methodologies including reverse osmosis, biological treatment, chemical precipitation and some of the newer innovative methods.

Following the technical program, we broke out into 7 groups to discuss 1) Sulfate research, 2) Regulatory and social strategies, 3) Source control, 4) Improving active treatment, 5) Improving passive treatment/pit lakes, 6) Brine handling and 7) Drivers for sulfate regulations. During these break-outs there were active discussions. At the end of the sessions, findings were reported to the full group. INAP has found these break-out sessions to be an excellent way to tap suggestions from very diverse groups of stakeholders. These suggestions have found fertile ground within INAP. In fact, the Sulfate Treatment Workshop is an example of a suggestion presented at the INAP “Path Forward” Symposium in Ottawa at the 9th ICARD.

At the conclusion of the workshop, there was a tour of the Kennecott Reverse Osmosis plant that is treating a sulfate-contaminated plume downstream from the Kennecott Bingham Pit and waste rock piles. We also observed the 100+ foot cutoff trench and dam that collects groundwater runoff from the Kennecott waste rock repositories.

6.0 9TH ICARD

ICARD is regarded as the leading global conference on the prediction and mitigation of issues associated with acid-rock, acid-mine, acid and metalliferous and neutral drainage. The 9th ICARD, hosted by MEND (Mine Environment Neutral Drainage), was held in Ottawa from May 20 to 26 and attracted 526 delegates from 19 different countries. The event featured eight pre-conference short courses with about 250 participants, a plenary session with experts representing industry, government and civil society, a trade show with 25 exhibitors plus a two-day field trip to Northwest Quebec where four mine sites were toured.

Of course, the most important element of this conference was the technical program that covered 3 days. It was outstanding with 127 oral and 43 poster presentations. Over 90 percent of the delegates felt that the conference was very good or excellent, and many attendees felt that this conference was one of the best that they had ever attended. Because of the size and venue design, the opportunities for networking were outstanding. You had the opportunity to meet everyone who was there.

The banquet and program was fabulous and over 95 percent of the attendees felt that the “Taste of Canada” Dinner and program was the highlight of the meeting. The program during this banquet involved many of the ICARD organizing committee. It was fantastic to see Elizabeth Gardiner, recently retired from MAC (Mining Association of Canada) and the ICARD conference chair, playing a carpenter’s saw with a violin bow. She was terrific with some professional help.

Some of the other comments of what attendees liked most about the 9th ICARD included: “Networking, networking, networking,” “Program content—presentations. Scope of topics covered,” “Extremely well organized, great availability of event planning group and absolutely amazing Taste of Canada Dinner,” “Technical program,” “Relevant topics of presentations,” and “Affordable.”

During the final luncheon, INAP recognized Gilles Tremblay with the 2012 Torch Award in recognition of his tireless efforts in promoting ARD prevention and mitigation world wide as the Co-chair of the 9th ICARD; Program Manager – Special Projects, NRCan; MEND and NOAMI Secretariats and a key member of the Global Alliance.

At the 9th ICARD, we received an outstanding proposal presentation for the 10th ICARD in Santiago, Chile. This venue was selected and plans are continuing for the next ICARD in Chile. Since this presentation, INAP was approached by the International Mine Water

Association (IMWA) with the request to join our meeting. After discussion with the INAP OpCom and Board, it was decided to invite IMWA to participate. The date for the 10th ICARD is April 20-25, 2015 in Santiago, Chile, so put it on your calendar.

The first call for Abstracts has been announced and will be due on 16 June 2014. This announcement is found at www.icard2015.com. The Technical Committee is open for membership. If you desire to participate in this important activity, please send me an e-mail including your resume and areas of interest at tchatwin@inap.com.au.

As one can see from this report, INAP has busy, since the last 7th Australian AMD Workshop, in Darwin in 2011.

SAVAGE RIVER REHABILITATION PROJECT – 2002 TO 2013

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ABSTRACT

The Savage River Rehabilitation Project (SRRP) was established by an Agreement between the Government of Tasmania and Goldamere in 1997. The objectives of the SRRP are promoting recovery of the Savage River ecosystem downstream of the Savage River Iron Ore Mine in north-west Tasmania, developing and implementing an agreed long-term strategic plan for the rehabilitation of historical acid and metalliferous drainage (AMD) at the mine, integrating rehabilitation with ongoing mining, and demonstrating best practice AMD management. The work of the SRRP Management Committee has included investigating historical AMD, developing cost-effective long-term solutions to mitigate and treat AMD, minimising acidic seepage flows, and adding alkalinity to Savage River. Regularly since 2002 the strategic plans and activities of the SRRP have been reviewed by a Review Panel. Ongoing key threats to achieving the goals of the SRRP are the potential premature closure of the mine, the loss of key project staff, the ongoing reduction in site AMD not continuing to diminish as expected, and the lack of sufficient funding to ensure cost-effective site AMD treatment. The paper describes the progress made by the SRRP between 2002 and 2013, the current best estimate of the residual historical site AMD, and the recommended further studies to address the challenges and opportunities faced by the SRRP.

1.0 INTRODUCTION

By way of introduction, the following sections provide the history and climatic setting of the Savage River Mine, and the Goldamere Agreement and the formation of the SRRP.

1.1 History and Climatic Setting of Savage River Mine

Savage River Mine has been producing iron magnetite concentrate since 1967, and pumping it 83 km to the Port Latta Pelletising Plant. The pellets produced are shipped to markets in Port Kembla and Korea, with a small quantity of unpelletised magnetite being sold to Eastern Australian coal producers for use in dense medium separation of fine coal.

The mine is located in steep, thickly vegetated terrain, surrounded by areas of high wilderness value, and the main elements of the site are shown in Fig. 1. The historical problems associated with AMD, the waste dumps, tailings dams and pits are typical of a number of old mine sites in western Tasmania in particular, and elsewhere in Australia and worldwide. The main elements of the site are the various pits, of which the North and South Deposit Pits remain active, various acid generating waste dumps, and the dis-used and acid generating Old Tailings Dam (OTD) above the current alkaline Main Creek Tailings Dam

(MCTD). The total surface areas of the pits, waste dumps and tailings dams are about 200 ha, 300 ha and 200 ha, respectively.

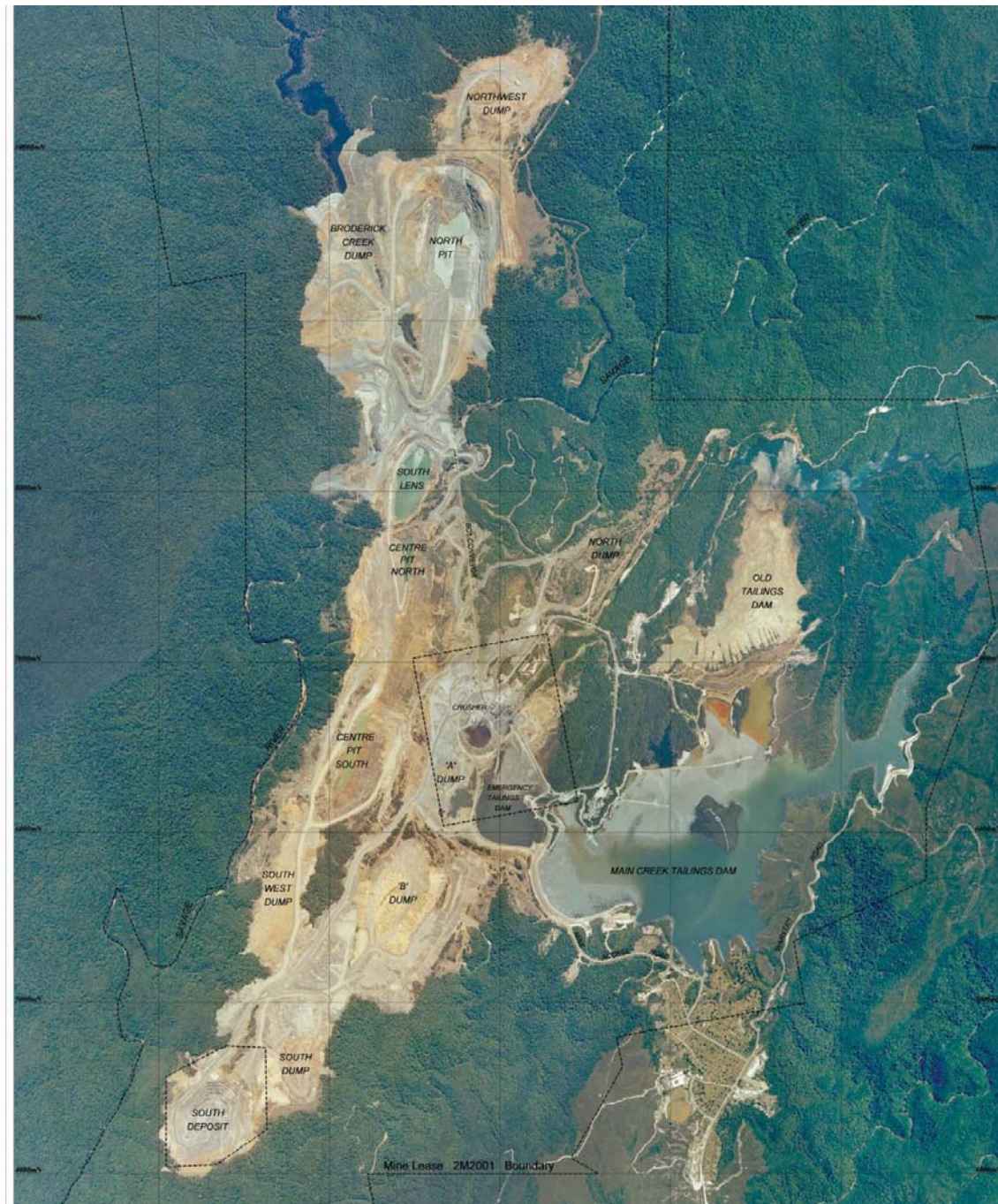


Fig. 1. Main elements of Savage River Mine site

The first 30 years of mine operations caused environmental harm to about a 30 km length of the Savage River. The reach downstream of confluence with Main Creek was found in 1995 to have lost 90% of its invertebrate diversity and 99% of its invertebrate abundance. The

principle cause was acidic drainage from the approximately 200 Mt of waste dumped on the site and exposed old tailings that had acidified, plus the effects of sedimentation.

The climatic setting of the site is cool temperate, with an average annual rainfall of 1,950 mm and an average annual pan evaporation of 650 mm. January to April are on average the driest months, and on average only in the month of January does monthly evaporation exceed monthly rainfall. Rain storms are typically of about an hour's duration, and come and go. About 90% of the rainfall comes in passing storms, rather than as continuous drizzle. Natural infiltration rates vary with the vegetation and slope. The rainforest is very damp, with little runoff. There is very much greater runoff from disturbed areas, and greater runoff from grassed areas, compared with the rainforest.

1.2 Goldamere Agreement and Formation of SRRP

The original mine operators, Picklands Mather and Co. International (PMI) ceased operations in 1996. On 26 March 1997, the Tasmanian State Government, the former operator (PMI) and incoming operator Goldamere Pty Ltd, trading as Australian Bulk Minerals (ABM, a subsidiary of Ivanhoe Mines Limited), reached a settlement assigning \$ 24 million for remediation of historical pollution. This was split between an "Environment Protection Fund", provided in cash to the Tasmanian Government by PMI, and a "Purchase Price Fund", plus interest to be provided over time by the new mine operator ABM. In January 2009, ABM merged with Grange Resources Limited (Grange).

The Goldamere Agreement between the Tasmanian Government and ABM was relaxed somewhat in 2000, accompanied by the formation of the SRRP Management Committee, comprising two representatives of the Tasmanian Department of Primary Industries, Parks, Water and Environment (DPIPWE), a representative of Mineral Resources Tasmania (MRT), and Bruce Hutchison and Stephen Kent of ABM (later Grange).

2.0 SRRP 2001 STRATEGIC PLAN

The initial SRRP objectives and their implementation, the focus of the SRRP, the initial engineering works supported in the 2001 Strategic Plan, and an overview of ongoing operations at the mine that are relevant to the SRRP, are described in the following sections.

2.1 Initial SRRP Objectives

The initial objectives of the SRRP were set in the 2001 Strategic Plan, as follows:

- To promote recovery of a modified but healthy ecosystem in the Savage River downstream of the mine, and permit fish migration into the upper Savage River.
- To develop and implement an agreed long-term strategic plan for the rehabilitation and remediation of historical disturbances at the Savage River Mine and Port Latta Plant.
- To integrate remediation works with ongoing mining operations wherever practical and to co-operate with ABM during the planning and implementation phases.
- To overtly demonstrate best practice in all aspects of the project and to communicate progress and findings to the community."

The implementation of the 2001 Strategic Plan involved the following iterative process.

- Identify the sources of contamination and contamination problems.
- Address the question: "Is remediation necessary?" – If the answer is no, then monitor the water quality to ensure that it remains acceptable.

- If remediation is deemed necessary, then address the question: “Is remediation possible through earthworks?” If the answer is no, then consider long-term water treatment.
- If remediation is deemed possible through earthworks, then identify and test the earthworks options.
- Based on the results of trial work, preferred earthworks-based remediation options should be recommended.
- The recommended options should then be implemented and monitored.
- Address the question: “Have the remediation objectives been met?”
- The earthworks-based remediation option that meets the remediation objectives is deemed to be a solution.

2.2 Focus of SRRP

The major focus of the SRRP was on the restoration of the aquatic environment downstream of the mine site. Davies *et al.* (2001) concluded that overall, toxicity in the Savage River appeared to be controlled by the combination of pH, dissolved Cu and hardness (particularly Ca). Alkalinity was assessed as being a relatively poor measure of toxic susceptibility. Aluminium and sulphate (SO₄) were also potential toxicants, but must exceed fairly high levels to pose toxic risks. Environmental Targets constitute the remediation objectives, and for the Savage River downstream of the mine these are given in Table 1 (based on Davies *et al.*, 2001), irrespective of discharge or pH. Use of these thresholds as initial “stage 1” remediation targets was deemed to result in a measurable degree of aquatic biological recovery from ARD-related toxicity. In addition, the airborne dust target for the Port Latta Pelletising Plant is PM₁₀<0.050 mg/L (based on a 24-hour average at Cowrie Point subjected to an easterly wind).

Table 1. Environmental Targets for Savage River downstream of mine (based on Davies *et al.*, 2001)

Parameter	Environmental Target Level			
	<i>At all times</i>	<i>When Ca<5 mg/L</i>	<i>When 5<Ca<10 mg/L</i>	<i>When Ca>10 mg/L</i>
Dissolved Cu		<0.035 mg/L	<0.050 mg/L	<0.070 mg/L
Ca:Mg mass ratio	~1:1			
Dissolved Al	<0.61 mg/L			
Total SO ₄	<450 mg/L			

Examination of the water quality data from Savage River at the South West Dump (Davies *et al.*, 2001) indicated that the toxic thresholds given in Table 1 were exceeded about 25% of the time, primarily by dissolved Cu, but occasionally also by high Al or SO₄. Examination of the water quality data from Savage River at Doodies’ Creek indicated that the toxic thresholds given in Table 1 are exceeded about 20% of the time, primarily by dissolved Cu, but occasionally also by high Al. Based on these assessments, the Performance Indicator Targets for the Savage River downstream of the mine given in Table 2 were set by the SRRP Management Committee.

It was estimated that about 65% of the dissolved Cu released from the mine would need to be treated. The estimated % contributions to the total dissolved Cu load from the different

elements of mine disturbance are given in Table 3, from which it is clear that the A and B Dumps and the Main Creek Tailings Dam (MCTD) combined contributed the largest proportion, followed by the North and South West Dumps.

2.3 Initial Engineering Works Supported

The engineering works supported in the SRRP Strategic Plan 2001 included clay capping, water covers (over tailings) in-pit where possible, diversion of clean water (to avoid contamination) to make long-term (15-year) treatment of polluted water viable, gravity drainage of acidic water, utilising on site sources of alkalinity (e.g. fresh tailings, magnesite, and carbonate-chlorite schist), trialling emerging technologies and revegetating non-polluting areas, carried out as far as possible as part of ABM's operations.

Table 2. Performance Indicator Targets for Savage River downstream of mine

Measure/Activity	Performance Indicator Targets (to year ending 26 March)		
	2002	2003	2004
Time proportion Savage River ETs achieved (average below SW Dump and Smithton Road Bridge)	78% of time	80% of time, assuming flow from Crusher Gully is collected and treated	82% of time, assuming North Dump Drain flow is collected and treated after early 2004
Fish and macro-invertebrate community downstream of mine	Statistically verifiable improvement over 1995	-	Statistically verifiable improvement over 2000/02
Number of days ET is exceeded at Port Latta	<10 days	<7 days	<7 days

Table 3. Estimated % contributions to total dissolved Cu load from different elements of mine

Source	Estimated % contributions to total dissolved Cu load
A & B Dumps and Main Creek Tailings Dam	38.5
North Dump	18.7
South West Dump	17
Crusher Gully	6+
Old Tailings Dam acidic seeps	0.5
Old Tailings Dam North	3.7
Pits combined	~3
Broderick Creek alkaline flow-through (~39% of all waste rock)	6.8

2.4 Relevant Ongoing Operations at Mine

ABM identified the following four waste types from the operating pits at Savage River:

- *A waste*: competent, NAF carbonate-chlorite “schist”, used for flow-throughs (formed by end-dumping at 45° from both abutments), and magnesite;
- *B waste*: low PAF material, used for road building;
- *C waste*: “clays”; and
- *D waste*: pyritic and Potentially Acid Forming (PAF) with up to 8% S, the proportion of which increases with increasing pit depth, which is end-dumped and encapsulated in paddock-dumped clay in typically 15 m high, compacted, flat lifts.

The tailings are initially alkaline, and would remain so provided they are flooded. Concern was expressed about the following:

- whether the Old Tailings Dam (OTD) acidic seeps would turn the Main Creek Tailings Dam (MCTD) acidic, particularly if the mine closes prematurely;
- whether the Broderick Creek flow-through would remain alkaline and continue to flow (or clog);
- whether or not clay covers on acid generating dumps would reduce their acid generation through limiting rainfall infiltration and possibly limiting oxygen ingress; and
- the possibility that mining operations would close prematurely due to low demand and low prices for pelletised magnetite, which would have potentially serious implications for the SRRP.

3.0 SRRP 2003 STRATEGIC PLAN

The 2001 SRRP objectives have been modified in successive Strategic Plans, prepared by the SRRP Management Committee in 2003 and 2012. The major changes since the SRRP 2001 Strategic Plan were the completion of a whole-of-site feasibility study, the 2002 Peer Review and a major extension to the expected mine life. The Strategic Plan involved the use of the cash under the “Environment Protection Fund” to pay for running the SRRP and for related R&D, and to use the in-kind “Purchase Price Fund” to build infrastructure.

Stage 1 of the 2003 Strategic Plan was required, within cost constraints with current technology, to at least retain the status quo after mine closure, include R&D for Stage 2, and not limit the options for Stage 2. Stage 2 was intended to design the North Dump Drain to South Lens Pit, design the B Dump cover, carry out a feasibility study for a mini-hydro scheme, and design a full-scale neutralisation facility, moving to construction if feasible.

4.0 SRRP 2012 STRATEGIC PLAN

The SRRP 2012 Strategic Plan described the major works completed since the 2003 Strategic Plan including the North Dump Drain and the B Dump covers. About a 47% reduction in copper load to the Savage River since 2003 was reported, mainly due to the Broderick Creek alkaline flow-through. An improved understanding of the behaviour of the OTD and B Dump was achieved, with the OTD, B Dump, South-West Dump and Crusher Gully being remaining sources of AMD.

5.0 KEY REVIEW PANEL FINDINGS AND RECOMMENDATIONS

The Review Panel has met with the SRRP Management Committee and the Savage River Mine Operators in 2002, 2005, 2009 and 2013. The Review Panel has comprised Dr David Williams (ongoing), Dr Ward Wilson (ongoing), Dr Stuart Miller (to 2005), John Miedecke (2002 and 2009), Dr Chris Humphries (in 2002), and Peter Scott (from 2009). The key findings and recommendations of the various Review Panel sessions are summarised in the following sections.

5.1 2002 Panel Review

The 2002 Review Panel concluded that the SRRP Management Committee is uniquely placed to establish a best practice model for AMD management at the Savage River Mine. The identified potential high risks facing the SRRP were the OTD acidic seeps turning the MCTD acidic and Broderick Creek flow-through turning acidic, particularly if the mine closes prematurely. It was also noted that the acidity from the historical waste dumps had essentially peaked, that the Savage River turbidity had improved since the mine installed an ore conveyor and contained runoff in the dis-used South Lens Pit, that revegetation of waste dumps was apparently not limited by AMD or the slope angle, and that dust potential at Port Latta was being addressed by ABM. Sulphate was postulated to have peaked in about 1982, and the minimum pH (and peak dissolved metal load) was postulated to have occurred in about 1990.

Discussion took place about future waste dumping options. These included a continuation of the Broderick Creek flow-through to maximise the addition of alkalinity to the Savage River, the extension of the South West and B Dumps above the historical waste dumps, with the final profile rehabilitated progressively, and the extension of B Dump eastwards into the Main Creek valley incorporating an alkaline flow-through on the Main Creek side slope. Discussion also took place about future tailings disposal options. These included the progressive raising of the MCTD maintaining the capability for a final water cover over the tailings, co-disposing tailings with acidic water in the MCTD as a treatment option for acidic water, and co-disposing tailings with waste into the Main Creek valley, which was not considered to be practicable.

Comments were made about the various waste dump cover options under consideration. These included a clay seal, a rainfall-shedding cover, an alkaline cover, a vegetated cover, a rip rap cover, and a water cover over tailings. The available "clays" were considered amenable to constructing seals on benches and short slopes, provided that good compaction was achieved. A shedding cover requires a slope of 2(horizontal):1(vertical) or flatter, but this would double the catchment compared with an angle of repose dump slope and hence would encourage erosion. Revegetation, armouring with rip rap, and sheeting with alkaline waste would be possible on angle of repose dump slopes. Overall, alkaline flow-throughs, sealing flat dump surfaces, and alkaline sheeting of angle of repose slopes were considered the preferred options. Comments were also made about various acidic water treatment options under consideration. These included treating the OTD acidic seeps with alkaline fresh tailings (in the MCTD), magnesite neutralisation either in a mobile or fixed fluidised bed or in drains lined with magnesite, and lime addition with the need to dispose of the resulting sludge. It was noted that none of these treatment options would reduce sulphate concentrations.

The recommendations of the 2002 Review Panel were as follows:

- To mitigate the OTD acidic seeps, by a combination of an expanded clay cover on the southern wall, a clay cover over the upper tailings beach, and the extension of the water cover over the lower tailings beach.

- To manage the Broderick Creek flow-through to maintain alkaline flow-through conditions.
- For the historical dumps, to divert clean water and to apply alkaline sheeting to angle of repose slopes, including the incorporation of historical dumps within ongoing dumps, while clay seals and rainfall-shedding covers on slopes were considered to offer little benefit.
- For the current dumps, alkaline flow-throughs surrounded by clay to isolate them from PAF waste and acidic water were preferred, incorporating clay seals on flat surfaces.

Issues considered by the 2002 Review Panel to warrant further studies included the following:

- a formalised qualitative risk assessment;
- a cost-effectiveness analysis of proven technologies for AMD mitigation and treatment;
- further development of empirically-based performance models for aquatic fauna;
- geochemical quantification of waste types and production schedules; and
- optimisation of waste placement and the utilisation of alkaline and clay materials.

5.2 2005 Panel Review

The 2005 Review Panel concluded that through the highly effective collaboration between DPIPWE and ABM staff, the SRRP had demonstrated industry best practice AMD management, including the identification and selective placement of waste types, operating to mitigate historical and future AMD, and acidic water management and treatment innovation.

The potential high risks facing the SRRP were revised in the 2005 Panel Review to include the following:

- the loss of key DPIPWE and/or ABM staff, which constitute an excellent and extremely committed team, who together command irreplaceable corporate knowledge;
- premature closure (before 2007) of the mine due to a catastrophic pit failure;
- failure to prevent the MCTD from going acidic, whether or not due to the OTD acidic seeps; and
- extreme rainfall during the operation or after the end of the operation.

Of the potential high risks identified in the 2002 Panel Review, the 2005 Review Panel concluded that the Broderick Creek flow-through was expected to produce neutral drainage in the long-term, and premature closure (before 2009) of the mine for reasons other than catastrophic pit failure was considered unlikely. Further, a trial rainfall-shedding cover on the Savage River side slope of the South West Dump was found to be beneficial in reducing copper loads, and the limited funds remained a key threat to the SRRP achieving its goals.

From 2002 to 2005, copper and sulphate fluxes at the pump station continued to decline, although at a decreasing rate. The copper flux reduced from 9 to 3 kg/day and the sulphate flux reduced from about 5 to 3 kg/day. Overall, the sulphate flux reduced by over 5-fold since the mid-1980s to 2005, consistent with the “burning out” of the North Dump since that time.

As a result of the Broderick Creek flow-through, the site copper flux decreased from about 30 to about 10 kg/day over the 2 years to 2005, while the site sulphate flux increased from about 9 to about 13 t/day, or 300 mg/L compared with a target of 500 mg/L and neutral drainage

sulphate saturation of about 2,000 mg/L, perhaps due to expansion of the flow-through. Alkalinity addition consistently achieved targets from 2000.

For acidic water treated in South Lens Pit, the dissolved copper concentration was low, and the sulphate and manganese concentrations did not exceed toxicity targets except for exceedingly low river flows ($<1 \text{ m}^3/\text{s}$). There was little change in the dissolved copper discharged from the MCTD, indicating that the OTD seeps are being neutralised by fresh alkaline tailings in the MCTD, but there was an increase in sulphate (from 200 to almost 300 mg/L).

Over the period 2002 to 2005, downstream toxicity targets for dissolved copper and calcium to protect fish were achieved under all flows, but targets to protect sensitive invertebrates have only been achieved during low flows ($<5 \text{ m}^3/\text{s}$), which occur about 40% of the time. Elevated Total Suspended Solids were recorded. Due to a lack of monitoring data in the Main Creek catchment, it remained not possible to identify water quality trends in this catchment.

Based on the water monitoring and flow data available to 2005, acid and copper loads from the site were postulated. It should be noted that site flows are not entirely known, due to the site's high topographic relief and high flows. Concentrations are somewhat better known, where they are measured. The site water quality monitoring data are incomplete in areal extent and over time. Not all historical acid loads are captured and there have been gaps in the monitoring time series. As a result, AMD loads (the product of flow and concentration) are only approximately known.

Figure 2 postulated that the historical waste dumps and pits, the OTD, and the MCTD in the future if it were allowed to go acidic, would contribute approximately equal worst case acid loads. The acid load from the waste dumps and pits were estimated to have peaked first, in about 2002, the OTD acid load was estimated to peak in about 2008, and the MCTD was assumed in the worst case (with no intervention) to go acidic and reach a peak acid load in about 2045.

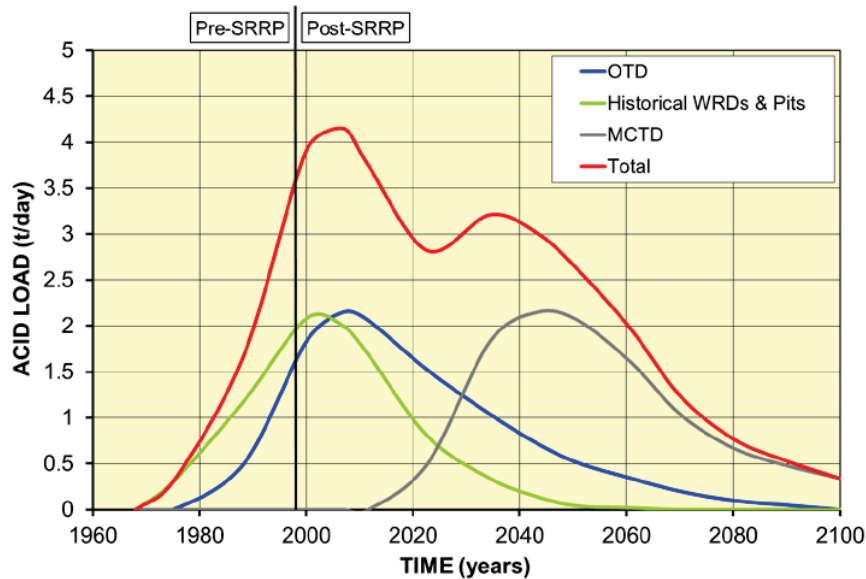


Fig. 2. 2005 worst case projections of acid loads

Figure 3 postulated that the historical waste dumps and pits were the main source of the copper load, estimated to have peaked in about 1995. There was also some copper load from the super-elevated OTD, estimated to have peaked in about 2000, with negligible copper load assumed from the MCTD, even with no intervention to prevent it from going acid.

5.3 2009 Panel Review

The 2009 Review Panel concluded that Grange and DIPWE had demonstrated excellent progress in achieving the objectives of the SRRP. Improvements were clearly evident in all aspects of waste and water management. The North Dump Drain diversion to South Lens Pit where it is passively neutralised had been successfully implemented, reducing loadings to Savage River. The rainfall-shedding cover on B Dump was successfully diverting rainfall runoff and reducing infiltration, and the alkaline sheeting on the Main Creek side of B Dump was adding alkalinity to Main Creek during flow-through following heavy rainfall.

By the time of the 2009 Panel Review, AMD generation from the historical dumps has peaked and loads were reducing. The water quality in Centre Pit was improving and the water quality in South Lens Pit was stable and alkaline. The OTD acidic seeps were being masked by passive neutralisation in the MCTD, but remained the major threat, together with the potential for the MCTD tailings to acidify post-closure.

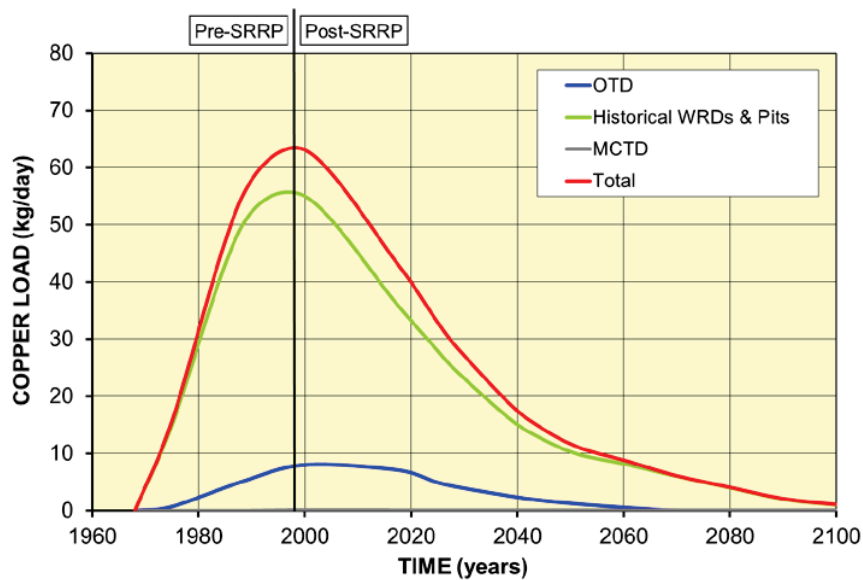


Fig. 3. 2005 worst case projections of copper loads

Key threats to the SRRP achieving its goals were identified in the 2009 Panel Review as being the following:

- the loss of key DIPWE and Grange staff and Consultants;
- the premature closure of the mine, due to the iron ore price and/or an adverse exchange rate;
- failure to effectively address the OTD acidic seeps;
- potential acidification of the MCTD post-closure;
- potential need for water treatment post-closure; and

- potential excessive water treatment expense and huge sludge volumes required to be stored.

Key opportunities for the SRRP were identified in 2009 as being the following:

- the extensive corporate knowledge and commitment;
- expected operation of the mine to 2023, with adequate financial backing and management;
- opportunity to address the OTD acidic seeps through ongoing operations;
- opportunity to engineer effective closure of the MCTD;
- passive treatment and scale-up of neutralisation; and
- cost-effective and minimised water treatment and sludge management.

Possible strategies identified in 2009 for mitigating the environmental impact of the historical waste dumps include the following:

- monitoring and reviewing the long-term effectiveness of water-shedding, non-acid generating top covers;
- establishing the split between runoff and infiltration in the long-term;
- potential for and impact of erosion and vegetation of the dumps;
- confirming the effectiveness of infiltrative, alkaline side slope covers;
- understanding the geochemistry of the dumps in the short and long-terms;
- better defining and minimising dump outflows; and
- taking advantage of opportunities for integrated waste and tailings storage.

After a few weeks of heavy rainfall, the water level upstream of the 20 m deep Broderick Creek alkaline flow-through rose from 16 to 26 m, and flow was observed downstream within a few hours of heavy rainfall. With no further rainfall, the flow-through began to drain down after a few days.

Possible strategies identified in 2009 for ongoing tailings storage include the following:

- raising the MCTD wall by 2.5 to 10 m (to a maximum RL of 338 m), insufficient in itself to accommodate tailings to 2023.
- constructing a new tailings storage in the lower Main Creek, should South Deposit be re-mined.
- raising the MCTD wall to RL 338 m and storing tailings against and over the flattened southern wall of the OTD to accommodate tailings to 2023 and effect an oxygen-limiting cover over the exposed acid generating OTD tailings.
- hydraulically placing covers over the MCTD tailings, using depyritised, thickened tailings.

5.4 2013 Panel Review

The 2013 Review Panel concluded that the site is continuing to perform well. The MCTD tailings continue to passively neutralise the OTD acidic seeps. Grange had commenced installing a pipeline to divert the OTD acidic seeps around the MCTD post-closure to ensure that they do not acidify the MCTD tailings, which it is proposed will have a water cover to maintain them saturated and non-acid. The Broderick Creek alkaline flow-through has dramatically raised the alkalinity being delivered to Savage River, substantially reducing copper loads, and is regulating flows. The North Dump Drain discharge continues to be passively neutralised in South Lens Pit, although South Lens Pit is recognised to have a finite neutralising capacity. The B Dump clay and alkaline covers and alkaline flow-through are performing better than expected, and the South-West Dump rainfall-shedding cover is a demonstration of good revegetation, but unproven acid-reduction performance.

Key threats to the SRRP achieving its goals were identified in 2013 as being the following:

- the limited project funding;
- the OTD acidic seeps;
- premature mine closure;
- the loss of key staff;
- inadequate timely tailings storage capacity;
- the residual B Dump copper load;
- the Crusher Gully copper load; and
- the limited acid neutralising capacity of South Lens Pit.

Key opportunities for the SRRP were identified in 2013 as being the following:

- ongoing integration with the Grange Life-of-Mine-Plan;
- ongoing MCTD neutralisation of the OTD acidic seeps;
- neutralisation of the B Dump acidic seeps by the proposed South Deposit Tailings Storage Facility (SDTSF);
- potential cash flow generated by a possible mini-hydro scheme; and
- affordable acid treatment.

On the basis of the improving site water quality, the effect of the reducing hydraulic gradient driving the OTD acidic seeps as the MCTD tailings level continues to rise, and assuming that the MCTD tailings are prevented from becoming acidic, the acid and copper loads from the site were re-postulated downwards, as shown in Figs 4 and 5.

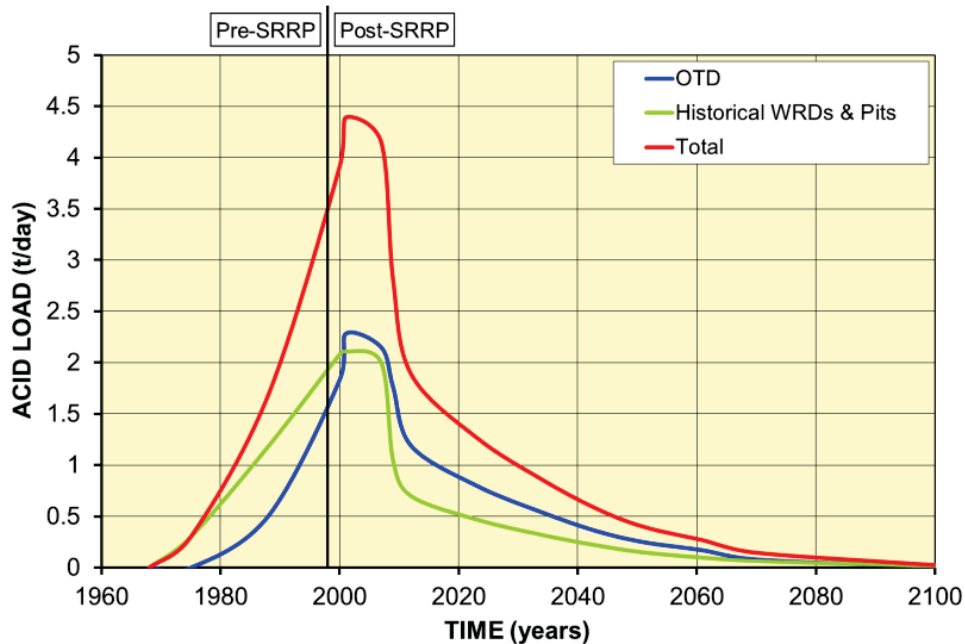


Fig.4. 2013 projections of acid loads

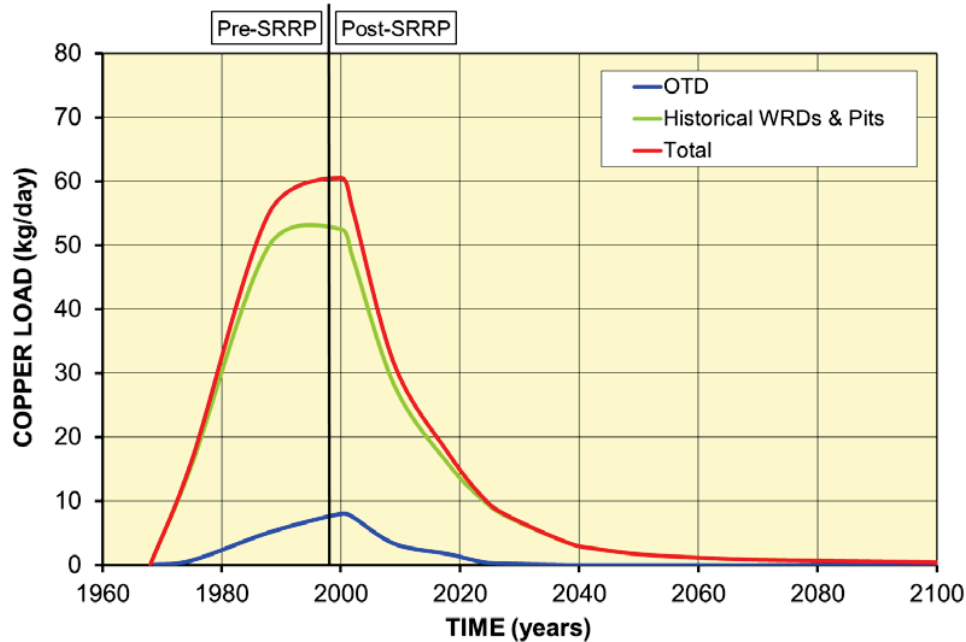


Fig.5. 2013 projections of copper loads

Further studies must be well-targeted and executed, and aimed at cost-effectiveness, and include the following recommendations of the 2013 Review Panel:

- geophysical techniques warrant further application to the OTD/MCTD interface (and MCTD wall studies);
- water balance of OTD tailings, including rainfall runoff monitoring and effect of hardpan and vegetation;
- consolidation testing of coarse and fine-grained tailings for dry density vs. saturated hydraulic conductivity, and Soil Water Characteristic Curve testing;
- further secondary mineralisation assessment; and
- Targeting integrated SRRP/Grange/Consulting/Research Masters Projects.

6.0 SOME PICTORIAL COMPARISONS OF SRRP ACHIEVEMENTS OVER TIME

Figure 6 shows the dramatic development of the Broderick Creek alkaline flow-through from 2005 to 2013, with the current end of the flow-through shown in Figure 6b indicated by the arrow in Figure 6a. Figure 7 shows the greatly increased revegetation of the thinly clay-capped southern wall of the OTD from 2002 to 2013. Figure 8 shows the greatly increased revegetation of the clay-capped fingers on the upper beach of the OTD from 2002 to 2013.

7.0 CONCLUSIONS

The progress made by the SRRP Management Committee over the period from 2002 to 2013 is summarised in Table 4.

Table 4. Summary of SRRP progress from 2002 to 2013

Year of Panel Review	SRRP Progress
2002	The SRRP Management Committee is uniquely placed to establish a best practice model for AMD management
2005	Through highly effective collaboration the SRRP has demonstrated industry best practice AMD management
2009	Continued highly effective collaboration between Alison Hughes, Bruce Hutchison and Stephen Kent, and their Consultants, demonstrating leading practice AMD management
2013	Ongoing highly effective collaboration and innovation between Alison Hughes, Bruce Hutchison and Stephen Kent, and their Consultants, achieving leading practice AMD management

8.0 ACKNOWLEDGEMENTS

The generous contributions of the staff of DPIPWE, ABM/Grange and their Consultants to the Review Teams is gratefully acknowledged.

9.0 REFERENCES

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Fig. 6. Development of Broderick Creek alkaline flow-through looking upstream from: (a) 2005 to (b) 2013



Fig. 7. Greatly increased revegetation of thinly clay-capped southern wall of OTD from: (a) 2002 to (b) 2013



Fig. 8. Greatly increased revegetation of clay-capped fingers on upper beach of OTD from: (a) 2002 to (b) 2013

COMMERCIAL IMPLEMENTATION OF MINE CLOSURE RESEARCH BRINGING TOOLS TO PRACTICE FOR MANAGEMENT DECISIONS

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ABSTRACT

It is common for closure research and closure planning support studies results to go uncommunicated and/or not be properly integrated into a mine site's closure objectives. The result is that information arising from the research is not implemented on a commercial scale, despite tens to hundreds of millions of dollars spent annually on the research and studies. Often, other impediments are at the heart of inaction, such as a lack of understanding, insufficient opportunity to distil and assess all of the implications of the body of research, corporate and regulatory hurdles, and/or a general resistance to change.

In order to make management decisions involving substantial investment, decision makers must have access to clear, concise, timely, and focused messages. Hence, technology transfer is not simply submission of a report, a thesis, a conference paper, or even convening a workshop. Rather, a systematic approach to moving research results to commercial scale must be utilised; otherwise there is a high risk that investment in research will not be realized, and research efforts will be minimised in the future.

1.0 INTRODUCTION

It is our experience that too often, the results of earnest, thoughtful, expensive, and successful mine reclamation research and development (R&D) is not effectively communicated, heard, evaluated, and implemented at mine sites. At all but the smallest and simplest mines, mine closure research is an important part of meeting mine closure objectives reliably and cost effectively. Much of applied research is ultimately aimed at developing new technologies for implementation at commercial scale, or incremental improvements to existing technologies.

Knowledge gaps or ideas for research usually come from the environmental staff in the mining industry, but may also be highlighted by engineering or operations staff, mine management, stakeholders, regulators, or indigenous groups; it may even come from consultants or third-party vendors. The resulting research project may be informal or formal, funded or not. Often research just involves people 'trying things' in the field (for example, vegetation plots with different kinds of seedlings), or a small group working together to try out a new operational idea (creating a rockpile and observing wildlife activity). Other times it is a formal project, with funding for staff, a consultant, and/or university researchers (including students), often with formal experimental design, a budget and schedule, and the anticipation of a report or thesis.

Many researchers and most students assume that publishing a paper, producing a report or a thesis is the final result, which will then somehow be discovered and adopted by the mine or mining industry and the idea will take off and spread far and wide; this is almost never the case. The research results usually languish on a shelf, test plots are forgotten because the

students or researchers move on, and the mine staff may move on during the research period. Moreover, the results can still languish for years or decades if the results are not well communicated to decision makers and those who would implement the R&D results. In some cases, the R&D is applied, but is then forgotten or falls out of favour.

In order to make management decisions involving substantial investment, decision makers must have access to clear, concise, timely, and focused messages. A technology transfer plan (or program) can bridge research results and decision makers. Research is focused on answering specific questions to address closure plan objectives and provisions set out for properly focused upscaling of research through pilot and demonstration phases in the field. Hence, technology transfer is not simply submission of a report, a thesis, a conference paper, or even convening a workshop. Rather, a systematic approach to moving research results to commercial scale must be utilized.

This paper presents an approach to streamline the process and develop a technology transfer plan within a mining company or a mining region. Simple strategies and methods for development and commercial implementation of promising research technologies can substantially multiply the return on investment of the initial research. Most importantly, the approach provides the opportunity to distil and communicate research results to practitioners and decision makers in ways that they can understand and execute in the field at a commercial scale. A simple case study demonstrates the process of technology screening in a technology transfer plan.

2.0 FROM RESEARCH TO COMMERCIAL IMPLEMENTATION – SCALE ISSUE

The most common challenge when applying a research result in a commercial scale is the issue of scale, both spatial and temporal. Promising outputs from a research project may not be favourable to a commercial project due to a number of factors such as different boundary conditions, construction constraints, time constraints, and/or economic constraints. Common problems for tailings and water treatment technologies include difficulty in converting from batch to continuous processes, retrofitting existing operations, operating under a wide variety of climatic conditions, or parts of the technology are too complex to be run in an ever changing mining environment. Examples of difficulties in upscaling reclamation research is the use of large mining equipment to place reclamation material and the inherent variability of the soil profiles that result, as well as difficulty in plant establishment, particularly in wetland situations where access is limited. In all of these situations, what is easy in the laboratory is much more challenging in the field, requiring substantial changes to the initial reclamation plan, and in some cases abandoning promising technologies as impractical or unworkable.

Technology development is usually designed to gradually upscale investigation scales at research levels to commercial levels. Figure 1 presents various test scales for research, development, and commercial waste rock piles (or dumps). During technology development processes, promising technologies screened from research results are further verified for commercial implementation. In addition to scale difference, technology development also has different objectives compared to research and commercial scale studies (Table 1).

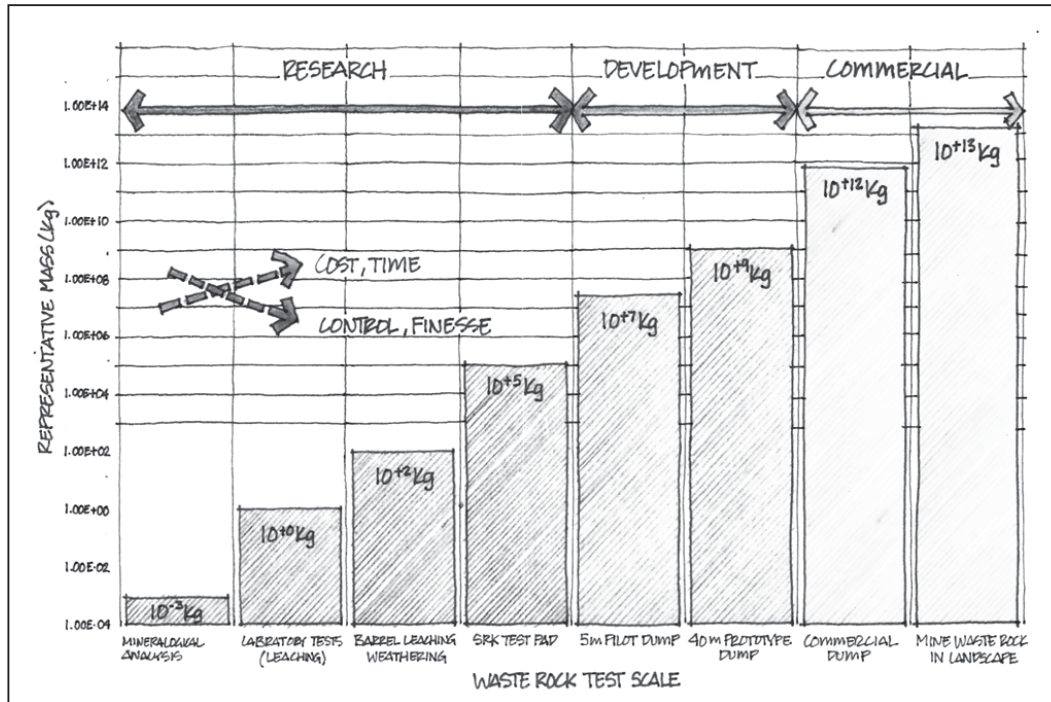


Fig. 1: Waste rock test scales

3.0 TECHNOLOGY TRANSFER PROGRAM

A technology transfer program or plan is critical in the process of transferring a promising research technology to commercial implementation. First and foremost, it helps the mining industry (decision makers). Through the technology transfer plan, technologies are screened and R&D programs are properly directed at the very beginning from the onset of the mine closure. Implications of R&D results to the mining company or mining industry can be accurately and concisely explained and interpreted, and hence, easily understood by decision makers. As a result, decision makers can obtain enough timely information to make decisions. Secondly, the technology transfer plan also helps researchers. Once R&D results and implications are understood by decision makers, there are more opportunities that R&D programs can keep going and the promising technologies finally reaching commercial implementation.

A technology transfer plan should not be conducted only at the end of a research or development program. It should be planned and applied before an R&D program. "Technology transfer" should be considered as a continuous process guiding the implementation of a mine closure plan. Figure 2 presents a flowchart incorporating main steps in a technology transfer plan.

Table 1. Comparison of R&D and commercialisation of technologies

Phase	Scale	Activities	Objectives
Research	Bench / Laboratory / Greenhouse scale	<ul style="list-style-type: none"> • develops a specific technology; • may be aimed at gaining a better understanding of fundamental processes 	i) focuses on understanding and assessing technologies; ii) concentrates on the potential upside of the technology; and iii) is not closely tied to economics.
Development	Pilot	<ul style="list-style-type: none"> • Involves a promising, well-researched technology that has moved from the research stage and is ready to be tested at field scale, typically many orders of magnitude greater size or fluxes than practical in the laboratory 	i) focuses on scale up, performance under continuous operation, and includes environmental and economic assessments; ii) aims to identify fatal flaws; iii) provides opportunity to learn how to design, construct, operate and implement technologies and characterizing the environmental impacts to the degree where it can be permitted at commercial scale
	Prototype	<ul style="list-style-type: none"> • Prototype field tests are permanent and reclaimed using commercial scale equipment. • The tests are large enough to gain confidence to move to commercial scale. • Prototype tests are designed to allow optimization of design for commercial, to fit into mining and closure plans, receive regulatory approval 	i) focuses on testing at full scale, developing parameters and optimizing design for commercial operation; ii) gains commercial-scale experience for operators
Commercial	Full size	<ul style="list-style-type: none"> • involves normal operations running at full scale for many years 	i) focuses on safe, reliable, cost effective operation, with continuous improvement

Ideally, a technology moves through these stages over several years, with the most promising technology making it all the way to commercial scale. There are risks in skipping any of these steps.

3.1 Develop Questions

A good question can provide direction to guide planning and conducting a research program. Questions from operators are likely the most practical to mines and may be less academic to researchers. For example, operators' questions might be similar to the following site-specific research questions: "What is the optimum reclamation material thickness for supporting good conifer tree growth?", "What cover design will minimize the net percolation into a waste rock dump over a 20 year period?", "What factors govern the chemical weathering of waste rock in a dump", "What is the highest fines content that a tailings slurry can be and still provide reasonably rapid consolidation in the field?".

Academically minded researchers might also be attempting to answer broader questions with the same work. Questions such as: "How does layering affect moisture movement in a cover system?", "How do we take into account natural climate cycles in the design of cover systems?", "What are the mechanics and rates of chemical weathering of waste rock result in geochemically stable clay mineral formation?", "Is there a predictable relationship between laboratory and field consolidation rates?".

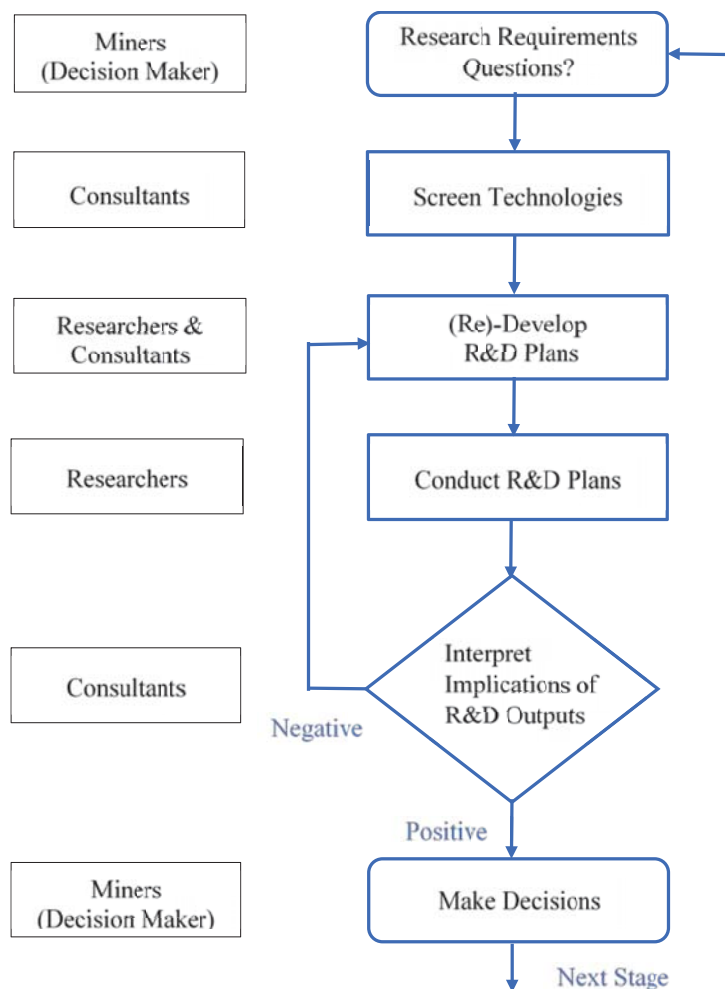


Fig. 2 Technology transfer flowchart

Sometimes researchers can tap into additional or matching research funding from other organizations and may have additional questions that they are researching that at first may not seem that applicable to the operations questions first posed. For example: “How can isotopes be used to understand the age of waters within a waste rock dump?” or “What are the mechanisms by which mycorrhizal fungi accelerate tree establishment on a newly reclaimed site?” Serendipitously, answers to these research questions often prove as useful to a mine as the original questions even though the links will often appear distant at first.

A technology transfer program starts from helping operators develop questions that are important to mines and have academic implications (i.e. can be academically answered). The benefits to transfer a practical question to a more academic one are that i) the answer of the academic question may be existing in literature, ii) relevant technologies (knowledge) associated with the academic question may be existing although there is no direct answer to the question, and iii) R&D plans can be framed in the proper technology context if there are no existing technologies to the question.

3.2 Screen Technologies

The purposes of technology screening are to i) exclude unsuitable technologies that cannot satisfy the miner's requirements or cannot answer the miner's questions, and ii) make a short list of technologies that are most promising to answer the operator's questions for further R&D plans. An unsuitable technology to a site-specific mine does not necessarily mean that the technology is not technically applicable to the mine site. Economical infeasibility can also make a technology improper to a specific mine site although the technology may have been successfully utilised in other mine sites. Climatic conditions, material constraints, construction difficulties, and other site-specific constraints can affect applicability of the technologies at the mine site.

Technology screening is crucial to both operators and researchers. On the one hand, some apparently unsuitable technologies (usually due to economical infeasibility) can be excluded at the very beginning of the mine closure planning and reclamation, thus the operator's available funds and time can be focused on more promising technologies. In contrast, R&D plans can obtain direct guidance and the right direction to proceed after technology screening, which will save researchers' time and quite possibly enhance implementation of R&D results at a commercial scale.

The group that conducts technology screening must have a comprehensive understanding of the technologies themselves and the mine site where the technologies will be applied. In addition, this group also needs a very good sense of technology development associated with mine closure and reclamation because the actual reclamation might be several years or decades later. It would usually be the best to have a panel of experts to do technology screening for a mining company or mining region.

Failure models and effects analysis (FMEA) is a useful tool that can be utilised to screen technologies for mine closure. FMEA (Robertson and Shaw, 2003) is generally an expert system approach for risk identification and quantification, and mitigation measure identification and prioritization. Through the process of conducting an FMEA, the mine closure plans and correspondingly applied technologies are evaluated, and promising technologies with low or controlled risks will be selected for further development and commercial implementation.

Promising technologies should be evaluated with a commercial eye in technology screening. What would commercial implementation look like? Using optimistic assumptions, would field application of the technology be practical, economical, or useful? Hence, a costing analysis must be completed at this stage; in essence, an analysis that is less about quantifying all benefits, but rather focuses on the technologies that make most economic sense and/or the group is most confident in with respect to implementation. The costing analysis would assume that each of the technologies are technically feasible, given the set of assumptions on performance, and evaluate what technologies have the most promise from a net present cost perspective. The question of whether it could be acceptable to regulators or stakeholders can be part of the FMEA evaluation.

3.3 Develop R&D Plans

Extracting more value and more results from R&D requires thoughtful design and systems. Arguably, the most important aspect of successful research is having a strong multidisciplinary team with a strong leader and visible support from mine management and operations. There are five conditions that are essential to success for R&D programs:

- A well-structured mission with clear questions to answer;
- Strong leadership to provide direction and support but also allow independence;
- Exceptional researchers (and students working closely as a team);
- Proper methodology employed in investigations including spatial and temporal scales; and
- Reliable funding not only for the students, but also the field support.

The authors' experience is that the best approaches for large-scale testing is a university team research approach, drawing upon strong, internationally recognized principal investigators from leading universities supervising a team of up to about a dozen graduate students working towards master's and doctoral theses. Work is supported by consultants and contractors and scientific and engineering mine staff (who may be also performing parallel research). Successful examples of this approach include studies at Syncrude Canada (Kelln et al, 2009), Chevron Questa Mine (Bucknan et al, 2009), and at Teck Coal (Strategic Advisory Panel on Selenium Management, 2010). Such university-based programs cost several million dollars per year and run five years or more. For these larger programs, it is often useful to have an independent review team of scientists and engineers (e.g. Bucknan et al., 2009). In short, allow each component of the team to be responsible for the tasks "they are good at". For example, in the case of developing a large scale waste rock prototype trial and/or cover system trial, it would make sense to utilise an experienced consultant and mine site staff to implement the design, which has been developed by the team as a whole. Then, the project is 'handed over' to the researchers. A common mistake is to have researchers be responsible for annual monitoring reports. In the authors' experience, students and researchers should focus on conducting the research in the field and 'mining' the data to a greater extent than is typically feasible from a consulting perspective. The consultants and mine site staff can then focus on providing the 'hard' deliverables (monitoring data management and high level assessment) associated with the research and researchers can focus on their key deliverable; namely, their thesis. Site staff, and especially consultants with a long term involvement with the site, should oversee the annual monitoring and data management and compilation to ensure that there is continuity in the knowledge base from student to student, and that all of the relevant data are remain available through time in a consistent form.

The above issues apply equally to smaller research programs; there are just fewer researchers involved, and generally answering a shorter list of questions.

3.4 Conduct R&D Plans

The execution of R&D plans must ensure that the key knowledge objectives are attained. However, it needs to be recognised by all parties that some flexibility will be required in precisely how certain logistical elements are implemented. Thus the responsibilities of conducting site-based R&D should not be totally taken by researchers. Whilst researchers may be skilled at undertaking the logistics of implementation of a research project at the laboratory scale, it is vital to engage the expertise of site staff, and accommodate constraints that operational logistics impose, for the design of a plan for upscaling the research to mine

site scale. These site constraints may at times outweigh what is considered by a researcher's focussed perspective to be the best way to install instrumentation or to implement a monitoring program.

The types of boundary conditions that were applied to research conducted at laboratory scale must be carefully considered as these will have an important bearing on how, and indeed whether, the research idea can be successfully upscaled. Sensitivity tests of the influence of boundary conditions on performance should always be considered and carefully handled in R&D programs.

3.5 Interpreting Implications Research Results

In general, an annual research report will be factual and focus on 'what has happened during the past year. However, what is said in this report may not be perceived to have much practical meaning by mine operations staff unless the operational implication of what has been found is delivered to them in an appropriate format. The implication interpretation of timely research results should be designed to give the miners results each year, in a form that enables them to be either used in current practice, or to be scheduled for implementation in future years. It is the authors' experience that whilst there are people who are reasonably patient, and understand the need for a research program to run its course, there are also those who believe they should be getting something tangible every year of the research, and not just a technical summary of the past year's research, or even worse a large and "incomprehensible" technical report after 5 years. The implication interpretation should address two questions:

- Is there anything from the previous year's work that can be put into commercial practice now?; and
- Does the existing conceptual model for mine closure need to be changed as a result of the year's research?

The purpose for asking the first question is to identify whether something of operational value can be concluded from the research in advance of completion of the whole program of work. Presuming that something of potential immediate value has been found the supplementary question would be what are the risks of changing commercial practice before the research questions have been fully addressed. The researchers should not be expected to 'step through the door' and present potentially overly optimistic conclusions about their research before all of the results are in. In these situations, it is likely more practical, and appropriate for a consultant to 'step through the door', which then allows the mine operator to 'push' the consultant to offer comments on the research results, and their implications to commercial practice.

For the second question, it means developing a clear and concise statement about what the conceptual model actually is, clear and concise statements about what the year's research means to changing, or not changing, the conceptual model, and if needed, an update on the conceptual model.

The outcomes from effective communication of the implication interpretation of research results can lead to different actions on the R&D plans:

- Re-develop the R&D plans or modify the existing R&D plans;
- Continue the existing R&D plans; and/or
- Induce the operators to make decisions (see below).

3.6 Making Decisions

Once operators obtain enough information, especially about the implications of commercialisation of the research results, decisions can be made. Although an earlier communication of initial positive research results may provide the operators with an indication of the implications of commercialisation, risks on decision-making would likely be higher at the beginning or in the early years of research. The actual and perceived risks would decrease with time provided the results continue to indicate ongoing good performance. The time when a research program can move to the next stage (i.e. development stage or commercialisation) usually depends on how large a risk the operator is willing to, or can, assume.

3.7 Roles of Miners, Consultants, and Researchers

A technology transfer program needs contributions and cooperation from miners, consultants, and researchers. Figure 3 shows relationship among the three components of the team. In general, consultants and researchers would have separate contracts with operators. Cooperation between the consultants and the researchers is a vital part of the relationship, and data sharing must occur with the operators' permission.

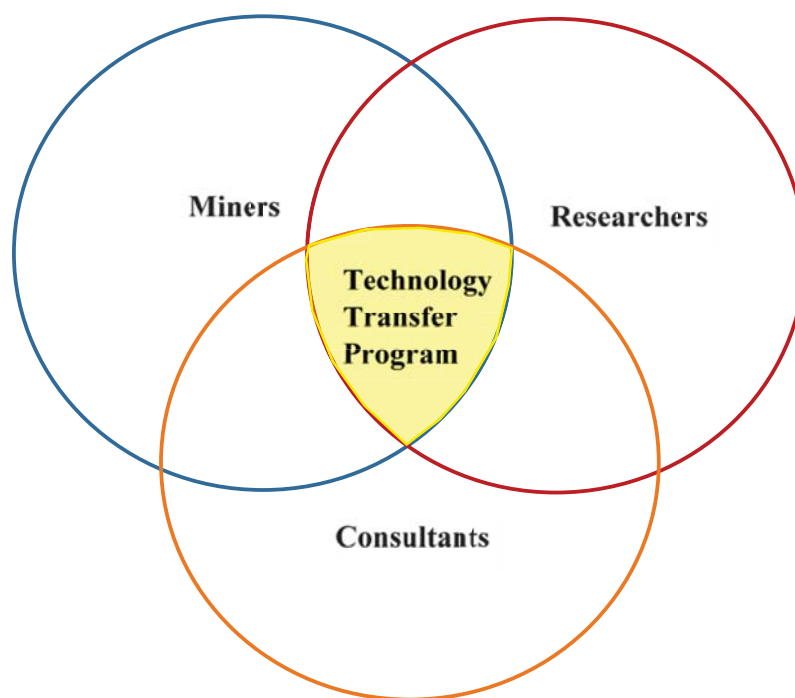


Fig. 3 Relationship among miners, consultants, and researchers in terms to a technology transfer program

As shown in Figure 2, the roles of the operators in a technology transfer program are to ask questions and make decisions. In addition, other supports such as access to mines, field sampling, and build-up of field trials provided by the operators are also crucial to successful R&D. The most important roles for the researchers in a technology transfer program are to develop and execute R&D plans, while the most important roles for the consultants are to

screen technologies and make timely implication interpretations of the research results. Incorporating consultants in a technology transfer program will allow the researchers to 'step back' from having to answer inevitable questions about different areas and plans on the site during the research. In essence, the technology transfer process needs to occur throughout the research program and allow the researchers to do what they are so good at; namely, delving deep into the details that are needed to address the research questions, and not be caught up in trying to answer questions before having sufficient information to be comfortable with doing so.

4.0 CASE DEMONSTRATION OF TECHNOLOGY SCREENING

This section presents a case study to demonstrate how the technology screening method has been used to help screen technologies based on site-specific constraints.

The mine site is located at a region in the northwest of Canada. At the site, waste rock dumps (WRDs) produced from open pit mining activities have net acid generation potential, which could contaminate the nearby creeks. The objective of the mine closure plan for the site is to put measures in place to ensure that any drainage from WRDs will not cause deterioration of water quality in the creeks; namely, the water quality of the creeks should satisfy the requirements of all federal and local government environmental laws. Under this situation, the operator may have the question "what is the best technology (methodology) that can be utilised in the mine closure to achieve the objective".

The first step of technology screening is to collect all possible technologies that have potential to solve the acid drainage problem at the site. The possible technologies for this case study include: 1) active water treatment systems; 2) water diversions; 3) pit design optimisation;

4) contact water collection systems; 5) cover systems; 6) sub-oxic zones; and 7) saturated zones. In each major technology category, there are a range of options. For example, there are options of a simple soil (layer) cover, complex soil cover, geosynthetic cover, imported soil cover, and vegetation in the cover systems.

The second step of technology screening is to assume where and in what context the technologies will be applied. Taking the cover systems as an example, they could be applied to the whole dump, dump surface only, dump face and crest only, dump face only, and dump toe only. Application scope of the technology will influence drainage quality and cost estimates, and hence the selection of the most promising technology type.

The third step of technology screening is to evaluate cumulative effects of the technologies that can be applied. For example, the effects of cover systems may include supporting reclamation objectives, limiting percolation, and limiting oxygen influx. The effects under the limiting percolation and oxygen influx categories can be further classified and specified. Figure 4 lists technologies, applications, and effects using the cover systems as an example.

After the above preparations, the collected technologies can be screened on the basis of site-specific constraints, which may include cost limitations (or cost benefits), regulatory requirements, environmental requirements, social benefits, health and safety requirements, and technological challenges. Table 2 summarises cost estimates for different technologies applied to the whole dump.

Table 2. Summary of cost estimates for selected technologies (values in \$ Million)

Cost Parameter	Base Case ¹	Topsoil Replacement	Simple Soil Cover	Complex Soil Cover	Geomembrane Cover	Saturated Toe	Internal Layer (Textural Discontinuities)	Passive Treatment in Backfilled Pit
Pre-Closure Cost ²	-	\$5	\$15	\$15	\$30	—	\$25	-
Closure Cost	\$100	\$105	\$110	\$110	\$175	\$45	\$90	\$30
Post-Closure Cost ³	\$140	\$140	\$135	\$115	\$45	\$20	\$50	\$25
Discounted Total Cost	\$240	\$255	\$275	\$250	\$270	\$65	\$185	\$55
Δ from Base Case		\$15	\$35	\$10	\$30	(\$175)	(\$55)	(\$185)

¹ The base case assumed that the dump would only be resloped to 2:1 and directly revegetated.

² The pre-closure cost refers to the cost occurring during operation but the activities will service closure, for example for stripping and stockpiling soil from the footprint of the dump so that it could be used in a soil cover.

³ The post-closure costs for each option and the base case assumed that seepage and groundwater below the toe of the dump would be collected and treated.

To compare the combined closure and post-closure costs, it was necessary to calculate a net present value (NPV) or net present cost (NPC). Those calculations require an assumed discount rate and they require all costs to be located in time. The discount rate used in the NPC calculations presented herein is 6%, and assumes that inflation is included in this rate (note that a great deal of caution is required when choosing a discount rate as it can have substantial influence on comparing NPC of alternatives). Simplifying assumptions were also used to locate the costs in time. It was assumed that the dump would be closed in Year 20, and that all closure costs would be borne in that year. It was also assumed that post-closure costs would commence in the following year and, for most cases, continue in perpetuity. The pre-closure costs were assumed to occur in Year 10. In Table 2, the discounted total costs were all expressed as closure costs (i.e. they were discounted forward or backward to Year 20).

Table 2 indicates that all cover systems of various designs do not appear to “pay for themselves” in terms of reduced water treatment costs when compared to the base case at the determined discount rate. Of course, the cost calculations are very dependent on assumptions about the cover system designs, their effectiveness in reducing infiltration, and specifics of the unit cost assumptions. Table 2 also shows that the passive treatment examples (Saturated Toe and Passive Treatment in Backfilled Pit) have extremely favourable net costs, with savings of \$175M, and \$185M relative to the base case. The net savings are largely attributable to the assumption that the options will be effective and a long-term active treatment will not be required. Given the fact that the cover systems offer significant additional benefits for reclamation and bio-diversity, soil covers seem likely to be worthy of further assessment, up to and including the large scale testing that would be necessary to really improve the design, performance and cost assumptions. Moreover, sensitivity analysis of closure costs with various discount rates indicates that some of the cover system designs can well “pay for themselves” when the discount rate decreases.

Technology screening costs with respect to technology application costs is only one aspect of the screening constraints (criteria). Similar screening processes can be applied to the other criteria. Outcomes from the technology screening may include favourite options, clearly less attractive options, and options with specific applications. In addition, key uncertainties are identified for each technology.

More importantly, further R&D requirements and/or R&D suggestions can be identified and recommended at the end of technology screening.

The technology screening case presented here is only part of the complete technology screening process. Technology screening provides opportunities to review potential technologies to solve operators' questions and identify promising technologies for further R&D. It is an important initial step in a technology transfer program.

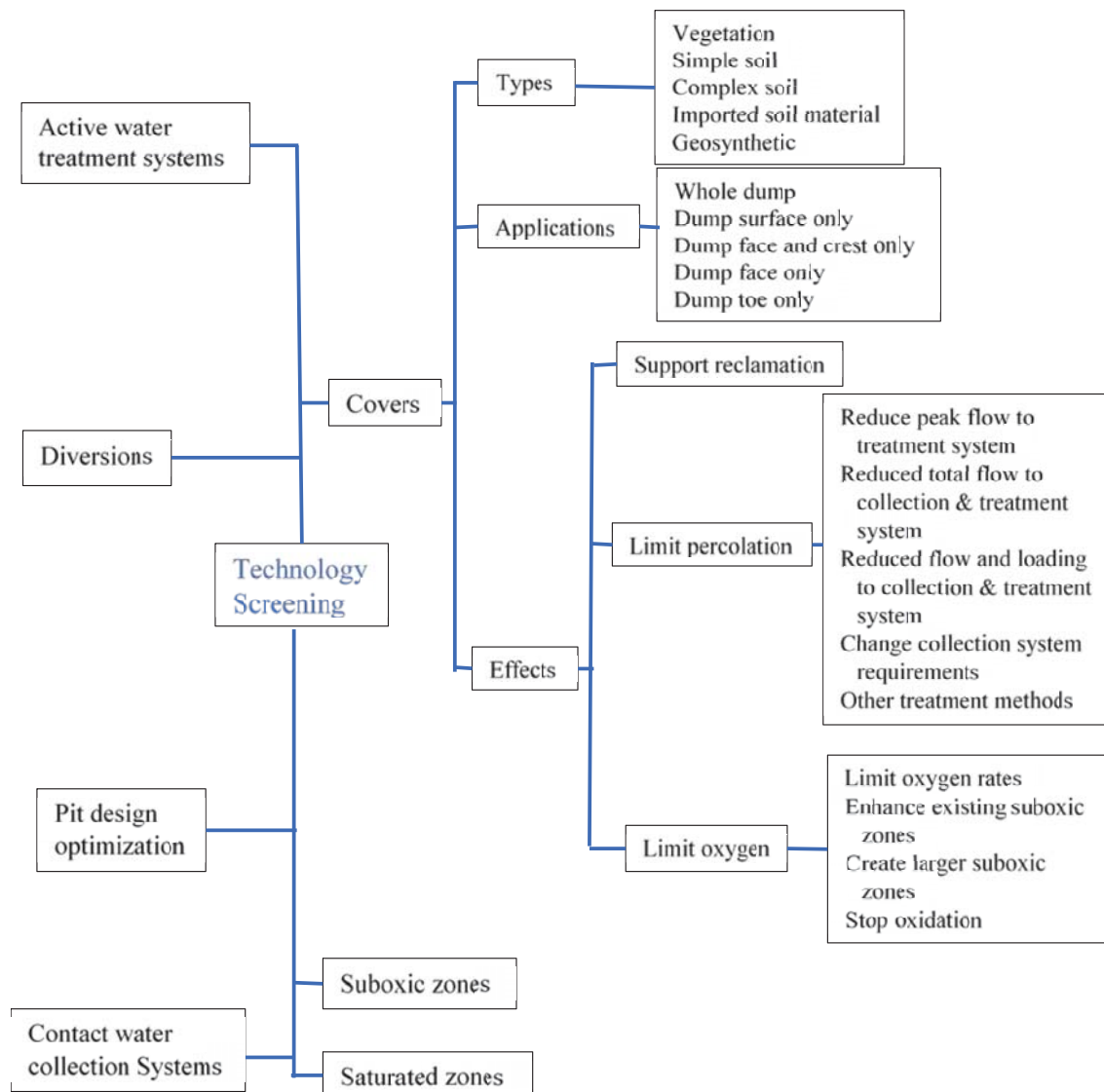


Fig. 5 Technology screening for cover systems (an example)

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5.0 SUMMARY

Commercial implementation of mine closure research requires the efforts of operators, researchers and consultants. There are some simple strategies and methods for development and commercial implementation of promising research technologies that can substantially multiply the return on investment of the initial research. One way to streamline the process is to develop a technology transfer plan within a mining company or a mining region. Involvement of experienced consultants in a technology transfer plan can help technology screening, research planning, and implication interpretation of research results, hence enhancing commercial implementation of promising research in mine closure.

6.0 ACKNOWLEDGEMENT

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LEADING PRACTICE SOLUTIONS FOR ACID ROCK DRAINAGE PREVENTION AND CONTROL: A KEY TO ACHIEVING A SUSTAINABLE FUTURE FOR MINERAL RESOURCE DEVELOPMENT

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ABSTRACT

The mineral resources industry has a poor record with a legacy of mines that continue to leach acid and contaminants making water unfit for human consumption and ecosystem health. As clean water is a human right and fundamental to sustainable development, the mining industries social licence to develop resources will depend on ensuring that ARD does not impact people's access to safe water.

An acid generating mine has the potential for long-term impacts on surface and ground water and aquatic life. In some cases the problems may be evident from the outset and steadily increase during the life of the mine. In others, ARD may appear later in the mine life or only after the mine has closed and the company has left the area. Once started, however, the process can endure for centuries.

It is apparent that ARD management strategies are not being effectively implemented at many new and operating mines and the ARD liability is likely to grow over the next decades. This is simply because the industry as a whole has not adequately adopted and implemented the knowledge and technologies that have been developed and proven over the past 30 years. As a result opportunities to incorporate effective ARD management into mine planning and day-to-day operations have been missed, and once ARD has started it is almost impossible to stop.

This paper presents the fundamentals of ARD prediction, prevention and control with site-specific experience from mine sites in the Asia Pacific region. These case studies demonstrate strategies for oxidation control, water flux control and acid-base management for ARD prevention and control.

1.0 INTRODUCTION

The performance by the minerals industry in ARD management is variable. Some companies are well advanced and continually improving, but the practices of others are out-dated and some are flawed. Action by companies individually and collectively is clearly required. In many areas, small companies are crucial to the standards of large ones. For example, projects near closure can be sold by multi-nationals to small companies that are not aware of a pending ARD liability thus opening up ways to avoid obligations and damage the industry as a whole. Multi-nationals and funding agencies can also fail in due diligence by not adequately accounting for ARD risks and liability in acquisitions. Collective action

must include companies of all sizes in order to produce positive results. INAP is one such industry organisation with a charter to facilitate this cooperation.

An acid generating mine has the potential for long-term impacts on surface and ground water and aquatic life. In some cases the problems may be evident from the outset and steadily increase during the life of the mine. In others, ARD may appear later in the mine life or only after the mine has closed and the company has left the area. Once started, however, the process can endure for centuries.

The financial liability of ARD is almost impossible to quantify but is currently likely to be many billions of dollars. It is also apparent that ARD management strategies are not being effectively implemented at many new and operating mines and the ARD liability is likely to grow over the next decades.

The reason for this is simply because the industry as a whole has not adequately adopted and implemented the knowledge and technologies that have been developed and proven over the past 30 years. As a result opportunities to incorporate effective ARD management into mine planning and day-to-day operations have been missed, and once ARD has started it is almost impossible to stop.

In addition to loss of productivity, sulphidic wastes can have a profound effect on the surrounding ecosystems. Where they are not physically stable, erosion or catastrophic failure may result in severe or long-term impacts. Where they are not chemically stable, they can serve as a permanent source of pollutants to natural water systems. These impacts have lasting environmental and socio-economic consequences and are extremely difficult and costly to address through remedial measures.

Historically the industry has a very poor record with a long legacy of mines that continue to leach acid and contaminated water. The combination of acidity and dissolved contaminants is toxic to most forms of aquatic life, and make water unfit for human consumption. As clean water is a human right and fundamental to sustainable development, the mining industries social licence to develop resources will depend on ensuring that ARD does not impact people's access to safe water.

2.0 ACID GENERATION IN SULPHIDIC MINE WASTE

ARD problems are established at the time of deposition or placement of sulphidic waste rock and tailings.

Typically when sulphidic mine materials are first exposed to atmospheric condition there is a lag before low pH conditions develop. This lag can vary from days to many years depending on the inherent geochemical lag (due to neutralising minerals such as a calcite that provide short term buffering), as well as construction and hydrological delaying mechanisms.

Figure 1 is a schematic of the time evolution of ARD generation and release from a waste storage facility. After the lag period, the pH drops and the acid load increases to a peak rate at some time later in the mine life or even post-closure. The acid generation rate (AGR) typically exceeds the acid release rate (ARR) due to the accumulation or storage of a significant amount of acid and oxidation products within the waste facility. The ARR can vary from less than 10% to more than 90% of the AGR depending on the climate and physical characteristics of the waste. The stored oxidation products include acid in minerals such as jarosite and melanterite and metal precipitates as sulphates and oxyhydroxides. The stored oxidation products are a concern for closure and post-closure as they can continue to

release acid salts and metals to the environment into the long term, even if oxygen and water influx to the waste has been reduced through placement of covers or other ARD mitigation strategies.

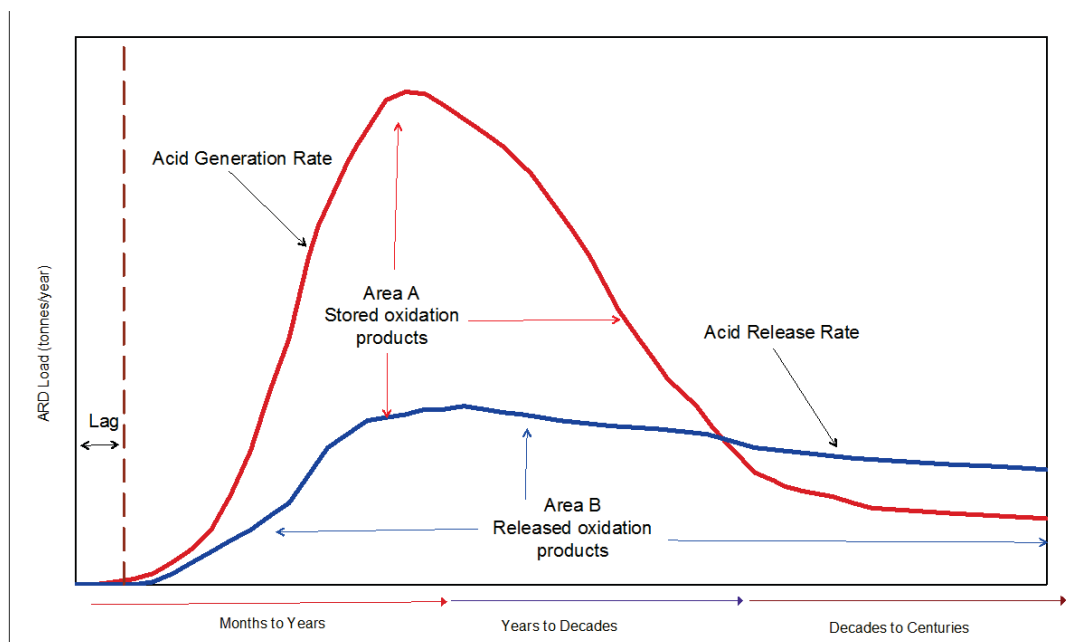


Fig. 1. Schematic of the time evolution of ARD generation and release from a waste storage facility

ARD is not a problem at every mine, even in sulphide-rich deposits. In some circumstances the inherent buffering capacity may be adequate to neutralise the acid and maintain near neutral pH conditions. But in these cases sulphates and elements such as Cu, As, Mn and Zn may still follow a similar evolution curve as shown in Figure 1 even though acid (low pH) conditions do not develop. Drainage with near neutral pH, which contains elevated concentrations of elements or sulphate is Neutral Mine Drainage (NMD) and can pose similar risks and liabilities as ARD. The term AMD (Acid and Metalliferous Drainage) can be used to cover both ARD and NMD.

Atmospheric oxygen is the main driver of oxidation and ARD generation and Figures 2 and 3 show how a dump built by the common practice of end tipping waste rock from a high tip head can promote the transfer of oxygen into the dump and vastly increase the mass of material exposed to oxygen.

Figure 2 shows how coarse and fine material segregates when dumped over an advancing face, creating a coarse base layer, chimney structures where the face angle changes and interbedded coarse and fine layers throughout.

Figure 3 is a section through this dump after 10 years and clearly shows the interbedded coarse and fine layers with extensive oxidation along the coarse layers extending out into the finer grey layers with stored oxidation products (mainly yellow jarosite). Oxygen movement

occurs by advection/convection along coarse layers driven by heat generated from the exothermic oxidation reaction with pyrite and by diffusion into the fine layers.

These Figures clearly show how the internal structure influences ARD generation and that alternative designs, such as bottom up construction in small compacted lifts, are necessary to prevent advective airflow when dealing with reactive mine waste.

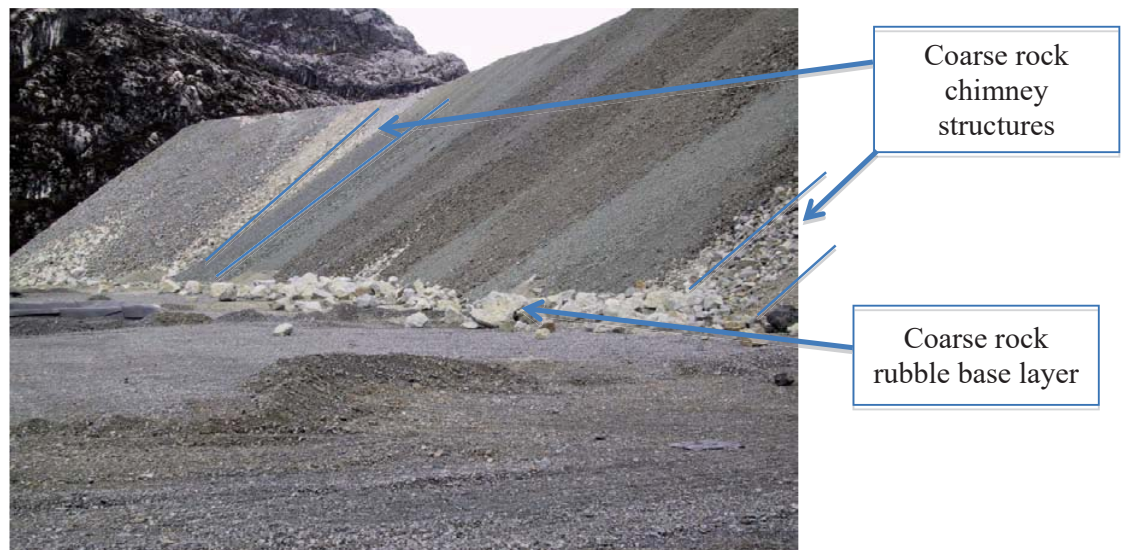


Fig. 2. Construction of a waste rock reactor

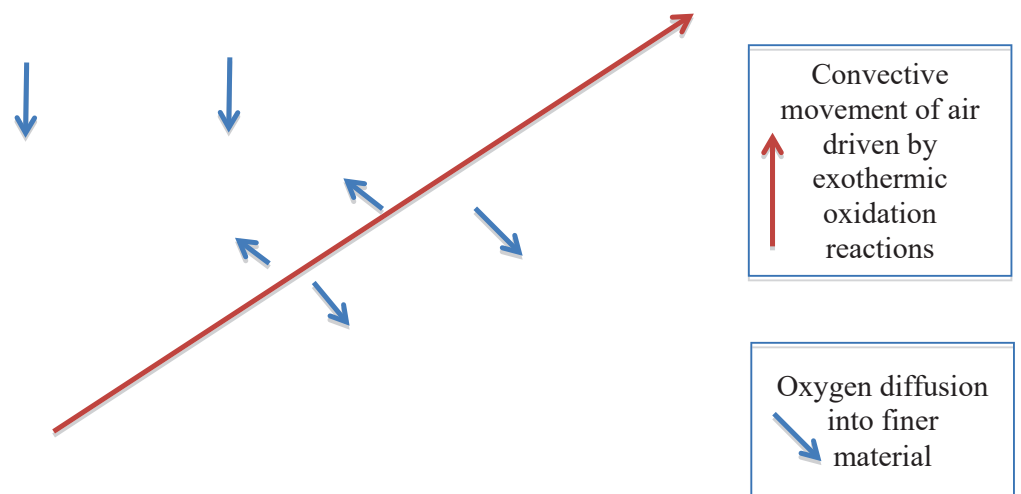


Fig. 3. Uncontrolled oxidation within waste rock reactor

3.0 PREDICTION AND GEOCHEMICAL CLASSIFICATION

The science for prediction of an ARD risk prior to mining is well advanced. Many management options can be considered with early identification and quantification of an ARD risk whereas option will be limited and costs highest if ARD is not addressed until later in the mine life.

Characterisation and identification of the ARD risk is the first step in developing an ARD management plan. Key to this is a comprehensive sulphur and geological database with associated ARD parameters such as carbonate, acid neutralising capacity (ANC), net acid generation (NAG) and associated mineralogical and elemental assays and kinetic test work programs such as leach columns or similar.

Compiling these data should commence as early as possible in the mine life cycle with adequate data available for consideration of ARD implications and costs at the feasibility stage. Once properly characterised and quantified, site-specific options can be fully evaluated and costed. There are many publications that can assist practitioners in this area including the GARD Guide (INAP), the ARD Test Handbook (AMIRA), MEND publications, the Leading Practice Handbooks on ARD (Australian Government) and numerous papers in proceedings of ARD specific conferences and workshops.

4.0 PREVENTION AND CONTROL

ARD prevention must be truly integrated with technical planning, design and operation. It must begin at exploration and be part of the process through to closure and beyond. If ARD is effectively managed, then closure costs and the risk of a long-term liability will be minimised, and access to future resources will not be compromised by poor performance.

Leading practice should not be defined by parameters that are set by regulation or read out of a manual, but there are specific task that are essential components of leading practice. These include:

- Geochemical classification;
- Quantification and production schedules by geochemical waste types;
- Selective mining, segregation and controlled placement; and
- Performance evaluation and monitoring

Leading practice at one site may not be appropriate at another and this is particularly evident across different climate and hydrological regimes. Water covers and oxygen barrier layers that prevent oxygen access to reactive sulphide mineral surfaces are best suited to climate regimes where precipitation exceeds the potential evapotranspiration. It is difficult to maintain saturation or near saturation in climates where the potential evapotranspiration exceeds precipitation and options must aim to minimise the generation (through oxygen flux control) and leaching (through water flux control) of oxidation products by segregation and selective placement of NAF and PAF materials, control of the internal structure of dumps to prevent advective/convective transfer of oxygen, infiltration control, water shedding, diversion and store and release mechanisms. In addition, blending of acid consuming (AC) and PAF rock to mitigate the ARD risk may be feasible where material geochemical properties and production schedules are favourable.

The following summarises some lessons from experience that are relevant to the development of leading practice solutions to ARD:

- Geochemical classification, quantification and production of waste types (e.g. potentially acid forming, non-acid forming, acid consuming) are critical for effective ARD management.
- ARD problems are established at the time of deposition and planning for closure in parallel with project feasibility and mine planning is fundamental to leading practice. Control options are fewer and costs higher the later ARD management is addressed in the project cycle (most options at feasibility stage; least options and highest cost at closure).
- Delayed implementation of controls results in the accumulation of oxidation products within the disposal facility and major issues for closure and long term liability.
- The internal structure, composition and properties of waste storage facilities must be designed and constructed to control gas and water fluxes for ARD control.
- Engineered soil covers:
 - Well demonstrated to be effective for ARD control on tailings storage facilities
 - Much less successful and difficult to manage on waste rock piles due to positive topography, unnatural geological structures, erosion, natural systems.

Soil covers have received significant attention and are a perceived low cost option at the feasibility stage of a project (due to NPV budgeting methods). However, covers are not necessarily the solution to long term ARD, especially when delayed until later in the mine life after oxidation has commenced and secondary products are being stored within the waste pile. Although the cover may limit water and oxygen entry, it may not prevent ongoing dissolution and leaching of secondary products and may not stop oxygen transfer sufficiently to meet environmental needs. Experience at sites such as Rum Jungle in Australia and the Equity Silver mine in Canada demonstrate the potential inadequacy of covers placed late in mine life or post-closure. As shown previously on Figure 2, there can be a significant store of soluble oxidation products that will continue to leach from the pile long after oxidation has slowed. This is particularly relevant to wetter environments where there will always be some percolation through a low permeability cover system.

- Water or elevated water table covers have been well demonstrated and are the best solution for geochemical control provided potential stability issues are addressed.
- Mitigation of an ARD risk through blending potentially acid forming and acid consuming material needs to have a substantial excess of the neutralising component. For ROM waste rock blends the NAPP needs to be less than minus 150 kgH₂SO₄/t; crusher/stacker built waste rock dumps require the reactive size fraction (<5mm) to have an ANC/MPA greater than 2; for tailings an ANC/MPA greater than 1.5 has been demonstrated.
- The climate and/or water balance influences control strategies, such as:
 - Focus on oxidation flux control where precipitation exceeds evaporation
 - Focus on water flux control where evaporation exceeds precipitation
- A comprehensive monitoring program including material geochemistry, oxygen concentrations and temperature within waste piles, drainage water quality and flow, as well as geotechnical and physical parameters associated with control structures

including encapsulating, containment or sealing layers is essential. This information is required to continually evaluate the performance of the ARD management plan and to refine designs and specifications as needed.

- Documented ARD Management Plans with clearly defined operating procedures, responsibilities and key performance indicators is required. Diligent supervision and quality control are critical to the effective implementation of an ARD management plan.

5.0 APPLICATION OF LEADING PRACTICE IN THE ASIA PACIFIC REGION

There are encouraging signs in the industry, including the Asia Pacific region, where leading practice ARD mitigation measures as well as leading practice for ARD control at the whole mine site level are being demonstrated. These practices include:

- ARD control through exclusion of atmospheric oxygen by design (water and engineered covers)
- Acid base balance control through use of high carbonate mine waste (blending/covers)
- Acid base balance control through sulphide recovery
- Segregation and selective placement

ARD control through exclusion of atmospheric oxygen by design has been demonstrated at the following sites:

- Martha Mine, New Zealand
- Golden Cross Mine, New Zealand
- Phu Kham, Lao PDR
- Ban Houayxai, Lao PDR
- Kelian, Indonesia

At the Martha Mine, PAF cells were constructed and encapsulated by compacted NAF waste. Tight engineering controls are in place and a comprehensive monitoring program including in-dump oxygen and temperature confirm the performance of this design. Figure 4 shows a PAF cell during construction and Figure 5 shows the placement and compaction of the sealing layers. Crushed limestone is broadcast on exposed PAF material prior to covering to control Mn solubility in runoff from active areas.



Fig. 4. PAF cell construction at Martha Mine, New Zealand



Fig. 5. Construction of encapsulation layers

The Phu Kham Copper Gold mine in Lao PDR incorporates many aspects of leading practice for ARD control in a wet environment. The Phu Kham site presents significant challenges for managing potentially acid forming mine waste in steep topography and a high rainfall environment. Early identification of the ARD risks prior to mining and integration of the geochemical requirements with the mine plan has enabled the company to managed ARD without any significant events.

The plan is based on the fundamental strategy of isolating sulphidic mine waste from atmospheric oxygen. This is achieved through placement of the higher sulphidic acid generating waste rock (Red Waste) within the tailings impoundment where it is progressively inundated by the supernatant pond. The lower sulphidic acid generating waste rock (Orange Waste) is isolated in cells within the downstream portion of the tailings storage facility embankment. Fine grained non-acid forming waste (Green Waste) and borrow material is used for encapsulation layers around PAF cells. Figure 6 is an overview of the TSF showing

the construction of PAF cells within the downstream zone of the embankment and placement of reactive PAF material within the TSF.

Performance monitoring of the geochemical and geotechnical characteristics of placed waste rock; oxygen and temperature monitoring within the PAF cells; and water quality data confirm that although there are significant environmental risks the results to date have clearly demonstrated that with good design, monitoring and management that ARD can be managed in a cost effective manner.

Key to the success of ARD management plan has been company wide awareness of the ARD risks and diligent operational management with regular technical review and evaluation of performance.



Fig. 6. Waste Rock Management for ARD Control at the Phu Kham Mine in PDR Lao

Pan Aust have applied the strategies developed at Phu Kham to their recently commissioned Ban Houayxai mine in Lao. This site is a lower risk ARD site, but segregation and selective placement of PAF and NAF material types is integrated with mine operations. The dumps are constructed from bottom up in small lifts to avoid segregation of fine and coarse materials and minimise the risk of developing advective/convective gas transfer between the atmosphere and interior of the dump.

The now closed Golden Cross mine utilised similar practices to Martha Mine with the segregation and compaction of PAF waste and compaction of NAF encapsulation layers to achieve a high degree of saturation for oxidation control. Golden Cross also closed the pit by partial backfilling with inert waste to create a free draining pit. Figure 7 is an aerial view of the mine post closure.



Fig. 7. Aerial view of Gold Cross Mine Post-Closure

The Kelian Equatorial Mine (KEM) in Kalimantan, Indonesia planned for water covers for waste rock, tailings and the final pit from early in the mine life. Waste rock was placed in a purpose built dam that was flooded at closure. Geochemical testing indicated that tailings were likely to have a lag period of at least a few years before low pH conditions would develop and hence the tailings were placed sub-aerially forming large beaches during operations. At closure the water level was raised to cover all tailings. The pit was filled with water to cover most of the PAF and problematic material types, however there is sufficient PAF above the final water level to result in dissolved Mn concentrations that exceeding guidelines for direct discharge. A constructed wetland system is used to remove the Mn and meet water quality standards for discharge.

Figures 8, 9 and 10 show the TSF and pit with final water level RL and waste rock dam prior to the final water level raise, respectively.



Fig. 8. Kelian TSF at closure



Fig. 9. Kelian Pit at closure



Fig. 8. Kelian Waste Rock Dam at closure

Acid base management for ARD control through use of high carbonate mine waste (blending/covers) is used at:

- Ok Tedi, Papua New Guinea
- Grasberg Mine, Indonesia
- Savage River Mine, Australia
- Beenup, Australia

In addition, acid base management by sulphide removal from tailings has been demonstrated at Ok Tedi and is proposed for Grasberg when the GRS block cave operation commences.

Experience at Ok Tedi and Grasberg demonstrate the need for a substantial excess of ANC to produce an effective blend for long term ARD control at the run-of-mine scale. At Ok Tedi, mine planning schedules mine production to deliver waste rock to the tip head at a NAPP of not less than minus 150 kg $\text{H}_2\text{SO}_4/\text{t}$ on a quarterly basis (known as the NAPP minus 150 plan). Annual sampling of the waste rock in the Harvey Creek and Ok Mani dumps is carried out to confirm that the material is NAF with a high factor of safety ($\text{ANC}/\text{MPA} > 2$ in the fine fraction).

Experience at Grasberg demonstrated that PAF rock and limestone could not be effectively blended using ROM size material with truck dumping from high tip heads due to segregation of the coarser limestone and finer PAF rock down the face. However, utilising a crusher and stacker method, effective blending was demonstrated and has been utilised for construction of the Lower Wanagon dump. The key operational target is to ensure that the reactive fraction in the blend (less than 5mm size) has an ANC/MPA of greater than 2. This requires a

substantial excess of ANC in the blend which is of similar magnitude to the NAPP minus 150 plan applied at Ok Tedi.

Figure 9 shows the scale of the Lower Wanagon dump with low tip faces (up to 600m). Figure 10 shows the average and median ANC/MPA for the bulk and fine fraction based on samples collected from down the face of the dump on a 6 monthly basis.



Fig. 9. Lower Wanagon stacker built dump at Grasberg Mine

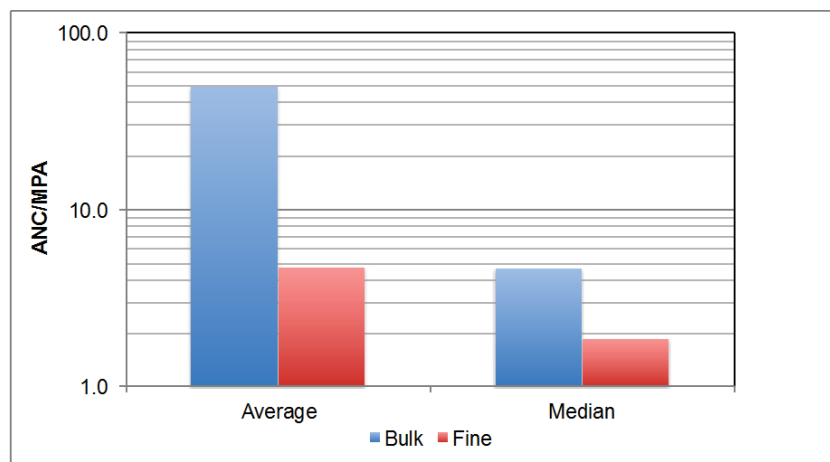


Fig. 10. ANC/MPA of Bulk and Fine (minus 5mm) material place in the Lower Wanagon Dump

At Ok Tedi, sulphur recovery from the tailings is carried out to ensure that mine derived sediments that are being dredge and stacked adjacent to the river at Bige will not be a long term ARD concern. The Pyrite concentrate is piped to the Bige area from the recovery plant and deposited below the water table with dredged sands stacked over the disposal cells to ensure permanent isolation from atmospheric oxygen.

Figure 11 shows the actual (sampled daily) and predicted (modelled) ANC/MPA of dredge sands. The sulphur recovery plant was commissioned in Q4 2008 resulting in an immediate increased in the ANC/MPA which is now generally greater than 1.5 and predicted to further increase as mining continues.

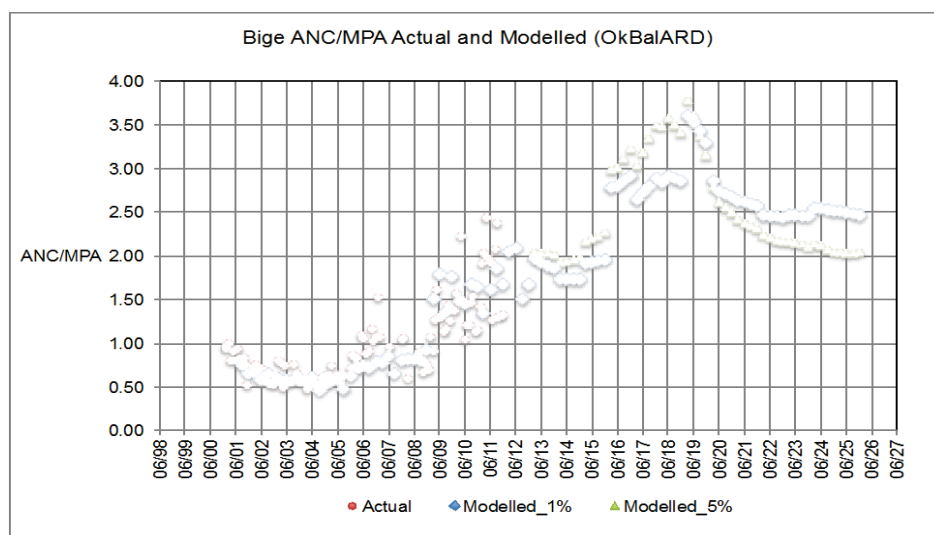


Fig. 11. ANC/MPA of Bige dredge sands (actual and modelled values shown). The 1% and 5% model runs reflect the expected range in annual erosion rates from the Harvey Creek waste rock dump.

Figure 12 shows the west back dredge sand stockpile and active PCon disposal cell in the background and dredge channel to the Ok Tedi. The dredge to access the PCon disposal area for cell construction uses the dredge channel. As each PCon cell is filled, they are covered progressively with dredge sands from the river. The PCon remains under the water table and is covered with at least 15 m of dredge sand.



Fig. 12. Bige Dredge sand stockpile showing a PCon disposal cell in the background (water filled pond).

5.0 CONCLUSION

The common thread at successfully managed ARD sites is early identification of the risk and incorporation of appropriate ARD control strategies throughout the project cycle from pre-feasibility to closure.

Performance evaluation and monitoring and a working ARD management plan with clearly defined operating procedures, department duties and responsibilities and key performance indicators are essential. Diligent supervision and regular technical review are necessary to ensure the ARD management plan evolves along with the ever changing mine and mineral processing plans at a site.

Designing and controlling the internal structure of waste rock dumps to minimise gas transfer and direct seepage away from problematic material is a major issue for the future of the industry when mining sulphidic materials. Constructing dumps from the bottom up in small lifts with compacted intermediate sealing layers may need to become the base case option, so that the true cost of ARD management is considered early in the project cycle. Differing these costs and relying on future mitigation actions such as covers, is likely to become less acceptable and difficult to permit.

MANAGEMENT OF PYRRHOTITE TAILINGS: A TEN YEAR CASE STUDY ON CONTINUOUS LEARNING AND IMPROVEMENT

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ABSTRACT

This paper presents case study information on the geochemical and mineralogical nature of tailing materials generated at Savannah Nickel Mine in the Kimberley region of Western Australia (WA). The mine is an underground operation, and tailings generated from processing ore are stored at an above-ground valley fill Tailings Storage Facility (TSF). Various studies have shown that leaving the tailings in situ at mine closure and using a cover system will produce a sustainable environmental outcome, and the mine operator has recently received regulatory approval to increase the capacity of the TSF and implement this closure strategy.

Over the past decade, geochemical and mineralogical studies have demonstrated that the geochemical behaviour of the tailings in the TSF is much more benign than initially predicted. Despite containing almost 30% pyrrhotite, the tailings generate only a fraction of the acidity predicted by classical static geochemical test techniques, and the rate of acidity generation is more than compensated by the inherent non-carbonate acid neutralising capacity available from specific silicate minerals within the tailings matrix.

The mine operator now has a very good understanding of the geochemistry of the tailings, although geochemical, mineralogical and other related studies are continuing, moving towards mine closure.

1.0 INTRODUCTION

Geochemical assessment of mine waste materials forms an integral part of the broad range of studies required for a mining company to obtain approval and finance for new projects (Parker and Robertson, 1998). However, the geochemical assessment process needs to continue throughout the life of mine. The results of the assessment provide a fundamental input into life- of- mine decision making processes for the various phases of mining and mineral processing activities, such as mine planning, operation and closure. Successful completion of geochemical assessment studies and the implementation of appropriate environmental management strategies can significantly reduce the risk of the development of Acid and Metalliferous Drainage (AMD) and associated environmental impacts.

Acquisition of timely geochemical information on the nature of mine waste materials can be used to develop innovative design solutions, limit the potential for environmental impacts, and promote sustainable closure (DITR, 2007; INAP, 2009; and Wilson, 2011). During the mine planning and regulatory approvals stage, there may be a limited amount of geochemical information available on the long- term behaviour of mine waste materials. This knowledge gap can lead to the development of impractical mine waste management design

solutions and prescriptive approval conditions, which may not result in the best environmental outcome. Some flexibility should remain during the mine operational phase to improve operational and mine closure plans, based on additional geochemical knowledge and experience gained throughout the mine life.

1.1 Geochemical and Mineralogical Studies

In Australia and internationally, technical guidelines for geochemical testing and assessment of mine waste materials have been developed (DITR, 2007; INAP, 2009), although it is important to consider the individual characteristics of sites and waste material types when selecting test methods to use. Classical geochemical characterisation techniques can sometimes produce inconclusive results, which are typically investigated further using a combination of more specialised geochemical tests and mineralogical tests. This is particularly true where some uncertainty remains over the long-term geochemical behaviour of mine waste materials.

The information acquired from geochemical and mineralogical tests on mine waste materials can assist with predicting the level of environmental risk and potential impacts. However, due to scale-up and other factors such as the heterogeneity of the mine waste materials and the redox conditions experienced by mine waste materials in the field, simplistic direct comparison of leachate chemistry with existing water quality guidelines should be treated with caution. Where uncertainty exists, modelling is often used to assess the complex behaviour of mine waste materials and potential impacts from mine waste storage facilities (Mayer *et al.*, 2003).

1.2 Focus of Paper

This paper presents a case study from Savannah Nickel Mine located in the Kimberley region of WA, which illustrates the benefits of ongoing geochemical and mineralogical characterisation and assessment studies on mine tailings materials throughout mine life and in planning for closure. These studies form an integral part of broader operational and closure planning studies completed for the site Tailings Storage Facility (TSF), such as geotechnical stability, cover design, geomorphology, solute transport modelling, ecotoxicology and ecological assessment of downstream aquatic fauna; however, these broader studies are not the focus of this paper.

It is demonstrated that classical geochemical classification techniques completed at the mine planning stage of the project produced results that were not replicated in the field and led to project conditions of consent, which were unlikely to produce the most effective long-term environmental outcome at the site. Geochemical, mineralogical and other related studies completed over the past ten years show that leaving the tailings *in situ* at mine closure and using a cover system will produce a sustainable environmental outcome, and the mine operator has recently received regulatory approval to increase the capacity of the TSF and implement this closure strategy.

2.0 CASE STUDY

2.1 Project Background

Savannah Nickel Mine is owned by Panoramic Resources Limited (Panoramic) and is located in the Kimberley region of WA in a semi-arid to sub-tropical climate, with average annual rainfall of approximately 557 mm (ranges from 280 to 1,310mm) and average annual

evaporation of approximately 3,200 mm. Rainfall variability is moderate, with most rainfall occurring in the hot summer months between December and March.

The mine commenced in 2005 as an open pit operation, but is now an underground operation, and tailings are currently disposed of as a cement-based backfill underground or stored at an above-ground valley fill TSF. The original mine approval required the mine operator to move the tailings from the TSF back into the underground workings and open pit at the end of mine life. Approximately 4.5 Mt of tailings had been placed at the TSF by the end of 2012.

Due to ongoing exploration success, additional mineral resources were identified at the project resulting in an extension of the mine life and a requirement for additional capacity to accommodate the additional tailings generated from mineral processing. In 2007, the mine operator began assessing options for the long- term management of tailings at the operation to match revised life- of- mine ore reserve estimates. The preferred option was to increase the capacity of the TSF in a series of embankment raises and leave the tailings *in situ* at mine closure and covering the TSF using the concept of an enhanced “store and release” cover system. The high evaporation to rainfall rate (over 5:1) essentially precludes the use of a permanent water cover system for the tailings. Geochemically benign waste rock adjacent to the TSF and other lower permeability materials were identified as suitable cover material as part of this closure option for the TSF. The cover was designed to store incident rainfall and then release this moisture through evapo-transpiration during dry periods, with excess surface runoff from extreme rainfall events directed off the cover via engineered drains and spillways. The cover design did not aim to stop all moisture infiltrating through the cover profile, and numerical modelling to determine net percolation rates assumed a ‘bare’ surface (*i.e.* no vegetation).

Geochemical information presented in the original approvals documentation (Notice of Intent) in 2002 predicted that the tailings would be Potentially Acid Forming (PAF) and could pose a significant long-term risk to the environment from potential seepage of AMD. However in 2008, an independent peer review found that removing tailings from the TSF and returning them to the open pit was unlikely to produce the best environmental outcome at closure. The peer review recommended a broad range of additional technical studies be completed associated with leaving the tailings *in situ* at the TSF at mine closure and covering with a suitable cover system. These studies included geochemistry, mineralogy, surface water and groundwater hydrology, cover design, geomorphology, solute transport modelling, ecotoxicology and ecological assessment of downstream aquatic fauna. A comprehensive network of surface and groundwater monitoring infrastructure was already in place at the site.

The additional technical studies outlined above were commissioned in 2008 and completed in 2013, and included consultation with relevant stakeholders, including State Government agencies. Site visits and risk workshops were held to communicate the ongoing findings of the technical studies and demonstrate a transparent process for seeking approval for the preferred operational and post-closure tailings storage option. In 2013, Panoramic received approval from the WA State Government to increase the capacity of the existing TSF and implement the preferred long- term tailings storage option at closure. Notwithstanding, Panoramic has continued to fund long- term geochemical and mineralogical studies on tailings, and some of the latest findings are included in this paper.

2.1 Tailings Characteristics

Geochemical studies completed on a simulated tailings sample in 2002 concluded that pyrite was present as the main sulfide species and classified the tailings as PAF. Subsequent geochemical and mineralogical studies completed over the past ten years have found that 'actual' tailings generated at the project and deposited in the TSF contain pyrrhotite rather than pyrite and react in a very different way to that originally assumed in 2002. Whilst the 'actual' tailings are still classified as PAF using static geochemical classification methods, the bulk tailings continue to generate very little acidity, and contain sufficient neutralising capacity to produce circum-neutral pH leachate with excess alkalinity. At the TSF surface, the tailings form a trafficable hardpan surface about 3 cm thick and the oxidation front has not progressed below the hardpan layer (Figure 1). Below the surface hardpan, the tailings remain relatively fresh/unoxidised and do not generate acid conditions after a storage period of up to seven years as evidenced by:

- The visual appearance of the tailings at the TSF,
- *In situ* geochemical and mineralogical data for tailings directly below the hardpan surface;
- The findings of a "tailings-at-depth" geochemical assessment down to 20m;
- Water quality seepage data at the downstream Water Storage Facility;
- Groundwater monitoring data downstream of the TSF; and
- Geochemical and mineralogical studies completed over the past ten years.



Fig. 1. Trafficable tailings hardpan surface at TSF

Water quality monitoring downstream of the TSF indicates that seepage water is typically pH-neutral, with excess alkalinity and elevated conductivity, mostly attributable to dissolved sulfate, calcium and magnesium and the concentration of trace metals is low. The elevated sulfate is attributable to a number of factors including the oxidation of near-surface tailings stored in the TSF (*i.e.* hardpan formation) and recycling of sulfate rich water from dewatering of the water storage facility seepage and underground workings at the mine.

Geochemical studies of *in situ* bulk tailings at the TSF demonstrate that below the hardpan surface, bulk tailings remain pH neutral, slightly brackish, and containing low concentrations of trace metals suggesting limited oxygen diffusion into the bulk tailing material (possibly

through a combination of surface hardpan formation, elevated tailings saturation levels, and maintenance of reducing/anoxic conditions at depth in the TSF. These results are mirrored by the results of ongoing Kinetic Leach Column studies on tailings completed over a six year period. A key finding is that under anoxic conditions, the tailings pore water contains very low concentrations of soluble reduced iron and sulfate sulfur species indicating limited potential for any latent acidity in TSF seepage. However, this does not entirely explain the geochemical behaviour of the *in situ* bulk tailing materials at the TSF. A significant amount of elemental sulfur was also found to be present in both the bulk tailings at depth and the oxidised hardpan, which suggests that alternative pyrrhotite reaction mechanisms may occur in the bulk tailings.

2.2 Tailings Hardpan

The relatively benign geochemical nature of the tailings solids at the TSF is unusual as this material can contain up to 30 wt. % pyrrhotite. A review of the available literature over the past 15 years suggests that pyrrhotite is less well studied than pyrite, but is commonly associated with nickel sulfide deposits (Schippers *et al.*, 2007, Heikkinen and Räsänen, 2008; and Robertson *et al.*, 2011 and 2012).

Several authors have found that the rate of pyrrhotite oxidation slows when a hardpan layer is present on tailings under both dry and saturated cover conditions (Gilbert *et al.*, 2003). McGregor and Blowes (2002) suggested that a TSF hardpan layer can act as a hydraulic and diffusive barrier against the migration of rainfall infiltration and oxygen. The authors presented a case study for uncovered pyrrhotite tailings at a TSF in Canada (Fault Lake), and highlighted that the hardpan layer had grown to 20 cm and the oxidation front had migrated only an additional 18 cm after 25 years of exposure to atmospheric conditions. Further numerical simulations of the Fault Lake tailings concluded that, over a time period of 1,000 years, only the top three metres of tailings would become oxidised (Romano *et al.*, 2006), which aligns with earlier tailing modelling work which indicated that the overall rate of oxidation of the near surface tailings was eventually controlled by the diffusion of oxygen into the tailings mass rather than by the reaction kinetics in the tailing (Elberling *et al.*, 1994). No information on water quality was presented in either of these references. These references illustrate the characteristics of a tailings surface hardpan and clarify the expected rate of progression of the oxidation front into the bulk tailings material over time.

At the Savannah Nickel Mine TSF, the tailings form a surface hardpan, which also appears to limit oxygen diffusion or at least maintain a high level of saturation in the bulk tailings and generally retains the bulk tailings in a mostly reducing/anoxic environment. This has been confirmed by recent work completed on the tailings, which indicates a slow and comparable rate of progression of the hardpan layer and oxidation front into the tailings (1cm per year if the tailings are maintained above 75% saturation). Another attribute of the tailings is that they contain significant amounts of potentially acid-neutralising gangue silicate minerals (e.g. enstatite), which can provide both short- and long-term neutralisation of acid generated through pyrrhotite oxidation (Ciccarelli *et al.*, 2008; Miller *et al.*, 2008).

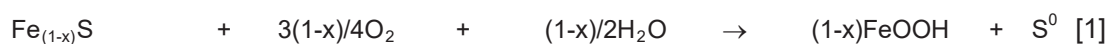
Soil-atmosphere modelling results associated with the proposed final cover design for the TSF demonstrate that a saturation level of 75 % in the tailings below the cover material is likely at depths of 5m or greater below the tailing-cover interface. At shallower depths in the tailings profile, a reduced level of saturation is expected, although at a depth of 1m, a saturation level in the range 62 to 68% is still likely. This modelling takes no account of the surface hardpan maintaining higher levels of saturation in the bulk tailings. Hence the predicted level of saturation, together with the surface hardpan, will serve to significantly slow

the pyrrhotite oxidation rate at the TSF to a rate that is likely to be matched by the silicate dissolution rate within the bulk tailings.

2.3 Reaction Pathways

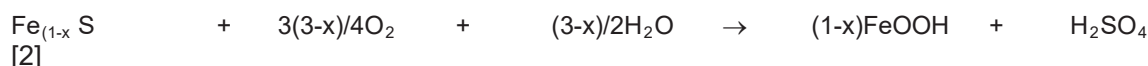
In order to explain the relatively benign nature of the project tailings, the reaction mechanisms of pyrrhotite should be considered. The dissolution behaviour of pyrrhotite is less studied than that of pyrite; however a number of studies from both laboratory and field investigations have been published (e.g. Belize *et al.*, 2004; Thomas *et al.* 1998, 2000; Nicholson and Scharer, 1994). Pyrrhotite can follow a number of reaction pathways that can be acid forming or non-acid forming (Thomas *et al.*, 2001). The two main reaction pathways which are considered to potentially operate within the tailings are provided below.

Reaction Pathway 1



Another potential pathway for pyrrhotite oxidation is represented in Eqn. [2], where iron is converted to goethite and in this situation sulfur is fully oxidised to produce sulfuric acid.

Reaction Pathway 2



Site evidence for sulfur formation in the bulk tailings was quantified by extraction of the solids with acetone followed by analysis of the extract by High Performance Liquid Chromatography (McGuire and Hamers, 2000). The presence of elemental sulfur suggests that pyrrhotite oxidation has occurred at least to some extent according to Reaction Pathway 1, where iron is principally converted to goethite or similar amorphous iron oxyhydroxides, and sulfide is primarily oxidised to produce elemental sulfur under conditions of less oxygen and water than Reaction Pathway 2 and does not involve any acid generation. In contrast, pyrrhotite oxidation under Reaction Pathway 2 also involves iron conversion to goethite, but in this situation sulfur is fully oxidised to produce sulfuric acid. It is noted that no reduced sulfur species (such as sulfite) were found in the bulk tailings pore water at depth and iron concentrations were very low suggesting that the potential for latent acidity to occur in seepage from bulk tailings at the TSF is low.

Whilst both reactions described above could be occurring in the tailings at the TSF, the KLC results, tailings at depth assessment, and seepage and groundwater quality observations downstream of the TSF suggest that, although pyrrhotite may be acid generating to some extent in the hardpan tailing material, it is acid neutral or NAF in the bulk tailing material. The presence of elemental sulfur in the bulk tailings hardpan and at depth, and lack of oxygen below the tailing hardpan surface (dissolved oxygen was measured in the tailing porewater during a drilling and sampling program at the TSF) suggests that Reaction Pathway 1 is likely to be favoured.

Another alternative adding to the complexity of interpretation, is that some acid may be generated according to Reaction Pathway 2 and then neutralised by alkalinity from residual

lime from the tailings or from mineral dissolution reactions or both. Recent mineralogical assessment work has established that the tailings contain enstatite, a magnesium silicate with a dissolution rate comparable to the measured oxidation rate of pyrrhotite in near saturated tailings and which provides a source of alkalinity at a rate comparable to the acid generation rate from pyrrhotite oxidation. This results in pH neutral drainage with excess alkalinity and very low levels of dissolved metals as well as elevated salinity from sulfate, calcium and magnesium. These results are consistent with observations reported in the literature regarding other pyrrhotite-bearing tailings wastes.

2.4 Kinetic Leach Column Tests

Long-term kinetic leach column (KLC) tests have been operated for bulk (25 kg) tailings samples from the project over the past six years to investigate the likely quality of seepage from tailings stored at the TSF. When operated under anoxic, saturated conditions the leachate chemistry closely resembles that of groundwater monitoring data for seepage from the TSF (i.e. pH neutral, excess alkalinity and low concentrations of trace metals). The KLC data has been used in other study components (e.g. post-closure TSF seepage modelling and potential groundwater impacts), to facilitate prediction of the likely quality of any long term TSF seepage.

More recently, three KLC tests have been established to determine the oxidative dissolution behaviour of pyrrhotite in the tailings as a function of water content and in the absence of additional alkalinity inputs (Fig 2). The three KLC tests have been operated with average levels of saturation of 50, 75 and 100% to investigate what geochemical processes are likely to occur in the tailings stored in the TSF in the post operational period, during which saturation levels and alkalinity inputs may be less than those which currently occur during operations.

The tailings used in the more recent KLC tests were fully characterised prior to commencement of the tests. The pyrrhotite content of the tailings is around 29 wt. %, and no pyrite was found. The tailings contain no carbonate minerals, but contain magnesium silicates such as enstatite and anthophyllite, which are likely to provide some neutralising capacity.



Figure 2. KLC test setup.

After 18 months “weathering” of the tailings under KLC test conditions, the following conclusions can be drawn:

For fully saturated conditions –

- Leachates from the column are pH neutral with no acidity;
- Leachates are composed essentially of calcium and sulfate with most metals being non-detectable;
- There is little evidence of pyrrhotite oxidation (~6% estimated oxidation);
- The main geochemical process occurring in the tailings under fully saturated conditions is dissolution of gypsum which is present in the tailings as a precipitated phase when the tailings are deposited in the TSF.

At 75% saturation –

- Leachate pH dropped to around 4.5 after 18 months with low levels of acidity;
- Leachates are composed essentially of calcium, magnesium and sulfate with low levels of metals;
- Around 15% of the pyrrhotite in the tailings has undergone oxidative dissolution;
- The main geochemical processes occurring in the tailings at approximately 75% saturation are oxidative dissolution of pyrrhotite to produce goethite and elemental sulfur as the major reaction (64%) and goethite and sulfate as the minor reaction pathway (36%). Less than 2% of the oxidised sulfur is leached from the column as soluble sulfate;

the majority of sulfate remaining within the tailings as partially soluble sulfates (e.g. gypsum, epsomite, melanterite) or less soluble salts such as jarosite or schwertmannite. Dissolution of magnesium silicates (enstatite and anthophyllite) is also occurring.

At 50% saturation –

- Leachate pH dropped to around 4 after 18 months with acidity around 10 times higher than that in leachate from the tailings weathered at 75% saturation;
- Leachates are composed essentially of magnesium, iron, calcium and sulfate with the concentrations of other metals being low;
- Around 57% of the pyrrhotite in the tailings has undergone oxidative dissolution over the 18 months of leaching;
- The main geochemical process occurring in the tailings at approximately 50% saturation are essentially the same as those occurring at 75% saturation, except that the rates are increased in the tailings with lower water content.

These results confirm that even under unsaturated conditions, the dominant reaction pathway for pyrrhotite oxidative dissolution in tailings produces elemental sulfur which, in the absence of further oxidation, is a non-acid producing reaction. Of the pyrrhotite, which is fully oxidised to sulfate, the majority appears to precipitate as poorly soluble sulfate salts which contain stored acidity. This stored acidity in compounds such as jarosite is likely to redissolve at a sufficiently slow rate such that dissolution of magnesium silicates in the tailings will match that of acid production eliminating or greatly reducing acidity.

The low amount of acidity generated during weathering of the tailings can clearly be seen from the data shown in Table 1. Even under conditions of 50% saturation where nearly 60% of the pyrrhotite in the tailings has oxidised, the acidity measured in leachate is only 0.2% of the expected acidity based on ABA analysis. Therefore it is likely that even if fully saturated conditions cannot be maintained post operation, oxidation of pyrrhotite will result in a significantly reduced acidity load than is predicted by classical static geochemical tests.

Table 1. Measured vs expected acidity during 18 months of weathering of tailings under controlled KLC test conditions

Saturation (%)	Amount pyrrhotite oxidised (%)	Expected acidity ^A (kg H ₂ SO ₄ /t)	Measured acidity (kg H ₂ SO ₄ /t)	Measured acidity as a % of expected acidity
50	57	145	0.34	0.23
75	16	41	0.039	0.10
100	6	15	0.035	0.23

A. Expected acidity was calculated by multiplying the measured net acid producing potential (NAPP) (255 kg H₂SO₄/t) by the percentage pyrrhotite oxidised.

The results of these KLC tests also demonstrate that there is an inverse correlation between the moisture content of the tailings and the rate of pyrrhotite oxidation, indicative of oxygen diffusion as the rate controlling process. These results suggest that at saturation levels likely to occur in the TSF following closure, the oxidation rate of pyrrhotite is likely to be sufficiently slow to be matched by the dissolution rate of magnesium silicates in these tailings resulting in pH neutral or low acidity leachate. Further testing is continuing to verify whether this situation is maintained.

3.0 CONCLUSIONS

Mine waste characterisation studies are an essential component of the total information package required to correctly design, operate and close sulfidic mine waste storage facilities to the standard required by international mining industry best practice. However, some flexibility should remain during the operational mining phase to develop and enhance mine rehabilitation and closure plans based on new geochemical, mineralogical and other knowledge and experience gained throughout the mine life. Decisions made by stakeholders during the planning stage should not preclude innovation and change during operation and towards closure as ongoing investigations uncover new information.

A case study has been used to illustrate the complexity of the geochemical and mineralogical processes associated with tailings material at a nickel mining operation in WA. Whilst detailed and targeted geochemical studies in the early stages of mine feasibility and planning can provide important information of the characteristics of mine waste materials, this paper illustrates that ongoing study is required throughout the life of mine to ensure that the most appropriate operational and closure options are developed and implemented for mine waste storages.

The results of the case study indicate that there is now a very good understanding of the geochemistry and mineralogy of tailings material at the mining operation based on a number of studies by recognised experts and peer reviews, where field evidence at the TSF matches predicted geochemical behaviour over time. The downward movement of the surface hardpan and oxidation front into the bulk tailings at the TSF is very slow and predicted to remain so even without a cover system in place. The final cover design for the TSF predicts a high level of saturation in the bulk tailings below the cover material, but does not take into account the positive effects of the surface hardpan on maintaining the elevated tailings saturation level. Mineralogical and kinetic geochemical studies show that the rate of pyrrhotite oxidation and acid generation in the tailings is likely to be matched by the magnesium silicate dissolution rate. At the TSF, this results in pH neutral drainage with excess alkalinity and very low levels of dissolved metals as well as elevated salinity from sulfate, calcium and magnesium. Further testing is continuing to confirm that this situation is maintained in the longer term.

4.0 ACKNOWLEDGEMENTS

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THE MT GORDON CHALLENGE: IMPACTS OF HISTORIC LEGACIES, HIGHLY VARIABLE CLIMATIC CONDITIONS AND TOUGH ECONOMIC CONDITIONS ON MANAGEMENT OF ACID MINE DRAINAGE

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ABSTRACT

Aditya Birla Minerals purchased the historic Mt Gordon Copper Project, previously known as Gunpowder Copper, in November 2003. Copper has been mined from two open cut pits and two underground mines periodically since the early 1900s with various processing methodologies used including heap leach, pressure leaching and copper sulfide flotation. Environmental protection requirements have undergone significant changes since the time of commencement of mining operations. In addition, the current project experiences considerable environmental legacies arising from actions taken that would not be allowed under modern environmental protection requirements.

During the 2008/09 wet season, higher than average rainfall was experienced in the Mt Isa region. By mid April 2009, the site had received 1,310 mm of rainfall equating to a 1 in 500 year ARI, two month rainfall event. This followed a period of four years of drought conditions where the site was experiencing severe water shortages. A decision to cease mining for economic reasons had already been made in early January 2009 prior to experiencing these large rainfall events. The subsequent above average rainfall forced closure of all operations and the site was formally placed on care and maintenance in February 2009.

The Esperanza Pit and Mill Creek Dam collected about 5 gigalitres of runoff from the project area during the 2008/09 wet season, effectively containing all potentially contaminated water on site. With a pH of 2.7, acidity of 3,200 mg CaCO₃ L⁻¹, copper concentration of 250 mg L⁻¹ and sulfate concentration of 5,500 mg L⁻¹, the challenge for Birla Mt Gordon to manage the water effectively and meet regulatory requirements began.

This paper will discuss:

- *Options identified for water treatment and water reduction given the site was on care and maintenance.*
- *Actions taken between 2009 and present to manage accumulated poor quality water.*
- *The impacts of climate on the options available for water reduction.*
- *When does stored water cease to be seen as a resource and become a liability?*
- *The use of ecological risk assessment as a tool in decision making.*

1.0 INTRODUCTION

Aditya Birla Minerals purchased the historic Mt Gordon Copper Project, previously known as Gunpowder Copper, in November 2003. Copper has been mined from two open cut pits and two underground mines periodically from the project since the early 1900s with various processing methodologies used including heap leach, pressure leaching, electrowinning and copper sulfide flotation.

The rugged landscape, highly variable subtropical climate and the highly sulfidic characteristics of ore and waste rock has resulted in considerable challenges for mine water management. Since mining commenced in 1927, the site has been subjected to successive cycles of drought and high rainfall. In recent periods of water surplus, mine operators have used controlled discharge of treated mine water to the receiving environment to reduce stored water volumes.

Exceptionally high rainfall was recorded in northwest Queensland during the 2008/09 wet season. This followed a period of drought where water supplies had reduced to record lows and contributed to a decision in late 2008/early 2009 to place the operation on care and maintenance owing to lack of water for processing operations. By mid-April 2009, the site had received 1,310 mm of rainfall equating to a 1 in 500 year ARI two month rainfall event. (Uncontrolled) discharges of acidic and metalliferous flood waters were reported for numerous operational and abandoned base metal mines in the Mt Isa and Cloncurry districts. Mt Gordon was the only one of nine affected mines that did not discharge poor quality water, with the majority of stormwater being contained in the mined out Esperanza Pit.

Since this time Birla Mt Gordon (Birla) has been liaising closely with Queensland Government regulators regarding reduction of the volume of stored low quality water to a level that presents minimal environmental risk. This paper discusses the processes adopted to manage these risks during a period in which the project was predominantly non-operational and on care and maintenance with limited available on site resources.

2.0 SITE DESCRIPTION

2.1 Location and Landform

The Mt Gordon Operations are located approximately 120 km northwest of Mt Isa in North West Queensland, in the Shire of Mt Isa near the former township of Gunpowder (Fig. 1). The mining operations are located entirely on the Calton Hills pastoral lease owned by the Kalkadoon Aboriginal Council.

The project is situated in the hilly and mountainous Mt Isa highlands. Elevations range from 190 metres (RL AHD) at Gunpowder Creek to 310 metres (RL AHD) at the mine site. Slopes range from moderately inclined to steep.

2.2 History

Copper mineralisation was first discovered near Gunpowder Creek in 1923 by two Afghan brothers. From 1927, the brothers worked a small open cut on the Mammoth Number One ore body. Between 1948 and 1969, Mammoth was worked as a small underground operation. In 1969, Surveys and Mining Ltd took control and embarked on a large scale development including the Mammoth open cut, exploration decline and a copper flotation plant. In 1971, a joint venture between Consolidated Gold Fields Australia Ltd and Mitsubishi took control, establishing Gunpowder Copper Ltd which further developed the Mammoth underground mine. In 1977, due to low copper prices, the operation was placed on care and maintenance.

In 1979 the mine re-opened and production was carried out by “in situ” leaching of broken ore until 1982 when the mine was abandoned. Between 1988 and 2003, the mine was operated in turn by Rension Goldfields Consolidated Ltd, Tremelling Pty Ltd, Adelaide Brighton Cement Holding Ltd, Aberfoyle Limited and Western Metals Ltd. During this period open cut mining of the Esperanza orebody commenced and underground mining of the

Mammoth orebody continued. Ore was processed in a pressure leaching plant to produce copper cathode and tails were disposed of in a purpose built valley fill TSF. In 2003 Western Metals went into administration and Mt Gordon along with other assets was put up for sale. In November 2003, just prior to the operation being closed due to lack of interested purchasers, Aditya Birla Minerals purchased the project narrowly preventing it from becoming another mine site where the government had insufficient funds to address environmental issues. Open cut mining of Esperanza ceased in 2005, underground mining continued and ore processing changed to a conventional sulfide flotation methodology to produce copper concentrate. The operation has largely been on care and maintenance since early 2009 with only short periods of operation in 2010-2011 and 2012-2013.



Fig. 1. Location of Mt Gordon operations

2.3 Climate

The Gunpowder region has a semi-arid climate with a short wet season between November and April that is influenced by the North West monsoon. The mean annual rainfall is 475 millimetres, with a distinct wet season from November to March and a dry season from May to early October. The wet season is characterised by periodic storms and heavy rainfall events. The dry season is characterised by little rainfall and a large diurnal variation in temperature. The average pan evaporation levels are expected to be similar to the 3,104 millimetres per year for the Mt Isa Aerodrome. This indicates a highly evaporative environment with a negative water balance of approximately 2,500 millimetres per annum.

2.5 Geology

The Esperanza copper-cobalt ore body is hosted by the carbonaceous shale and chert breccia of the Esperanza Formation near the convergence of the Mammoth and Mammoth extended faults. This Formation is a sequence of well bedded to locally massive, black carbonaceous to locally grey or grey-green, weakly dolomitic siltstone and pyritic shale. Carbonaceous rocks are dominant, especially in the vicinity of mineralisation. The mineralised unit of the Esperanza Formation has bands of laminated chert and bodies of chert breccia that are approximately 100 metres thick and conformably overlie the silt-shale sequence beneath. Less brecciated units of the Esperanza formation are located north of the Mammoth Extended Fault, south of the Mammoth Fault and immediately east of the highly brecciated mineralised zone. Paradise Creek Formation comprising dolomitic siltstone lies east and south of the orebody (AIMM 1998).

Mineralisation at the Mammoth deposit is hosted by the brecciated units of the Myally Subgroup. The lowermost exposed unit is a pink massive to weakly bedded quartzite which abuts the Portal Fault and is at least 60 metres thick. Overlying the quartzite is a complex sequence of interbedded laminated sandstone, arkose, siltstone and minor quartzite that is 60 metres thick nearest the surface and increases in thickness with depth. Overlying the Myally Subgroup to the surface is the conglomerate, sandstone and siltstone of the Surprise Creek Formation (AIMM 1998).

Pyritic black shale from the Esperanza deposit is the major source of acid and metalliferous drainage (AMD) at Mt Gordon. Mammoth waste rock is characterised by lower sulfide concentrations (as pyrite and associated chalcopyrite and chalcocite) and higher acid neutralising capacity (ANC).

2.6 Hydrology

The operations are located within the catchment of Gunpowder Creek. Gunpowder Creek is the main regional waterway which discharges to the Leichhardt River and subsequently to the Gulf of Carpentaria. Sub catchments potentially affected by the Mt Gordon operations include Greenstone Creek, Mill Creek, Magazine Creek and Esperanza Creek.

2.7 Hydrogeology

The main aquifer at Mt Gordon is located within fractured rock at depths 30 metres below ground level. The baseline groundwater flow regime is that of a relatively flat water table with an elevation of about 220 metres AHD at the Mammoth and Esperanza ore bodies and a gradient of about 1% towards Gunpowder Creek.

This aquifer has a very low permeability (between 10^{-7} and 10^{-9} m s⁻¹) and groundwater flow is generally along fault lines and fracturing. Faulting may give rise to hydraulic isolation and compartmentalisation where the area is virtually isolated from the regional groundwater flow system. Higher permeability brecciated fault zones act as conduits for groundwater flow. The direction of flow is dictated by lithology and structural features. Due to the low permeability of the fractured rock aquifer, groundwater flow follows a tortuous path and the speed of groundwater flow is slow, in the order of 0.3 m yr⁻¹.

2.8 Mt Gordon Receiving Environment

The ephemeral nature of watercourses within Gunpowder Creek catchment controls the amount of water in the creek systems throughout the year. Storm flow within the creeks occurs rapidly with wet season rains and is characterised by highly turbid water. Over the dry season, vast stretches of the Gunpowder Creek watercourse dry out, creating a series of pooled reaches that are subject to high impacts from visiting wildlife and cattle. Minor creeks generally dry completely.

The main receptor of poor quality water from Mt Gordon Operations is Gunpowder Creek, a major tributary of the Leichardt River (Fig. 2).

An assessment by the Australian Centre for Tropical Freshwater Research (ACTFR) in December 2002 (ACTFR 2003) identified Gunpowder Creek as a highly disturbed system as a result of both current and past land use activities, especially mining and cattle grazing. The area near Mt Gordon is also highly mineralised, which results in occasional naturally elevated concentrations of toxicants. For example, copper levels recorded at sites upstream of Mt Gordon Operations have been as high as 3 mg L⁻¹. Also, photosynthetic activity during the day by algae can increase pH values to levels exceeding 10. Gunpowder Creek supports a significant algal population, most likely as a result of nutrient enrichment of water by grazing cattle.

Recharge to the aquifer is via direct infiltration into the fractures and by infiltration through gullies and creeks where they cross these fractures. Due to the low permeability of the surrounding rock, the water table may rise by up to 10 m following recharge periods.

2.8 Mining Infrastructure

2.8.1 Mines

Two pits and two underground mines have been developed since 1927 with a third decline at Esperanza South commenced in 2007/08, but not fully developed.

Mammoth Pit and Underground was mined sporadically by a number of previous project owners. Mining of the pit ceased about 1972 with it being backfilled from 1999 to 2002 with waste from historic heap leach operations and waste rock from Esperanza Pit and Mammoth Underground. Mining in the Mammoth Underground occurred most recently between 2002 – 2009 and 2011-2013 following a period of disuse and consequent extensive period of de-watering. Mining is currently on care and maintenance, but resources in the underground are still considered economic.

Esperanza Pit was mined from 1997 to 2005. Since closure of the pit, it has been used as a water storage. This has provided contingency for capture of potentially contaminated water from the Esperanza catchment during the wet season. A water treatment plant was commissioned in 2007/08 to recover copper and cobalt from pit waters. The plant operated

until January 2009 when it was partially flooded by cyclonic rainfall. The plant was not operated after this time due to technical and commercial difficulties.

2.8.2 Waste dumps

Three main waste rock dumps are present across site. A number of other likely waste rock disposal areas also exist as a result of the natural paucity of flat land and the historical use of waste to form access roads, operational platforms and other mining related infrastructure.

The North Waste Dump is formed from waste from initial development of the Esperanza Pit and received waste from 1997 to 2002. The dump has been constructed such that it only has one open face, sloping north. The southern face forms the embankment of the Esperanza TSF. The dump effectively blocks the valley between the TSF and Gunpowder Creek. The northern face of the dump has been rehabilitated and appears to be geotechnically stable and supports some vegetation, particularly on the berms. The upper surface has not been rehabilitated as it contains stockpiles of waste materials for eventual covering of the upper surface of the Esperanza TSF.

The Mammoth Waste Dump contains waste materials from historic mining of the Mammoth Pit and underground mine as well as small volumes of waste from Esperanza underground and open pit. The Mammoth Pit has been backfilled and now forms part of the Mammoth Dump. The dump fills a valley leading to Esperanza Creek, blocking a small ephemeral tributary referred to as Esperanza Creek East. Waste materials from Mammoth Underground continued to be placed on the dump until suspension of mining activities. The upper sections of the dump have been completed and the surface compacted and covered with non-mineralised waste rock. The middle and lower sections remain active.

The Esperanza Waste Dump primarily contains materials from Esperanza Pit and small volumes from the Mammoth underground and Esperanza South decline development. The dump was commenced in 1997 and contains the majority of the transitional and fresh waste rock from Esperanza Pit. The dump is located on the southern boundary of the mining tenement and blocks the valley immediately upstream of the Esperanza Pit, including Esperanza Creek. Disposal of waste to this dump ceased in 2005 when open pit mining ceased. Small volumes of waste from development of the Esperanza South decline were placed on the upper surface of the dump in 2008. Re-mining of the upper surface commenced on a trial basis in 2010 as low grade ore was identified. The trial was stopped due to processing difficulties.

Esperanza waste rock is typically highly sulfidic (up to 45%) with minor amounts of acid neutralisation capacity (ANC) provided by dolomitic strata. Pyrite and chalcopyrite are the dominant acid producing sulfide minerals and much of the waste is classified as potentially acid forming (PAF).

Mammoth waste rock is characterised by much lower sulfide contents (typically 0.2 to 2%) and variable ANC. The majority of the waste is classified as either benign or potentially acid forming – low capacity (PAF-LC).

2.8.3 Tailings storage

The operation has one historic and one operational tailings storage facility.

The Old TSF was operated up to 1985. It is located immediately north of Gunpowder Creek with the northern embankment constructed on the creek floodplain. Records indicate the

facility leaked poor quality water to the environment since it was commissioned. Since closure and subsequent rehabilitation of the facility, the upper surface has been used for administration offices, laydown areas, evaporation ponds and concentrate drying bays.

The Esperanza TSF is a valley fill style facility located west of Esperanza Pit in a parallel valley. It has a footprint of about 22.6 ha with a storage volume of about 5.6 million m³. The embankment wall was constructed in 1998 using waste materials from Esperanza Pit. The facility has been raised with natural saddles in the adjoining ridges being filled to increase containment capacity. Gunpowder Creek is located north and hydrologically down gradient of the TSF. The facility remained in use until recent suspension of operations.

Tailings produced from Esperanza ore are characterised by high sulfide contents and oxidise rapidly to form seepage with low pH and elevated concentrations of sulfate, copper, cobalt and other metals. Tailings produced from Mammoth ore, which account for most of the tailings deposited in Esperanza TSF in recent years have much lower sulfide contents, although seepage still contains elevated sulfate concentrations.

2.8.3 Water storages

Various water storages have been constructed at Mt Gordon for both process and storm water management purposes.

Raw water for the operation comes from Lake Waggaboonya (capacity 5.775 GL), which was purpose built in 1969 to supply water to the mine and associated township of Gunpowder. Smaller purpose built structures for process water storage, storm water retention and evaporation purposes are located on the Mining Lease. Many have been upgraded over time as operations have varied and environmental management expectations changed. Esperanza Pit (capacity 6.4 GL) and Mill Creek Dam (capacity 1.4 GL) are the largest of the storages with both used for retention of poor quality storm water. Two retention ponds (combined capacity of about 250 ML) and an associated diversion channel were constructed in the Upper Esperanza catchment in 2010 and 2012 to reduce clean water inflows into the Mining Lease.

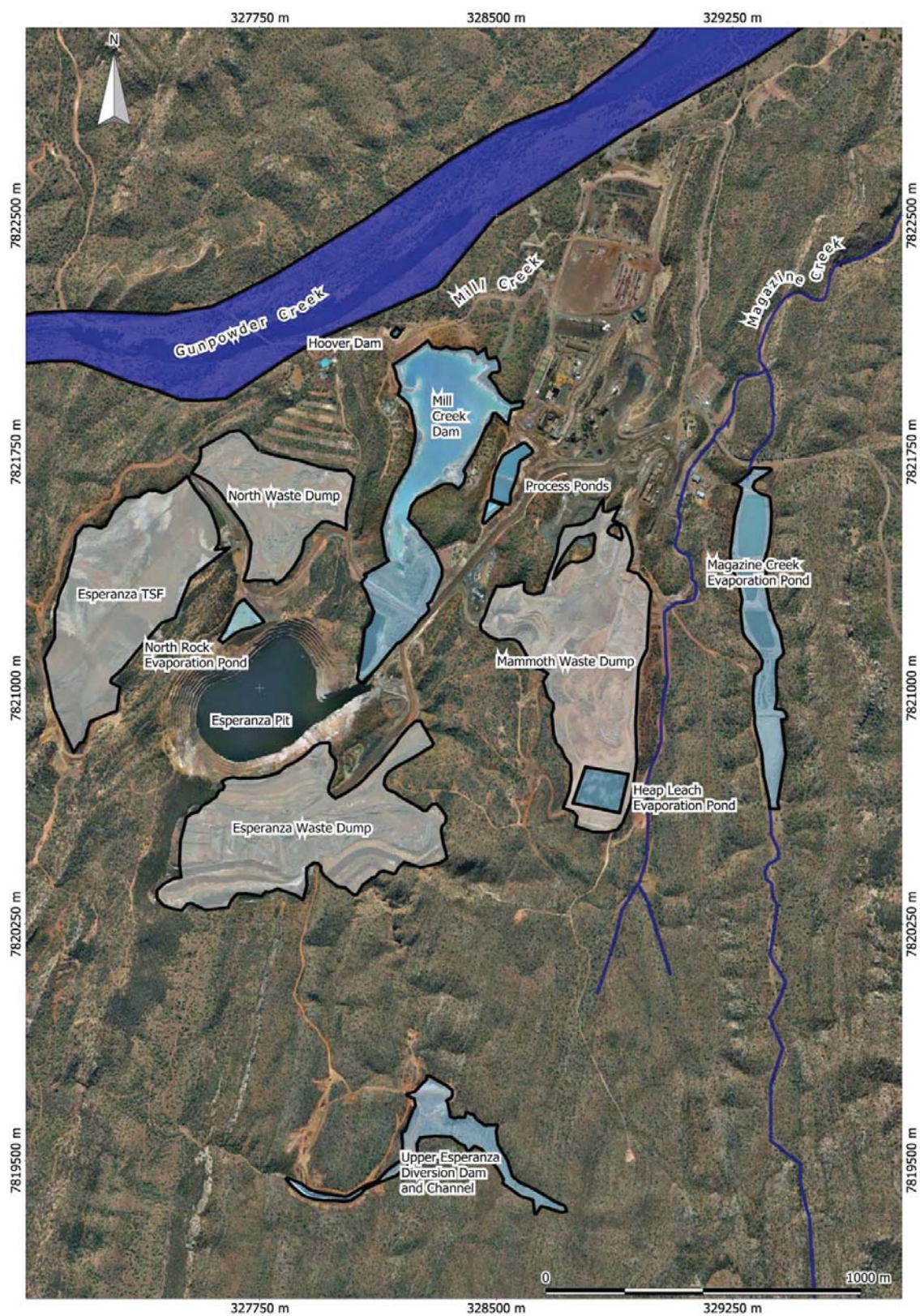


Fig. 2. Site plan layout

3.0 2008/09 WET SEASON EVENTS

3.1 Regional

During the 2008/09 wet season, much higher than average rainfall was experienced in the Mt Isa region with Birla Mt Gordon receiving a total of 1,310 mm. The majority of rainfall was recorded during early and late January 2009 with some significant events also occurring during late December 2008 and early February 2009 (Figure 3). Review of the rainfall record shows:

- Seven days of rainfall above 50 mm were experienced.
- Four days of rainfall above 100 mm were experienced.
- One day of rainfall above 200 mm was experienced.
- There were two major rainfall events of 521 mm over eight days and 434 mm over nine days, separated by 16 days of lighter rainfall in January 2009. This is not long enough for the catchment to dry out so the two events had a combined effect.

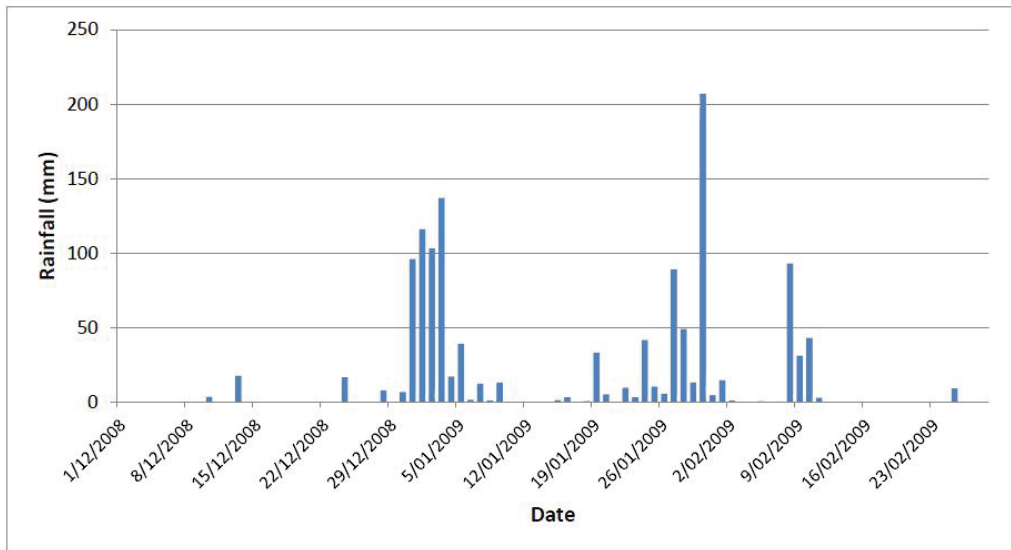


Fig. 3. 2008/09 wet season rainfall by day at Mt Gordon

Based on hydrological model information developed for the project, the rainfall is estimated to be in excess of a one in 500 year ARI two month wet season. DEHP maintain a flow gauging station on Gunpowder Creek adjacent to the mine site. Runoff recorded at this station for the wet season was 470 mm. The highest previously recorded (1971 to present) runoff for a wet season was 407 mm in 1973.

Almost all of the runoff within Birla Mt Gordon mining leases was retained on site, with minimal release of contaminants to the environment. This resulted in onsite water storages being filled to almost capacity.

3.2 Mine Site

3.2.1 Volume of poor quality water

Poor quality water captured during the 2008/09 wet season was stored in two locations – Esperanza Pit and Mill Creek Dam.

A survey on 1 October 2008 recorded the Esperanza Pit water level at RL 183.8 m (volume of 2.7 GL). By 13 February 2009, Esperanza Pit water level was at RL 221 which equates to a volume of about 5.94 GL. As of 28 August 2009, water levels in Esperanza Pit were recorded to be RL 223.01 (6.01 GL) compared to maximum pit capacity of 6.4 GL at RL228.3.

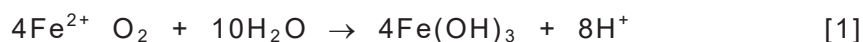
Mill Creek Dam was effectively empty prior to the 2008/09 wet season. It was estimated to contain about 700 ML of water as of May 2009, about 50% of its total capacity.

3.2.1 Water quality

Water quality data for untreated water in Esperanza Pit and Mill Creek Dam as of October 2009 is presented in Table 1. Water quality was similar in the two storages, indicating either hydraulic connection or similar sources of acidic and metalliferous drainage (AMD). Sulfidic waste rock stored in the Esperanza and Mammoth waste rock dumps and exposed sulfidic black shales in the walls of Esperanza Pit are likely sources of AMD.

Acidity in poor quality water at Mt Gordon exists in three distinct forms:

- Hydrogen ion (H⁺) acidity as free sulphuric acid. This form of acidity is highly reactive and therefore rapidly neutralised by hydrated lime until sufficient calcium dissolves to initiate precipitation of gypsum.
- Metal ion acidity associated with metallic ions including aluminium, copper, manganese and iron. Initial reactivity with hydrated lime is relatively rapid, but decreases as a result of surface coating by precipitated metal hydroxides.
- 'Latent' or 'stored' acidity that can be released by oxidation of reduced metal ions such as iron (Fe²⁺), manganese (Mn²⁺) and cobalt (Co²⁺).
- Equation [1] demonstrates how oxidation of reduced iron generates additional acidity. The rate of generation (and hence neutralisation) of this form of acidity is controlled by the availability of oxygen (aeration).



4.0 WATER REDUCTION STRATEGY

A variety of environmental risk reduction options were explored by Birla in response to regulatory concerns and potential legal action if an uncontrolled discharge was to occur. It was recognised the strategy needed to include both water treatment as well as water volume reduction mechanisms. The options for pit water reduction activities were significantly constrained by the project being on care and maintenance since January 2009. Recommencement of operations in late 2009 allowed other water reduction options to be re-evaluated. Considerable pressure was applied by regulatory authorities for rapid volume reduction in stored poor quality water. Decisions made regarding the water reduction strategy were strongly influenced by the operation being on care and maintenance and regulatory actions.

Table 1. Water quality in Esperanza Pit and Mill Creek Dam as of October 2009

Parameter	Units	Esperanza Pit	Mill Creek Dam
pH		2.8	2.7
Acidity	mg CaCO ₃ /L	2,400	3,200
Alkalinity	mg CaCO ₃ /L	-	-
TDS	mg/L	8,900	8,600
Sulfate	mg/L	5,800	5,500
Calcium	mg/L		220
Cu	mg/L	250	250
Co	mg/L	16	28
Cr	mg/L		0.014
Cd	mg/L		0.0023
As	mg/L	<0.003	0.007
Pb	mg/L	0.004	<0.005
Al	mg/L		270
Fe	mg/L		130
Mn	mg/L		110
Zn	mg/L	5.9	5.2

4.1 Objectives

The primary objective of the water reduction strategy was to restore sufficient water storage capacity to reduce the risk of uncontrolled release of poor quality water to the receiving environment in following wet seasons.

Longer term objectives are to establish a sustainable water balance for the site and to minimise generation of AMD from waste storages and historic infrastructure prior to closure.

4.2 Regulatory Requirements

A regulatory framework for developing a practical and cost-effective program for water reduction was established by agreement between Birla and the Queensland Department of Environment and Heritage Protection (DEHP, formerly Department of Environment and Resource Management, DERM). The process commenced with DEHP issuing Environmental Protection Order (EPO) (STAT457) – Water Reduction and Other Actions - on 17 April 2009 in response to the volume of poor quality water on site at that time.

Birla responded by submitting a proposal for a Transitional Environmental Program (TEP) similar to earlier efforts to reduce surplus mine water in 2006 through a Voluntary Environmental Management Program (VEMP). DEHP advocated more stringent water quality targets to those achieved in the 2006 VEMP and recommended technologies including Nanofiltration, Reverse Osmosis and High Density Sludge (HDS) chemical treatment. Although these technologies were capable of producing very high quality water, the required capital expenditure and operating costs were considered excessive, especially as these options were unlikely to be required for long term water management strategies for Mt Gordon.

The end result of the process was the issue of a negotiated Court Order in September 2011, with the prime mechanism for water reduction being controlled release of treated water during the wet season according to the following criteria:

- Flows in Gunpowder Creek must exceed the 25th percentile for wet season flows.
- Treated water quality must comply with the limits listed in Table 2.
- The flow volume in Gunpowder Creek at the time of release must be at least 20 times the volume of water discharged.

Apart from water treatment methodology and appropriate treated water quality targets, the main point of discussion was the acceptable volume of untreated water that could be stored in Esperanza Pit. The maximum relative level (MRL) was set at RL 184 m at 1 November each year, which is 10 m below the RL of 194 m level of the invert of Gunpowder Creek. This was based on application of the precautionary principle and assumes good hydraulic connectivity between Esperanza Pit and Gunpowder Creek between the RL 184 and 194 elevations. Birla presented a case for increasing the MRL stating that there was no evidence of hydraulic connectivity between Esperanza Pit and Gunpowder Creek until RL of about 220 m was reached. Birla felt strongly that untreated water was a potential resource in this highly variable rainfall environment and that by limiting storage volumes to RL 184 m, regulatory agencies were potentially compromising future operations.

Table 2. Water quality discharge criteria

Parameter	Units	Limit	Limit Type
pH		7.0 to 9.0	Range
TDS	mg/L	9,000	Maximum
Sulfate	mg/L	6,000	Maximum
Cu	mg/L	2.5	Maximum
Co	mg/L	5	Maximum
Cr VI	mg/L	0.001	Maximum
Cd	mg/L	0.002	Maximum
As	mg/L	0.25	Maximum
Pb	mg/L	0.05	Maximum
TSS	mg/L	30 mg/L or 10% above background	Maximum
Dissolved oxygen	mg/L	2	Minimum
Oil and grease	mg/L	20	Maximum

4.3 Water Treatment Strategy

Of the various water treatment processes evaluated by Birla, dosing with alkali was considered to be the preferred method to meet the criteria in Table 2, taking into account timeframes for implementation and optimisation, safety to operators and the environment, and suitability of available infrastructure.

4.4.1 Lime dosing

Dosing untreated water with hydrated lime will neutralise acidity, precipitate metals and reduce soluble sulfate concentrations by precipitation as gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$). Birla had prior experience at Mt Gordon for treating poor quality water using lime dosing. Existing infrastructure included lime storage facilities, a lime slaking plant for preparing hydrated lime slurry ($\text{Ca}(\text{OH})_2$) from quicklime (CaO) and a 60 m³ capacity lime neutralisation tank. As the operation was on care and maintenance, this infrastructure was readily available for use.

It was understood that it is important that the relative amounts of free and metalliferous acidity are known to allow development of an effective neutralisation strategy. Analysis indicated that approximately half of the total acidity was present as hydrogen ion (H^+) acidity and half as metal acidity, dominated by aluminium, iron and manganese.

Dosing with other alkaline materials such as caustic soda, soda ash and magnesium oxide was considered, but was not implemented because of greater costs, reagent storage and handling issues. Laboratory experiments were conducted to determine the minimum time required to neutralise acidity and precipitate metals. Review of results showed that:

- The minimum dosing rate to neutralise the free acidity and remove most of the copper and cobalt was at least 2.5 grams of hydrated lime per litre (Table 3), probably closer to 3.0 grams per litre.
- At the trialled dosing rates, metal removal was effective, but only modest reductions in TDS and sulfate were realised. It would be necessary to increase the pH to 9 to 10 to achieve maximum reduction of TDS and sulfate.
- The reaction was quite rapid, with most of the neutralisation and metal precipitation occurring within one or two hours of contact time.
- Aeration did not appear to be necessary. Slightly higher pH values were achieved, but variable results were achieved in metal removal; in some cases the residual concentrations of soluble metals increased slightly.
- The effectiveness of metal removal is mainly controlled by pH:
 - Iron was reduced to insignificant levels by pH = 5.
 - Aluminium removal was maximised between pH = 6 to 7. Its solubility increased again under more alkaline conditions, although these levels were not problematic.
 - Copper removal was effectively complete by pH = 6.5.
 - Zinc removal was effectively complete by pH = 7.0.
 - Cobalt removal required a higher pH, effectively complete at pH = 7.0 to 7.5.
 - Lime dosing at these rates did not reduce manganese levels below the EIL value of 1.9 mg L^{-1} for freshwater.

In summary the results of the lime trial demonstrated that the best outcome that could be expected from lime dosing was to increase the pH to a value between 7.5 and 8.0. Water quality at this point would be suitable for natural evaporation, mechanical evaporation or irrigation to land. Based on the known acidity values of untreated water, the minimum dosing rate required to achieve this would be approximately 2,500 to 3,000 tonnes of hydrated lime per ML. At this point, sulfate levels would have been reduced by only 15% at pH 7.5, i.e. from 4,200 to 3,600 mg L^{-1} . More significant reductions were expected by increasing the target pH to values between 8.0 and 8.5. It was recognised that this treatment process would produce a significant amount of sludge consisting of gypsum and metal hydroxides.

Consideration was given to using a 60 m^3 capacity lime neutralisation tank, construction of a new HDS treatment system, *in situ* lime dosing of Mill Creek Dam and *in situ* lime dosing of Esperanza Pit. The adopted strategy was a combination of the 60 m^3 capacity lime neutralisation tank and an enhanced *in situ* lime dosing of Mill Creek Dam that utilised a natural feature known as the “Bat Cave” for lime addition and settling of water treatment sludges.

Water in Mill Creek Dam was successfully neutralised in the 2009 dry season, but re-acidified as a result of acidic inflows during the 2009/2010 wet season. It was neutralised again in the 2010 dry season and was maintained at neutrality until February 2011 at which time lime dosing was interrupted as a result of supply issues associated with adverse weather

conditions elsewhere in Queensland. The dam contents were neutralised again once supplies resumed some 4 weeks later.

Table 3. Acidity and metal ion concentrations of Mt Gordon wastewater

Parameter	Concentration mg L ⁻¹	Hydrated Lime Consumption mg L ⁻¹ (equivalent to kg ML ⁻¹)
Total Acidity to pH 8.3	2,400 (as CaCO ₃)	1,780
Aluminium	110	453
Iron	110	146
Copper	93	108
Cobalt	18	23
Manganese	97	131
Zinc	5.9	6.7

In situ dosing of Esperanza Pit water was not attempted since untreated water is preferable as a resource for mining operations because of the potential for recovery of copper and other metals and problems associated with re use of treated (lime dosed) water because of scaling of pumps and pipes. By treating the 55% of the stored water needed to be removed to comply with the Environmental Authority, the remainder would be significantly more difficult to re-use in future operations. Efficient application of lime to the deep, steep sided pit was also considered problematic.

Over 3 GL of poor quality water was treated between September 2010 and December 2011. Treated water was stored in Mill Creek Dam before either being discharged to Gunpowder creek when suitable conditions were present or being used as part of other water reduction methods.

4.4 Water Volume Reduction Strategy

A considerable risk factor for the operation was the occurrence of another above average wet season. It had been recognised that runoff from upper Esperanza catchment flowing through the Esperanza Waste Dump and into Esperanza Pit was the major contributor to the volume of poor quality water being accumulated on site. In 2009/10, a 320 m diversion channel and a 200 ML dam was constructed upstream of the mining lease to divert clean runoff to Gunpowder Creek upstream of the operations. This has reduced the Esperanza Catchment area reporting to Mt Gordon operations by 68% and thus has significantly reduced the risk of future accumulations of large volumes of poor quality water on the Birla Mt Gordon site.

Options for reducing the volume of stored water were primarily based on discharge of appropriately treated water to Gunpowder Creek, passive and enhanced evaporation and beneficial re-use options such as substitution for raw water in the process plant, dust suppression and land irrigation.

An Ecological Risk Assessment (ERA) was undertaken to understand the likely impacts of releases of varying qualities and volumes of water to Gunpowder Creek. ERA is a set of formal, scientific methods used for defining and estimating the probabilities and magnitudes of adverse impacts on plants, animals and/or whole ecosystems posed by a particular stressor. The ERA process identifies the ecological receptors of concern, estimates the

concentration that the ecological receptors are exposed to and, based on the magnitude of this concentration and determines whether the ecological receptors and values are at risk. ERA is a tool advocated for use by the National Environmental Protection Council and in Chapter 3 of the ANZECC 2000 Water quality Guidelines, but in practice has been little used in Australia by the mining industry. Regulatory authorities have historically preferred to use conservative approaches applying ANZECC water quality guidelines as strict criteria. This approach was taken by DEHP with the outcomes of the ERA used by Birla to help inform decision making, but not accepted by DEHP.

4.4.1 Controlled discharge

Controlled discharge of treated water to Gunpowder Creek was considered the preferred option in terms of cost, available infrastructure and acceptable level of risk to the receiving environment.

It was recognised, however, that this option was highly dependent on flows in Gunpowder Creek, which were known to be highly variable in terms of volume and duration. Options for early wet season discharge of small volumes of treated water to Gunpowder Creek and discharge of treated water to Lake Waggaboonya were rejected by DEHP. Reasons centred on concerns regarding sulfate concentrations exceeding ANZECC livestock drinking water standards.

Controlled discharge options were thus restricted to discharge during flow events in Gunpowder Creek where treated water quality met strict quality criteria and discharge volume was tied to flow conditions.

Volumes of treated water discharged to Gunpowder Creek are shown in Table 4. Controlled discharge of 3,260 ML has occurred from 2009 to date. Drought conditions experienced in the 2012/13 wet season resulted in very little water being able to be discharged.

Table 4. Controlled Discharge Volumes 2009-2013

Wet Season	Discharge Volume (ML)
2008/09	0
2009/10	97
2010/11	1,097
2011/12	1,903
2012/13	163

4.4.2 Evaporation

A trial was established in 2009/10 to assess the effectiveness of high capacity evaporators for water reduction. Although the nominal capacity of the evaporators was 2 ML day⁻¹, results from the trial indicated much lower evaporation rates which were attributed to low evaporation efficiencies and extended downtime required for maintenance of the pumps of pipes due to scaling, especially when treated water was used.

In the negotiation process with DEHP, Birla agreed to install and operate nine high capacity evaporation units. These operated year round using predominantly untreated water from Esperanza Pit to overcome high maintenance requirements associated with use of treated water. The average dry season reduction potential was estimated to be approximately 380 ML. Total actual evaporation from the high capacity evaporators for 2011 was about 560 ML. Other passive evaporation processes implemented were application of treated water to the Esperanza TSF during periods when the operations were on care and maintenance, use of treated water for dust suppression, construction of a low pressure land irrigation sprinkler system on the banks of Mill Creek Dam and construction of lined evaporation ponds at North Rock Dump (between Esperanza Pit and Esperanza TSF) and on a decommissioned Heap Leach Pad.

4.5 Water Re-use

Substitution of untreated and treated water in the ore processing plant were investigated and implemented when mining and ore processing operations resumed in late 2009 and then again in 2010. This improved the overall site water balance and assisted with water reduction. Use of this option was limited to when the project was operational.

A dry season land irrigation system was trialled to investigate use of treated water for pasture irrigation on Calton Hills Station. The results indicated application of treated water had not resulted in build-up of soil contaminants and warranted further investigation if ongoing needs for water reduction existed.

4.7 Current Situation

At the end of the 2013 dry season, Esperanza Pit contained about 4.3 GL of poor quality water and Mill Creek Dam contained about 600 ML of treated water suitable for discharge to the environment if water flows in Gunpowder Creek occur.

Water treatment activities have largely been suspended until water currently stored in Mill Creek Dam can be discharged. Use of high capacity evaporators is continuing and is anticipated to result in further reductions in stored water volumes in Esperanza Pit. The wet season forecast indicates the Mt Isa region is likely to experience a below average wet season for 2013/14 and thus the opportunities for controlled discharge are likely to be very low. Discharge infrastructure is maintained ready for use in accordance with DEHP Court order requirements.

5.0 CONCLUSIONS

Within a six week period in January and February 2009, water management at the Birla Mt Gordon operation changed from one of facing extreme measures for procurement of sufficient water to maintain production to one of narrowly avoiding discharge of poor quality mine water to the environment, as was experienced at eight other mine sites in the Mt Isa region. The challenges for reducing the volume of stored poor quality water to reduce risk of overflow in following wet seasons was compounded by the mine being placed on a care and maintenance basis following these floods and a legacy of AMD issues arising from a mining history of more than 80 years.

A regulatory framework for treating and discharging excess water was established following a negotiated settlement between Birla and DEHP. Birla's position was that the process should be managed in a staged approach with the primary objective to be avoidance of the risk of overtopping in subsequent wet seasons in a cost-effective manner. It argued that some of

the excess water should be considered an asset rather than a liability and offered a means of “drought proofing” the project during expected periods of below average rainfall. Ecological risk assessment was adopted to provide an improved understanding of potential impacts of uncontrolled discharge, which were found to be less severe than first anticipated. Being primarily responsible for the receiving environment, DEHP advocated stringent conditions for treated water quality and discharge volumes, and considered stored poor quality mine water as an environmental liability.

A program based on water treatment with lime dosing to neutralise acidity and reduce concentrations of sulfate and metals (soluble and particulate) to levels that have been shown to present minimal risk to key receptors in Gunpowder Creek was negotiated and successfully implemented at Mt Gordon. Water reduction has been achieved by a combination of high capacity evaporators (about 1.1GL to date), low tech passive evaporation systems and controlled discharge of treated water to Gunpowder Creek during subsequent wet seasons (3.2 GL).

Over a four year period and a cost of approximately \$20 million, water volumes have been reduced to a point where there is more than sufficient capacity to store wet season runoff from a 1 in 100 year three month ARI wet season event in Esperanza Pit without using other onsite storages. Although the volume of remaining water currently exceeds limits stipulated in the current site Environmental Authority, it is regarded by Birla as a substantial asset to mine operations in a drought-prone environment. A case is currently being assessed by DEHP for what Birla considers a more appropriate MRL for water storage that considers both liability of excess water storage against the benefits of re-use of mine water.

The risk of the operation accumulating such a large volume of poor quality runoff in the future has also been substantially reduced through construction of a number of upstream diversions to prevent water from contacting oxidised waste rock and ongoing works to reduce the production of contaminated water from historic infrastructure.

6.0 REFERENCES

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BEYOND THE PAF CELL

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ABSTRACT

For many mining operations the PAF cell is the default solution to dealing with problematic waste. Historically this solution has allowed mine planners to adopt a position in the short term that any problematic material can be dealt with at any point in the mine schedule. In addition the blanket use of "engineered covers" as long term closure solutions allows the deferral of acid mine drainage (AMD) risk mitigation to mine closure. The net result is that the mine schedule is often prepared with little consideration of AMD issues which have been "risked" away.

However this approach may in many cases be leading operators to misprice the level of risk that AMD issues can present, even in the short term, due to a number of reasons such as early closure and the presence of fast reacting materials. In addition this approach also seems to be at odds with the general international principles of risk management which are grounded in the accepted view that prevention is better than mitigation.

To assess alternative options for waste management at a site in WA, O'Kane Consultants (OKC) carried out an assessment and design for co-disposal options of PAF. With an integrated approach to AMD management and working closely with site geologists and mine planners, co-disposal options have been proven to be compatible with the mine schedule. In addition detailed geochemical testing including kinetic testing carried out at the planning stage to support co-disposal has allowed the AMD risk profile to be better quantified over life of mine.

Passivation of sulfide oxidation reactions (leading to reduction in short term AMD risks), is for example a key benefit of co-disposal that can be demonstrated as a result of kinetic testwork. Additionally long term risks are shown to be reduced due to less reliance being placed on a "final cover solution" as a result of progressive compact and lift construction techniques.

1.0 INTRODUCTION

Acid and metalliferous drainage (AMD) is arguably the most significant "liability" that mining companies retain post closure. Management costs for AMD have been estimated globally at approximately \$US 1.5 billion per annum. (Lottermoser 2003). The total worldwide acid mine drainage liability for both historic and operational mines has been estimated at in excess of US \$10 billion (Lottermoser 2003). Total costs in Australia have exceeded \$US 500 million to date (Australian Nuclear Science and Technology Organisation 2003).

There are two major factors that have resulted, and arguably still are resulting, in AMD liabilities being created at mine sites.

- AMD problems often reveal themselves at a late stage in the development of a mine, in some cases they may not be evident for many years after closure (or potentially after abandonment). As a result of the 'lag' that may occur before AMD impacts are evident, environmental and financial risks are heavily weighted to the mine closure

rather than operations period of the life cycle of the mine. A result of this lag time AMD risk is often underestimated by operators as there is a link made between the observation that there is “no problem at this time” to a conclusion that there “will be no problem”. The presence of this lag effect therefore tends to foster complacency at sites where significant future problems may be a few years around the corner. The management of AMD is consequently impacted by the effect this lag time has on decision making as short term thinking does not translate well to long term issues like mine closure.

- In the planning stages of a mining project, fundamental decisions are made about the operation including waste storage and management. Although the fine detail has not yet been developed, conceptual designs are drawn up which commonly include footprints and designs for waste rock dumps (WRDs) and tailings dam (TSFs). This can prove to be a problem as it is still not routine for the use of geochemical modelling of AMD-classified material in conjunction with the early planning and scheduling of mining activities. A result of this is that concept designs are conceived before detailed geochemical assessments have been completed. A consequence of this “design before facts” approach is that retro fitting of the concept designs is then required if an AMD problem is identified later on.

The consequence of these two factors has been the proliferation of the use of PAF cells for management of PAF materials. This is because:

- A PAF cell can be “retro fitted” into almost any waste rock dump
- Poorly designed PAF cells may not result in any notable impacts for many years therefore problems can be “hidden”

The use of PAF cells for these reasons could be described as reverse engineering. The primary design consideration for the storage of PAF material should be aligned with industry best practice which, based on the international principles of risk management, are grounded in the accepted view that prevention is better than mitigation. The PAF cell in many cases does not meet this most basic principle.

2.0 PLANNING FOR FAILURE

Some planning issues that commonly result in poor decision making with respect to waste materials management include:

- Designs from a previous operation, or stage of mining at the site are “copy and pasted” using a “one size fits all” approach. This may then result in many stages of retro fitting and reverse engineering to make the design “fit for purpose”.
- Material(s) that were assumed to be suitable for construction purposes are later classified as unsuitable due to AMD risks. This may result in a significant loss of construction materials
- The volume of material that requires management as a result of AMD risks is significantly underestimated at the planning stage. This can result in the design of containment facilities at a late stage of planning. This in turn may result in significant cost and/or time delays as design metrics like footprints and/or height restrictions for waste dumps may already be set by planning consents.
- A key assumption is made during the approvals process that effective management of AMD material can be achieved by construction of a PAF cell. This may not be an appropriate means of management however if PAF material will be produced in small

volumes over a long time period. The construction and management of a single PAF cell is not practical over this timeframe. As a result the management measure proposed may be ineffective with respect to managing AMD risks.

- An early stage mining proposal is introduced during the planning process (for example an early ore mining option to generate positive financial returns). Planning decisions for AMD management may therefore be expedited. This may result in management measures being proposed that are not compatible with further stages of mining.

As previously stated, these issues generally conspire to the adoption of the PAF cell as the “solution” to PAF management. This would not be an issue if the PAF cell could be ubiquitously used as an effective solution, however there are a number of key reasons why this is not the case:

1. PAF cells are not generally engineered containment facilities. The common term used in industry is “encapsulation” however this is misleading as it implies a degree of engineered containment. In reality PAF cells are merely buried waste masses.
2. PAF cells rely on cover systems that reduce infiltration to function as they are not engineered structures. Any infiltration into the WRD will flow into a PAF cell unimpeded. However cover systems are placed on the surface of WRDs which is the most common failure point for these structures due to erosion/vegetation loss/weathering etc. The factor of safety for a PAF cell is often only as high as that of the cover system which in many cases does not offer a high factor of safety over long term time frames. (Wilson ref)
3. The internal structure of a WRD as a result of crude dump construction methods results in the formation of internal hydrology/gas regime that does not conform to the assumptions made when the PAF cell was “designed”. In simple terms oxygen comes from under the PAF cell in the dump not from the surface and water can come in from the sides.
4. Waste dumps are not routinely equipped with internal monitoring equipment making it difficult to validate the performance of the design
5. Construction of waste dumps at many sites is not completed with detailed QA/QC control and documentation measures. Cover systems are often proposed to limit infiltration into the WRD, however the most common failure point on a WRD is at the surface. As a result cover systems have an inherently low factor of safety in design in many instances.
6. The waste schedule is not compatible with the PAF cell design resulting in PAF materials that come out of the pit:
 - a. Too early
 - b. Too late
 - c. In too much volume
 - d. Are very reactive and so can’t be left exposed for long periods

3.0 WRD DESIGN ISSUES

Some of the key issues that have an impact on the design of PAF storage facilities within WRDs and the success (or otherwise) are summarized below:

3.1 WRD structure

Generic conceptual models for WRDs have been published by many authors, and many guidance documents contain “industry standard” schematics. However, what may appear to

be a useful concise summary “picture” is potentially dangerous if these are assumed to be based on complete and robust technical models. Some authors have attempted to incorporate field data and scaled laboratory experiments to make these models more technically robust (notably the work of Ward Wilson, e.g. Wilson 2011), however this approach has yet to translate to the more generic “industry standard guidance”.

Figure 1 is taken from Australian Government DITR (2007) and clearly shows the limits of a generic model. For example the oxidized zone is shown as a simple halo around the edges of the waste, and infiltration is indicated to be vertically uniform. Whilst potentially applicable to homogeneous waste materials such as tailings, this model is not likely to be reflective of waste rock dumps. The location of a standard PAF cell is indicated on this Figure which is positioned to reflect the understanding of this conceptual model, i.e. that the middle of the dump is the best place as it is “encapsulated” from oxygen and water ingress.

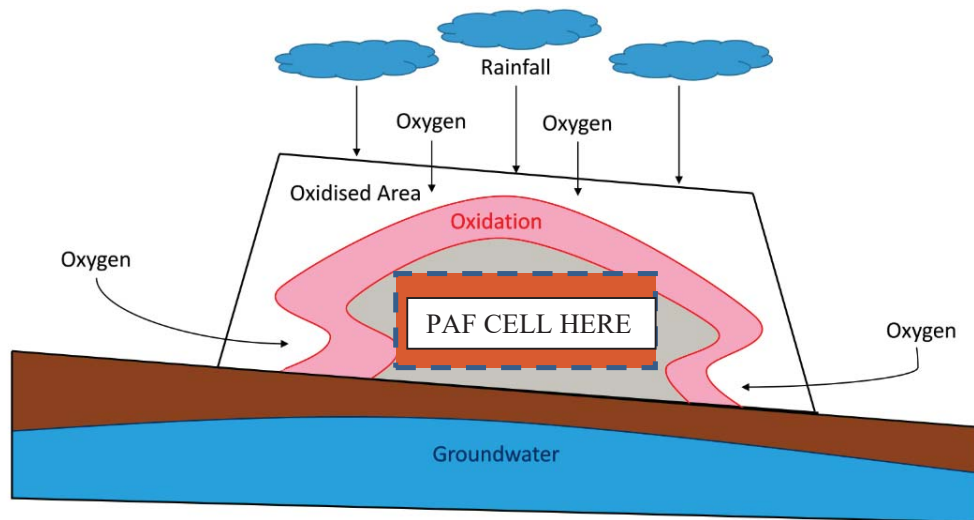


Fig. 1. Generic model of AMD production and contaminant migration from a waste rock dump after Australian Government DITR (2007).

The conceptual model shown in Figure 1 is commonly used for decisions on PAF cell placements however this model does not reflect the internal structures of dumps created as a consequence of the prevalence of end tipping material. These structures have intrinsic hydrologic characteristics which control oxygen and water flow throughout the waste material (Wilson 2011). Figure 2 is a conceptual model showing a cross section of a typical WRD constructed by end dumping material. The segregation of coarse and fine grained material into parallel bands along tip faces is a common feature of end-dumped WRDs. The segregation of material as shown in Figure 2 has been confirmed by WRD excavations in the work of Wilson (2011). The conceptual cross section shows that the infiltration of water enters the WRD at the top of the pile, percolating down through areas of fine grained materials. During heavy rain events such as cyclones, water will also enter the WRD in coarse grained sections quickly percolating to depth. Oxygen ingress primarily will enter the WRD at the bottom of the pile moving upwards through the free draining course material layers.

The presented conceptual internal structure of WRDs constructed by the common practice of end dumping is an ideal scenario for the production of AMD given the ample supply of atmospheric oxygen and water.

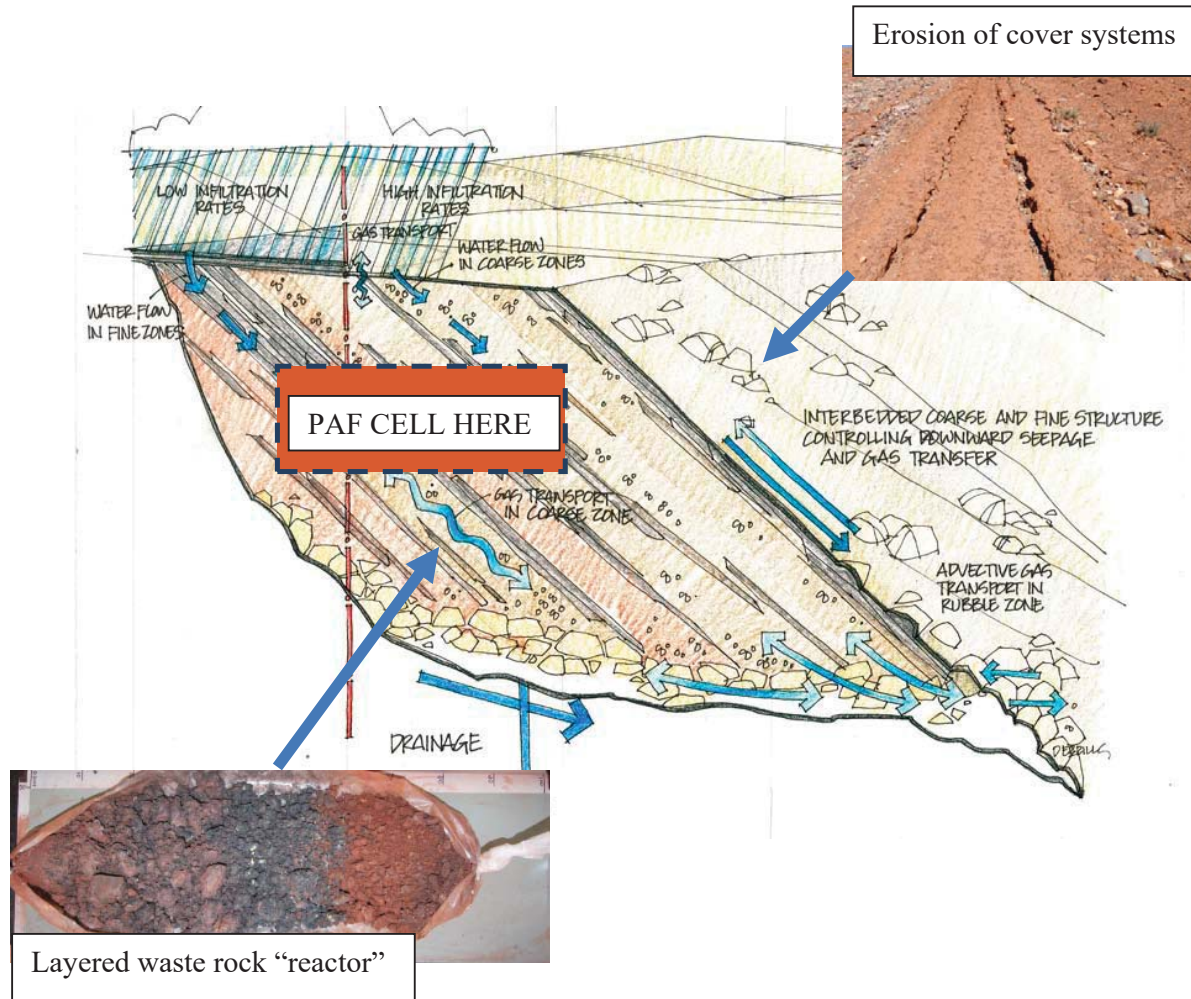


Fig. 2. Conceptual cross section of end dumped WRDs illustrating oxygen and water transport

3.2 WRD design

In contrast to other engineered waste management structures like domestic landfills, construction and regulator approval of WRDs commonly does not include detailed engineering specifications, as built drawings and QA/QC documentation. This has resulted in many WRDs being constructed poorly and often with limited planning/thought having been made with respect to materials management and placement. This is no clearer than the prevalence of end tipping in the construction of WRDs. End tipping as has been shown previously creates the perfect conditions for AMD to occur within a dump. Even when an un-engineered unlined PAF cell is proposed within a WRD, end tipping is still used to construct the dump, and in some instances the PAF cell itself.

In general PAF cells rely on cover systems that reduce infiltration into the WRD to function as they are not engineered structures and so are vulnerable to ingress of water and/or oxygen. This means that PAF cells are not actually the management measure to protect against AMD in their own right, rather it is the cover system that is the management measure. This is because any infiltration, or oxygen ingress into a WRD will flow into a buried PAF cell unimpeded as there is generally no “containment structure”. Many successful engineered cover designs have been implemented so the reliance on the cover system cannot be seen as a significant risk in all instances, however the fact remains that the most common failure point on a WRD is at the surface. As a result many cover systems have an inherently low factor of safety in design compared to alternatives such as truly engineered containment systems and layered co disposal options (discussed herein).

3.3 WRD monitoring

In contrast to other waste management industries such as domestic landfill, WRDs are generally not subject to active monitoring by way of internal instrumentation being installed (with the exception of cover systems). This “walk away” approach has resulted in the creation of potential environmental hazards as processes like AMD take many years to develop and often will go undetected for decades until a significant AMD discharge occurs without warning.

With the advancement of sonic drilling technology WRD can be assessed at depth to evaluate:

- Internal hydrologic conditions and the response of the WRD to climatic variables such as incident precipitation and pressure
- Internal processes such as heat generated through sulfide oxidation
- Movement and replenishment of oxygen through dump structure
- The location of dump structure such as rubble zones and underlying historical drainage pathways etc

The placement of oxygen probes, and pore pressure piezometers would allow the performance of the PAF cell to be monitored over time, although this at present is not industry standard practice.

3.4 Waste schedule

There are a number of planning factors related to the mine schedule that can make PAF cell use challenging:

- Waste schedules are not always produced, or are generally not very accurate with respect to AMD volume calculation or the time over which PAF will be produced
- PAF may be excavated early in the mine life before there is a suitable place to locate a PAF cell due to WRD construction limitations
- PAF may be excavated late in the mine life and disposal requires re-excavation of the waste or near surface disposal
- Reactive material requires immediate cover and, PAF cells may have been designed to be open for long periods which increases the exposure of the material to oxidation reactions
- A PAF cell looks simple on a schematic but the designation of one (or a few) areas for PAF deposition will restrict the operator to placement of PAF in very small area of the dump that may not be compatible with the mine schedule.

- The production of PAF may not be as planned in the schedule and so a designated location in the WRD may not be accessible.
- More PAF may be excavated than thought and so the cell will be filled faster than thought. Additional “cells” are then required in areas not previously planned for PAF disposal.
- A detailed waste schedule may not have been created at the time the design needs to be completed for the dump. The location and size of PAF cells may therefore not be appropriately designed.

An example schedule is shown on Figure 3. Although the volumes of PAF are not large, it is only once the waste schedule is viewed that the management challenges become apparent. From this figure it is clear that if PAF production occurs later on in the schedule and therefore a PAF cell is proposed, it may have to be close to the surface of the WRD if construction is from the bottom up as is standard practice.

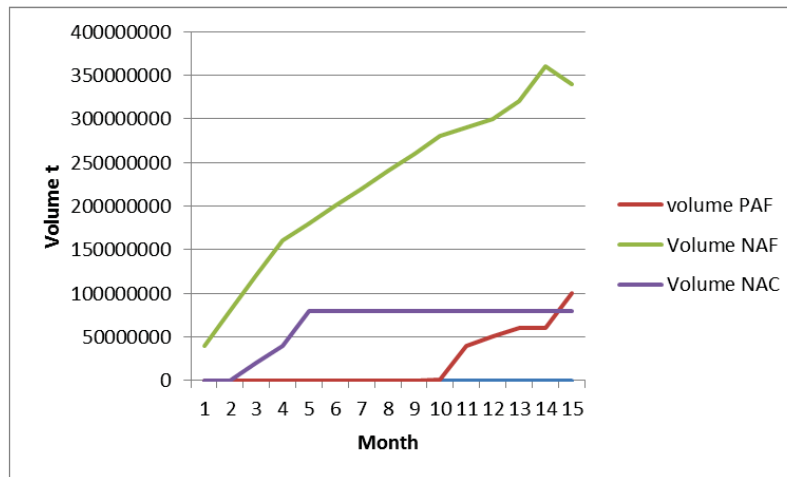


Fig. 3. Example of mine waste schedule from mine waste model (Pearce et al 2012)

4.0 CO DISPOSAL AS AN ALTERNATIVE

Co disposal has been used as a design alternative to cells for a number of years however the practice is not wide spread. The main reasons for this are that co disposal options are considered to be:

1. More expensive than a “containment cell”
2. Not compatible with the mine schedule
3. Too complicated to build

These are mainly issues relating to practical site management and planning issues, all three of these pre conceived “problems” are however not necessarily correct.

1. It is true that a co disposal option is more expensive in the short term and requires a more complex engineering design than a traditional PAF cell. However given a crude PAF cell is only one step above do nothing option then the cost base is not an applicable comparison. In addition the cost of placing and maintaining a complex

- cover system is often not taken into account, nor are the long term closure liability costs of the “failure” of the PAF cell that could result in AMD.
2. A co disposal option may in fact be more flexible and therefore compatible with the mine schedule than a PAF cell as multiple depths/areas can be used for PAF deposition. Scheduling problems with PAF cells include:
 3. Design complexity and engineering challenges of placing and maintaining a complex cover system to perform over decades/100s years to reduce infiltration (as PAF cells rely on this assumption) are often not taken into account.

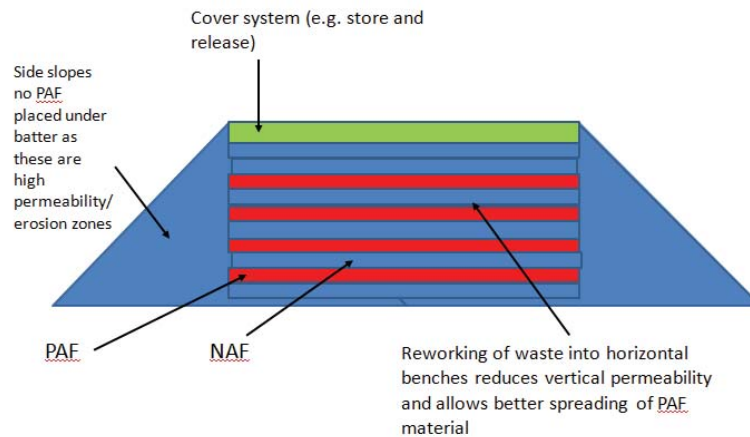


Fig. 3. Concept co disposal option

The general concept of co-disposal is to include:

- Horizontal layering to reduce vertical permeability and consequently oxygen and water ingress
- Co mingling of PAF and non acid forming (NAF), preferably with buffering properties to reduce the reaction rates of the PAF material
- Less reliance on the cover system to perform adequately as a mitigation measure for AMD

5.0 EXAMPLE OF CO DISPOSAL OPTION

OKC assisted with the management concept for a site in WA which required management of PAF materials. Details of the site are as follows:

- PAF materials were identified in the geological block model as pyritic banded iron formation (BIF)/chert in relatively small quantities <10%
- Kinetic testing indicated the materials were fast reacting as they have no buffering capacity
- Leach testing identified metals release as a low but potential risk if oxidation of the PAF materials occurred
- The mine schedule incorporated a geochemical block model which predicted that PAF materials would be produced over much of the life of the operation due to the location of the material in the pit shell

- Disposal under the slopes of WRDs was required

A PAF cell was not deemed to be a suitable option given the parameters of the site, instead a co disposal option was investigated as a potential design solution.

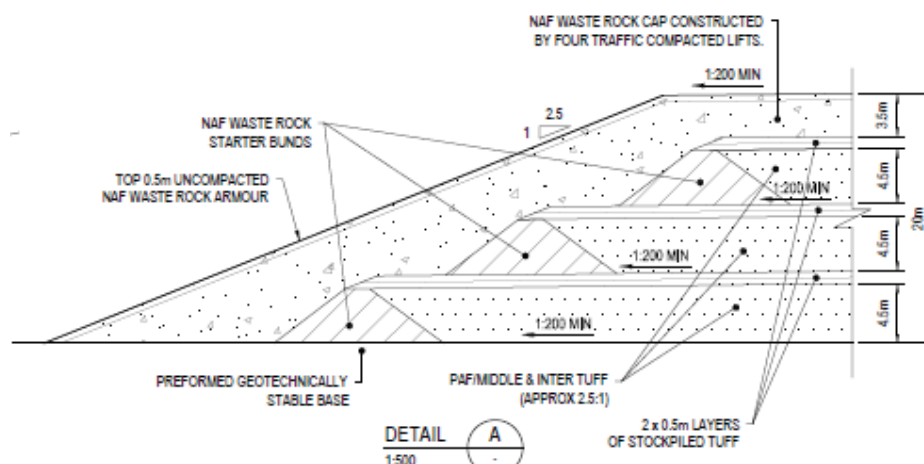


Fig. 4. Example of detailed engineered co disposal where under-slope PAF disposal is required

The co disposal option was preferable over a PAF cell for the following reasons:

- There is flexibility in the disposal of PAF over the life of the WRD as there is not a restricted area in which PAF has to be placed. Rather PAF disposal can be completed in a progressive manner. This is important as PAF was predicted to be produced over the life of the operation.
- Disposal under slopes was required due to site constraints and dump land forming therefore engineered containment was required as infiltration could not be guaranteed to be prevented by a cover system
- There was an availability of NAF Tuff material on site that can be easily compacted into low permeability layers. Tuff material was also likely to be produced progressively over the life of the operation
- Geochemical assessment indicated the Tuff materials may act to reduce AMD discharges as a result of passivation of pyrite oxidation, retardation of metals leaching, and buffering of pH.

The design allowed for co disposal of PAF and NAF Tuff materials together in paddock dumped lifts of 4.5m. In between lifts, traffic-compacted layers of Tuff were to be placed to form horizontal low permeability breaks to oxygen and water ingress.

The design was operationally simple as PAF can be placed at the rate it comes out of the pit. Some areas of the dump may have no PAF as a result of excavation schedule but this does not matter in context as the design does not rely on PAF being placed in specific areas.

6.0 GEOCHEMICAL TESTING OF CO DISPOSAL OPTION

A series of detailed geochemical tests were completed to validate the co disposal option.

Two materials were identified as potential co disposal agents:

- NAF Tuff (with no appreciable PAF component)
- NAF BIF with no appreciable PAF component and appreciable acid neutralizing capacity (ANC) as a result of presence of carbonates

The tuff was found in discrete layers and so could more easily be separated and recovered while the BIF was more difficult to recover.

Material that required management included pyritic material with an elevated PAF potential which was found in both BIF and in a Tuff layer. Kinetic net acid generating (NAG) tests indicated that this Tuff material with a similar (net acid producing potential) NAPP to BIF material was not as fast reacting. The chart below shows a kinetic NAG test carried out on:

- PAF Tuff NAPP: 66 $\text{H}_2\text{SO}_4/\text{t}$
- PAF BIF NAPP: 43 $\text{H}_2\text{SO}_4/\text{t}$

As can be seen the Tuff sample has slower reaction kinetics than the BIF sample indicating that it may have some buffering/passivation properties.

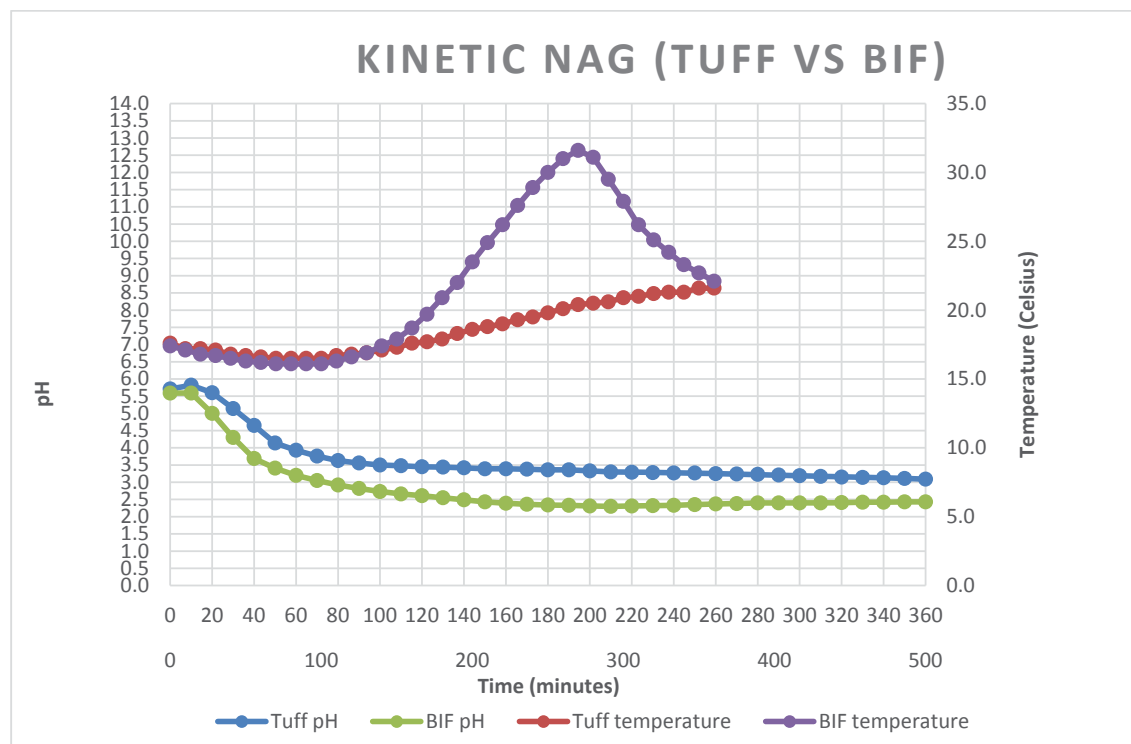


Fig. 5. Results of Kinetic NAG testing

The Tuff was identified from this initial geochemical testing as potentially having the following properties:

- Potential passivation of pyrite oxidation
- Potential retardation of metals leaching
- Potential buffering of acidic pH.

An acid base characterization (curve) ABCC test was carried out to determine the relative properties of the NAF Tuff and NAF BIF:

Table 1. ABCC testing results

Sample ID	ANC	ABCCUnits
	kg H ₂ SO ₄ /t	
Tuff (S1)	1.745	<1
BIF (TP 315606)	18.76	20.00

As can be seen from Table 1 the NAF BIF has all ANC present in an available form, however the NAF Tuff appears to have very limited buffering capacity.

An x ray diffraction (XRD) was carried out on the NAF Tuff to determine its mineralogy which returned major minerals as quartz, illite, kaolinite and muscovite. It is possible that although the material has low buffering capacity to the ABCC test, the clay minerals may offer passivation of AMD reaction and sorption of metals as a result of cation exchange.

To determine the effectiveness of the NAF Tuff and the NAF BIF as co disposal agents a leaching test was carried out which included the following rationale:

- Experiment A: A sample (200g) of PAF material from site with high soluble sulfate minerals was subject to a 24hr 1:1 leach using water (sample 315609). This produced an acidic leachate (200ml) with dissolved metals that was considered to be reflective of leachate that may be generated within the waste rock dump (WRD). The pH of this leachate was 3.3. This pH was constant with the Kinetic NAG test of the pyritic tuff which has an endpoint of pH 3.1 (Figure 5).
- Experiment B: Three samples of PAF material of the pyritic BIF were chosen and subjected to a NAG test. The liquor (200ml) from the NAG test was retained as an example of the leachate that may be generated from the WRD. The pH of these leaches were consistent with the endpoint of the Kinetic NAG test of the pyritic BIF which was pH 2.4 (Figure 5), the results for the NAG tests are shown below:
 - 106223: 2.29
 - 106224: 2.31
 - 315611: 2.71

Results from the experiments below are shown in Figures 6 and 7

- Experiment C: 100ml of the water-leached PAF sample was then mixed with 100g of the NAF Tuff (material was not pulverized, sand to gravel size material used) and left for 24 hrs

- Experiment D: 100ml of the water leached PAF sample was then mixed with 100g of the NAF BIF (material was not pulverized, sand to gravel size material used) and left for 24 hrs
- Experiment E: 100ml of the NAG liquor from PAF sample 106223 was then mixed with 100g of the NAF Tuff (material was not pulverized, sand to gravel size material used) and left for 24 hrs
- Experiment F: 100ml of the NAG liquor from PAF sample 106223 was then mixed with 100g of the NAF BIF (material was not pulverized, sand to gravel size material used) and left for 24 hrs
- Experiment G: 100ml of the NAG liquor from PAF sample 106224 was then mixed with 100g of the NAF BIF (material was not pulverized, sand to gravel size material used) and left for 24 hrs
- Experiment H: 100ml of the NAG liquor from PAF sample 106224 was then mixed with 100g of the NAF Tuff (material was not pulverized, sand to gravel size material used) and left for 24 hrs
- Experiment I: 100ml of the NAG liquor from PAF sample 315611 was then mixed with 100g of the NAF BIF (material was not pulverized, sand to gravel size material used) and left for 24 hrs
- Experiment J: 100ml of the NAG liquor from PAF sample 315611 was then mixed with 100g of the NAF Tuff (material was not pulverized, sand to gravel size material used) and left for 24 hrs

The results from the experiments are shown as follows:

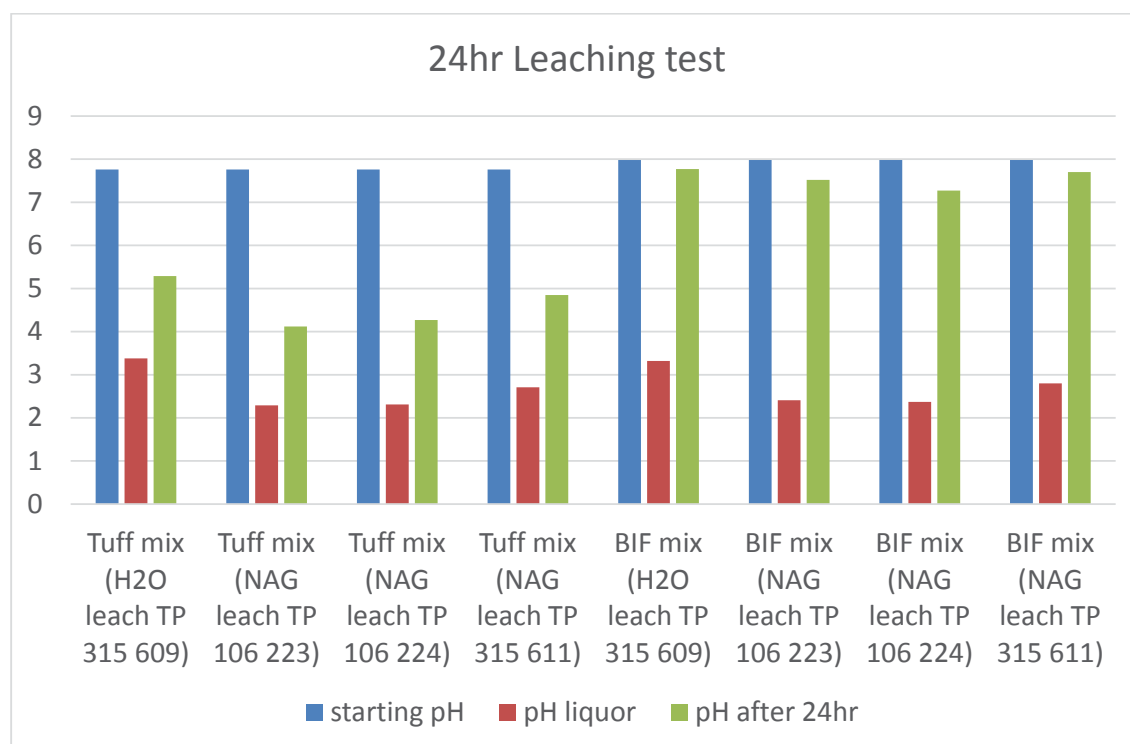


Fig. 6. Results of leach testing of co disposal materials (pH)

Figure 6 shows:

- The NAF Tuff material buffers the acidic pH of all the leachates with the pH raised to between 4-5 from 2-3 after 24 hours. This is still slightly acidic however it is within the range of iron and aluminum precipitation and therefore if replicated on site is likely to reduce AMD impacts.
- The NAF BIF provides a higher degree of buffering than the NAF Tuff, with all results being above pH 7, this would be expected from the ABCC testing which indicated that the material contains readily available buffering capacity.

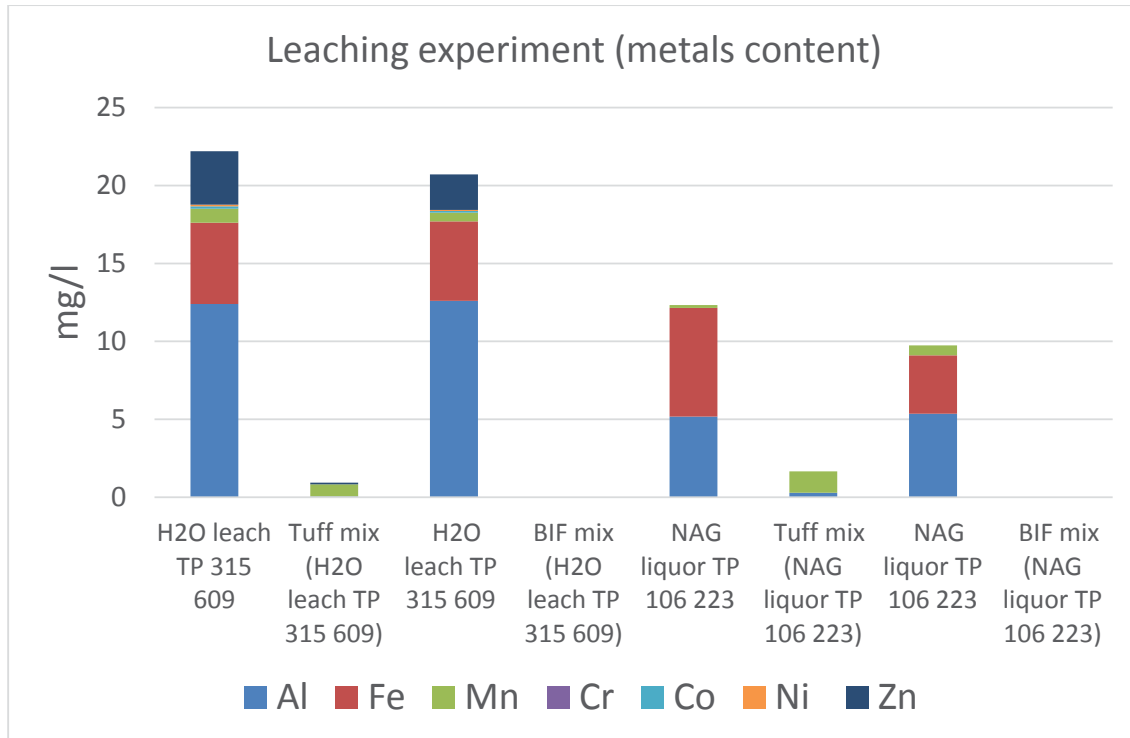


Fig. 7. Results of leach testing of co disposal materials (dissolved metals)

Figure 7 shows the results of the analysis of the liquors from the experiments. The figure shows:

- Both the NAF Tuff and the NAF BIF are very effective in removing dissolved metals from the leachate solution. Aluminum iron and zinc were found in the initial liquor from 315609 at a combined concentration over 20mg/l, these elements were reduced to less than 1mg/l combined after the 24hr leaching test.
- Manganese appears to not be removed from solution by the NAF Tuff, but it is by the NAF BIF. It is likely that this relates to the higher pH achieved by the NAF BIF due to higher buffering capacity.

The results from Figure 6 and 7 are perhaps surprising given that the NAF Tuff from testing had negligible buffering capacity and yet was very effective in both raising the pH of the solution and removing metals from solution. It is likely that cation exchange in the clay

minerals is responsible for the buffering and metal removal seen in the experiments. This property along with the fact that the material can be traffic-compacted into low permeability layers makes the Tuff a very useful co-disposal agent.

On this basis it was determined that the NAF Tuff would be the more suitable co-disposal agent given:

- The NAF Tuff contains clay minerals and when traffic-compacted will form a low permeability layer. The NAF BIF is hard rock and so will not form an effective low permeability layer
- The NAF Tuff appears to perform effectively to both buffer pH and to remove metals from solution
- The NAF Tuff was found in the block model to be constrained to specific geological horizon and is visually identifiable and therefore it was more practical to segregate for use.

7.0 CONCLUSIONS

The use of PAF cells is ubiquitous in the mining industry as a means to provide a convenient management solution for PAF waste. However the concept has fundamental flaws that makes the practice questionable from a mine closure and liability perspective. Co disposal is presented as a feasible alternative that can both be compatible with operations on site and can offer a management solution that has a higher factor of safety than conventional PAF cells.

Detailed geochemical testing has been carried out to demonstrate the effectiveness of co disposal. In addition the testing demonstrates that even materials that appear from basic static tests to have limited use as co disposal agents can in fact possess advantageous geochemical properties. The NAF Tuff material has been demonstrated to possess a number of properties that make it suitable for use as a co disposal agent. This is from a geochemical, physical, planning and operational viewpoint.

The identification of co disposal as an alternative option to the PAF cell on sites is only possible with collaboration and coordination between mine planners, engineers, site geologist, consulting geochemists and regulators, and some lateral thinking. It is only once the paradigm of the “PAF cell as the solution” has been replaced from thinking that progress can be made to improve waste management solutions. For this to happen alternatives have to be presented in a manner that makes logical and financial sense to all of these stakeholders.

8.0 REFERENCES

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OVERALL ACID PRODUCTION RATES IN MINE WASTES

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ABSTRACT

Mine waste landform design has the potential to limit rates of acid production within sulfide bearing landforms such as waste rock dumps and tailings storage facilities. Rates of acid production in mine wastes are dependent on the sulfide oxidation rates. Within a dump or tailings storage facility rates will vary in space and over time. Prediction of overall rates of acid generation therefore relies on an understanding of the oxidation rate distribution within the landform.

A number of methods are used to estimate oxidation rates, including:

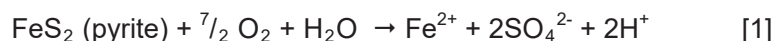
- Calculation based on sulfate release from column leach tests and humidity cells;*
- Measurement of oxygen consumption rates in laboratory scale sample containers;*
- Oxygen concentration and temperature profiles measured in waste rock dumps and tailings dams.*

Two of these provide rates applicable to laboratory conditions and if used to extrapolate to the scale of the entire waste landform, without consideration of other factors, could overestimate the overall acid generation rates. Laboratory rates can however be used to derive the distribution of oxygen throughout the landform and thereby provide an estimate of the overall acid production rate, time to depletion of sulfides and the effectiveness of waste management strategies.

This paper presents examples of measured and derived oxygen distributions within different landforms, and indicates how laboratory-derived measurements could be used to estimate overall rates of acid generation from such landforms.

1.0 INTRODUCTION

Mine waste landforms are complex. Amongst other things, the landforms vary in scale and geometry, and the materials contained within the landforms may show a range of chemical and physical characteristics. Landforms that contain sulfidic materials are known to represent potential sources of acid production. For example, pyrite in the landform could oxidise as follows:



If the landform contains neutralising potential, then some or all acidity produced may be neutralised. However, for such cases, pH neutral, saline (sulfate-rich) drainage may occur and could still prove detrimental to local water quality.

The rate at which acid (and sulfate) is produced in the landform is proportional to the rate at which sulfidic materials oxidise. Within the landform, oxidation rates will vary in space and time, principally in response to changes in oxygen distribution and supply rates. At a

particular time, the overall oxidation rate within a landform is the result of multiple individual rates that apply at different internal locations.

To estimate the overall performance of a waste landform it is therefore necessary to:

- (i) estimate possible oxidation rates for key materials (ideally as a function of key parameters such as oxygen content, temperature and moisture content);
- (ii) develop an understanding of how oxygen will be distributed within the landform, as a function of time.

A key challenge facing mine operators is to design and manage waste landforms so that oxidation rates are controlled and minimised. This paper, after a brief review of methods available for the estimation of oxidation rates, focuses on how estimates of oxidation rates can be combined with models of oxygen supply and gas transport to build a picture of overall oxidation rates within the landform. For different landform designs, these overall rates have been used to calculate long-term performance.

2.0 MEASURING OXIDATION RATES

A range of laboratory-scale methods are available for measuring oxidation rates. The most common of these are humidity cell tests (Price, 2009) and free-draining column tests (AMIRA, 2002). In summary, these methods are based on placing a crushed sample in a column or cell and maintaining the sample in a moist, aerated condition (i.e. optimal for oxidation). This oxidation rate is referred to here as the intrinsic oxidation rate.

The column/cell is rinsed regularly with de-ionised water and oxidation rates are calculated on a basis of dissolved sulfate in leachates. An alternative method involves measurement of oxygen consumption rates (Nicholson et al, 1997; Bennett et al, 2005; Bourgot et al, 2011). Crushed samples are placed in a sealed container and the oxygen concentration in gas is monitored. Oxidation rates are calculated on the basis of rates at which oxygen concentrations decrease.

Laboratory-derived estimates pertain to a specific set of conditions, e.g. relatively fine particle size and a small range of moisture contents and temperatures. Importantly, laboratory conditions are controlled so that oxygen supply is not a limiting factor.

Field-scale testing includes:

- Operation of large-scale columns, cribs and test piles (e.g. Bailey et al. 2012; Peterson et al. 2012) – to generate data for coarser-grained, run-of-mine materials. Such tests can be artificially irrigated or left open to local rainfall. Drainage is collected and analysed and the sulfate content used to estimate oxidation rates. These methods can be conducted prior to construction of the heap.
- Measurement of oxygen concentration and temperature profiles - achieved by installing monitoring equipment in the landform, either by incorporating equipment during construction, or by placing in drilled holes. Data is collected as a function of depth, and over time. The shapes of profiles can be interpreted to estimate the distribution of oxidising material and rates of local oxidation (Bennett et al, 1994).
- Measurement of oxygen flux across surfaces (Nicholson, 1997) – achieved by sealing a container to an external surface (more readily achieved for fine-grained materials such as tailings). Oxygen concentrations within the container are monitored so that an oxygen consumption rate can be determined. This method

provides a measure of oxygen consumption integrated over depth and must be combined with an estimate of oxygen penetration depth to provide oxidation rates per unit volume.

Estimates of oxidation rate form the basis of predictions of long term waste landform performance. A commonly adopted approach is to measure oxidation rates in the laboratory, and then extrapolate the values to the field system by adopting 'scaling' factors to account for expected differences between laboratory and field conditions. In the current paper an approach is described where oxidation rate estimates are combined with representation of oxygen transport in the landform.

3.0 OXYGEN TRANSPORT

There are three significant oxygen transport mechanisms in waste landforms. One of these, diffusion, is driven by spatial gradients in the oxygen concentration, whereas the other two, advection and convection, are driven by gradients in the total gas pressure. Convection results from pressure gradients produced by differences in gas temperature. Pressure gradients driving advection may be the result of external factors such as wind (Ann and Pantelis, 1997).

Figure 3-1 shows oxygen concentrations measured in a waste rock dump 20 years after dump construction (Timms and Bennett, 2002). Oxygen is supplied into a batter by air convection. To the right of the batter oxygen transport is by oxygen diffusion from the top surface.

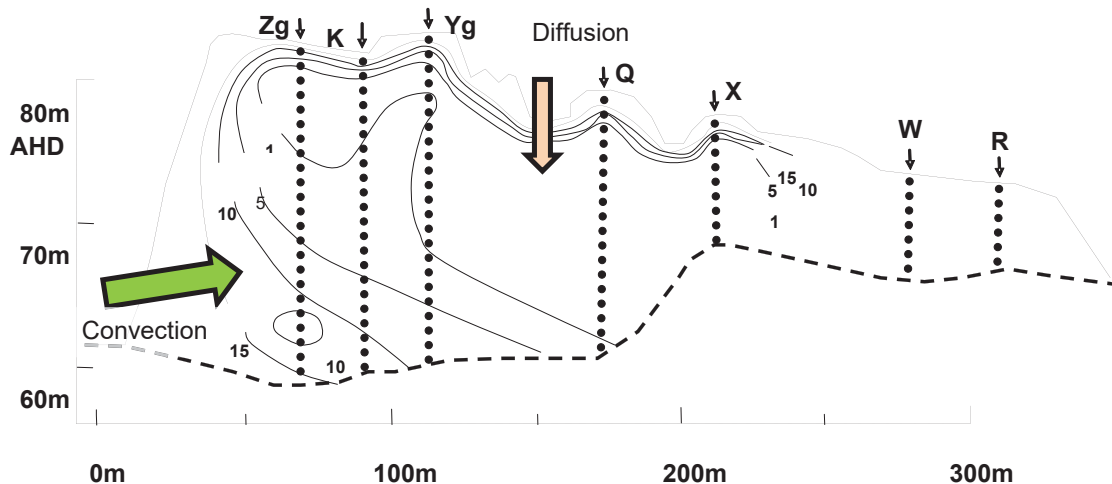


Fig. 3.1 Oxygen concentrations in Intermediate Dump at Rum Jungle mine

Notes: Timms and Bennett, 2002
Line are contours of oxygen concentration (v%).

The left column of Figure 3-2 presents schematic representations sulfide sulfur and oxygen concentration distributions for a range of landform designs and material types. In the right

column, for cases (a) to (e), examples of measured one or two dimensional oxygen concentration distributions are presented. Note that oxygen concentration distributions evolve with time as sulfides are consumed, and temperatures and boundary conditions change.

In case (a) sulfide sulfur and oxygen are distributed throughout the dump supporting oxidation in materials of varying oxidation rates. Such distributions were observed in the North Arm of the West Lyell dumps at Mt. Lyell copper mine (Garvie, et al. 1997).

Convective oxygen supply to the batter and diffusive oxygen supply from the top surface is shown in case (b). This is similar to the oxygen concentration distribution measured in Intermediate Dump. Diffusion is the only oxygen transport mechanism in case (c), a condition expected in waste rock dumps with batters of low permeability to gas. It is also the condition expected for fine-grained, low intrinsic permeability materials such as tailings. The profile shown was measured in tailings.

Clay covers of low intrinsic permeability and oxygen diffusion coefficient can limit oxygen diffusion and slow the rate of advance of the oxidation front (case (d)). These conditions were established in Storage 2 at the Martha Mine and have been maintained for more than 17 years (Garvie, et al., 2012).

Case (e) illustrates selective emplacement of sulfide bearing waste in a location away from the batter and top surface of a dump and under non-sulfide bearing waste. The measured profile is for dump with 15 m of non-acid forming (NAF) waste placed over 30 m of potentially acid forming waste rock (PAF). The addition of low diffusion coefficient cover between sulfide bearing and non-sulfide bearing waste is presented in Case (f).

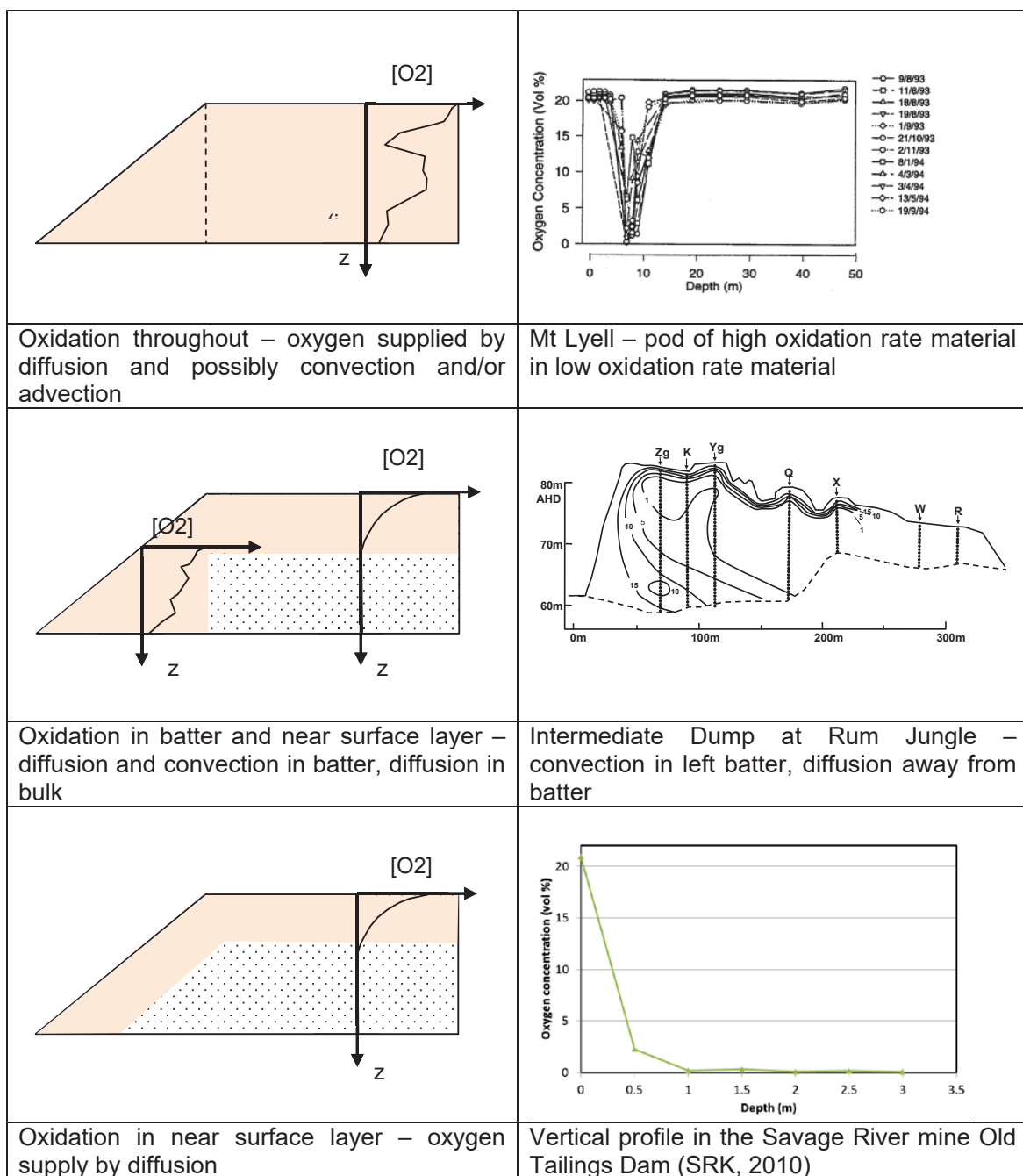


Fig. 3.1 Sulfide sulfur and oxygen concentration distributions in waste landforms

Note: In the left column pink colouring indicates regions where oxidation is occurring; dots indicate the presence of sulfides that are not oxidising, white represents the absence of sulfides and low diffusion coefficient covers are coloured brown.

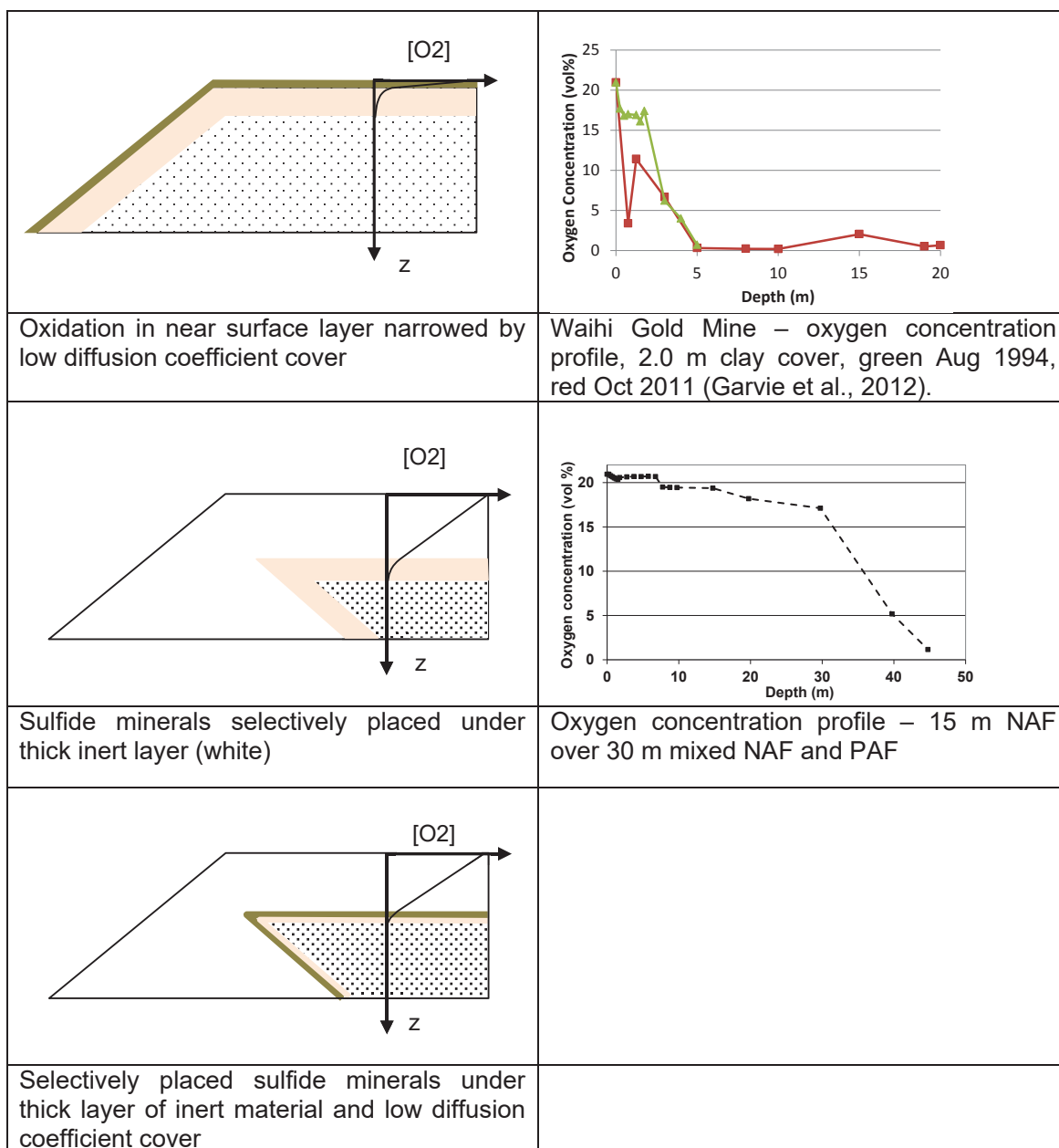


Fig. 3.3 Continued: Sulfide sulfur and oxygen concentration distributions in waste landforms

4.0 MODELLING - A DESIGN TOOL

4.1 Mathematical Representation of Gas Transport

Gas transport processes are described by mathematical expressions for mass and momentum conservation (Equation 1 and Equation 2). A detailed description of the mathematical representation of the processes is provided by Pantelis and Ritchie, 1991. Table 4-1 defines notation used in the equations, and gives typical parameter values.

Advection, convection and oxygen diffusion are represented in the equations. Equation [1] accounts for multiple gas species, as indicated by the index i . One of these species is oxygen. The term $\rho\omega_i$ represents the concentration of the i th component of the gas.

When the i th component is oxygen the terms of Eqn. [1] describe the following:

- (i) Total rate of change of gas density (concentration) in an element of the dump
- (ii) Convective and advective oxygen flux (1st term in brackets)
- (iii) Diffusive oxygen flux (2nd term in brackets)
- (iv) Rate of oxygen consumption per unit volume of dump

$$\underbrace{\partial\rho/\partial t}_{(i)} + \underbrace{\nabla\cdot(\rho\omega_i\mathbf{v})}_{(ii)} - \underbrace{D_i\nabla(\rho\omega_i)}_{(iii)} = \underbrace{S}_{(iv)} \quad [1]$$

$$\varepsilon_g\mathbf{v} = \frac{-Kk_g(\varepsilon_g)}{\mu}(\nabla P + \rho g\nabla z) \quad [2]$$

$$\tau = \frac{\xi\rho_s}{S} \quad [3]$$

The convective and advective flux are dependent on the mass averaged velocity of the gas \mathbf{v} , which is dependent on the intrinsic permeability K and gas pressure gradient, ∇P (Equation 2). The oxygen diffusion flux is dependent on the oxygen diffusion coefficient, D_{O_2} . Both parameters, K and D_{O_2} , are characteristics of the landform materials. Values of these parameters can be modified by waste and cover material compaction and changes to the degree of saturation. Higher degrees of saturation lead to lower oxygen diffusion coefficients.

Equation 3 defines the characteristic time, i.e. the time required for all sulfides in an oxygen supplied volume to oxidise.

Various research groups have developed methods of solving the Equation 1 and Equation 2 under differing simplifying assumptions (e.g. Pantelis and Ritchie, 1991, Lefebvre, 2001). The constant intrinsic oxidation rate model (Ritchie, 1995) is an analytic model that accounts for diffusion but not convection or advection and has the advantage of relative simplicity.

Table 4.1 Definition of notation and typical parameter values

Symbol	Physical property	Units	Typical values
ρ	Intrinsic density of gas in pore space	kg/m ³	
ρ_s	Sulfide sulfur density	kg/m ³	0.05 to 3
ω_i	Mass fraction of ith component in the gas phase		
D_i	Diffusion coefficient of the component of the gas phase	m ² s ⁻¹	4×10 ⁻⁶ in waste rock
S	Maximum value of intrinsic oxidation rate	kg(O ₂)/m ³ s	5 × 10 ⁻¹⁰ to 5 × 10 ⁻⁶
ε_g	Gas filled porosity		0.18
v	Mass averaged velocity of gas phase	ms ⁻¹	
K	Intrinsic permeability of the waste	m ²	
$k_g(\varepsilon_g)$	Relative coefficient of permeability		0 to 1
P	Pressure of the gas phase	Pa	
z	Vertical direction coordinate	m	
μ	Absolute viscosity of gas phase	Pa s	
ξ	Stoichiometric ratio of oxygen to sulfide sulfur		1.75 for pyrite

4.2 Illustrative Calculations of Waste Landform Performance

A constant intrinsic oxidation rate model was used to calculate the accumulated sulfate produced over time within a landform. Oxidation was modelled in the batter by assuming that convection was established at the time of completion of the dump, occurred throughout the batter and continued until all sulfides were consumed. Away from the batter oxidation was modelled using a constant intrinsic oxidation rate model.

The model was used to calculate the sulfate production for cases (a) through to (f) of Figure 3-2, over a period of 150 years following landform construction. The calculated results are for a cross-section 1 m thick (e.g. in section A to A' in Figure 4-1).

So that the results could be directly comparable each case was based on:

- management of the same overall volume of sulfidic material. For some cases the material was evenly distributed throughout the landform (Cases (a) to (d)); in others, the material was selectively emplaced (Cases (e), (f));
- a single, and constant, value for the oxidation rate when the sulfidic material was supplied by oxygen. For cases (a) to (d), because the sulfidic material was distributed throughout the landform, a 'dilution factor' was applied to the oxidation rate. The use of a constant oxidation rate is a simplification as rates are known to be dependent on, amongst other things, temperature, oxygen and sulfide concentration.

The conditions assumed in the modelling are summarised in Table 4-2. Two scenarios were modelled for case (d) using two different oxygen diffusion coefficients. One scenario was modelled for all other cases. The two diffusion coefficients were 1/10th and 1/100th of the oxygen diffusion coefficient expected for as mined waste rock ($D \sim 4 \times 10^{-6} \text{ m}^2\text{s}^{-1}$).

Generally the conditions in a landform will differ from those indicated in the table. The objective is to indicate the types of changes that could be expected due to choice of landform design.

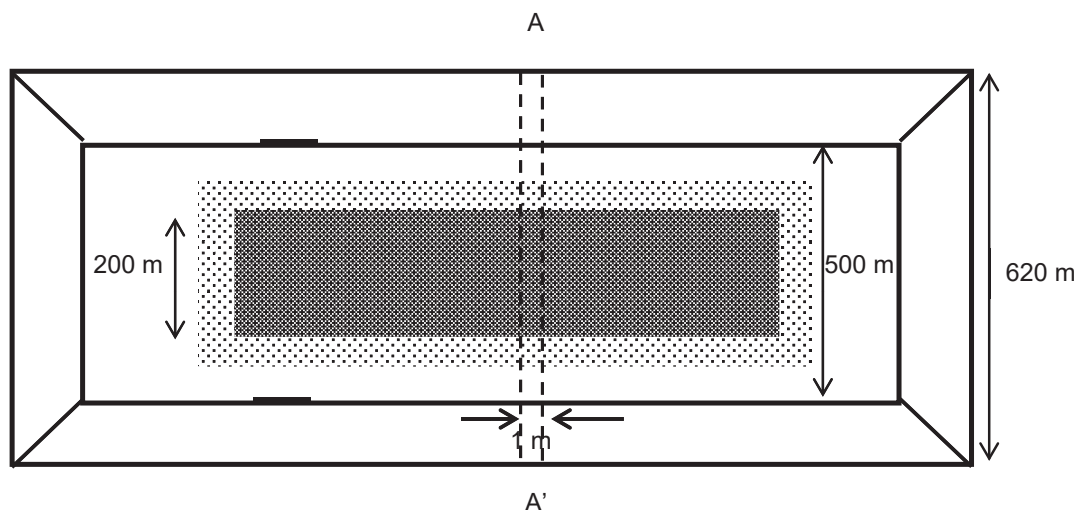


Fig. 4.1 Plan view of modelled dump

Note: Shade area is 0.5 wt% sulfide sulfur corresponding to cases E and F.

Table 4.2 Value of parameter values used in the model

Parameter	Description	Units	Value	Applicable to case
H	Overall height	m	45	All
x_i	Thickness of inert layer	m	30	e, f
x_c	Cover thickness	m	1	d, f
θ	Batter angle	degrees	37	All
ρ_s	Sulfide sulfur density – diluted	kg/m ³	1.04	a to d
	– undiluted		8	
S	Oxidation rate – diluted	kg(O ₂)m ³ s ⁻¹	1.3×10^{-9}	a to d
	– undiluted		10^{-8}	e, f
D_{O_2}	Diffusion coefficient of gas phase oxygen in the wast	m ² s ⁻¹	4×10^{-6}	all
D_c	Diffusion coefficient of a cover	m ² s ⁻¹	4×10^{-7}	d1
			4×10^{-8}	d2, f

5.0 RESULTS AND DISCUSSION

5.1 Outcomes of Modelling

The accumulated mass of sulfate produced over time is presented in Figure 5-1 for a 1 m thick cross-section of the landform. The rates of sulfate production are provided by the slope of the curves.

Elimination of convective oxygen supply in the landform reduces the rate of sulfate production and accumulation at early times. This is seen by comparison of cases (a), (b) and (c). Convection contributes to sulfate accumulation only during early times because the characteristic time is 44 years.

A low diffusion coefficient placed on the top surface of the dump reduces the sulfate accumulation rate (compare cases (c), (d1) and (d2)). For example, at 44 years the total quantity of sulfate produced is reduced to about 82% and 22% for the covers with oxygen diffusion coefficient of 4×10^{-7} and $4 \times 10^{-8} \text{ m}^2 \text{ s}^{-1}$ respectively. Thus, there is a disproportionate reduction in sulfate production rate for lower oxygen diffusion coefficient for the dump geometry and intrinsic oxidation rate modelled.

The 1 m thick cover with the lower diffusion coefficient results in a sulfate accumulation rate lower than that for a 30 m thick layer of inert waste covering the selectively placed sulfide bearing waste (case (e)).

The selectively placed waste that is also covered by a lower diffusion coefficient layer has the lowest sulfate generation rate. At 44 years the accumulated sulfate is about 16% of that generated for the uncovered diffusively supplied waste.

For cases (a) to (d1) all sulfides are oxidised in about 88 years.

These results indicate in changes in sulfate generation rates as a result of changing the landform design. Although the results are for a specific landform of specified characteristics similar responses to changes in landform design would be expected for other waste materials and landforms of different geometries. The rates of accumulation of sulfate would differ from landform to landform depending on, for example, the oxidation rate of the waste when oxygenated, the oxygen diffusion coefficients and intrinsic permeabilities.

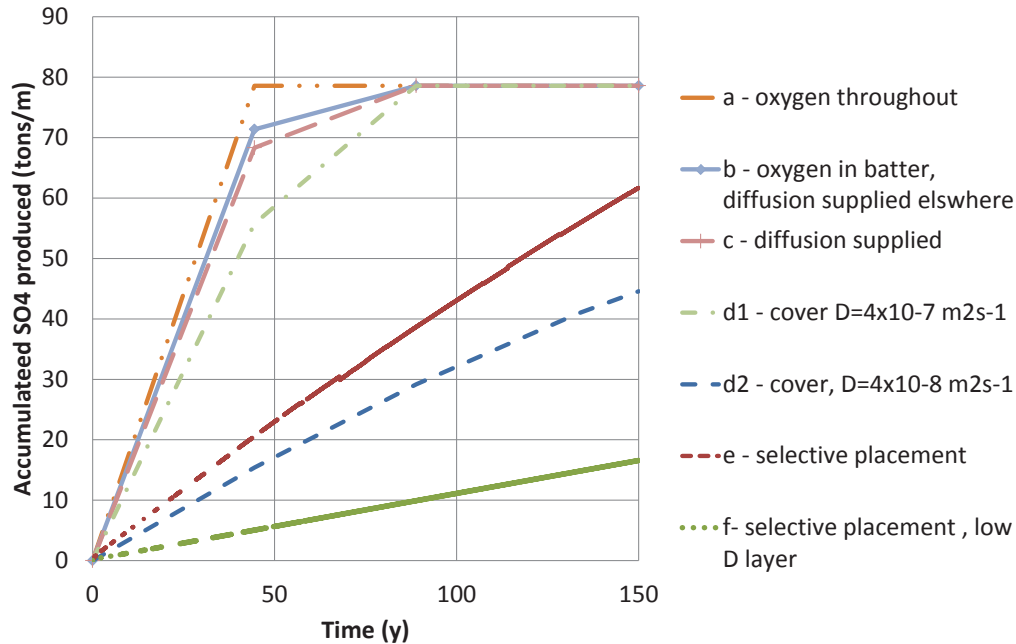


Fig. 5.1 Sulfate accumulation versus time for landforms similar to those of figure 3.2.

5.2 Implications for Construction

Cost effective landform designs that meet performance criteria should be selected based on:

- A good understanding of the chemical and physical characteristics of the materials to be managed, and
- Due consideration of local weather conditions and available construction materials. For example, in drier climates surface clay covers intended to maintain high degrees of saturation to achieve low oxygen diffusion coefficients may not be sustainable in the long-term.

The timing of implementation of management strategies could have a significant impact on the evolution of sulfate in the landform and potentially on effluent quality. For example, preventing convection and applying a low diffusion coefficient cover at early times could reduce the quantity of sulfates available at a given time to dissolve and appear as a load in drainage waters.

Adequate quality assurance and control must be implemented during construction to ensure that the as-built facility characteristics conform to design specifications (e.g. diffusion coefficients and permeabilities should be within a defined range).

5.3 Evidence of Landform Performance

Mine closure planning generally includes measures to monitor the performance of waste landforms. Such measures are often a regulatory requirement, and evidence that landforms are performing satisfactorily will be necessary prior to mine relinquishment.

Performance measures can be based on monitoring of seepage water quality. However, it should be remembered acid and sulfate produced near the top surface of a facility must be transported to the base before it will impact seepage quality. Depending on the scale of the facility, water residence times may be long. Thus lag times before base seepage chemistry reflects internal acid production may be several years, even decades.

Installation of oxygen and temperature monitoring equipment would facilitate more timely collection of data on landform performance, possibly enabling earlier compliance with regulatory obligations and increasing the likelihood that poorly performing designs can be identified early.

6.0 CONCLUSIONS

Combining the measurement of the oxidation rates of sulfide bearing wastes with the modelling of oxygen supply mechanisms provides a means of identifying landform designs that would reduce acid and sulfate production rates taking account of site specific conditions.

The reduction in production rates is a function of both the thickness and the oxygen diffusion coefficient of material placed over the sulfide bearing waste. For the geometry and intrinsic oxidation rate considered:

- A 30 m thick inert layer of waste rock reduced the sulfate generation rate more than a 1 m thick cover with an oxygen diffusion coefficient $1/10^{\text{th}}$ of that expected for waste rock but less than a 1 m thick cover with an oxygen diffusion coefficient $1/100^{\text{th}}$ that of waste rock.
- The lowest sulfate production rate was for segregated sulfidic waste under a two layer cover. The top layer was 30 m of inert waste rock and the second layer was a 1 m thick layer with an oxygen diffusion coefficient of $1/100^{\text{th}}$ that of as mined waste rock. For dry climates this design has the advantage that the cover is deep in the landform and not susceptible to evaporation effects which would increase the diffusion coefficient.

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THE RUM JUNGLE NATIONAL PARTNERSHIP AGREEMENT

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ABSTRACT

On the 7 October 2009 the Northern Territory and Commonwealth governments entered into a four-year \$7.048 million National Partnership Agreement on the management of the former Rum Jungle mine site (NPA). The objectives of the NPA were to improve site maintenance and environmental monitoring activities and to develop an improved rehabilitation strategy for the site consistent with the views and interest of stakeholders particularly the joint traditional Aboriginal owners of the site – Kungarakana and Warai.

The outcomes of these investigations and consultation with key stakeholders have formed the basis for the development of a conceptual rehabilitation strategy for the site. This has included the assessment of long-term monitoring data and previously undertaken technical investigations on the site and downstream to identify areas of technical certainty and uncertainty. The key information gaps that were identified were used to scope and commission new investigations to inform the options being developed for rehabilitation for the site. In addition, both Aboriginal and non-Aboriginal cultural heritage attributes have been taken into account because they are fundamentally important to achieving a successful rehabilitation outcome for the site. Finally, industry-specific leading practice principles, together with leading practice guidance from relevant national and international agencies has been incorporated into the project.

A total of five potential rehabilitation options were developed for consideration. The options ranged from re-covering the existing waste rock dumps in-situ to backfilling the water filled pits and consolidating residual waste to a new waste rock dump. In early 2013 a rigorous evaluation process was undertaken using the Multiple Accounts Analysis (MAA) tool. The MAA process was structured around four key issues; environmental, technical feasibility, cultural and financial and each individual component of these issues were scored on its relative projected performance against the rehabilitation objectives.

A Conceptual Rehabilitation Plan which documents the findings of the NPA including the preferred rehabilitation strategy has now been completed. This paper describes the results of the NPA and ongoing rehabilitation planning that is part of the second phase of the NPA.
<http://www.rumjungle.nt.gov.au>

1.0 INTRODUCTION

1.1 Background on the Former Rum Jungle Mine Site

The former Rum Jungle mine site (Rum Jungle) is near Batchelor in the Northern Territory (NT), approximately 105 km, by road, south of Darwin. Between 1954 and 1971, Rum Jungle produced 3,530 tonnes of uranium oxide and 20,000 tonnes of copper concentrate (Davy, 1975). The mine site consists of:

- three waste rock dumps – Main, Intermediate and Dysons
- two water-filled mine pits – Main and Intermediate

- one mine pit backfilled with tailings and overlaid with contaminated soil – Dysons.

The mining and placement methods used for waste rock and process tailing during this time resulted in substantial volumes of acid and metalliferous drainage (AMD). Ongoing oxidation of sulfide minerals in the waste, followed by annual wet season leaching of soluble oxidation products released large concentrations of copper and other heavy metals and acid into the surrounding environment. The Commonwealth of Australia (Commonwealth) provided financial assistance to the Northern Territory Government (NTG) to implement a four-year rehabilitation program between 1982 and 1986, to address these issues, followed by a 12-year monitoring program (Northern Territory Government, 2013).

After the rehabilitation works were completed, a technical assessment determined the engineering and environmental criteria set for the rehabilitation project were successfully met. However, works did not result in a final condition for the site that would meet contemporary water quality standards. When funding for the 12-year monitoring program ceased, additional problems also arose with how to manage wildfire, weeds, feral animals and access (Kraatz, 2004).

The Finnis River Land Claim No.39 was lodged by the Northern Land Council on behalf of claimants on 20 July 1979, under section 50(1)(a) of the Aboriginal Land Rights (Northern Territory) Act 1976 (ALRA). Rum Jungle formed part of the areas subject to the claim. An inquiry into the claim was conducted by the Aboriginal Land Commissioner, who recommended that the majority of land subject to the claim, including Rum Jungle, be granted to Aboriginal Land Trusts established under ALRA. Kungarakana and Warai people were found to be traditional Aboriginal owners of Rum Jungle and other areas subject to claim.

The majority of the land recommended for grant was vested in two Aboriginal Land Trusts. No decision on the potential grant of the Rum Jungle site has yet been made, pending the outcome of negotiations between the Commonwealth, the Northern Land Council and Kungarakana and Warai people about the future of the site, including rehabilitation.

1.2 Site Description & Layout

Rum Jungle comprises approximately 650 hectares of relatively elevated ground, bisected by wet season ephemeral streams that feed into the East Branch of the Finnis River. The East Branch of the Finnis River joins the Finnis River about eight kilometers downstream of the mine site. The Finnis River then flows west for about 60 kilometers before emptying into Fog Bay.

The region is characterized by a tropical savannah-like climate and typically receives about 1500 mm of annual rainfall. 90% or more of rainfall occurs during a distinct wet season that lasts from November to April and no sustained rainfall occurs from May to October. Mean monthly maximum temperatures at the Batchelor Airport range from 31.2°C in June to 36.8°C in October (during the 'build up' to the wet season).

1.3 National Partnership Agreement

As part of the 2009-10 budget, the Australian Government committed \$7 million over a four-year period for the environmental management of Rum Jungle (Commonwealth Government, 2009). To manage this commitment, the NTG and the Commonwealth entered into a National Partnership Agreement on the management of the former Rum Jungle mine site (NPA). The NPA defined the overall objectives and outcomes for the program of works at

Rum Jungle. Under the terms of the NPA, the Commonwealth and NTG committed to improving the management of Rum Jungle in a way that is consistent with the interests of stakeholders, particularly the traditional Aboriginal owners by:

- improving the understanding of the current state of the environment
- improving site management
- developing an improved rehabilitation planning, which may lead to future rehabilitation works under new arrangements.

The NPA has been driven by the NT Department of Mines and Energy (DME), with technical oversight from the Rum Jungle Working Group, comprised of NT and Commonwealth Government agencies and the Northern Land Council.

2.0 REHABILITATION PLANNING

2.1 Technical Investigations

A significant number of technical investigations have been conducted under the NPA. These investigations have identified that all waste materials stored on-site contain sulfides and elevated concentrations of metals (aluminum, cadmium, cobalt, copper, magnesium, nickel, selenium, uranium, sulphate and zinc), which generate AMD. AMD affects the East Branch of the Finnis River and the Finnis River downstream of the mine; concentrations of cobalt, copper and nickel exceed ANZECC water quality guideline trigger values for the protection of aquatic ecosystems during low-flow periods in the river. Copper concentrations exceed the trigger values in the East Branch of the Finnis River under all flow conditions. A comprehensive groundwater investigation identified localized groundwater contamination on the mine site (RGC, 2012).

Investigations also identified localized contamination of soil with metals that requires cleaning up, including the operations area, the old tailings dam area, and the fluvial areas. These areas may also include radiological contamination (e.g. unclaimed tailings, boulders).

The vegetation on the mine site is composed of mostly grass species, with significant weed infestations, leading to erosion and degradation of the existing covers on the waste rock dumps.

2.2 Environmental Monitoring

A comprehensive ground and surface water monitoring program has been a key component of the NPA. The monitoring was undertaken to gain a better understanding of the water quality conditions on-site and off-site, and also to provide data for the hydrogeological investigations being conducted. The network of groundwater monitoring bores was also substantively expanded to address identified knowledge gaps.

2.3 Stakeholder Engagement

In parallel with progressing the technical investigations, DME developed and implemented a stakeholder consultation plan. The key stakeholder groups for this project are: traditional Aboriginal owners, the Rum Jungle Working Group, state and federal government organisations, affected landowners, the community of Batchelor, non-government organisations, tourists, scientific research organisations and universities, prospective contractors and consultants and the media.

2.4 Potential Rehabilitation Scenarios

Five potential rehabilitation scenarios for Rum Jungle were developed and assessed to evaluate how each scenario addressed the key rehabilitation objectives. These objectives were developed through a comprehensive process of internal analysis of technical requirements and extensive consultation with stakeholders, including the two traditional Aboriginal owner groups. In summary the rehabilitation process should create a landscape which:

- is safe for people and wildlife
- is physically, chemically, and radiologically stable
- has a significantly reduced contaminant load (associated with AMD) travelling beyond the boundaries of the site
- supports sustainable land uses by traditional Aboriginal owners of the area with few, if any, limitations
- encourages beneficial post-rehabilitation land uses.

The five scenarios evaluated ranged from re-covering the existing waste rock dumps and the Dysons backfilled pit insitu, to backfilling the two existing water-filled pits and relocating and consolidating the remaining surface material in a new waste rock dump.

The proposed alternative rehabilitation scenarios are summarized as follows:

- Scenario 1. Install new covers on the existing waste rock dumps (WRDs) and Dyson's (backfilled) Open Pit;
- Scenario 2. Backfill the Main and Intermediate Open Pits and re-cover residual waste rock from the Main WRD & waste rock from Dyson's WRD in the central mining area;
- Scenario 3. Backfill the Main and Intermediate Open Pits and re-locate residual waste rock from the Main WRD to Dyson's Area before re-covering;
- Scenario 4. Backfill the Main and Intermediate Open Pits and re-locate residual waste rock from the Main WRD & Dyson's WRD to a new WRD in the Old Tailings Dam area; and
- Scenario 5. Backfill the Main Open Pit and re-cover residual waste rock from the Main WRD & Dyson's WRD separately.

2.5 Evaluation Process

The proposed rehabilitation scenarios were evaluated using Multiple Accounts Analysis (MAA). MAA is a tool for evaluating different options or alternatives for a project, or specific components of a project, by weighing the relative benefits and costs (or losses of a variety of independent factors (Shaw, et al. 2001; RGC, 2013). MAA is particularly useful when socio-economic considerations are a major component of a rehabilitation planning exercise, as well as technical and environmental dimensions. In the case of Rum Jungle, the findings from the large number of technical investigations had to be considered, alongside diverse stakeholder perspectives.

The MAA evaluation process was undertaken over five days in February 2013. Robertson GeoConsultants Inc. led the process, with input from DME, the Rum Jungle Working Group and traditional Aboriginal owners. The evaluation process focused on four key categories or accounts:

- predicted environmental performance
- cultural considerations
- technical feasibility
- financial cost to implement the scenario.

Each account had a number of sub-accounts called 'issues'. For example, minimising the potential for AMD is an issue relating to the Predicted Environmental Performance Account. Each issue was then further broken down into specific 'indicators' that clearly described the impacts of each issue. The performance of each scenario against every indicators was scored on a scale from 1-9, with 9 assigned the best performer and the remaining items scored relatively. However, the worst performing scenario was not necessarily assigned a value of 1. Each level of the MAA (account, issues, and indicator) was assigned a weighting (from 1-9), with the highest score of 9 assigned to the aspect judged by the group to be the most important contributor to a successful rehabilitation outcome.

The accounts, issues, and indicators were initially developed by consultation between DME and Robertson GeoConsultants. The accounts, issues and indicators were further refined by the Rum Jungle Working Group and traditional Aboriginal owners as the workshop progressed through each account.

2.6 Preferred Rehabilitation Strategy

Based on the MAA, the preferred rehabilitation strategy needed to include backfilling of one or both the pits with waste currently stored at the surface and consolidating the remaining waste into a single landform located as far away as practically possible from water courses. The conceptual rehabilitation scenario that addressed these requirements best is scenario 4 which:

- backfills the Main and Intermediate pits
- relocates remaining waste rock from the main and Dysons waste rock dumps to a new facility in the old Tailings Dam Area
- the estimated capital cost of scenario 4 is \$109 million (in \$AUD 2013).

2.7 New Project Agreement – Stage 2

A new Project Agreement has now been signed between the Commonwealth and NTG in August 2013. The activities under this new agreement include detailed engineering design (including supporting investigations); scheduling arrangements (project management); preparation of detailed procurement packages; attainment of relevant environmental approvals; stakeholder engagement; and ongoing site monitoring and maintenance. These tasks are necessary to allow for costing the preferred rehabilitation strategy to a satisfactory level of accuracy to secure capital works funding, and for all regulatory and preparatory work to be completed for implementation in Stage 3.

3.0 SUMMARY

The technical investigations commissioned as part of the rehabilitation planning process have filled key knowledge gaps about the current environmental status of the Rum Jungle site. This knowledge was enhanced by assessment of both the long-term monitoring data and previous technical investigation undertaken onsite and offsite.

In addition, both Aboriginal and non-Aboriginal cultural heritage attributes and the outcomes of consultation with key stakeholders have been taken into account because they are fundamentally important to achieving a successful rehabilitation outcome for the site.

Finally industry-specific leading practice principles, together with leading practice guidance from relevant national and international agencies were incorporated into the rehabilitation planning process.

Five rehabilitation scenarios were developed for evaluation. A rigorous evaluation was undertaken in February 2013 with input from the Rum Jungle Working Group and traditional Aboriginal owners. The evaluation process, known as Multiple Accounts Analysis, focused on four key categories: environmental, technical, cultural and financial.

The preferred strategy addresses the issue of AMD by, where possible, returning waste material back to the former mine pits and consolidating the remaining waste into a single location. Traditional Aboriginal owners have identified a strong preference for this strategy, as they have identified that they would like the site to return, where possible to its pre-mining topography and allow sustainable traditional land uses.

Completion of the current Project Agreement will deliver a rigorously designed and fully costed rehabilitation project ready for implementation.

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REHABILITATION PLANNING AT THE FORMER RUM JUNGLE MINE SITE, NORTHERN AUSTRALIA

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ABSTRACT

Acid and metalliferous drainage (AMD) at the former Rum Jungle mine site in northern Australia has led to significant impacts on local groundwater and on the aquatic environment of the East Branch of the Finniss River. Although rehabilitation was undertaken in the 1980s this was not to a level that meets current best practice for mine closure. In addition the works were done without input from Traditional Owners of the area.

The Northern Territory and Commonwealth governments have entered into a National Partnership Agreement (NPA) on the management of Rum Jungle. The NPA objectives are to improve site maintenance and environmental monitoring activities and to develop an improved rehabilitation strategy for the site that is beneficial to stakeholders and consistent with the views and interests of the traditional Aboriginal Owners of the site.

Five alternative rehabilitation scenarios were proposed during the initial phase of the NPA and ranged from re-covering the waste rock dumps to backfilling the flooded open pits with waste rock and consolidating residual waste rock in a new facility. The scenarios were comparatively evaluated using Multiple Accounts Analysis (MAA), a systematic and transparent decision analysis tool that involves evaluating each scenario on predicted environmental performance, cultural considerations, technical feasibility, and financial cost.

The value-based decision component of the MAA process was conducted in February 2013 and resulted in the selection of a preferred rehabilitation scenario for Rum Jungle. This involves submerging the majority of waste rock in the flooded open pits and then consolidating the residual waste rock in a new facility. This scenario best addresses the various environmental issues at Rum Jungle and is consistent with land-use aspirations of the local Traditional Owners. This paper describes the MAA process and key findings that have informed further rehabilitation planning at the site.

1.0 INTRODUCTION

1.1 Background on Rum Jungle

The former Rum Jungle mine site is located 105 km by road south of Darwin in the headwaters of the East Branch of the Finniss River near the township of Batchelor. Rum Jungle was one of Australia's first major uranium mines and produced approximately 3,500 tonnes of uranium and 20,000 tonnes of copper concentrate between 1954 and 1971 (Davy, 1975).

After mining ceased, acid and metalliferous drainage (AMD) impacted local groundwater and caused substantial impacts on the aquatic environment of the East Branch of the Finniss River. The serious impacts on the Finniss River persisted for decades until rehabilitation works were undertaken in the mid-1980s, resulting in less than half the annual load of copper

being delivered to the East Branch of the Finnis River. The work involved re-shaping and covering the waste rock dumps (WRDs), treating highly-contaminated water that had filled the two largest pits, and then backfilling a third pit with tailings and contaminated soils collected from the former heap leach area and Old Tailings Dam footprint (Allen and Verhoeven, 1986).

Despite rehabilitation, groundwater at Rum Jungle and the East Branch of the Finnis River downstream remain adversely affected by AMD. Consequently, the Commonwealth and Northern Territory (NT) Governments entered into a four-year National Partnership Agreement (NPA) on the management of Rum Jungle in 2009. As part of the NPA, the Mining Performance Division of the NT Government's Department of Resources (now the Department of Mines & Energy, DME) was tasked with characterizing current conditions of the site and developing a revised rehabilitation strategy that would both meet the current more stringent environmental standards and be sensitive to cultural issues that specifically pertain to the Rum Jungle site.

1.2 Site Description & Layout

The region near Rum Jungle is characterized by a wet-dry tropical monsoonal climate and typically receives about 1,500 mm of annual rainfall. Annual evaporation is about 2500 mm. 90% or more of rainfall occurs during a distinct wet season that lasts from November to April, with no sustained rainfall occurring from May to October. Mean monthly maximum temperatures at the Batchelor Airport range from 31°C in June to 37°C in October (during the 'build up' to the wet season).

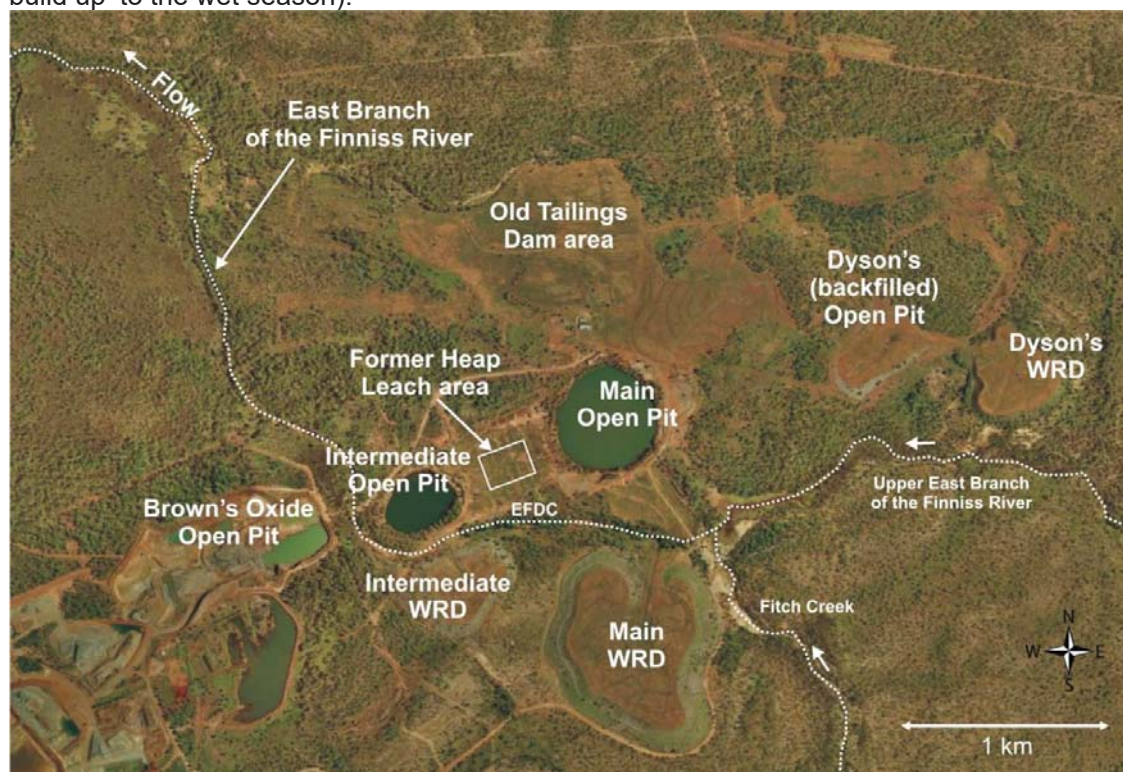


Fig. 1. Layout of the Rum Jungle Mine Site

Rum Jungle is located along the East Branch of the Finnis River about 8.5 km upstream of its confluence with the West Branch of the Finnis River (Figure 1). Unimpacted surface water enters the mine site from the east via the upper East Branch of the Finnis River and from the southeast via Fitch Creek. Before mining, these creeks met in the central mine area and subsequently flowed eastward via the natural river course. During mining, the river was diverted to the East Finnis Diversion Channel (EFDC) to allow access to the Main and Intermediate ore bodies (see Figure 1).

Since initial rehabilitation in the 1980s, flows from the upper East Branch of the Finnis River and Fitch Creek have flowed directly into the EFDC and the Main Open Pit near the former Acid Dam. Water then flows from the Main Open Pit to the Intermediate Open Pit via a channel that roughly follows the pre-mining river course. Outflow from the Intermediate Open Pit to the EFDC occurs near the western boundary of the mine site and combined flows from the flooded open pits and EFDC continues northward via the natural river course. Flows in the East Branch of the Finnis River are currently monitored downstream of the site at multiple locations.

A brief description of the main features of the site and some pertinent aspects of historic mining operations and initial rehabilitation works completed in the 1980s is provided in the sub-sections below.

1.2.1 Main and Intermediate Open Pits

The Main, Intermediate, and Dyson's ore bodies were the principal ore deposits at Rum Jungle. Each of these ore bodies was mined out by the early 1970s and the pits had each flooded with contaminated groundwater and seepage by the time mining operations ceased (Davy, 1975). The Browns Oxide open pit was constructed in recent times to access another orebody lying to the west of the original Rum Jungle mine lease area. It is of potential relevance to the groundwater flow paths around the Intermediate Open Pit as it lies along the strike of the fault that runs through the Intermediate and Main Pits. The Browns pit is currently de-watered by the tenement holders of the mine lease adjacent to Rum Jungle.

The Main Open Pit was mined to about 111 m below ground surface (bgs) from 1954 to 1958 and became flooded with highly-contaminated groundwater after mining operations ceased (Davey, 1975). The majority of the pit profile currently has relatively good water quality as a combined result of extensive lime treatment during rehabilitation in 1985 and subsequent annual wet season flushing flows from the East Branch of the Finnis River since that time. However, a thin layer of highly-contaminated water still remains at the bottom of the pit (see RGC, 2012).

The Intermediate Open Pit was mined to about 57 m bgs in the 1960s and was also flooded with highly-contaminated groundwater and seepage from the Copper Extraction Pad area. Pit water from the Intermediate Open Pit was not treated to the same extent as water from the Main Open Pit but water quality conditions have substantially improved through time by annual inflows of clean water from the East Branch of the Finnis River (via the Main Open Pit). In common with the Main Pit some highly-contaminated water remains at the bottom of the Intermediate Open Pit (see RGC, 2012).

1.2.2 Dyson's (backfilled) Open Pit

Dyson's Open Pit was mined to a depth of about 46 m bgs in the late 1950s and was partially backfilled with tailings during mining operations. This pit was later backfilled entirely with

additional tailings removed from the Old Tailings Dam and leached low-grade ore and contaminated soils removed from the Copper Extraction Pad area in 1984/1985. The backfilled open pit was also covered at this time to reduce rainfall infiltration and further oxidation of backfill materials (Allen and Verhoeven, 1986).

1.2.3 Waste Rock Dumps & Dyson's Backfilled Pit

Waste rock removed while mining the Main, Intermediate, and Dyson's ore bodies was stored in three waste rock dumps (WRDs) (see Figure 1). These WRDs were re-shaped and covered in 1984/1985 as part of the rehabilitation works at the site (Allen and Verhoeven, 1986). The WRDs (and shallow backfill materials in Dyson's Open Pit) represent the main sources of contaminants to groundwater at Rum Jungle and the East Branch of the Finnis River (see Davy, 1975; Kraatz, 2004; RGC, 2012).

Waste rock from the Intermediate WRD and shallow backfill in Dyson's Open Pit have the highest net acid-generating potential (NAGP) and contribute disproportionately large loads of Cu and other metals to groundwater. These high-priority materials consist of waste rock (Intermediate WRD) and contaminated soils and spent heap leach material removed from the Copper Extraction Pad area during initial rehabilitation in the 1980s (Dyson's Open Pit) that would likely be placed first into the Main Open Pit during rehabilitation (see RGC, 2013). Waste rock from the Main WRD is considered moderate-to-low priority material, whilst waste rock from Dyson's WRD is the lowest priority material at Rum Jungle (see RGC, 2013).

1.2.4 Copper Heap Leach Area

Copper from sub-grade ore and the oxidized capping from the Intermediate ore body was extracted via heap leaching on a pad located between the Main and Intermediate Open Pits. The heap leach procedure involved piling sulphide and oxide ore into separate piles atop a low-permeability pad and then spraying the ores with a highly-acidic mixture of raffinate, barren liquor, and water from Main Open Pit to dissolve soluble copper in the ore pile (Andersen, 1966).

Pregnant liquor was intended to drain from the heap primarily via a series of lined channels and culverts but the process was inefficient and seepage losses were substantial (Davy, 1975). These losses led to significant impacts on groundwater between the Main and Intermediate Open Pits and this impacted groundwater ultimately filled the pits when de-watering ceased in the 1960s.

Spent heap leach material and soils that had been contaminated during the leaching process were removed during rehabilitation in the 1980s and used to backfill shallow portions of Dyson's Open Pit. Some heap leach material and contaminated soils remain in the Copper Extraction Pad area and deep groundwater in this area is characterized by very high concentrations of dissolved copper (see RGC, 2012).

1.2.5 Old Tailings Dam Area

Slurried tailings were discharged during mining operations to a relatively flat area to the north of the Main Open Pit. Drainage from this area entered a small creek that eventually flows into the East Branch of the Finnis River (Watson, 1979). Perimeter walls were later built towards the eastern end of the creek to form a series of small dams commonly referred to as the 'Old Tailings Dam' (Davy, 1975).

Most of the tailings were later removed from this area and placed in Dyson's Open Pit during site rehabilitation in the 1980s. The area was then limed and re-shaped to control drainage (Allen and Verhoeven, 1986). A one-layer soil cover was also installed to enable the establishment of vegetation. According to Fawcett (2007), some residual tailings remain in this area and would require removal during implementation of a revised rehabilitation plan.

2.0 CURRENT REHABILITATION PLANNING

2.1 MAA Framework & Development

In the context of mine closure, a Multiple Accounts Analysis (MAA) is a decision analysis process that provides a framework for an owner or regulator to consult key stakeholders during the process of selecting or optimizing a closure plan. At Rum Jungle, the DME elected to utilize a MAA to ensure that environmental, technical, and cultural issues are considered transparently during the process of rehabilitation planning and that a preferred rehabilitation scenario is selected in a systemized and defensible manner.

The seven steps involved in a MAA are summarized as follows:

- Step 1. Identification of Candidate Rehabilitation Scenarios
- Step 2. Pre-Screening Assessment
- Step 3. Selection of Characterization Criteria
- Step 4. Development of a Multiple Accounts Ledger
- Step 5. Value-Based Decision Process
- Step 6. Sensitivity Analysis
- Step 7. Summary & Recommendations

Steps 1 to 2 were undertaken by RGC and key DME personnel, and five alternative rehabilitation scenarios were ultimately retained for the MAA. These scenarios principally involve backfilling the flooded open pits with the highest-priority waste rock to eliminate further generation of AMD and then either re-covering residual waste rock or consolidating it in a new facility before re-covering. The proposed alternative rehabilitation scenarios are summarized as follows:

- Scenario 1. Install new covers on the existing waste rock dumps (WRDs) and Dyson's (backfilled) Open Pit
- Scenario 2. Backfill the Main and Intermediate Open Pits and re-cover residual waste rock from the Main WRD & waste rock from Dyson's WRD in the central mining area
- Scenario 3. Backfill the Main and Intermediate Open Pits and re-locate residual waste rock from the Main WRD to Dyson's Area before re-covering
- Scenario 4. Backfill the Main and Intermediate Open Pits and re-locate residual waste rock from the Main WRD & Dyson's WRD to a new WRD in the Old Tailings Dam area
- Scenario 5. Backfill the Main Open Pit and re-cover residual waste rock from the Main WRD & Dyson's WRD separately

Step 3 involved the identification of key issues and characterization criteria that allow for a relative comparison of the proposed alternative scenarios during the subsequent value-based decision process. In this context, an issue is defined as any material impact (either positive or negative) associated with implementing any of the alternative scenarios being evaluated. As part of Step 4, issues are typically grouped into one of the following categories (or 'Accounts'):

- Environmental;
- Socio-Economic;
- Technical; and
- Financial.

The value-based decision process (Step 5) of the MAA involves systematically considering each criterion in the multiple accounts ledger and evaluating positive and negative impacts related to a particular rehabilitation scenario. Initially, this process entails ranking the scenarios sequentially from best to worst for each indicator by deciding which scenario(s) maximizes positive impacts. Any scenario that maximizes positive impacts is then assigned a scalar value of 9 (out of 9). Scenarios that are perceived to maximize positive impacts to a lesser extent are assigned a scalar value of 8 or less (with a scalar value of '1' indicating that a proposed scenario does not provide any discernible positive impact).

After the scenarios have been assigned scalar values for each criterion, the importance of each criterion to the issue being considered is decided upon by participants in the value-based decision process and a weighting factor between 1 and 9 is assigned. A score for each issue is then determined by adding together the products of the scalar values and weighting factors for each criterion and dividing by the sum of the weighting factors for that issue. To determine an account score, issue scores are added together in the same way after the weight of the issue to the account is assigned. Similarly, the importance of each of the Environmental, Socio Economics/Cultural, and Technical Accounts to the overall MAA is eventually weighted by assigning a weighting factor for each account from 1 to 9 and account scores are then used to calculate the overall MAA score for each scenario.

Issue, account, and overall MAA scores are each expressed on a scale of 1 to 9. A higher score for particular issue or account indicates that a proposed scenario is more favorably regarded in regards to that issue or account. The scenario with the highest MAA score is considered the preferred rehabilitation scenario.

Upon completion of the value-based decision process, each of the proposed alternative scenarios is ranked based on a MAA score that reflects value judgments by the meeting participants. A sensitivity analysis (Step 6) is normally conducted at this stage in order to evaluate the sensitivity of the MAA scores to the weightings of the different accounts. This is an important step to identify any subjectivity or bias in the selection of a preferred rehabilitation scenario. Key conclusions from the MAA process and a series of recommendations on how to proceed with rehabilitation planning are then provided as part of Step 7.

3.0 MAA RESULTS

3.1 Overall Scores & Rankings

Scenario 4 received an overall MAA score of 7.8 (out of 9) and was selected as the preferred rehabilitation scenario for Rum Jungle (see Table 1). This scenario involves backfilling the pits and re-locating residual waste rock to a new facility in the Old Tailings Dam.

Table 1. Summary of MAA Results

Account	Alternative Rehabilitation Scenario					
	0	1	2	3	4	5
Environmental	3.3	4.3	6.6	4.4	8.0	6.9

(Weight = 9)						
Cultural	1.3	2.3	4.6	5.0	8.1	3.9
(Weight = 9)						
Technical	4.3	3.9	5.4	5.4	3.3	5.7
(Weight = 7)						
OVERALL SCORE	2.9	3.5	5.5	4.3	7.3	5.5
Ranking	-	5	3	4	1	2
Capital Cost	n/a	\$54	\$94	\$111	\$103	\$74

This scenario had the highest scores for each of the Environmental, Cultural, and Technical Accounts so was very highly-regarded in all aspects considered. Scenarios 2 and 5 were ranked second with an overall MAA score of 5.5. However, these scores lagged significantly behind the top score of Scenario 4. Key findings from the three individual accounts considered are summarized below.

3.2 Environmental Account

For Rum Jungle, issues in the Environmental Account pertain to the predicted environmental performance of a proposed rehabilitation scenario and include evaluating how effectively a scenario is predicted to minimize the potential for AMD and improve surface water quality conditions in the East Branch of the Finnis River.

Summaries of the key findings for the Environmental Account are provided below.

- Scenario 4 received a score of 8.0 (out of 9) and was ranked the highest of the proposed rehabilitation scenarios. This scenario was predicted to be particularly effective in terms of improving surface water quality in the East Branch of the Finnis River, limiting further generation of AMD, and minimizing further groundwater contamination because it involved consolidating waste rock in a new WRD in the Old Tailings Dam area.
- Both of the scenarios that involved backfilling one or both of the flooded open pits and then re-covering residual waste rock *in situ* near Fitch Creek (Scenarios 2 and 5) received scores between 6.5 and 7.0. These scenarios are still considered protective in terms of environmental performance but less so than Scenario 4 (due primarily to the likelihood of ongoing contamination of groundwater and surface water from residual WRD in the central mining area).
- The scenarios that involved installing new covers on the existing WRDs and backfilled pit in Dyson's Area (Scenario 1), or backfilling the pits and consolidating residual waste rock in Dyson's Area (Scenario 3) scored poorly in terms of environmental performance (less than 4.5 out of 9 for this account).

3.3 Socio-Economics/Cultural Account

Issues identified for the Socio-Economic Account were originally intended to characterize the potential socio-economic impacts of rehabilitation on local stakeholders and the TO representative's cultural aspirations for Rum Jungle. This account was, however, eventually re-labeled 'Cultural' because only potential impacts on Traditional Owners were considered during the value-based decision process. Moreover, the Cultural Account was eventually weighted the same as the Environmental Account. From experience elsewhere, this is an unusually high weighting for cultural issues but highlights the value that the DME has assigned to cultural aspects of rehabilitation planning.

Some key findings from the Socio-Economics/Cultural Account are summarized below.

- Any scenario that involved leaving a residual WRD near Fitch Creek or re-locating waste rock to Dyson's Area was unacceptable to the representatives of the Traditional Owners and received scores of 5 or less for the Socio-Economics/Cultural Account. Scenario 4, on the other hand, was very well-received by the Traditional Owners and received a score of 8.1 (out of 9). This scenario was overwhelmingly preferred by Traditional Owners because not only was waste rock removed from (or not re-located to) the most culturally-sensitive areas of the site but a significant improvement in surface water quality conditions in the East Branch of the Finnis River was assured by implementing this scenario (see Environmental Account above).
- The Traditional Owners expressed a very strong preference for restoring the pre-mining course of the East Branch of the Finnis River by directing the river through a water cover over the backfilled pit(s). Backfilling the Main Open Pit was considered a crucial aspect of rehabilitation by the Traditional Owners, whereas backfilling the Intermediate Open Pit was not a high-priority.

3.4 Technical Account

With respect to the Technical Account, which considers technical feasibility, engineering issues, and the future burden on society, Scenario 4 ranked the highest (7.3 out of 9) and other scenarios were scored much lower by comparison (i.e. 5.7 or less). The high score for Scenario 4 reflects in particular the relatively small WRD that would remain in the Old Tailings Dam area and the expected ease with which seepage from this new facility could be captured. Note, however, that this account was weighted lower than the Environmental and Socio Economics/Cultural Accounts because none of the technical issues were considered critical to the selection of a preferred rehabilitation scenario.

The financial cost to implement the different scenarios (capital costs only) were estimated from waste rock/pit volumes and accepted unit rates for major earthworks, such as re-locating waste rock and installing new covers. Also considered were lime costs (for backfilling the pits), the cost to install a composite liner under a new WRD in the Old Tailings Dam area, the cost to build a water treatment plant, and costs related to supervising the implementation of a scenario. In terms of expected capital costs (in \$AUS), Scenario 3 is the most expensive to implement (at \$111M) and Scenario 5 was the least expensive (at \$74M). The initial estimated capital cost to implement the preferred rehabilitation scenario (Scenario 4) was \$103M.

4.0 SUMMARY

The following proposed alternative rehabilitation scenarios for Rum Jungle were evaluated using an MAA approach:

- Scenario 1. Install new covers on the existing waste rock dumps (WRDs) and Dyson's (backfilled) Open Pit
- Scenario 2. Backfill the Main and Intermediate Open Pits and re-cover residual waste rock from the Main WRD & waste rock from Dyson's WRD in the central mining area
- Scenario 3. Backfill the Main and Intermediate Open Pits and re-locate residual waste rock from the Main WRD to Dyson's Area before re-covering
- Scenario 4. Backfill the Main and Intermediate Open Pits and re-locate residual waste rock from the Main WRD & Dyson's WRD to a new WRD in the Old Tailings Dam area

- Scenario 5. Backfill the Main Open Pit and re-cover residual waste rock from the Main WRD & Dyson's WRD separately

The MAA involved identifying key environmental, cultural, and technical issues and then weighing these issues based on expert advice from consultants and members of the Rum Jungle Working Group and input on cultural issues and post-rehabilitation land-use aspirations from representatives of the Traditional Owners. The alternative rehabilitation scenarios were then ranked (from 1 to 5) based on overall MAA score.

Backfilling the flooded open pits and re-locating residual waste rock to a new facility (Scenario 4) was selected as the preferred rehabilitation scenario for Rum Jungle. This scenario is very well-regarded from an environmental perspective because it maximizes the amount of the waste rock submerged underwater in the pits and consolidates waste rock in new WRD that is located relatively far from the East Branch of the Finniss River. Scenario 4 is also the only scenario that is well-regarded by the Traditional Owners. For these reasons, implementation of this scenario will likely establish a benchmark for the rehabilitation of abandoned mines in Australia.

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THE INFLUENCE OF BIOTITE AND PHLOGOPITE DISSOLUTION ON AMD NEUTRALISATION AT BRUKUNGA PYRITE MINE, SOUTH AUSTRALIA

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ABSTRACT

Brukunga Pyrite Mine is an abandoned pyrite mine in South Australia. As a result of mining, the sulphide ore was exposed to the atmosphere, which caused acid and metalliferous drainage (AMD) production that has unusually high magnesium levels (250 - 2000 ppm). Analysis of water, waste rock and tailings samples showed that the waste rock dumps are responsible for the abundance of Mg in the contaminated water, through acid attack on phlogopite and biotite, which are the only magnesium-bearing minerals in the waste rock. Dissolution of these minerals by the AMD releases not only magnesium, but also iron, potassium, aluminium and silica into solution. However, Al, Fe, and Mg are being removed through acid releasing precipitation, which reduces the neutralising capacity of the phlogopite and biotite dissolution. The sludge formed by lime neutralisation of the AMD at Brukung contains ~2 wt% magnesium, accounting for the reduction in Mg concentration of the water as it passes through the treatment plant. XPS analysis of the sludge indicates that the magnesium is bonded to carbonate.

1. INTRODUCTION

Prediction and management of AMD (acid and metalliferous drainage) from mining processes are significant in assessing the environmental and economic feasibility of any mining operations, and any neutralising capacity inherent in the rocks at the minesite can be beneficial in this regard.

A site that contains carbonate minerals usually has a good neutralisation capacity due to the rapid reaction of carbonate minerals (Eary and Williamson, 2006), but silicate minerals like chlorite, biotite and plagioclase also offer an acid neutralisation capacity, which is actually potentially greater than that of calcite (Weber et al., 2005, Sherlock, 1995, Nesbitt and Jambor, 1998, Banwart and Malmstrom, 2001).

At the abandoned Brukung Pyrite Mine in South Australia, the AMD contains unusually high levels of magnesium (250 - 2000 mg/L). If this comes from the dissolution of biotite in the bedrock it indicates that silicate dissolution may be neutralising significant amounts of acidity at the site. The magnesium is removed in the lime neutralisation treatment plant, but the mechanism of this removal is uncertain. This paper investigates biotite dissolution at the mine site, in order to determine whether this process is acting as an acid neutralising agent, reducing the overall costs of remediation at the mine.

2. DESCRIPTION OF THE STUDY AREA

Brukung is situated approximately 40 km east of Adelaide in the Mt Lofty Ranges of South Australia. The Brukung Pyrite Mine was established in the early 1950s, and the mining of pyrite and pyrrhotite for the production of sulphuric acid continued until operations ceased in

1972. 8 Mt of waste rock was placed in two large waste rock piles and 3.5 Mt of tailings were placed in a former valley (Armstrong and Cox, 2005).

The pyritic beds which were mined form the Nairne Pyrite Member within the shallow marine Lower Cambrian Talisker Calc-siltstone (Daily and Milnes, 1972). The sediments were subjected to intermediate grade metamorphism, forming chlorite and biotite (Offler and Fleming, 1968).

Not all of the iron sulphides were removed during mining and processing, and a considerable amount still remains in quarry walls, waste rock piles and tailings, and is oxidising and producing acid and metalliferous drainage (AMD). To remediate the AMD problems, a treatment plant using hydrated lime was built. This was upgraded in 2005 to double the plant's capacity, as well as changing to the high density sludge (HDS) process, which produces less voluminous sludges.

3. METHODS

Water was sampled from surface water and groundwater around the Brukunga Pyrite Mine and from different stages in the treatment plant. Some parameters were measured in situ - pH, conductivity, temperature, redox potential and dissolved oxygen, and the samples were analysed for Al, Si, Ca, Mg and SO_4^{2-} as S using a GBC Quantima Inductively Coupled Plasma - Optical Emission Spectrometer (ICP-OES) and Fe, K, Mn and Zn using a Varian AA240 Atomic Absorption Spectrometer. Na was measured on the Sherwood Mode 420 Flame Photometer and Cl analysis was conducted by Ion Chromatography (IC) using a Metrohm 883 Basic IC instrument. Total alkalinity was determined by titration, and samples that had an initial pH of less than 4.5 were assumed to contain no alkalinity.

Thirteen rock samples were taken at random from across the Brukunga site, along with a precipitate from Dawsley Creek. Sludge samples were taken from each of the reactor tanks in the treatment plant, and a precipitate from the tailings storage facility (TSF) wall was collected, along with samples from different depths in the TSF (0.5 m and 3 m). Mineral analysis was conducted with X-ray powder diffraction (XRD) and geological major and minor elemental analysis was conducted with X-ray fluorescence (XRF). X-ray photoelectron spectroscopy (XPS) was conducted on two treatment plant sludge samples. Carbon-coated polished thin sections of rock samples were analysed by a Cameca SX50 Electron Microprobe. Scanning Electron Microscopy coupled with Energy Dispersive Spectrometry was conducted on a sludge sample.

4. RESULTS

The bedrock in the Brukunga area consists of mica schists, composed of muscovite, feldspars, biotite and phlogopite (the Mg-rich end member of biotite with Mg:Fe ratio > 3). Pyrite is found in some samples and jarosite can be seen forming around some pyrite crystals.

The yellow precipitate collected from the wall of the tailings storage facility is composed of 50 wt% jarosite, along with small amounts of pyrite and muscovite. The upper tailings are oxidized and contain less biotite and no phlogopite compared to the unoxidised tailings located deeper in the storage facility.

The predominant minerals in the sludge formed at the treatment plant are gypsum (60-80 wt%), ettringite ($\text{Ca}_6\text{Al}_2(\text{SO}_4)_3(\text{OH})_{12}\cdot 26\text{H}_2\text{O}$, 10-30 wt%) and amorphous ferric hydroxide (10-30 wt%). There are substantial amounts of Mg in the sludge (3.23% as MgO), which is

present as an amorphous material and/or a crystalline phase in quantities too low to be detected.

X-ray photoelectron spectroscopic (XPS) data for the sludge (Fig. 1) shows a C peak at 289.5 ± 0.1 eV indicating carbonate compounds (Liu et al., 2009), and a Mg 2p peak at 50.1 eV, which is identical to that in magnesium carbonate (Liu et al., 2009). Mg 2p peaks for magnesium-sulphate and magnesium-hydroxide have different binding energies: 51.0 - 52.0 and 49.6 - 49.8 eV respectively. (Ardizzone et al., 1997, Fournier et al., 2002, Haycock et al., 1978, Liu et al., 2009).

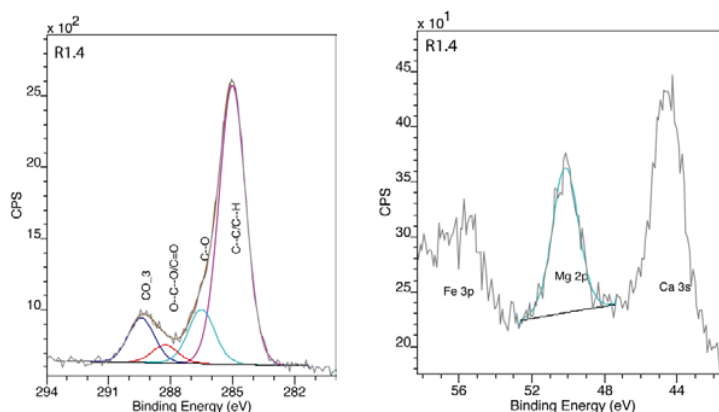


Fig. 1. XPS scans for carbon and magnesium phases in The treatment plant sludge.

The water analyses were standardised to the highest Cl^- concentration to eliminate any differences due to evapotranspiration, assuming that chloride is a tracer and is unaffected by any water-rock reactions or mineral dissolution/precipitation. The analyses were then grouped by cluster analysis and six clusters were identified. The median analysis of each was plotted as a Schoeller plot (Fig. 2).

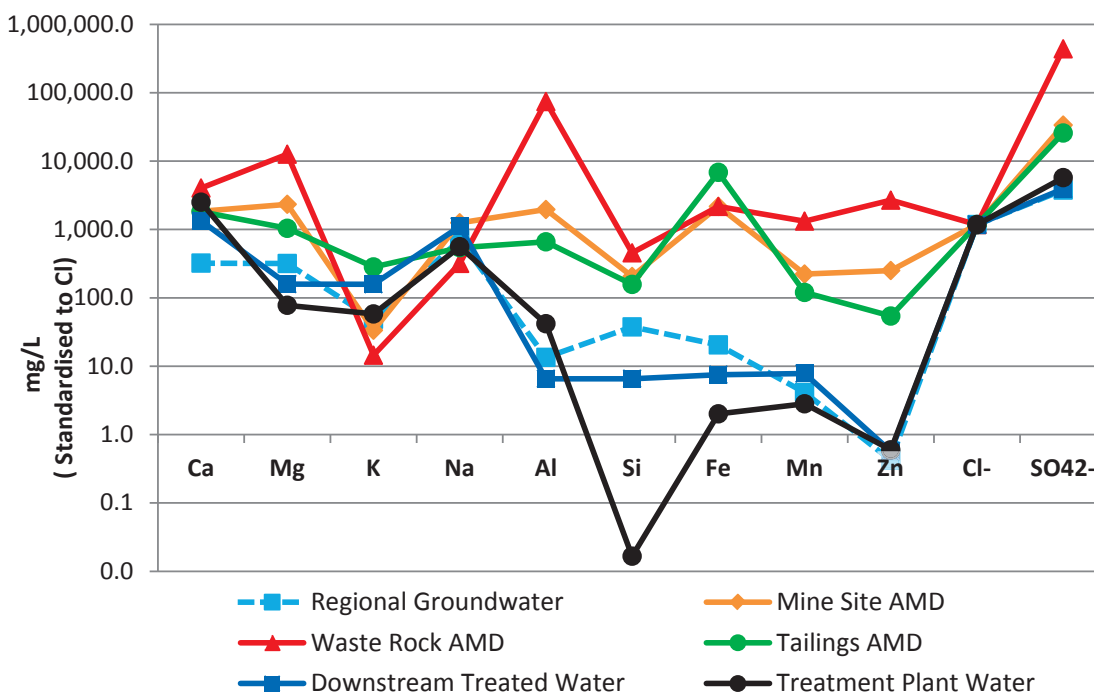


Fig. 2. Schoeller plot of median analyses of each group of water samples

Uncontaminated regional groundwater has consistently low concentrations of Al, Fe and SO_4^{2-} , and a median pH of 6.3, and comes from bores across the mine at depths of at least 12-14 m, which is below the depth at which groundwater contaminated by AMD is encountered.

The pH for shallow groundwater contaminated by AMD is 2 – 3 and the median EC is 7.5 mS/cm. Ground and surface water from around the waste rock piles is very acidic and has the highest EC (12 - 25 mS/cm) and the highest levels of Mg, indicating that the waste rock piles are the main source of Mg in the AMD at the mine site.

In the treatment plant Al, Fe, Mn, Mg, Si, Zn and SO_4^{2-} are strongly removed during the neutralisation process, but K is not. Ca concentrations are kept high due to the addition of hydrolysed lime.

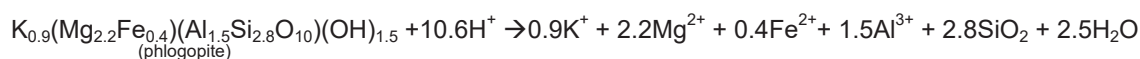
5. DISCUSSION

The pyrite and pyrrhotite at the mine site oxidise to produce ferrous iron, sulphate and acid. The ferrous iron oxidises to ferric iron which hydrolyses to form ferric hydroxide or oxidises pyrite, producing more ferrous iron and acid.

The oxidation of pyrite and pyrrhotite account for the high levels of Fe and SO_4^{2-} found in the acid and metalliferous drainage (AMD) at Brukunga. The AMD from around the waste rock piles shows higher concentrations of Fe than in the AMD elsewhere at the mine site, where Fe may be precipitating as iron hydroxide, goethite or lepidocrocite which were identified in some rock samples.

The acid produced attacks minerals in the surrounding schists, resulting in the high concentrations of metals in the mine waters. The high levels of calcium and sodium in the AMD are due to feldspar weathering, which also releases aluminium; the low pH prevents Al from precipitating. Biotite, phlogopite and muscovite have also undergone acid attack, releasing more Al as well as Fe, Mg and K. Because micas can incorporate manganese in their crystal structures (Deer et al., 1992), the high Mn concentrations in the AMD were probably released from the micas.

Biotite and phlogopite are affected to a much larger extent by decreasing pH than muscovite (Kalinowski and Schweda, 1996). From the empirical formulas derived by electron microprobe analyses of biotite and phlogopite at the mine site $((K_{0.7}(Mg_{0.9}Fe_{1.1})(Al_{1.7}Si_{2.9}O_{10})(OH)_{1.5})$ and $(K_{0.9}(Mg_{2.2}Fe_{0.4})(Al_{1.5}Si_{2.8}O_{10})(OH)_{1.5})$ respectively, the dissolution of one mole of phlogopite consumes 10.6 moles of H^+ , and dissolution of one mole of biotite consumes 9.8 moles of H^+ . Biotite and phlogopite dissolution releases potassium and magnesium in the molar ratio of 1:1.3 to 1:2.4 but the ratio in the AMD at Brukunga is 1:126 to 1:2154, because the precipitation of jarosite is sequestering K. Potassium is in relatively higher concentrations in the collection ponds located directly in front of the tailings storage facility, then the mine site AMD.



Jarosite precipitation not only uses potassium and iron from biotite/phlogopite dissolution, but requires extra iron released by pyrite oxidation, and releases 3.9 moles of acid per mole of potassium (from the empirical formula derived by electron microprobe analyses), which counteracts some of the initial neutralisation:



When the AMD reaches the treatment plant, the pH rises and effectively all the aluminium and iron in solution from biotite/phlogopite dissolution precipitates as $Al(OH)_3$ and $Fe(OH)_3$, releasing another 3 moles of acid per mole of Al and Fe.

Magnesium is strongly removed during neutralization in the treatment plant. X-ray photoelectron spectroscopy (XPS) analysis of the sludges suggested that the Mg is bonded to carbonate and the treatment water is supersaturated with respect to magnesite ($MgCO_3$), but undersaturated with respect to $MgSO_4$ and $Mg(OH)_2$. Analyses conducted by SEM showed that the Mg is uniformly distributed through the sludge and did not reveal any magnesite crystals, so the Mg may be absorbed onto the surface of sludge particles forming a complex with CO_3^{2-} . The removal of Mg bonded to carbonate will release an additional 2 moles of acid per mole of magnesium.

Thus although dissolution of biotite/phlogopite consumes 9.8/10.6 moles of H^+ , the precipitation of effectively all the Al, Fe and Mg released cancels out the bulk of this acid neutralization (9.1/9.7 moles of H^+ per mole of biotite/phlogopite). Some potassium released by biotite/phlogopite dissolution remains in solution, so a small proportion of the acid neutralization by biotite/phlogopite dissolution remains effective, but this represents <1% of total acidity that is neutralised at the treatment plant.

6. CONCLUSION

At the Brukunga Mine the dissolution of biotite and phlogopite releases potassium, iron, aluminium and magnesium, and all the Fe, Al, Mg and most of the K precipitate as secondary minerals that release acid.

So while the dissolution of biotite and phlogopite does initially neutralise acidity, it has little overall effect due to the acid-releasing precipitations largely occurring in the neutralisation plant as a result of the increased pH.

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WHAT DO ELEVATED BACKGROUND CONTAMINANT CONCENTRATIONS MEAN FOR AMD RISK ASSESSMENT AND MANAGEMENT IN WESTERN AUSTRALIA?

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ABSTRACT

Water quality contaminants include a range of naturally occurring chemicals that can cause degradation of aquatic ecosystem water values when concentration ranges exceed biological tolerances. Both acid and metalliferous drainage (AMD) and acid sulfate soil (ASS) can increase contaminant concentrations through reduced pH and increased solute concentrations especially of toxic metals and metalloids. Water quality guideline criteria are typically used to maintain existing end use value objectives when managing AMD/ASS-affected waters.

However, surface and ground waters of catchments comprising mining resources often show elevated solute concentrations in baseline conditions due to their unique geologies. From an AMD and ASS risk assessment perspective, regional water quality may therefore be unique and locally-relevant such that site-specific water quality guidelines may therefore be required to most reasonably manage water quality objectives.

We provide case study examples from iron ore and coal mining from the Western Australian regions of the Pilbara, and the South-west to show that defining water quality criteria for closure is more than just using generic national guidelines, but an explicit consideration of the baseline regional bio-physico-chemical context.

1.0 INTRODUCTION

Physical and chemical stressors are naturally occurring parameters that can cause significant impacts to aquatic ecosystems when concentrations are either too high or too low. Physical and chemical stressors may also present an additive toxic risk or may modify the effects of toxicants through synergism and antagonism of other stressors, e.g. pH can alter the dissolved fraction of metals and their speciation affecting their toxicity (Neil et al. 2009). Acid and metalliferous drainage (AMD) can increase contaminant concentrations through reduced pH and increased solute concentrations, especially of toxic metals and metalloids.

1.1 Water Quality Guidelines

The role of water quality guideline criteria in managing waters actually or potentially affected by AMD is typically to maintain existing end use value objectives. As for much of the developed world (Jones 2012), mine closure planning in Western Australia requires development of closure objectives and commensurate criteria to demonstrate achievement of these objectives (DMP/EPA 2011). There are general acid and metalliferous drainage (AMD) monitoring protocols and standards available which can help guide mine water quality management and monitoring. The International Network for Acid Prevention (INAP) has

produced a global acid rock drainage (ARD) guide (GARD Guide) which summarises the technical and management practices for industry and stakeholder use (Verburg et al. 2009). There are also many scientific and technical organisations working on AMD and heavy metal pollution from mine operations, such as the International Network for Acid Prevention (INAP), Mine Environment Neutral Drainage (MEND), the International Mine Water Association (IMWA), the Acid Drainage Technology Initiative (ADTI), the South African Water Research Commission (WRC), and the Partnership for Acid Drainage Remediation in Europe (PADRE). These organisations have all published guidelines which, together, represent an overview of strategies on monitoring water quality of surface water relevant to the extractive industries from international and national level of governments and organisations.

However, selection of related guidelines and appropriate sampling protocols still must depend on site-specific characteristics, permitting standard limits and requirements, and required data accuracy and precision. These in turn must explicitly depend upon stakeholder values such as long-term end uses for the water resource as well as accounting for regional water quality and ecological tolerances, e.g. extremely soft receiving waters (Van Dam et al. 2010) and any existing disturbances.

Across Australasia, a common default position for water quality management criteria by regulators is the use of the Australasian Water Quality Guidelines (A/A) (Batley et al. 2003a); particular those for 95% protection of aquatic ecosystems (McCullough and Van Etten 2011). The purpose of water quality guidelines is to maintain, or improve, upon current water quality values. Three broad ecosystem condition classifications are recognised by the A/A guidelines (ANZECC/ARMCANZ 2000). All ecosystem condition classifications explicitly recognise the long-term goals desired by stakeholders and implied by the selected management goals and water quality objectives for the water resource.

1. High value systems (99% biodiversity protection) — effectively unmodified or other highly-valued ecosystems that are of conservation or other ecological value with ecological integrity intact. Unlikely to occur outside of national parks or other conservation reserves, or outside of remote and undisturbed locations.
2. Slightly to moderately disturbed systems (95% biodiversity protection) — ecosystems in which aquatic biological diversity may have been slightly adversely affected by human activity. However, biological communities remain in a healthy condition and ecosystem integrity is largely retained.
3. Highly disturbed systems (80% biodiversity protection) — degraded ecosystems of low ecological value. Although this classification implies that degraded aquatic ecosystems still retain potential, or actual, ecological or conservation values, practical considerations mean it may often not be feasible to return them to a slightly–moderately disturbed condition.

1.2 Water Quality Guideline Development

Water quality criteria used in ecosystem protection guidelines are to protect aquatic biota (the end use value in this circumstance) and are not an end in themselves. These default values are prepared by analysis of a comprehensive set of available ecotoxicological data (Aldenberg and Slob 1993). Specifically, the default values are prepared by analysis of a comprehensive set of available ecotoxicological data from a number of regions and species; which may, or may not be, relevant to the area of interest.

Biological methods of assessing toxicity in aquatic environments have found support in developing water quality criteria because they have the capacity to integrate effects through continuous exposure, and, more specifically because they measure directly the level of

change at which a particular substance becomes toxic. This approach supplants earlier efforts at indirectly estimating toxicity, using chemical and physical surrogate measurements alone (Auer et al. 1990; Karr and Chu 1997, 1999). Ecotoxicological testing with bioassays is valuable for ranking the toxicity of different chemicals and other stressors, for determining their acceptable concentrations in receiving systems, and for elucidating cause and effect relationships in the environment (Chapman 1995).

However, the data for the A/A toxicant trigger values are generally not locally-derived both in a national and almost certainly not in a regional sense; they do not include any local species in the toxicity testing array and are largely based on overseas species, e.g. rainbow trout, fathead minnow, *Daphnia magna*, etc. Hence, ecological responses such as toxicity-testing data from relevant test-species and local receiving waters are of significantly more value than broadly encompassing water chemistry guidelines such as the A/A in determining what concentrations are acceptable for protecting ecosystem values.

The A/A data for physico-chemical stressors do not incorporate toxicity-test data at all; just the 80th percentile of the amalgamation of a number of historical data sets across broad geographical regions. Therefore, there are no explicit data that these values will protect 95% of the species found in this area's ecosystem. Indeed, in the aquatic ecosystems of most remote locations in which much mining occurs, we do not understand the assemblages well enough to know what 95% of the species (they have not yet been described) and therefore site-specific trigger values are not available.

1.3 Regional Differences: or one site's 'contaminated state' is another site's 'baseline condition'

In the absence of site-specific trigger values for aquatic ecosystem protection, drainage quality and leaching test results require comparison to default guideline values. These generic values may not be appropriate for many mining regions whose catchments are expected contain enriched geological materials. Indeed, from a mining perspective, what *Contaminated Sites* practitioners may call 'above ambient levels', miners refer to as 'resource indicators' such as elevated metal concentrations in waterway sediments (Averill 2013).

This natural geochemical enrichment may lead to elevated water body solute concentrations when compared with non-site-specific assessment criteria for water bodies and aquatic ecosystems. Use of generic guidelines not accommodating this regional variability may result in over management of mine drainage not commensurate with the risk it presents (Batley et al. 2003b).

1.4 Water Quality Monitoring

Critically, monitoring of AMD receiving waters is not management; it is the process by which the need for management intervention is determined. In application of monitoring data to the guidelines, contaminant concentrations are usually compared with a single default trigger value, developed from a modelled protection value arising from the data generated by these tests. In the absence of local toxicity-response test data (by far the most common circumstance worldwide), the default trigger values are assumed to provide a reasonable upper limit. However, only rarely is water quality monitoring data compared with either water quality distributions from either a baseline dataset or from a reference site.

Trigger values are defined as concentrations that, if exceeded, would indicate a potential environmental problem, and consequently should 'trigger' a management response.

ANZECC/ARMCANZ (2000) further defines low-risk trigger values as “*concentrations (or loads) of key performance indicators below which there is a low risk that adverse biological effects will occur*”. The physical and chemical trigger values are not designed to be used as ‘compliance’ or threshold values at which an environmental problem is inferred if exceeded, rather, they are designed to be used in conjunction with professional judgement in order to provide an initial assessment of the state of the water body (Hart et al. 1999; ANZECC/ARMCANZ 2000).

1.5 Trigger Values

The role of the 80th percentile at the reference site is to simply quantify the notion of a ‘measurable perturbation’ at the test site. These 80th percentile site-specific trigger values provide an upper limit for stressors that cause problems at high values, while the 20th percentile provides a lower limit for stressors that cause problems at low values. These site-specific trigger values set a benchmark for ‘current status’ against which future variation in water quality at these sites can be assessed.

Toxicant default values are based upon actual biological effects data and so by implication, exceedance of the value indicates the potential for ecological harm, rather than a percentile system which has intuitive appeal among experts but is otherwise largely arbitrary. Hence, the A/A guidelines recommend toxicity-test values take precedence over local reference conditions until more robust (e.g., local test species) toxicity-testing data can show otherwise. As a consequence, for toxicity test-derived trigger value concentrations to be increased, demonstration of no-significant ecological effect would generally have to be carried out using local species or similar site-specific data.

Trigger values are not an instrument to assess ‘compliance’, trigger values are an ‘early warning’ mechanism to alert managers of a potential problem. Compliance values will generally be set by regulators based upon national or regional water quality guidelines, or from an arbitrary deviation from regional/median expected water quality.

Trigger values are derived preferably from locally appropriate control or reference data, although the Water Quality Guidelines provide default values where such data do not exist or cannot be gathered (Barmuta et al. 2001).

In formal terms the trigger-base approach is as follows: A trigger for further investigation will be deemed to have occurred, when the median concentration of *n* independent samples taken at a test site exceeds the eightieth percentile of the same indicator at a suitably chosen reference site or from the relevant guideline value in the Water Quality Guidelines. This approach acknowledges natural background variation through comparison to a reference site’ baseline conditions or an analogue site(s). The locally derived water quality triggers therefore accommodate site- specific anomalies.

Because the reference site is being monitored over time, the trigger criterion should be constantly updated to reflect temporal trends and the effects of extraneous effectors (e.g., climate variability, seasonality).

2.0 CASE STUDIES

We present two case studies for site-specific derivation of trigger values from Western Australia’s South-west and Pilbara and biogeographic regions (Fig. 1).

Site-specific trigger values were derived following the derivation protocol presented in the flow chart (Fig. 2). This protocol is fundamentally based on Section 3.3.2 'Defining low-risk guideline trigger values' of the Australasian Water Quality Guidelines for Fresh and Marine Waters (ANZECC/ARMCANZ 2000). Outliers are measurements that are extremely large or small relative to the dataset distribution and, therefore, are suspected of being sampling errors that may misrepresent the population from which they were collected (USEPA 2006). Significant outliers were identified and removed from the datasets prior to analyses.

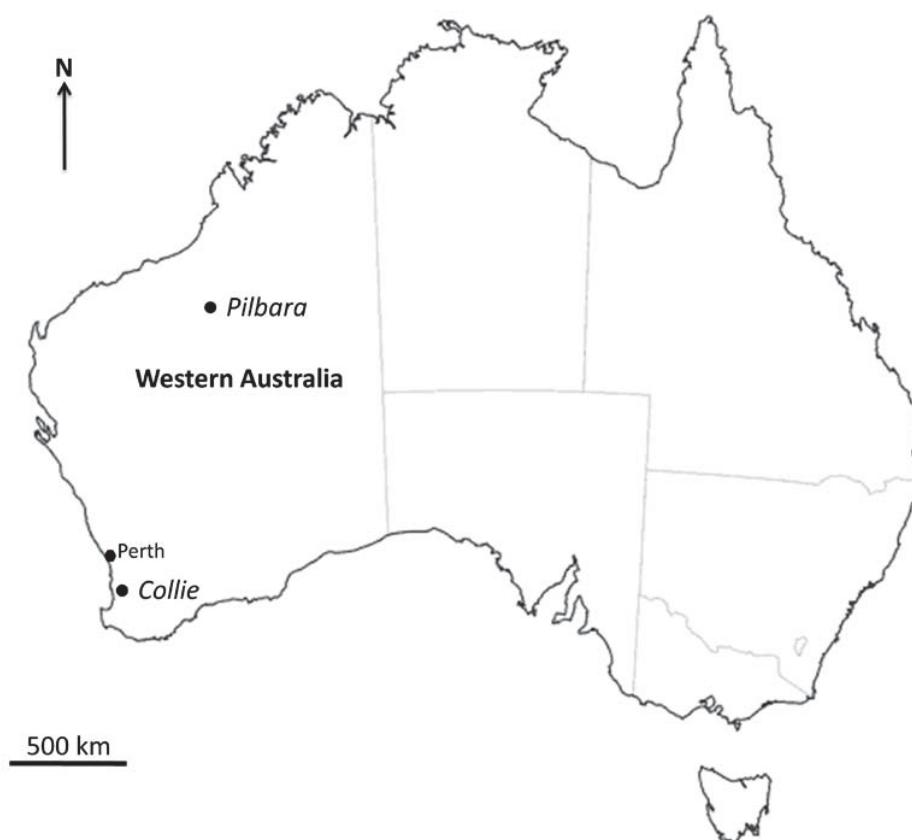
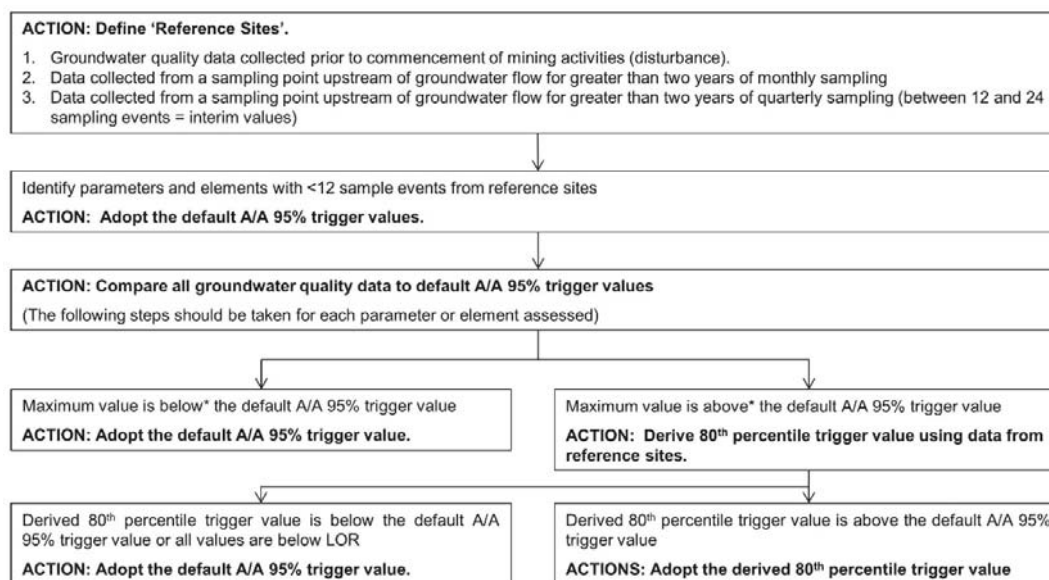


Fig. 1. Case study locations in Western Australia



*For parameters with an upper and lower default A/A 95% trigger value (pH, EC, etc), derive 20th and 80th percentile values

Fig. 2. Site-specific trigger value derivation protocol used for the case studies

2.1 South-West Acid and Metalliferous Discharge

Mining of the Lake Kepwari void in south-west Australia (WO5B) began with diversion of the seasonal Collie River South Branch (CRSB) away from the pit site and around the western margin, and ceased in 1997. During rehabilitation, reactive overburden dumps and exposed coal seams were covered with waste rock, battered and topsoil replaced, and revegetated with native plants. To reduce wall exposure and acid production, the lake was rapid-filled by a brackish first-flush diversion from the CRSB over three winters from 2003–2005 (Salmon et al. 2008).

Although river water initially raised water pH to above pH 5, lake pH subsequently declined to below pH 4 by 2011 and displayed elevated solute concentrations as a result of AMD inputs, most likely though in-catchment and in-lake acidity generation from PAF, and acidic groundwater inflow (Müller et al. 2011). The volume of the lake is now around $32 \times 10^6 \text{ m}^3$, with a maximum depth of 65 m and surface area of 103 ha.

During the third week of August 2011, a rainfall of 85.6 mm (BOM, 25/12/2012) in Collie over 48 h led to high flows in the CRSB of a 1:8 year magnitude (DOW 2013). The water level in the CRSB rose, overtopping and then eroding the engineered northern dyke wall that separated the CRSB diversion from Lake Kepwari. As a result, water levels in Lake Kepwari rose 1.7 m adding 3 ha to the surface area of 103 ha (Premier Coal unpublished data) and increasing lake volume by around 6% (McCullough et al. 2012). Lake water then decanted through a previously designed outlet before overtopping this and then decanting back through the breach as CRSB levels dropped (McCullough et al. 2013). There is now a permanent connection to the river and lake at the breach point and through a newly designed lake outflow weir. Planned long-term flow-through of the pit lake via the weir presents a risk to downstream water quality that needed to be understood in the regional context of naturally and historically elevated CRSB solute concentrations.

Laboratory data collected through the Premier Water Quality Monitoring Program (PC-CRSWMP) were used to develop site-specific trigger values for the CRSB from a reference site above Lake Kepwari influence. Regular exceedance of water quality guidelines by a reference South Branch site above the pit lake decant indicates that Collie River background concentrations were already elevated for many of these parameters (McCullough et al. 2013). Although there are no known mine influences above this point, there are known catchment activities that have degraded water quality through eutrophication such as farming (Wetland Research & Management 2009) and salinisation through forest clearance (Tingey and Sparks 2006).

A single measurement meter (Hanna Instruments) was used to sample water quality at each CRSB site for pH and dissolved oxygen (DO). Water samples were also collected and laboratory analyses undertaken for dissolved Al, Fe, Mn, Zn and suspended sediment (SS) concentrations. Water samples were filtered and then acidified with reagent grade nitric acid (1%) until analysed for selected elements by Inductively Coupled Plasma Mass Spectrophotometry (ICP-MS, Varian). These and other water quality analyses were undertaken in a NATA-accredited laboratory and followed standard methods (APHA 1998).

PC-CRSWMP data were statistically analysed using SPSS (2011) statistical software. Prior to derivation of 80% Trigger Values, data sets were explored. Exploratory techniques include numerical summaries, data visualisation, transformations, detection of outliers, checking for censored data, trend detection and smoothing. Relative fits to normal distributions as a test of dataset skewness and median data values were calculated in addition to 80th percentile values for stressors that cause problems at high concentrations and 20th percentile values for stressors that cause problems at low concentrations.

The site-specific trigger values derived for the Collie River South Branch are shown in Table 1. Where available, derived water quality results were compared against 80% A/A Aquatic Ecosystem Protection guidelines as the CRSB is described as “highly disturbed” (Wetland Research & Management 2009). A/A 80% trigger values were used as these are based on actual environmental responses to these variables (as toxicity-test data). Where 80% Ecosystem Protection guidelines were not available, water quality results were conservatively compared against default A/A 95% Ecosystem Protection guidelines (slightly disturbed ecosystems).

Important factors influencing the interpretation of monitoring water quality data are as follows. Site-specific trigger values should be considered in context of other information relevant to individual physical and chemical variables that may serve to moderate toxicity. At this stage, site-specific triggers relate to base-flow conditions as they were only derived from ebb and base-flow conditions. Consequently, they should not be compared with monitoring during high flow events. Some site-specific trigger values, e.g. TDS also differ greatly from respective A/A (2000) default low-risk trigger values. This is due to the limitations of the regional approach adopted by A/A for developing default low-risk trigger values that, by definition, does not account for local variability and catchment geochemical differences. For example, it is also likely that the site-specific trigger value for these sites reflects the existing impact that catchment activities such as deforestation leading to increased sediment and solute loads have had on the CRSB (Mauger et al. 2001).

Table 1. 80% Ecosystem Protection A/A (2000), derived Collie River South Branch (CRSB) and final trigger values. All values mg L⁻¹ unless otherwise stated.

Parameter	Fe	Mn ¹	pH	TDS	DO ³	SS ³
Ecosystem protection ¹	0.31	3.6	—	—	—	—
South-west, WA lowland river ²	—	—	6.5–8.0	77–192	—	—
Livestock watering ⁴	—	—	—	2.0 ⁴	—	—
Reference ⁵	0.05–4.3	0.02–5.1	3.2–8.6	580–5,700	-	1–97
Derived 20/80%	0.83	1.2	6.5–7.3	1,700–3,700	5.7–8.4	17
Site-specific trigger values	0.83	3.6	6.5–8.0	77–3,700	5.7–8.4	17

¹ANZECC (2000) 95% ecosystem protection, ²as base river flow, ³as diurnal range, ⁴poultry no-effect limit, ⁵CRSB PML09 reference monitoring site, — = no guideline available.

2.2 Case Study 2: Pilbara

Iron ore mining in the Pilbara has been occurring since the mid 1960's to exploit the vast mineralised banded iron formations, namely the Brockman, Marra Mamba and Nimingarra Iron Formations. Natural waters that contact these enriched geological materials may accumulate higher concentrations of solutes when compared with non-site-specific assessment criteria for water bodies and aquatic ecosystems.

Water quality databases were provided by the client for each mining operation and each operation was assessed individually. An assessment of appropriate reference sites was undertaken to identify monitoring or production wells:

- With groundwater quality data collected prior to commencement of mining activities (disturbance).
- Within undisturbed locations (i.e. from upstream of possible environmental impacts).
- Within a local but different system.

Sites required data for greater than two years of monthly sampling (≥24 sampling events) for the derivation of site-specific trigger values. Interim values were derived for datasets comprising between 12 and 24 sampling events.

Where available, water quality results were compared with default A/A 95% trigger values for aquatic ecosystem protection and default trigger values for physical and chemical stressors for tropical Australia for slightly disturbed ecosystems (ANZECC/ARMCANZ 2000). A/A 95% trigger values were used as these are based on actual environmental responses to these variables (as toxicity-test data). Where default A/A 95% trigger values were not available, water quality results were compared against interim or low-reliability values (ANZECC/ARMCANZ 2000).

Prior to the derivation of the 80th percentile trigger values and other statistical analysis, treatment of data treatment was employed. Groundwater quality measurements reported as below the limit of reporting were considered equal to the limit of reporting (ANZECC/ARMCANZ 2000). This particular treatment of the data may influence results when

a higher than usual limit of reporting was provided. Data outliers are then omitted if the raised limit of reporting is greater than any other measured concentration in the dataset.

As presented in Fig. 2, maximum values for As, Co, Mn, Sb, Se and Sn were below the default A/A 95% trigger values. Therefore, site specific trigger values were not derived for these elements and the default criteria were applied. Default A/A criteria were also applied to those parameters and elements that recorded maximum values above the default assessment criteria but lower derived 80th percentile values (pH, Ag, Al, B, Be, Fe, Hg, Mo and Ni).

Although the maximum values for Ag and Be were above the A/A 95% trigger values, they were not measured above the limit of reporting. Therefore, developing 80th percentile trigger values for Ag and Be would not be appropriate.

Derived site-specific trigger values for EC, NO₃, total P, Cd, Cr, Cu, Pb and Zn were above the default A/A 95% trigger values for aquatic ecosystem protection and default trigger values for physical and chemical stressors for tropical Australia for slightly disturbed ecosystems (ANZECC/ARMCANZ 2000). A subset of these parameters/elements is presented in Table 2. This exceedance of trigger values by baseline water quality may be due to intrinsic limitations of the regional approach adopted by ANZECC/ARMCANZ for developing default trigger values that, by definition, do not account for finer regional variability and catchment geochemical differences, even if they are well known, e.g. elevated Pilbara NO₃ concentrations (Magee 2009).

Table 2. 95% Ecosystem Protection A/A (2000) and derived site specific trigger values. All values mg L⁻¹ unless otherwise stated

Parameter	NO ₃	Cd	Cr	Cu	Pb	Zn
Ecosystem protection	0.7 ¹	0.0002	0.001 ^{2,3}	0.0014	0.0034	0.008 ²
Reference Sites	1.5–62	0.0001– 0.01	0.001– 0.05	0.001– 0.089	0.001– 0.006	0.005– 0.1
Derived 20/80%	12.4	0.001	0.005	0.006	0.005	0.013
Site-specific trigger values	12.4	0.001	0.005	0.006	0.005	0.013

¹ Figure protects against toxicity and does not relate to eutrophication issues, ²Figure may not protect key test species from chronic toxicity, ³as chromium VI, — = no guideline available.

Some site-specific trigger values were much higher than the respective default A/A 95% trigger values (e.g. NO₃). This may be due to intrinsic limitations of the regional approach adopted by ANZECC/ARMCANZ for developing default trigger values that, by definition, do not account for local variability and catchment geochemical differences.

As outlined in Case Study 1, site-specific trigger values should be considered in context of other information relevant to individual physical and chemical variables that may serve to moderate toxicity. They should also be reassessed when more data from various production and monitoring wells becomes available; this will work towards addressing any limitations identified in the dataset. The suitability of locations selected as reference sites should also be reassessed once new data becomes available.

3.0 CONCLUSIONS

Prior to the development by a new mining project, many sites may not have water quality conditions defined as pristine by the A/A water quality guidelines (99% biodiversity protection). An increase in contaminant concentrations is reasonably expected where the geochemistry of a site is already elevated for certain elements; characteristics that make this region of interest for mineral extraction.

Trigger values are solely intended to provide an 'early warning' mechanism to alert managers to a potential risk. The appropriate response may be further site-specific investigation, or immediate remedial action. They are not explicitly designed as compliance points in themselves; this would involve a more complete assessment of the toxicant source – pathway – receptor model (Hart et al. 2006).

In many cases 80% trigger values have values that are more sensitive than ANZECC/ARMCANZ (2000) 90% Ecosystem Protection values for the same variable. In this case we recommend that the ANZECC values be used instead of the 80% trigger values as the criteria and data quality to determine this value has been more rigorously been evaluated.

For ecosystems that can be classified as highly disturbed, the 95% protection trigger values can still apply. However, depending on the state of the ecosystem, the management goals and the approval of the appropriate state or regional authority in consultation with the community, it can be appropriate to apply a less stringent guideline trigger value, say protection of 90% of species, or perhaps even 80%. These values are provided as intermediate targets for water quality improvement. If the trigger values have been calculated using assessment factors, there is no reliable way to predict what changes in ecosystem protection are provided by an arbitrary reduction in the factor.

Comprehensive and effective assessment and management of water quality relies on integrating biological approaches with the more traditional chemical and physical-based approaches, where chemical data provide explanatory variables for trends observed for biota ("cause for consequence") (Chapman 1990). We propose that maintenance of the value of water bodies potentially impacted by AMD is best served by more explicit consideration of regional water quality norms and objectives and site-specific baseline condition than by over-simplistic application of national or regional guidance.

4.0 REFERENCES

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FIELD AND DESKTOP WASTE ROCK CLASSIFICATION GUIDE FOR A METALLIFEROUS MINE IN THE NORTHERN TERRITORY, AUSTRALIA

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ABSTRACT

A field and laboratory geochemical investigation was conducted to assess the potential for heavy metal leachate release, saline drainage and acid rock drainage (ARD) from the overburden materials generated from a metalliferous mine in the Northern Territory. The results from the geochemical investigations were used to derive a field and desktop waste rock classification guide. The guide is needed for mine geologists to classify the materials as being potentially acid forming (PAF), non-acid forming (NAF) and saline/ metalliferous NAF while in the field, such that materials of environmental concern can be stored in the appropriate storage facilities. Previously there were discrepancies in waste types classified via the geological block model, laboratory results and in-field observations, which had implications for the accuracy of mine planning, waste handling and waste management. The field classification criteria is based on insitu sampling of materials obtained from blasting charge drill pattern and analysis via handheld X-ray fluorescence (XRF) spectrometer. The verification of the handheld XRF results and the desktop waste classification guide is based on extensive static and kinetic testing on the materials.

1.0 INTRODUCTION

The McArthur River Mine (MRM) is located in the Northern Territory, approximately half way between Darwin and Mount Isa (Fig. 1). A geochemical investigation is underway to assess the potential for metal leachate release and Acid Rock Drainage (ARD) from the overburden material generated from the open pit.

Previously several aspects were identified on site that required further investigation in relation to the classification of mine waste. These include: i) there are discrepancies in the volume of non - acid forming (NAF) material and potentially acid forming (PAF) material quantified though either the 'block model', quantified by static geochemical testing or identified in the pit by the mine's pit technician classification processes; ii) the existing classification of PAF and NAF is quite broad and did not take into consideration saline NAF, neutral mine drainage (NMD; metalliferous NAF) and acid consuming (AC) materials types. There is also a small proportion of material that spontaneously combusts while in the pit or after placed on the waste rock dump. These combustible materials are referred to as highly reactive PAF materials (or, "smokers") and involve rapid oxidation of sulphides (mostly pyrite) and the combustion of carbon-rich material in the waste. The classification criteria for smokers is currently being developed and is therefore not discussed in this paper. A methodology is needed that can be applied by mine geologists to classify waste types *in situ* so that PAF, NMD, SD and particularly reactive PAF materials are restricted to the appropriate storage facilities.

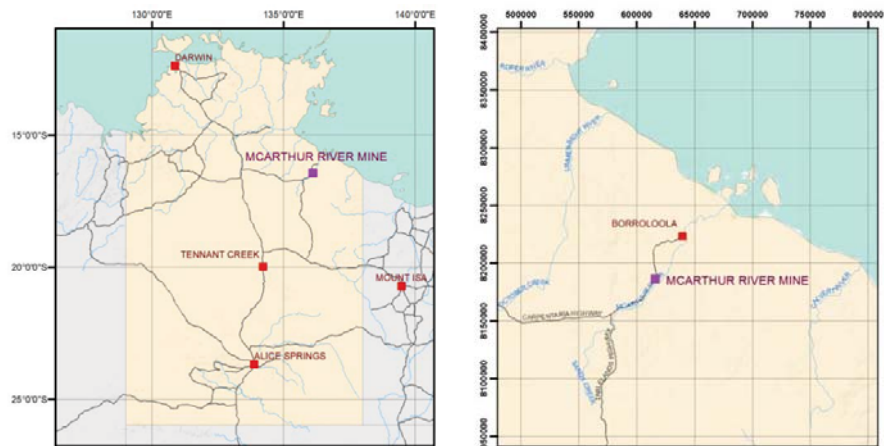


Fig. 1. McArthur River Mine site location.

1.1 Historical Waste Classification: Static testing results

The historical net acid generation (NAG) pH results and Fe assay results are currently used on site to differentiate PAF from NAF. The mineralogical analysis indicated that iron in the samples is mostly in the form of pyrite (FeS_2), with the exception of some sphalerite ($(\text{Zn}, \text{Fe})\text{S}_2$) which has iron-inclusions. Framboidal pyrite in the PAF was also commonly observed in the field. The relationship between Fe and S concentration, due to the presence of pyrite, is highlighted in Fig. 2. The positive linear relationship for the laboratory data (geochemical database) shows how the two elements are discretely related.

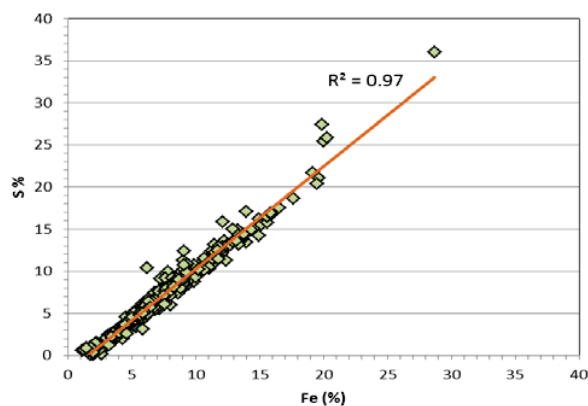


Fig. 2. Iron (%) versus Sulphur (%)

The NAG pH versus Fe concentration graph (Fig. 3) provides an indication of the Fe % cutoff for materials that are likely to be PAF based on the NAG pH. The current site Fe % cutoff is $> 7.5\%$ Fe for PAF (or $> 7.5\%$ S), based on the assumption that NAG pH < 4 is PAF.

There are several issues with the historical classification criteria, principally that the 7.5% Fe cutoff misclassifies a number of samples and the classification excludes the samples that are likely to create NMD and Saline drainage (SD).

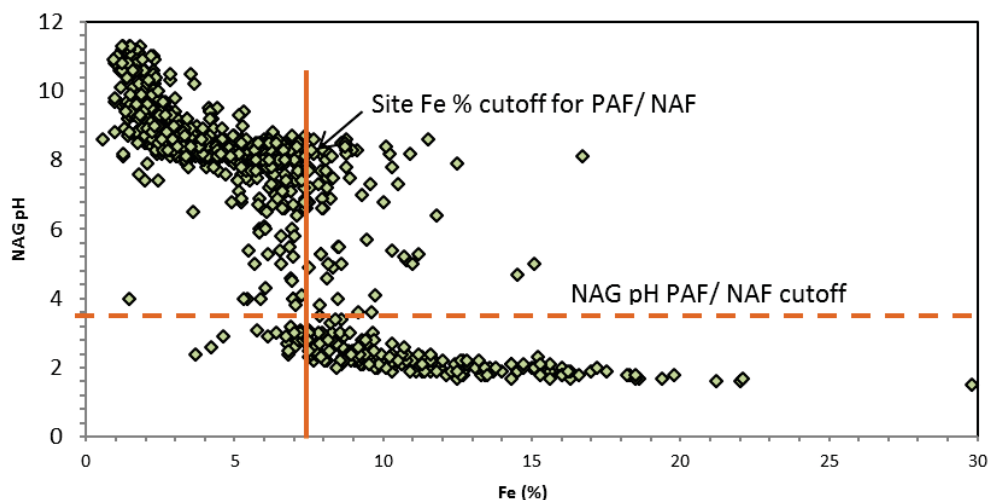


Fig. 3. NAG pH versus Fe % indicating the site Fe % cutoff for PAF (i.e. > 7.5% Fe)

2.0 METHODS

The geochemical characterisation program is based on a combination of field investigations, static laboratory testing and consideration of previous kinetic testing. The field program entailed field investigations including: pH measurements; hydrogen peroxide oxidised pH measurements (pH_{fox}); and, handheld X-ray fluorescence (XRF) testing (using a Niton XL3t GOLDD+ instrument).

Samples were obtained from the drilling residue which was placed next to the holes drilled as part of the blasting patterns (i.e. “drill mounds”). The drill mounds include the powdered material obtained from the drill hole, which, in the case of this field work, was ~ 4 - 7 meters deep into the base or pit wall. At least 8 drill mounds were sampled and analysed randomly across five different drill patterns. For each drill mound at least four handheld XRF spectra were acquired; the four spectra were taken from different areas of the drill mound along with the collection of four ~ 125g subsamples taken from the zones immediately underneath the zone where the four handheld XRF measurements were made. The subsamples were combined and sent to the MRM laboratory for total elemental analysis (via desktop XRF). Repeat scans were collected for sample statistics and to determine the variability in metal concentration within each drill mound.

The previous laboratory analysis has been conducted on ~1800 samples which were hand-picked from core to characterise the lithological units associated with the mining overburden materials. This analysis included, *inter alia*, a combination of acid base accounting (ABA), net acid generation (NAG) pH testing, total metal analysis, X-ray diffraction (XRD) and shake flask extractions (SFE).

3.0 RESULTS AND DISCUSSION

3.1 Field Results

The results from the laboratory XRF spectroscopy analysis are summarised in Fig. 4. Generally, there is a positive linear relationship when the measured metal concentrations from the field (handheld XRF) are graphed with the laboratory measured concentrations

(XRF) (Fig. 4). There is some variability in the results between elements, drill mounds and drill patterns; however, generally the field and laboratory results are in agreement.

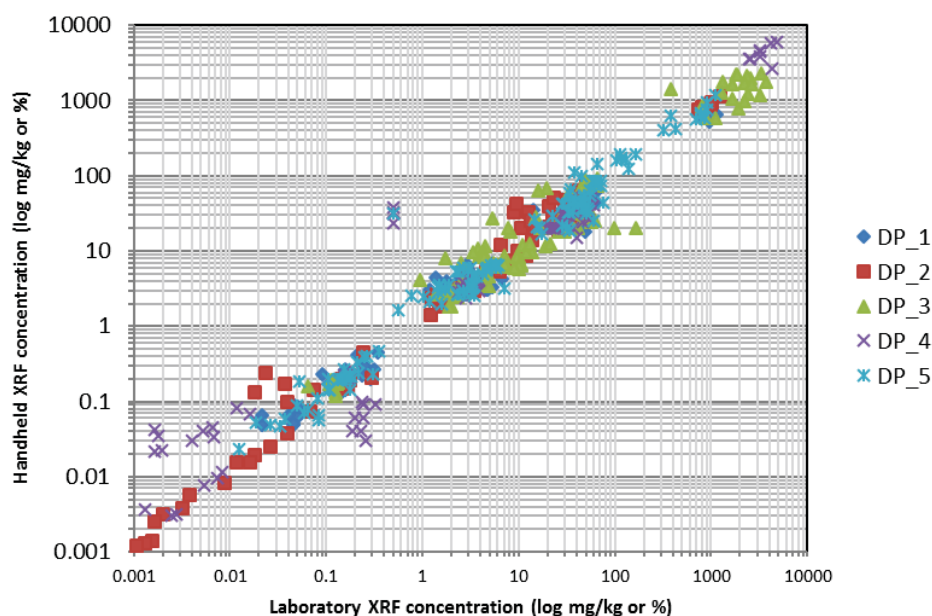


Fig. 4. Handheld XRF results versus laboratory XRF results for drill mound samples (i.e. for different samples and different elements: Cu, Fe, S As, Pb and Zn).

The weakest and strongest relationships found between handheld and laboratory XRF measurements of Zn concentration are displayed in Fig. 5. There is a high degree of scatter and corresponding poor r^2 value for the Zn concentration data for drill pattern 1, while a strong relationship was evident for drill pattern 2 results. The variability in the results for drill pattern 1 may occur due to down hole variability since the samples are obtained from the 4 – 7m interval which may cross lithological boundaries for DP1. Batching of the four samples may create some uncertainty when comparing the laboratory data for the composite sample with the discrete field measured data. Ideally all four samples should be analysed separately by the laboratory to minimise this effect of sample compositing. The second potential source of variation relates to differences in the volume of material that is measured by the field and laboratory instruments. The beam size of the handheld XRF is only small (<1 cm) which results in a very small sample size when compared to the laboratory XRF which uses ~100 g of a well-mixed and representative sample. Thirdly, particular lithological units may be more heterogeneous due to mineralogical differences and particle size variations which can create discrete ‘pockets’ of enrichment (e.g. galena and sphalerite). Finally, the sensitivity and accuracy of the two instruments; the desktop XRF is more accurate than the handheld XRF and certain elements are more susceptible to this accuracy.

Despite these limitations, the results indicate that the handheld XRF is an appropriate tool for rapidly determining metal concentration while in the field based on the strong relationship for metal concentrations in most of the drill mounds. These results may be used for initial resource definition, or, determining whether material is elevated in a particular metal of environmental concern (i.e. low versus medium versus high concentration). The rapid

determination facilitates better assessment and direction of where and how waste material should be stored and handled. Materials that are considered waste and significantly enriched in Zn (amongst other metals) may result in metalliferous drainage at this site, and need to be managed to limit the impact on the receiving environment.

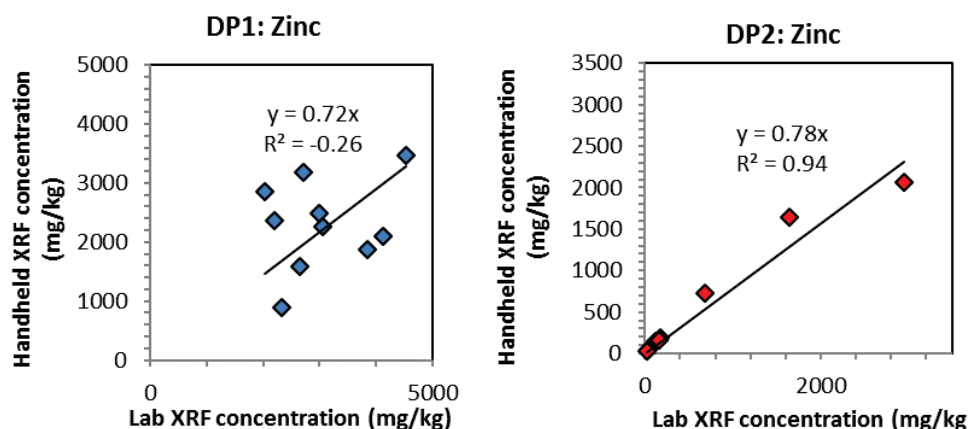


Fig. 5. Lab measured zinc concentration versus field measured zinc concentration for drill pattern 1 (DP1; worst case relationship) and drill pattern 2 (DP2; best case relationship).

The weakest and strongest linear relationships for Fe and S concentration are displayed in Fig. 6 and Fig. 7, respectively. The relationship between the results is generally very good in terms of the trend; however, the laboratory XRF results are substantially higher (~1.8 times) than the handheld XRF results. This may be a reflection of the different sensitivities of the instruments or it may be due to issues with the factory calibration of the handheld XRF instrument. The handheld XRF sulphur values are easily corrected by applying a correction factor. Iron has previously been used as an analogue for the presence of pyrite and thus acid potential. The relationship between sulphur and iron is highlighted in Fig. 8 for the field measured values.

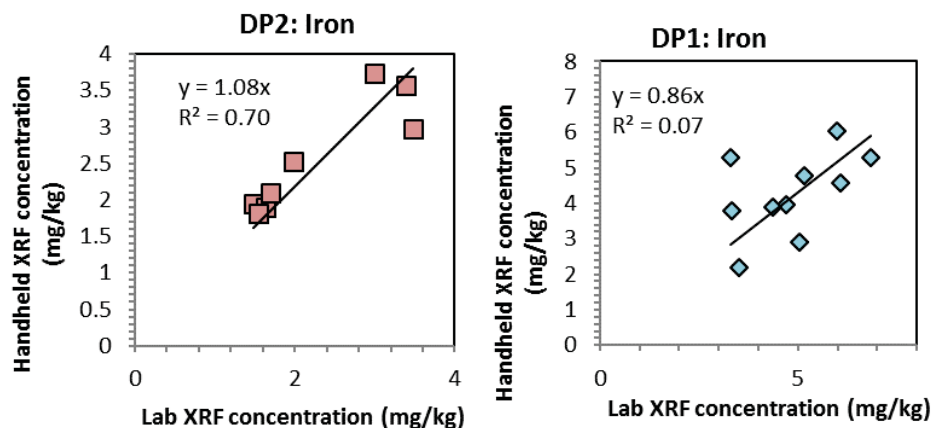


Fig. 6. Laboratory measured XRF iron concentration versus field measured iron concentration for drill pattern 2 (DP2; best relationship case) and drill pattern 5 (DP1; worst case relationship caused by one outlier).

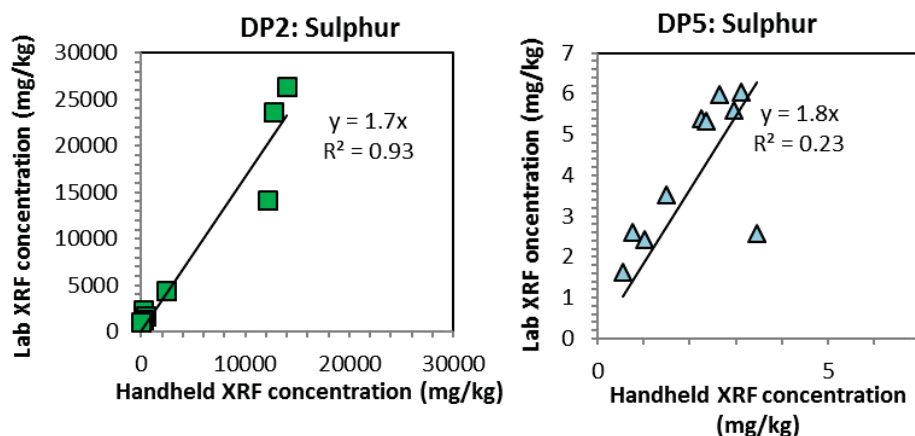


Fig. 7. Laboratory measured XRF sulphur concentration versus field measured sulphur concentration for drill pattern 2 (DP2; best relationship case) and drill pattern 5 (DP5; worst case relationship caused by one outlier).

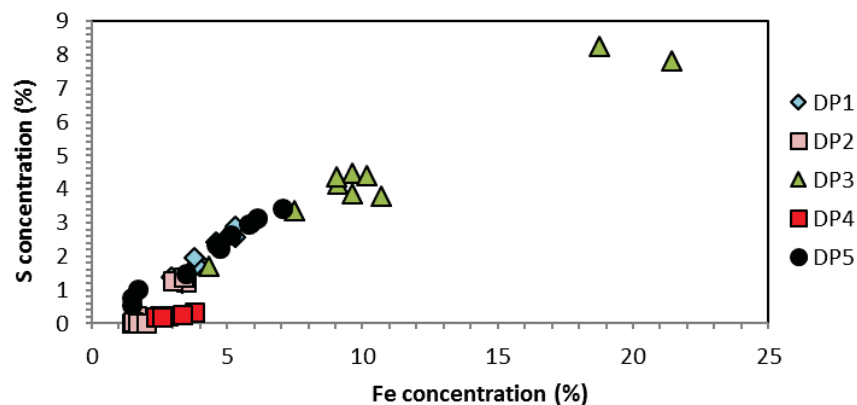


Fig. 8. Field measured (handheld XRF) sulphur (S) concentration versus field measured iron (Fe) concentration (no correction factor).

3.2 AMD Site Classification Based On Previous Static and Kinetic Testing Results

Results from overburden static testwork were used to assign an AMD risk classification to the samples and to identify correlations with available ABA data and geochemical drill-core assays. The aim is to use the interpretations of this data to devise a refined AMD classification scheme which encompasses NMD and SD which has previously being excluded.

A good correlation was observed between assigned AMD risk, geology, sulphur and iron content of the samples. These correlations were used to produce a conservative and initial high level AMD classification. The previous Fe% cutoff was shifted from 7.5% to 5% (e.g. Fig. 9; Black Bituminous Shale lithology type). This is also applicable for S % as there is strong correlation between S % and Fe %. This shift in cutoff greatly reduces the possibility of mis-classifying PAF material. In addition, materials classified as NAF based on NAG pH values, but with elevated sulphur concentrations fall within the field of NMD materials due to their potential for producing higher salinity upon oxidation (e.g. Fig. 10). Samples which

contain sulphur values > 0.5% generally result in the generation of NAG test liquor containing sulphate > in excess of 200 mg/L, which is a water quality trigger value for the site.

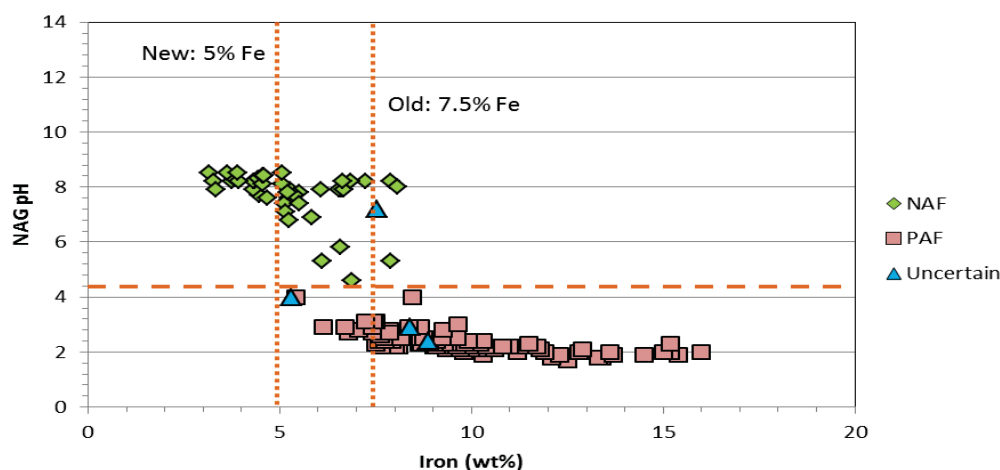


Fig. 9. Comparison of AMD classification system proposed for the BbH lithology with the classification assigned to the samples.

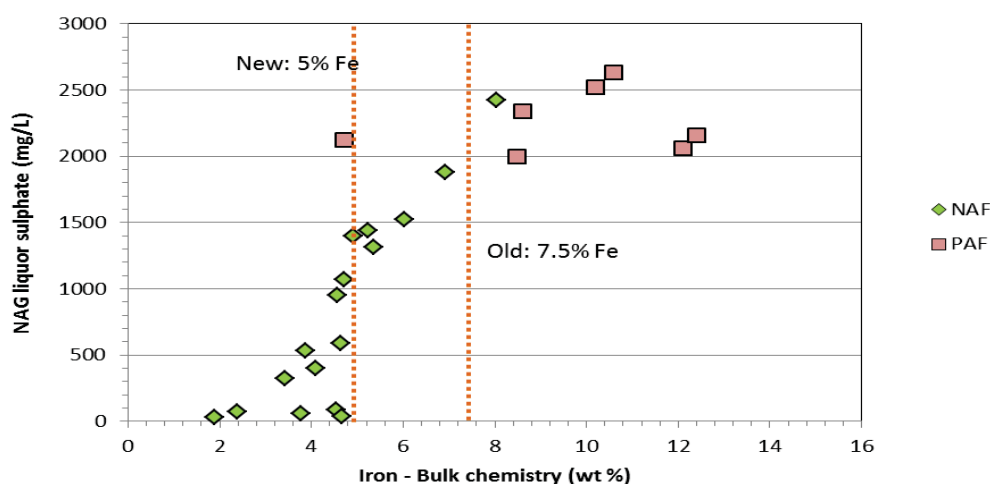


Fig. 10. NAG liquor sulphate concentration as a function of the proposed Fe cutoff grade for all lithology types.

The high level classification does not entirely take into consideration the potential for neutral metalliferous or saline drainage. The concentration of metals liberated as a result of oxidation (i.e. NAG liquors) was plotted against the corresponding total metal concentration for a particular sample. Zinc is provided as an example in Fig. 10. The site environmental water quality trigger values were compared to the NAG liquor concentrations to determine what the approximate total metal concentration of a particular sample is that may result in a particular metal exceeding leachate threshold values for site. These values are considered conservative as the NAG liquor concentrations may over-estimate oxidation.

For the classification scheme, zinc, iron, lead, silver, copper and sulphur were utilised since these parameters are included in the current site assay suite for resource definition. Due to

the variability in mineralogy of different samples, more than one metal needs to be incorporated into the classification coding to specify samples that are likely to result in production of metalliferous leachate. Based on the larger analysis suite, particularly the previous shake flask extraction (SFE), NAG extract analysis and the kinetic leaching results, the following total metal concentrations were considered to best define NAF waste most likely to produce NMD: Zinc > 500 mg/kg or Lead > 400 mg/kg or Copper > 700 mg/kg. Further kinetic testing is necessary, to increase confidence in the metal cutoff concentrations, particularly for NMD.

NAG liquor sulphate values which exceed site water quality threshold limits (i.e. 200 mg/L) are most likely to be associated with samples which contain > 1% total sulphur. Based on this, 1% has been assigned as the saline drainage cutoff value.

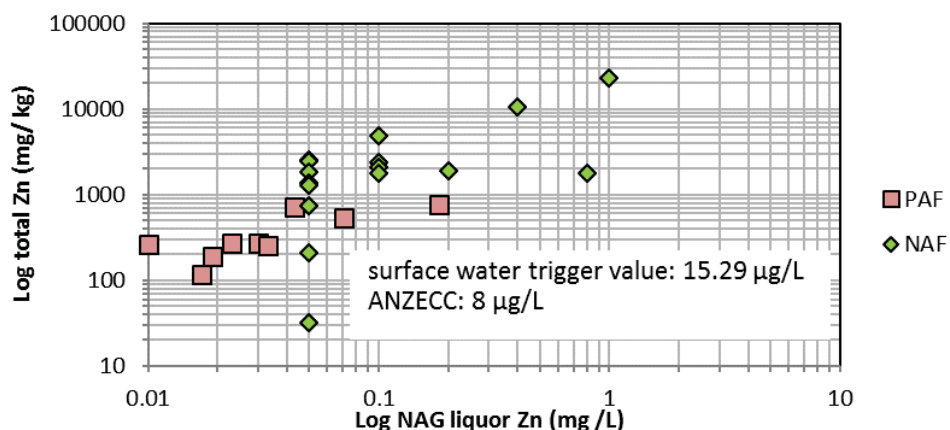


Fig. 11. Total zinc (Zn) concentration versus NAG liquor (oxidised leach) Zn concentration.

Based on the saline drainage cut off value for sulphur and the NMD cutoff values for Zn, Pb and Cu, the following waste classification criteria sufficiently classifies the NAF waste materials into NAF (non metalliferous and non-saline) metalliferous NAF (NMD) and saline NAF producing types for this site:

- NAF – S% < 0.5 OR Fe % < 5%. AND zinc ≤ 500 mg/kg AND lead ≤ 400 mg/kg AND copper ≤ 700 mg/kg.
- Saline NAF - S% ≥ 0.5 AND S% < 5%.
- Metalliferous NAF - S% < 0.5 OR Fe % < 5%. AND zinc > 500 mg/kg OR lead > 400 mg/kg OR copper > 700 mg/kg

The classification scheme has been applied to a set of drill hole assay data to illustrate the approach. Initially the waste is classified only using iron and sulphur concentration, to give the NAF, saline NAF and PAF classes (Fig. 11). Then the metal cutoff values are applied to the NAF subset of samples (e.g. Fig. 12) to identify those that are potentially metalliferous NAF (NMD); only one metal may trigger this classification. The results indicate that no samples would be classified as only NMD since As cutoff values of 200 mg/kg is consistently exceeded and therefore As will not be used in the proposed waste classification criteria until further data is available.

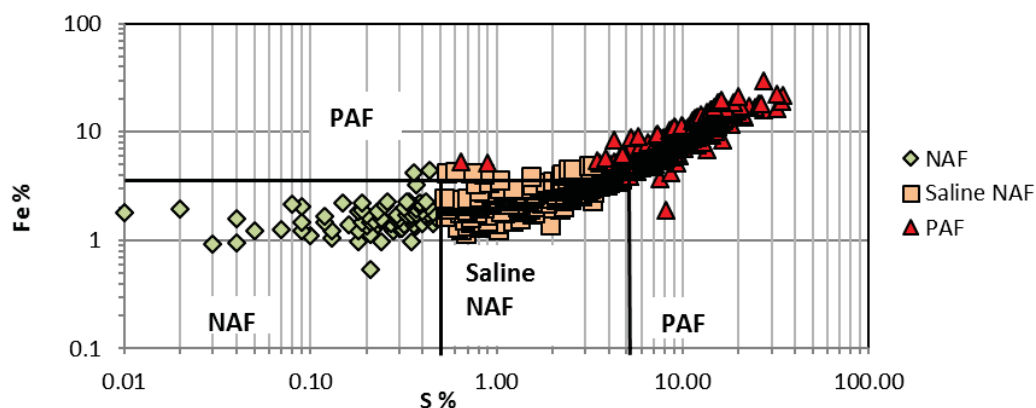


Fig. 12. Waste classification according to Fe% and S%: NAF, saline NAF and PAF.

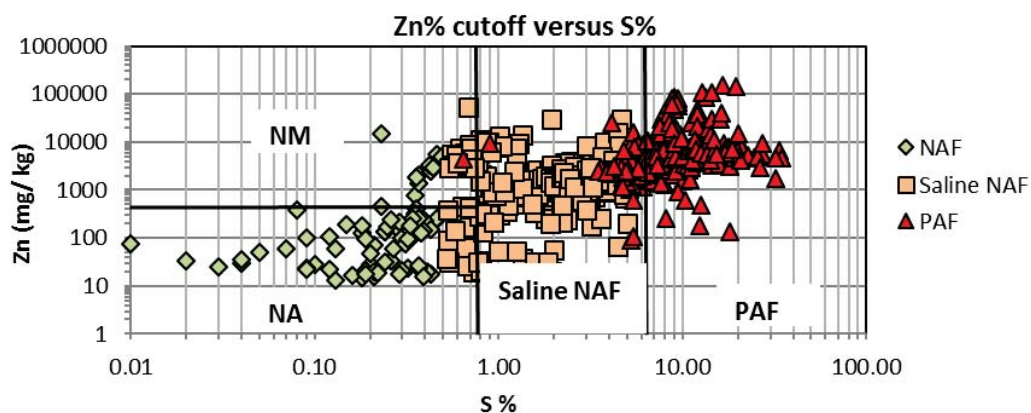
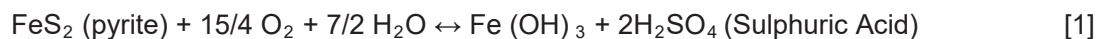


Fig. 13. Waste classification including Zn cutoff concentration (400mg/kg) to illustrate the identification of the NMD field for this metal: NAF, saline NAF and PAF.

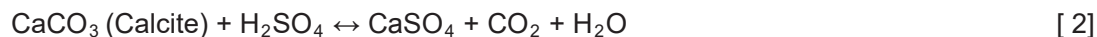
3.3 Consideration of Acid Neutralising Capacity

The classification scheme developed above provides the primary method for identifying the different classes of mine drainage that may be produced. However, in order to provide some indication of the intensity (i.e. content of metals and/or acidity) of the drainage, the ratio of neutralising capacity to acid generating potential provides a further level of refinement to guide the management of these various categories of wastes on site.

Some basic equations need to be understood in order to calculate the ANC and maximum potential acidity (MPA) for a sample with the data provided in the assay suite. The S % provides a measure of the MPA by assuming all the sulphur is in the form of pyrite and that these sulphides release acidity when oxidised. Therefore the MPA of a sample is calculated according to the stoichiometry of the below equation (Eqn. [1]). According to Eqn 1, for a sample containing 1% S as pyrite, then MPA would be equal to 30.6 kg H₂SO₄/t; MPA = 30.6*S%.



Conversely, if the carbonate portion of the sample is dominated by calcite, then the ANC of a sample can be calculated according to the below equation (Eqn. [2]).



There is a poor linear relationship between Ca % and ANC (Fig. 14), $r^2 = -0.3$. The X-ray diffraction (XRD) data indicates that only a small proportion of the samples contained Ca in the form of calcite. The trend strongly suggests significant contribution of Ca to non-carbonate minerals. Therefore, using Ca % as an analogue for total carbonate in the materials would likely lead to large errors in the ANC estimated by this approach.

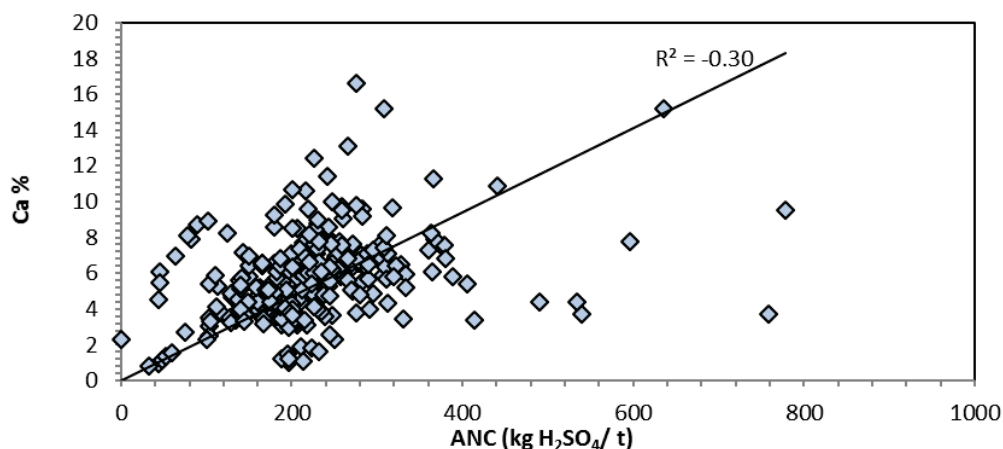


Fig. 14. ANC versus Ca %.

For MRM, the results indicate that the carbonate portion (ANC) of the materials is dominated by dolomite rather than calcite; therefore equation 1 (Eqn. [3]) may be more applicable. Fig. 15 displays the correlation between Mg % and ANC. There is a linear relationship between Mg and ANC for the bulk of the data (population 1), showing that Mg is likely largely present in the form of dolomite. However, there is a separate population of Mg (population 2; Fig. 15), that is not associated with ANC. This suggests the presence of another carbonate, non-Mg bearing mineral(s), which is likely the calcite contributing to the ANC.

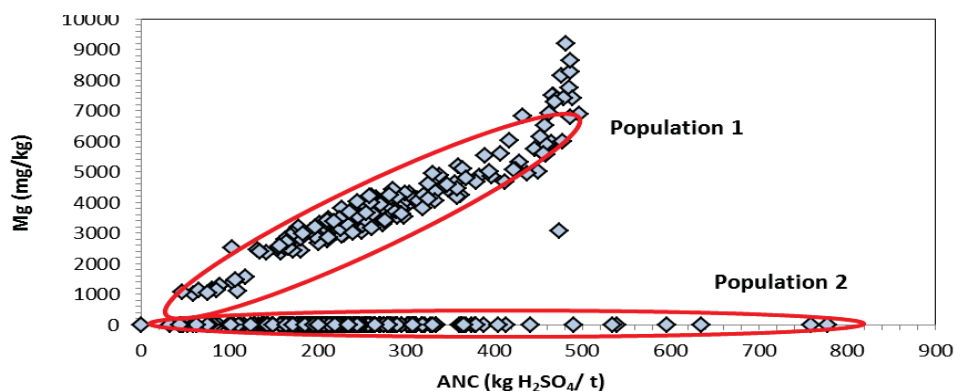


Fig. 15. ANC versus Mg %; two populations of data identified.

Although there is a good linear relationship between Mg content and ANC, the second population (due to calcite) means using Mg content as an analogue for total carbonate in the waste materials may lead to misclassification of the ANC. An alternative to using Ca and Mg concentration for determining ANC is the use of carbon data. This method relies on the assumption that all the inorganic carbon in a particular sample is present as acid neutralising carbonate minerals. For the MRM rock types, this is a reasonable approximation as the organic carbon content in these rocks is expected to be relatively low, and carbonate minerals are likely to represent the bulk of the carbon content.

Fig. 16 shows the strong linear relationship between C % and ANC. There is a small population of data which is an outlier to the general trend (Fig. 16); therefore, for this set of data there may be an overestimation of the available ANC. According to Eqn 2, for a sample containing 1% C as CaCO_3 , then ANC would be equal to 83.3 kg $\text{H}_2\text{SO}_4/\text{t}$; $\text{ANC} = 83.3 \times \text{Total C\%}$. Therefore, the Net Potential Ratio (NPR; MPA/ ANC) criterion is calculated as:

$$\text{NPR criteria} = \frac{(\text{C\%} \times 83.3)}{(\text{S\%} \times 30.6)} = \frac{\text{ANC}}{\text{MPA}} \quad [4]$$

A comparison of the laboratory measured ANC to the calculated ANC from Eqn. [4] is shown in Fig. 17. There is a good linear relationship between measured and calculated values; however, further investigations are necessary to differentiate the various carbon species (particularly organic carbon content) so that the classification criteria can be refined.

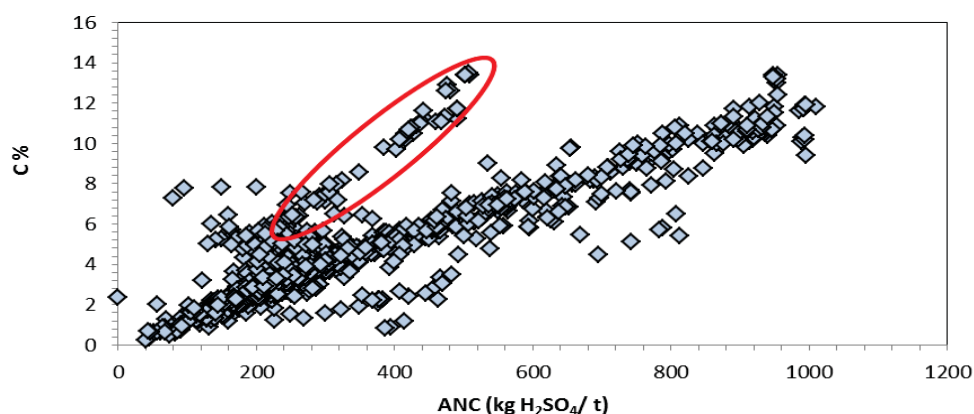


Fig. 16. ANC versus C%.

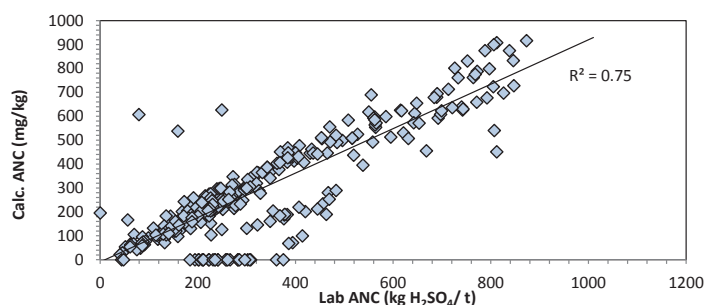


Fig. 17. Laboratory measured ANC versus calculated ANC.

3.4 Revised Classification Criteria

The revised classification framework takes into consideration the above calculations and results. The final set of criteria identifying the six categories of waste are summarised below noting that these criteria are applied sequentially (i.e. first identify high capacity NAF, then NAF- S/M etc) to the data in the block model.

Criteria 1: High capacity NAF (acid consuming)

- Inferred NPR > 2:1, AND
- Sulphur content $\leq 0.5\%$, AND
- Zinc ≤ 4000 mg/kg AND Lead ≤ 400 mg/kg AND Copper ≤ 700 mg/kg.

This material is considered environmentally low risk in terms of AMD, NMD and SD. Similarly, the material has the geochemical properties to consume acid and may possibly be used to negate the effects of AMD for certain materials.

Criteria 2: NAF – S/ M: Saline/ metalliferous NAF

- NPR > 2:1, AND
- Sulphur content > 1% but < 5%, AND/OR
- Zinc > 4000 mg/kg OR Lead > 400 mg/kg OR Copper > 700 mg/kg.

The material classified under Criteria (2) may provide a small quantity of acid consuming capacity; however, the materials are likely to generate saline or neutral metalliferous drainage over time if measures are not taken to minimise ingress of water into the stockpile waste.

Criteria 3: NAF – LC: Low Capacity NAF

- NPR > 2:1, AND
- Sulphur content < 5%, AND
- Zinc ≤ 4000 mg/kg AND Lead ≤ 400 mg/kg AND Copper ≤ 700 mg/kg.

This material is considered environmentally lower risk in terms of AMD, NMD and SD, so would not require any storage considerations or capping requirements.

Criteria 4: PAF – LC: Low capacity PAF

- NPR > 1:1, AND
- Sulphur content $\geq 0.5\%$.

This material has a low to moderate probability of generating acid mine drainage and can be associated with poor drainage water quality in terms of acidity, salinity and elevated metals.

Criteria 5: PAF – HC: High Capacity PAF

- NPR < 1:1, AND
- Sulphur content $\geq 0.5\%$

This material has a high probability of generating acid mine drainage and can be associated with poor drainage water quality in terms of acidity, salinity and elevated metals.

Criteria 6: PAF – RE: Reactive PAF

The criteria for highly reactive PAF (“smokers”) is currently being developed. The material is characterised by rapid oxidation and generation of SO₂ gas either in-pit or shortly after dumping.

4.0 CONCLUSION

A classification guide has been developed that can be applied by mine geologists to classify waste types *in situ* so that PAF, NMD, SD and particularly reactive PAF materials can be restricted to the appropriate storage facilities. The revised waste classification uses field measurements and the routine assay suite for the mine. The updated scheme also allows for the classification of saline and metalliferous waste types which were not previously included as specific considerations in the waste classification guide. This revised classification guide has been implemented on site and the initial results are showing promise for large improvements compared to the historical classification guide. Inclusion of aspects other than PAF and NAF has raised awareness of the importance of holistic mine waste management and of the environmental considerations for this material at MRM. Ongoing geochemical characterisation and field assessment will allow refinement of the classification criteria if necessary.

DESIGNER TAILINGS: IMPROVING THE MANAGEMENT OF TAILINGS THROUGH COLLABORATIVE RESEARCH

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ABSTRACT

The physicochemical properties of mine tailings contribute to weathering and soil formation processes which in turn are fundamental determinants of biological activity, vegetation establishment and ecosystem functioning. Despite this, the complex role of ore and gangue mineralogy, fine grinding, and flotation chemistry in shaping the physical and geochemical properties of tailings and their storage, are not well understood.

This project seeks to improve the management of tailings, through proactively engineering those physicochemical properties, as well as the chemistry and quantity of tailings water, during the course of ore body characterisation and mineral processing. This will be achieved through: the development of novel tailings management models; benchmarking the success of proactive initiatives; and identifying pathways for realistic re-use opportunities for valuable by-products in tailings streams.

The study involves feeding data gathered on the biogeochemical processes in tailings storage facilities and immediate receiving environments, back into the design and activities of mineral processing plants and upstream ore characterization, to produce integrated value chain models. The project aims to eliminate environmental and social legacies of future mine tailings by proactively engineering the physical, mineralogical and geochemical properties of tailings across the entire mining process, including initial ore characterization, mineral processing and rehabilitation. This research project is part of the Sustainable Minerals Institute's NextMine™ Initiative.

1.0 INTRODUCTION

Most of the environmental impacts of tailings and tailings storage facilities (TSFs) have their origins in the physical and chemical actions and additions imposed on mineral-bearing rock in process plants, which in turn are determined by orebody characteristics. However, historically most research efforts in mineral processing to maximise recovery and research on tailings rehabilitation aimed at minimising negative environmental consequences have been unconnected. The complex role of ore and gangue mineralogy, fine grinding, and flotation chemistry in shaping the physical and geochemical properties that present the ecological and rehabilitation challenges for tailings and their storage are not well understood. With increasingly lower grade ores being extracted and consequently higher

volumes of waste materials produced, it becomes imperative to have a multidisciplinary proactive approach for improved valuable mineral extraction as well as tailings management.

The physico-chemical properties of tailings such as pyrite content, pH, and concentration of deleterious elements, oxidation state and salinity all contribute to how the materials respond to weathering and to the soil formation processes that are fundamental determinants of how biological activity, through microbial communities, and ultimately vegetation establishment and ecosystem functions, will be achieved. For successful rehabilitation of tailings storage facilities, those properties, as well as the chemistry and quantity of tailings water, can potentially be engineered proactively and favourably during the course of orebody characterisation and mineral processing. For example, it might be possible to minimise, manage or eliminate the ecological effects of arsenic in tailings storage facilities by following the source mineralogy, chemical transformations in the process plant, and pathways to ecological receptors.

Examples of more integrated processes, while rare, do exist and there is potential to build on these examples. Mbamba et al (2012) proposed a two-stage process, coal and pyrite flotation, with a view to both environmental and economic benefit. This development could be incorporated into an integrated waste management scheme, to ensure both the efficient recovery of valuable coal from the ultrafines and the production of the majority component of the reduced volume of tailings that can be disposed of without environmental harm.

Montenegro et al (2013) developed a “superstructure” (a framework that included the capacity to consider both process and environmental issues), which included all possible flotation circuits to be assessed. The parameters examined included the flotation efficiency, capacity, product quality, economics and environmental impact to produce a concentrate with a high As grade and a low Cu grade, and a concentrate with a low As grade and a high Cu grade. The method provided information on the relationship between the circuit structures with indicators of efficiency, quality, and economic and environmental impact, allowing the best flotation circuit to be selected based on achieving multiple goals.

Hesketh et al (2010) proposed a systematic or integrated approach to tailings management in which conventional tailings are separated with the use of flotation into a sulfide-rich and a largely benign fraction of tailings. The technical feasibility of pyrite removal using standard copper flotation circuits is presented, whereby typical porphyry copper tailings is desulfurized by manipulation of collector dosage to reduce the acid forming fraction to 9% of the total tailings, and the sulfur content of the benign tailings to 0.2%.

Options for the disposal of the sulfide rich-phase include cemented paste backfilling or as a possible source of sulfuric acid for use in other (bio)hydrometallurgical operations such as heap leaching and metal extraction. Sulfide-lean tailings could be used to reduce infiltration and ingress of oxygen when co-disposed with potentially acid generating waste rock.

A fully integrated sulfide tailings management strategy that involves both surface and underground disposal is generally recommended for operating mines that are able to add a flotation circuit at the end of their existing treatment process. The challenge with environmental desulfurisation is to float sulfides selectively from a pulp already treated which contains various chemicals and ions that interfere with the flotation process. Kawatra and Eisele (2001), for example, have investigated the ability of several microorganisms to depress pyrite flotation under different pH conditions. This approach can reduce substantially the volume of acid generating mine tailings to manage at the surface, especially when the concentrate can be returned as cemented paste backfill underground, and it has a better

outcome for higher sulfur content (>12%) (Benzazoua et al, 2008). However, the environmental effects of backfill are not well understood.

Integrated methods are particularly attractive for places such as equatorial countries where due to seismic activity, frequent torrential rain and rugged topography, the risk of constructing tailings dams is high. The Ok Tedi copper mine in Papua New Guinea has introduced a new Tailings Pyrite Plant (TPP) to remove a sulfur-containing pyrite concentrate. The ex-mill (Cu flotation) tailings formerly discharged into the river have now become feed materials for the TPP. A publicly available report (OK Tedi Mining, 2009) shows that the majority of the pyrite mill tailings samples are non-acid forming (NAF). The average sulfur recovered from the tailings within this period December 2008 to June 2009 was 81%, which is still below the target 90% recovery. Freeport Indonesia, has adopted a strategy to mix abundant carbonate rocks from skarn lithologies with sulfidic ore prior to flotation. The results of more than 10 years column leaching of tailings from blended feed showed stable geochemistry. This is also supported by monitoring data which showed improvement of geochemical stability of tailings in disposal areas of the low lands (Rusdinar et al, 2013).

Although the conventional approaches to tailings management may look more attractive at the early stages of an operation, the benefits of a proactive integrated approach over the life of a mine, particularly in the post-closure period, may significantly exceed the initial extra investment that will be needed. There is a notable lack of integration of sustainable development principles in design of new plants. The work that has been done focuses on complying with regulations or comes at the end of the design process, when most key decisions have already been made (McLellan, 2009). Some of the challenges in environmental design are technical challenges such as the limited availability of thermodynamic information covering the full spectrum of chemical conversions, the low grade of ores and the presence of impurities, while other challenges are sociological, such as the relative conservatism within the industry for change. The environmental impacts of minerals processing often manifest themselves over a wider range of spatial and temporal scales than those of other industry chemicals.

The authors are currently involved in a research program at the Sustainable Minerals Institute, where the experimental and observational information gathered on the biogeochemical processes in tailings storage facilities and immediate receiving environments are fed back into the design and modification of the range of activities within mineral processing plants, and upstream characterization ore and gangue. This information will assist the development of new total integrated value chain models ranging from orebody characterization, treatment and rehabilitation models that can maximize the success of rehabilitation and mitigate environmental risks. The current paper presents some preliminary modelling results from this project.

2.0 METHODOLOGY

The current case study is an undeveloped Copper-Gold Project. The ore deposit contains 15 million tonnes of copper and almost 18 million ounces of gold and is hosted by intermediate porphyritic volcanic rocks affected by silica and silica-clay alteration, advanced-argillic and argillic hydrothermal alteration. The mining will involve a large scale open pit producing copper concentrates with associated gold. One of the key features is the association of copper with arsenic minerals (enargite). The presence of arsenic in ore, the associated pyrite, and the clay content are important for the environmental management of tailings and hence the focus of this case study was on these attributes.

Selected drill core samples from the high arsenic (HA), low arsenic (LA), and no arsenic (NoAs) sections of the deposit were used for grinding and flotation tests as well as environmental tests. Samples were grinded to three size fractions: 150 μm , 75 μm and 25 μm . The base case flotation was performed at the JKMRC using the following conditions: Grind time 7-11 min (depending on ore), Grind pH 10, Flotation pH 11, Flotation time 9-11 min, PAX 160-420 g/t, MIBC 15-60 g/t, Lime 500-1700 g/t. The feed and tailings samples were characterised by a Mineral Liberation Analyser, X-ray diffraction mineralogy and wet geochemical analysis for elemental concentrations and for acid base accounting. The purpose of the test works was to ascertain if there was a potential “point of intervention” along the entire value chain to see if anything can be done differently at that particular point for better tailings management outcomes. The test results provide data for the validation of the models.

The flowsheet development started with using geometallurgy data to create classes and domains of throughput-recovery in the orebody using Principal Components Analysis. Then the classes were used to select targeted samples for mineralogical analysis. In the next step, the particle size-by-composition and grain distribution data generated by MLA (and X-ray tomography in the next stage) were used as input data for an in-house Model Development Kit (MDK) to simulate comminution and flotation and the consequent tailings stream (Figure 1 and Figure 2).

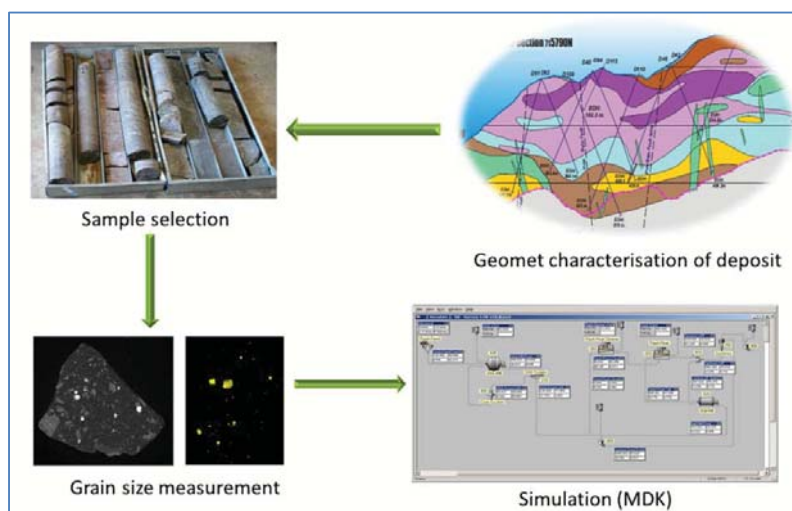


Fig. 1. General modelling approach

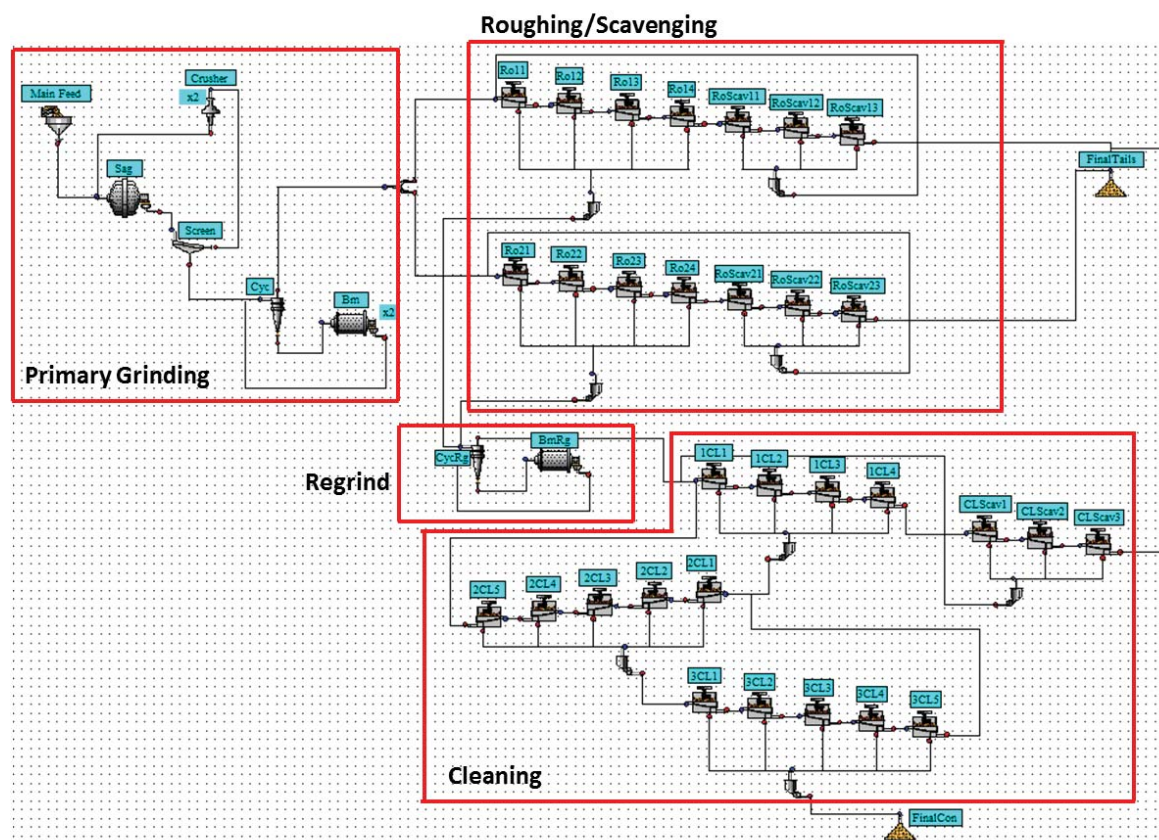


Fig. 2. Possessing flow sheet

3.0 RESULT AND DISCUSSION

3.1 Geochemical characterisation

Figure 3 shows modal mineralogy by MLA for tailings from high arsenic and low arsenic ores. The gangue mineralogy in porphyry ore is dominated by quartz followed by muscovite, feldspar, pyrophyllite, and diaspore. Skarn contains carbonate. The main sulfides are iron (pyrite), copper (chalcopyrite and bornite) and copper-arsenic (mainly enargite). Tailings from the low arsenic ore contained higher concentrations of Pyrophyllite (up to 13.5%) and Muscovite (5%) compared to those of tailings from high arsenic ore.

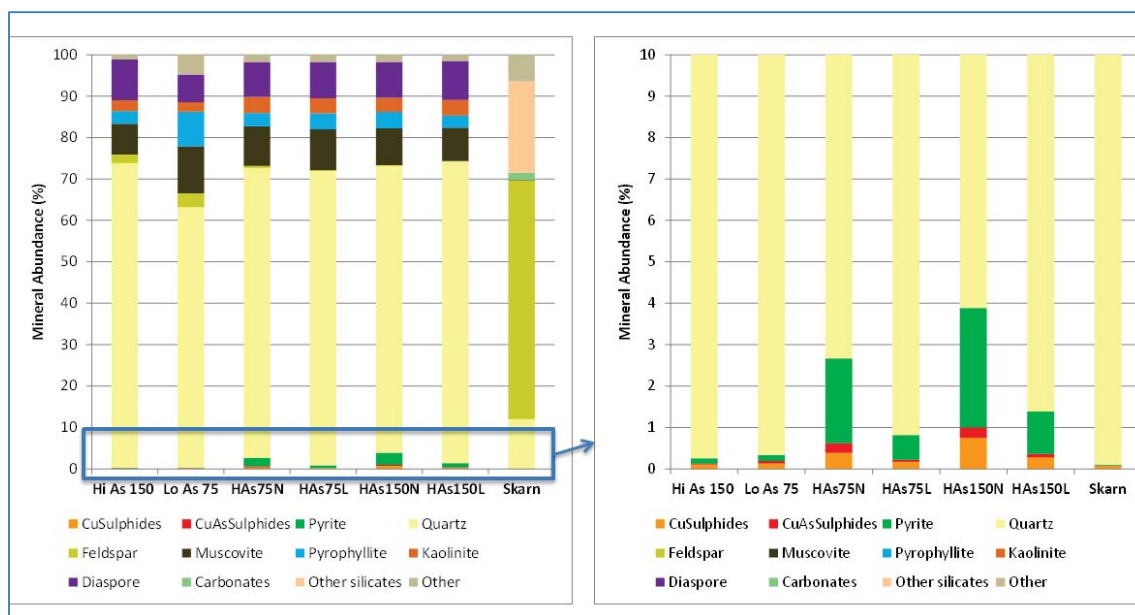


Fig. 3. Modal mineralogy of tailings by MLA (LAs=low arsenic, HAs=high arsenic, N=natural flotation conditions, L=lime added)

Figure 4 shows Net Acid production Potential (NAPP) values of feed and tailings with particle sizes from 20 μm to 150 μm . Tailings samples number 1 to 4 (T1-T4) were collected after 10 minutes and tailings 5 and 6 (T5 and T6) were collected after 10 seconds. The NAPP values of low arsenic tailings were higher (up to 23.7 kg $\text{H}_2\text{SO}_4/\text{t}$), compared to 4.4 and 2.6 kg $\text{H}_2\text{SO}_4/\text{t}$ for the high arsenic tailings. This is a reflection of pyrite content in the feed. However, high arsenic tailings collected at shorter times also have high NAPP values (e.g. T3=46.6 kg $\text{H}_2\text{SO}_4/\text{t}$).

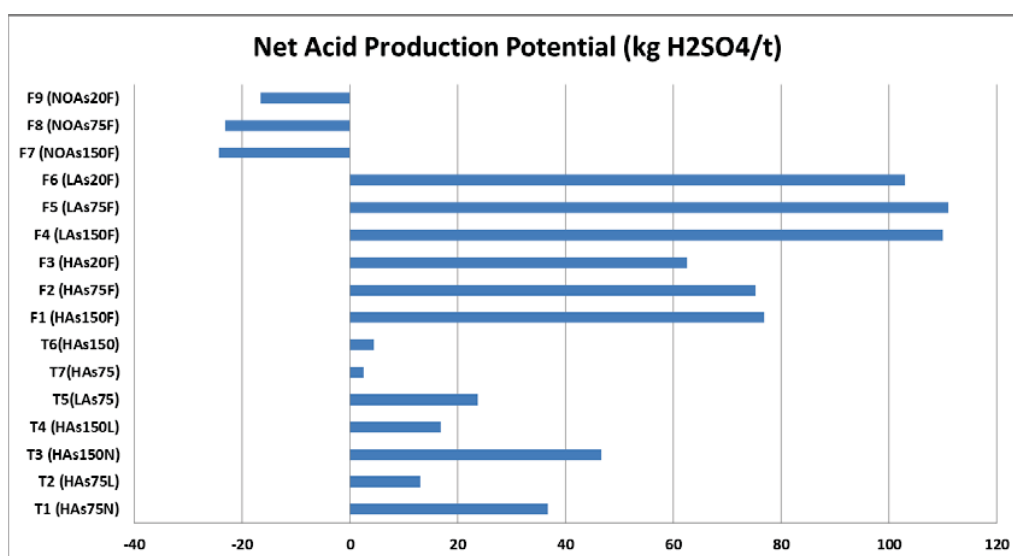


Fig. 4. Net Acid Production Potential (NAPP) values of feed and tailings. NoAs= no arsenic, LAs=low arsenic, HAs=high arsenic, N=natural flotation conditions (base case), L=lime added, F=feed, and T=tailings

Based on the MLA data the low arsenic feed is generally less liberated as it has finer grain size and a shifted grade-recovery curve. The low arsenic samples also contain higher percentages of clay minerals particularly pyrophyllite (10-30%) and higher concentrations of pyrite (2-10%).

The relationships between the three important parameters of residence time, grinding size, and flotation pH can be better investigated with MLA. With the decrease of residence time, the amount of pyrite and arsenic in tailings will increase and the pyrite grains are more liberated. With the decrease in grinding size, the amount of pyrite and arsenic will decrease in tailings, and the minerals are more liberated. Higher flotation pH will result in reduced amount of pyrite and arsenic in tailings and less liberation of minerals (Figure 5).

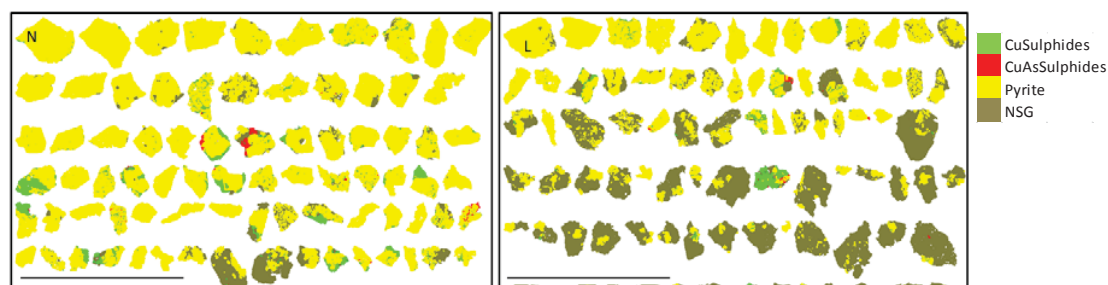


Fig. 5. MLA image of 150 µm tailings with (L) and without lime (N) addition in flotation (Scale bar=1000µm)

The high arsenic ore contains 600 ppm arsenic and the low arsenic ore <20 ppm arsenic. The base case flotation experiment resulted in the effective removal from the tailings stream of arsenic associated with enargite ore. However, it was still possible to use the differences in the composition of high and low arsenic to assess potential variation in bioavailable fraction and toxic effects (Huynh et al, 2013).

3.2 Examples of modelling scenarios

The initial modelling results show the value of using MLA mineralogical textural data for developing processing models. The liberation of copper minerals for the low arsenic feed, shows the presence of these minerals in a much wider range of particle sizes predominantly in the finer fraction, and also association of copper minerals with particles with lower density. This is consistent with the MLA results which show the dominance of clay minerals in the low arsenic ore, and also less liberated textures of copper minerals in the low arsenic ore. Once the models are completed such information can be extended to predict the reactivity of tailings particles under various feed type and grinding and flotation conditions. Two examples are presented here. In the first example (Table 1 and Table2), which is focused at the concentration of pyrite and the generation of AMD in tailings, the flotation conditions are changed and the grade and recovery of tailings is predicted compared to the base case. For the high arsenic ore, first pyrite is depressed (scenario1) and then pyrite and enargite are both depressed (scenario 2) in the process. In the case of no arsenic ore, pyrite is depressed (scenario1).

In the second example (Figure 6), the focus is on the concentration of clay minerals, particularly kaolinite, muscovite and pyrophyllite, in tailings. The base case is compared with

a modelled scenario in which clays are dispersed to 12 microns by a scrubber and are concentrated in the tailings by a flotation column.

In both examples the modelling results clearly show the expected outcomes for changed conditions. In the first example, the amount of pyrite increased in tailings and in the second example, the amount of clays increased. Similarly, other scenarios can be used to predict the particle size of tailings and concentrations of toxic elements (e.g. arsenic) in tailings.

Table 1 Modelling results for high arsenic ore

Base Case	Cu		As		Pyrite	
	Grade	Recovery	Grade	Recovery	Grade	Recovery
Final Conc	37.20	89.76	2.21	89.60	12.52	7.24
Final Tails	0.15	10.24	0.01	10.40	5.51	92.76

Scenario 1	Cu		As		Pyrite	
	Grade	Recovery	Grade	Recovery	Grade	Recovery
Final Conc	41.18	90.04	2.44	89.87	2.74	1.44
Final Tails	0.14	9.96	0.01	10.13	5.83	98.56

Scenario 2	Cu		As		Pyrite	
	Grade	Recovery	Grade	Recovery	Grade	Recovery
Final Conc	40.37	78.09	0.34	11.03	3.10	1.44
Final Tails	0.31	21.91	0.08	88.98	5.81	98.56

Table 2 Modelling results for no arsenic ore

Base Case	Cu		Pyrite	
	Grade	Recovery	Grade	Recovery
Final Conc	8.87	84.27	6.21	11.13
Final Tails	0.03	15.73	1.00	88.87

Scenario 1	Cu		Pyrite	
	Grade	Recovery	Grade	Recovery
Final Conc	9.38	82.69	0.58	0.95
Final Tails	0.04	17.31	1.11	99.05

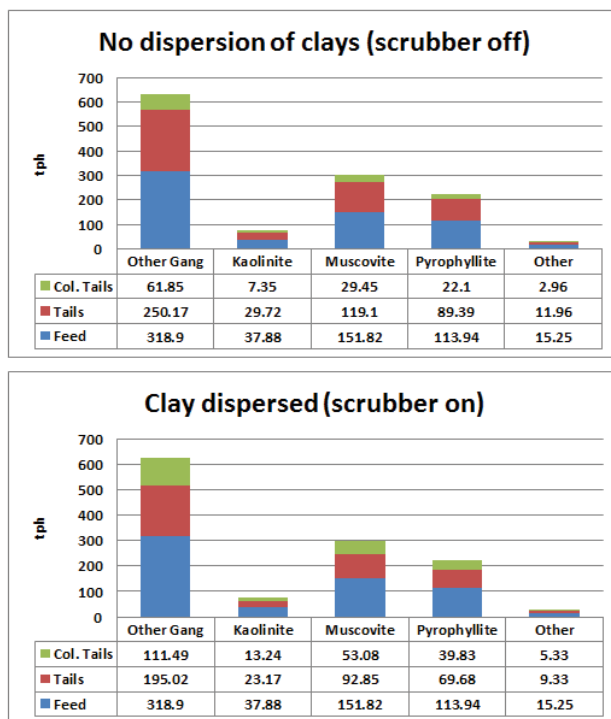


Fig. 6. The effect on throughput if clay is dispersed to finer than 12 microns

4.0 CONCLUSIONS

The reproducibility of modelling results shows the robustness of current integrated grinding and flotation models. The ultimate goal is to produce a simulator that can be used to model the whole value chain of a minerals processing operation with respect to process (specific operations) and duration (exploration to closure), with a focus on in this case managing the tailings. The modelling output showed the value of adding detail mineralogical textural data as an input parameter for simulating mineral processing. Currently other flotation conditions are being evaluated to produce different concentrations of arsenic minerals in the flotation product which in turn will change the geochemistry of tailings.

The main benefit to be gained from this project is the development of strategies in mineral processing plants that would deliver tailings with properties that are less problematic and more amenable for rehabilitation. The modelling scenarios based on the case study data provide the capacity to predict the composition of tailings and the resulting environmental management implications. For example, the type and content of clay minerals in tailings will affect the geotechnical stability and water recovery. On the other hand, certain clays, if separated in the process, can be used as impermeable layers to prevent AMD. Clay content will also influence decisions made for paste or thickened tailings and underground backfilling.

This approach will lead to an outcome that makes TSFs more cost-effective in the long-term through reduced liabilities, and at levels of contaminant security and stability that will minimise closure and residual risks and protect environmental and social values for the future.

5.0 ACKNOWLEDGEMENT

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COVER SYSTEMS.... ARE WE THERE YET?

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ABSTRACT

Cover systems are an accepted prevention/mitigation alternative for managing mine waste seepage impacted by acid and metalliferous drainage. The mining industry have advanced theory, practice, construction, and performance monitoring of cover systems to achieve this level of acceptance. However, uncertainties and misunderstandings remain as to benefits provided by cover systems. There is a tendency to ‘over-sell’ benefits, with cover systems seen as a ‘one size fits all’ closure strategy, often leading to lack of characterisation and management of reactive waste prior to/during operations, or at least to the extent that should be undertaken. The mining industry is also challenged at times to demonstrate benefits, or rather the magnitude of benefits, provided by cover systems to prevent/mitigate adverse impacts to receiving environments and/or management of flow/water quality to collection/treatment systems. There are numerous benefits of cover systems in regards to meeting closure objectives; for example, meeting land use objective, wildlife habitat, etc. However, this paper focuses on demonstrating cover system benefits in regards to developing a better understanding for the relationship between generation of acidity loads from the waste storage facility and net percolation rates offered by the cover system.

1.0 INTRODUCTION AND BACKGROUND

The mining industry is challenged by increased environmental requirements for reclamation of disturbed mining sites and mine wastes including waste rock dumps, heap leach piles, and tailing storage facilities. The increased environmental requirements may come from a more stringent regulatory requirement for the mine company’s business license and/or a general stakeholder perspective and the mine company’s social license. Management and control of acid and metalliferous drainage (AMD) produced along with almost all mining activities as a result of disturbing sulphide-bearing overburden is usually an important aspect when considering the design of cover systems within a mine closure plan.

Cover systems are an accepted prevention/mitigation strategy for managing AMD. The mining industry have advanced theory, practice, construction, and performance monitoring of cover systems to achieve this level of acceptance. Cover systems can have numerous design functions, including but not limited to isolation of waste, limiting the influx of atmospheric water and/or oxygen, controlling the upward movement of process-water constituents / oxidation products, and providing a medium for establishing sustainable vegetation (MEND 2004; 2012; INAP 2011). According to the cover system design functions, cover system design alternatives may include store-and-release or ET (evapotranspiration) cover systems, enhanced store-and-release cover systems, barrier-type cover systems, and cover systems with engineered layers (Ayres and O’Kane 2011). Selection of an appropriate cover system design requires completion of a receiving environment impacts analysis.

The mining industry is also challenged at times to demonstrate benefits, or rather the magnitude of benefits, provided by cover systems to prevent/mitigate adverse impacts to receiving environments and/or management of flow/water quality to collection/treatment systems. The challenges are mainly attributed to the following factors:

- dynamic variations of acidity loading with/without cover systems;
- difficulty with predicting the time period in which post-closure costs continue;
- a discount rate for net present cost (NPC) calculations agreed to by all stakeholders; and

- uncertainties associated with all processes (values) used for pre-closure, closure, and post-closure costs estimation.

In addition, there are differing opinions, evidence, and research with respect to benefits a cover system provides in reducing net percolation to waste rock, which increases the complexity with demonstrating benefits that a cover system provides compared to a direct AMD collection/treatment system. The same challenges also exist when demonstrating relative benefits of various cover system design alternatives. Even given these challenges, cover systems are still constructed for any number of ancillary benefits (driven by social and ecological needs).

This paper focuses on understanding of the acidity load generation (and solute concentration) and net percolation relationship, which should be facility-specific for each waste storage facility (WSF) and cover system design alternative. Mitigation of AMD is discussed based on oxygen ingress and net percolation controls. A technology transfer program and NPC evaluation method are considered useful tools for selection of an optimum cover system design alternative and demonstration of cover system benefits compared to the direct AMD collection and treatment alternative.

2.0 WATER BALANCE IN COVER SYSTEMS – BACKGROUND

The primary purpose of a cover system is to restore the disturbed mine sites into a stable, natural condition while minimising / preventing degradation of the receiving environment following mine closure. Figure 1 illustrates water balance components through the cover system above a waste rock dump. Here net percolation (NP) is defined as the net result of meteoric water infiltrating into the underlying waste rock. Hence, the water balance through cover systems is expressed as:

$$\text{Storage change} = \text{Precipitation} + \text{Run-on} - \text{Evapotranspiration} - \text{Runoff} - \text{Interflow} - \text{NP}$$

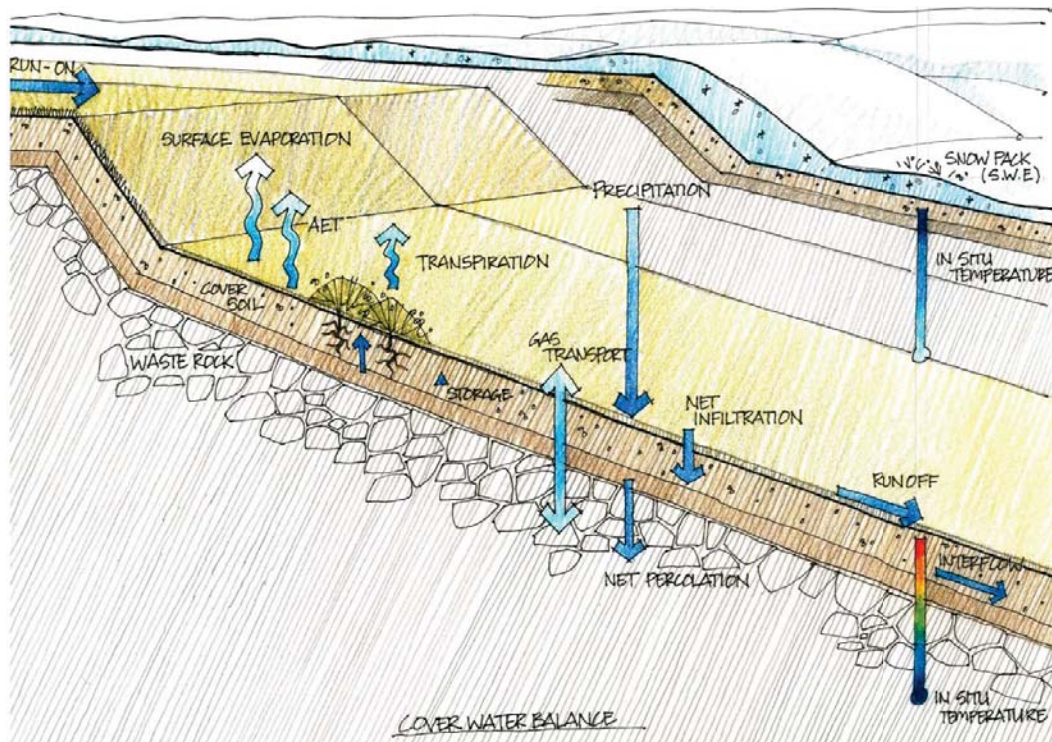


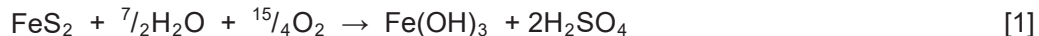
Fig. 1. Schematic of water migration processes that influence net percolation infiltrating from a cover system into the underlying waste rock (from MEND 2012).

Acidity load generated from mine wastes is considered a major risk for degradation of the receiving environment, in particular the downstream aquatic ecosystem. Acidity load is a mass concept, which is associated with both drainage volume (flow rates or NP) and acidity concentration (mg/L), where acidity load is determined by rate multiplied by concentration. From a closure perspective, the flow rate from a mine waste facility is often directly controlled by NP infiltrating from the cover systems into the underlying WSF. This flow has the ability to transport recent acidic oxidation products

(e.g., from ongoing oxygen flux through the cover) and stored oxidation products associated with secondary acidic salts (generated prior to cover emplacement) to the receiving environment. However, the acidity concentration per unit volume can be variable at times. This concept is discussed below.

3.0 THE STATE-OF-PRACTICE, CURRENT ACIDITY LOAD MODEL FOR COVER SYSTEM DESIGN

Acid generation principles due to the oxidation of sulphides (e.g. pyrite) in waste rock or tailings are well known in the mining industry. Water and oxygen are two reagents within the pyrite oxidation reaction that contribute to the formation of acidity (Eqn. [1]). In addition to being a reagent, water is also responsible for transport of the acidity (AMD) generated out of waste storage facilities. It is generally accepted that water is usually available and sufficient to satisfy pyrite oxidation within any WSF. However, solute concentration and AMD volume are directly affected by available water flushing through the waste rock. The flushing water is dependent on NP once a cover system is placed on the WSF. Therefore, NP through cover systems affects acidity loading to the receiving environment.



The mining industry is often challenged to choose between the AMD management strategies of treatment using active or passive systems, and prevention / minimisation using cover systems (often with reduced treatment requirements). Typically, two conceptual models for acidity load generation exist when options are evaluated to manage AMD (Figure 2). Model 1 assumes that acidity load increases linearly with an increase in NP, while Model 2 assumes that acidity load does not change with NP. Although not shown in Figure 2, one should be clear that even with Model 2, the acidity load must be zero when the NP rate is zero. Hence, Model 2 inherently assumes that there is a 'jump' to a high acidity load with very little net percolation.

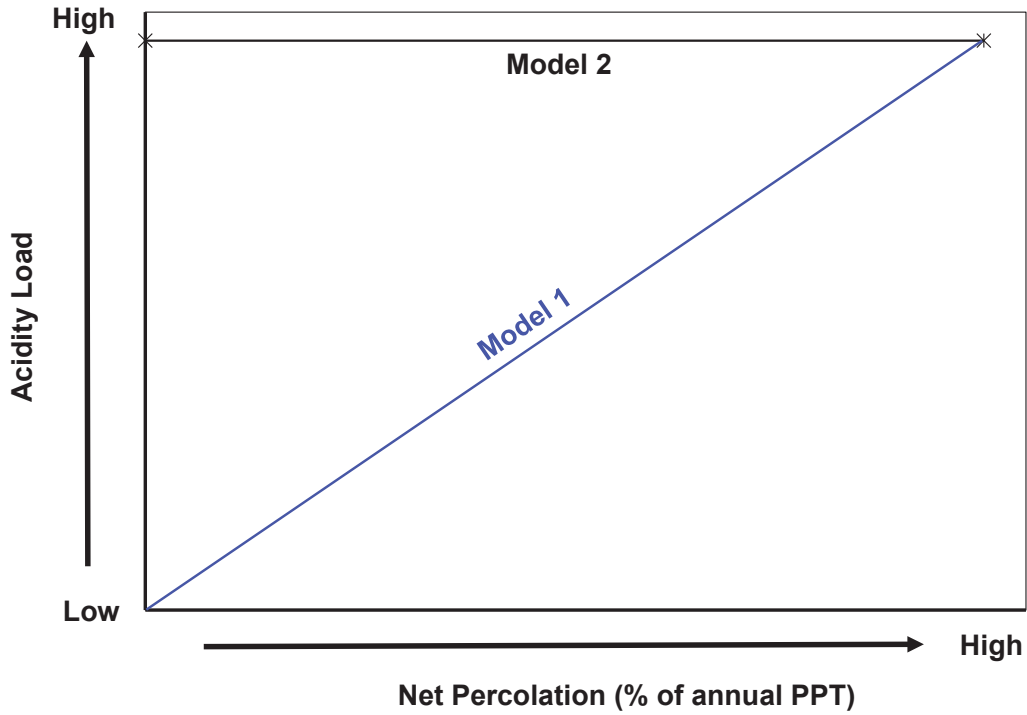


Fig. 2. Conventional conceptual models showing relationship between acidity load and net percolation (NP).

Given that: $\text{Acidity Load} = [\text{Solute Concentration}] \times \text{Flow (i.e. NP)}$, Model 1 results in a constant solute concentration, i.e. the solute concentration in AMD is independent of NP (Concentration 1 in Figure 3). Model 2 assumes that with very low NP, acidity concentration will be very high. Conversely, at high values of NP, acidity concentrations will be very low due to dilution (Concentration 2 in Figure 3).

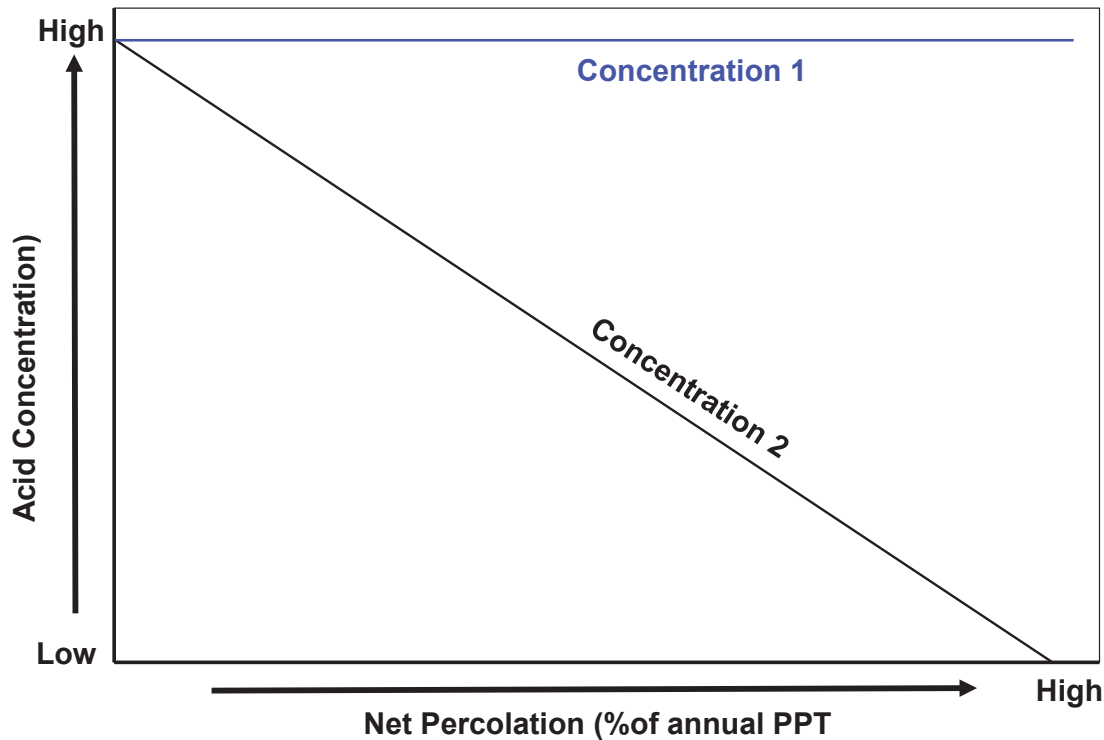


Fig. 3. Acid concentration and net percolation relationship resulted from conceptual models.

Model 1 (acidity load increases linearly with NP) can represent two situations where the WSF contains stored acidic oxidation products that are flushed out with increasing NP where:

- i) Ongoing sulfide oxidation generates secondary acidic oxidation products where the oxidation rate is a function of the oxygen flux into the WSF and that with increasing NP sufficient sulphides have oxidised to deliver an ever increasing acidity load; and
- ii) The secondary acidic oxidation products were formed prior to the construction of an oxygen excluding cover where increased flow rate transports increased stored oxidation products. This process occurs until these secondary acidic oxidation products are exhausted (at which point the model does not apply).

Where significant secondary acidic oxidation products exist the NP must be reduced to control acidity loads.

Model 2 represents a WSF where the acidity load is constant with increasing NP. In this situation the sulphide oxidation rate is low and considerably lower than the NP such that increased flow through the WSF dilutes the available acidity. There are a number of assumptions in this model, for instance, even if NP can be reduced to an infinitely small drop of water, this conceptual model still dictates the entire load available within the WSF will be transported within that drop of water. In reality, a drop of water, or small volumes of water, cannot flush over the entire WSF to produce a high acidity load.

It is likely that a combination of the two models will be present in a WSF, where Model 1 could be an early stage to Model 2. Understanding the models or any transition between the models is important for cover system design and understanding acidity loads.

4.0 UNDERSTANDING OF THE ACID LOADING AND NET PERCOLATION RELATIONSHIP

Considering both oxygen ingress and net percolation could affect acidity load generation in waste storage facilities, an alternate conceptual acidity load model is proposed (Figure 4). The alternate conceptual model has the following features:

- 1) When NP infiltrating into a WSF is low (i.e. lower than the threshold NP at Point A), the acidity load generated from the underlying WSF increases with increased NP. During this stage, acidity load is controlled by net percolation (water) limitation; this is Model 1.
- 2) When NP infiltrating into a WSF is high (i.e. greater than the threshold NP at Point B), the acidity load generation decreases with increase of NP. In this stage, acidity load is controlled, in general, by oxygen availability and/or the removal of stored oxidation products; this is Model 2.
- 3) The acidity load is mostly independent of NP when the NP value is between the two threshold points A and B. During this stage, the acidity load generation is unchanged because the increased net percolation is balanced by the decreased solute concentration due to dilution.

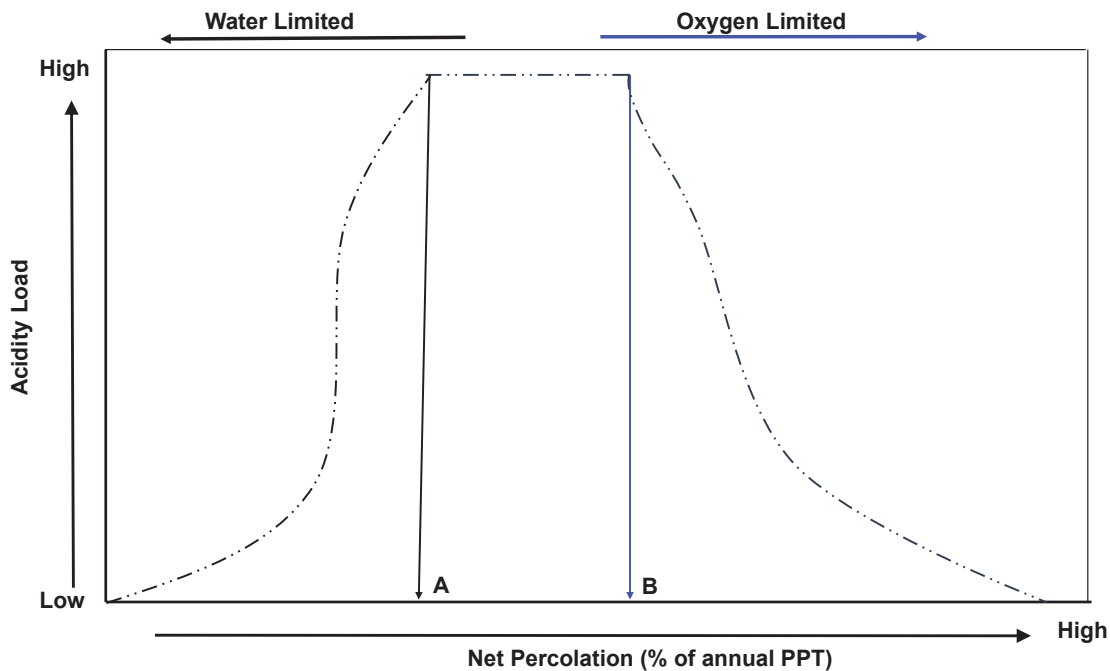


Fig. 4. The proposed alternate conceptual model showing acidity load generation and net percolation relationship.

The system can be complex and is a function of the geochemistry of the waste rock and internal pore-gas conditions. For WSFs that have significant stored acidic oxidation products, the NP required to reach Point A could be high compared to another site with rapid exclusion of sulphide bearing overburden to oxygen where the rock has a low quantity of acidic salts and low ongoing sulphide oxidation rates.

The rationale and justification for the proposed alternate conceptual model are:

- 1) Tests by Gosselin et al. (2007), and Song and Yanful (2010) indicate that the oxidation rate of pyrite attained a maximum value at a certain water content (5%-15%) when oxygen was available in sufficient quantities;

- 2) The oxygen diffusion coefficient in water is lower than that in air by approximately four orders of magnitude (Aubertin et al. 2000), which means that oxygen ingress will be greatly limited when the degree of saturation of the waste material and/or cover system is high; and
- 3) It is the author's opinion that release of AMD would be minimal (if it is not zero) as NP approaches zero.

It should be noted that the segment shapes of the curve in Figure 4 are only for illustration purposes; the model curves will differ from site to site, and from one WSF to another. The key 'message' is that as an industry, there is an opportunity to optimise design, and therefore costs, which is not being realised as a result of utilising the 'default' model (or generalised approach), as outlined in Figures 2 and 3.

In general, one could argue that the mechanisms which influence the shape of the alternate conceptual model are understood, or at least reasonably well understood. Clearly, this is a broad generalisation and research is required for identifying additional mechanisms, and improving understanding for mechanisms that are known. However, taking advantage of the opportunity discussed above requires a site specific approach to developing an understanding for external and internal controls on the known mechanisms, which can affect the shape of the model curve (Figure 4). For example, the location of the WSF as well as the site-specific climatic conditions of the site are key external controls, which in reality cannot be 'engineered'; at least not to any substantial effect. Internal controls can include, for example:

- Physical characteristics of the waste material;
- Geochemical characteristics of the waste material (including secondary acidic oxidation products);
- Geometry of the waste storage facility; and
- Internal structure and configuration of the waste storage facility.

The site-specific external and internal controls, will strongly influence the mechanisms that in turn influence sulphide oxidation, acidity generation, and acidity load emanating from the WSF. In general, these external and internal controls will influence the physical, chemical, and biological conditions within the WSF. For example, the physical structure and geometry of a waste rock dump (WRD), as well as the physical and geochemical characteristics of the waste rock strongly influence gas, temperature, and moisture conditions; all three of which influence sulphide oxidation rates. Or, lower net percolation rates can reduce seepage rates within a WRD, and thus potentially result in saturation indices being reached. Mineral solubility is dependent on conditions within the WRD, particularly pH, which may change with time as the geochemical regime in the waste rock evolves. Stockwell et al. (2006) found that, with the exception of gypsum and jarosite, minerals were more soluble with decreasing pH within a deconstructed WRD. At higher pH levels, iron concentrations in seepage were controlled by precipitation of iron oxyhydroxides.

Solute loading from WRDs is also influenced by biological conditions and these must be addressed otherwise results may be biased (Dockrey, 2010). This is especially true as ferrous iron oxidation has been identified as a rate determining step in the production of oxidation products. This process is mediated by microorganisms such as *Thiobacillus ferrooxidans* and other chemoautotrophs that catalyse oxidation of ferrous to ferric iron (Song, 2010). The ferric iron produced subsequently accelerates the oxidation of sulphide minerals under acidic pH conditions (Dockrey, 2010).

Dockrey (2010) found correlations between weathering rates during the wet season and the number of microorganisms within a trial WRD. This indicates that the amount of substrate available for colonization is an important control on microbial growth and points to thermodynamic influences on their growth. An exponential relationship was found between microbial population size and sulphate loadings, and a linear relationship between sulphide mineral content and

loadings on an annual basis. This was attributed to larger numbers of microorganisms being able to use energy from sulphide oxidation more efficiently.

In environments with neutral pH conditions, neutrophilic sulphur-oxidising bacteria will be the most prevalent in microbial communities, although acidophilic iron- and sulphur-oxidising bacteria can also be present in small numbers (Dockrey, 2010). Sulphur-oxidising species oxidise intermediate compounds in order to prevent armouring of sulphide minerals, and weathering rates can be increased by 30% to 100% in this way. Iron-oxidising species have been shown to increase acid generation by 10% to 100%, making the presence of microbial activity significant for these processes.

5.0 CONTROLLING ACIDITY LOAD GENERATION

Although the specific objectives of cover systems may differ from site to site, two objectives are usually evaluated in a cover system design; namely, 1) limiting oxygen ingress into the WSF, hence reducing the potential for ongoing long-term production of acidity, and 2) reducing NP into the WSF, hence reducing the potential drainage volume from the WSF and the flushing of stored oxidation products (recent and pre-cover).

5.1 Controls of Oxygen Ingress

The principal mechanisms contributing to airflow and gas transport in waste piles (i.e. gaseous oxygen in this context) include diffusion, convection due to thermal gradients, and advection due to wind gradients or barometric pumping; diffusion might be the major gaseous oxygen transport mechanism into tailings facilities. A detailed discussion of gas transport mechanisms is available in GARD (INAP 2011) and Wels et al. (2003). Diffusive processes are typically limited to the near-surface zone, while advection and barometric pumping have the potential to transmit gaseous oxygen (O_2) to greater depths, and greater lateral distances, within the reactive mine waste rock. The reactivity of the waste material in combination with the air permeability has a strong influence on the magnitude of thermally induced convection. Conversely, the magnitude of barometric pumping is controlled by the waste material porosity and changes in ambient air pressure.

Climatic conditions in which the WSFs are located and the geometry of the WSFs can affect oxygen ingress. Advective transport of oxygen will be higher in climates that are generally moisture limited due to increased air permeability. Climate also dictates temperature controls on advective mechanisms by where large changes in ambient temperatures establish pressure gradients leading to large advective fluxes, provided sufficient air permeability exists. WRDs pose the greatest susceptibility constraining oxygen ingress, largely due their constructed geometry. Pile geometry, namely sloping of the waste rock piles can induce convection regardless of the specific distribution or permeability of the materials contained within the pile (Lefebvre et al. 2001). Generally, the more permeable the waste material, and the greater the height-to-width ratio of the waste facility, the greater is the potential for advective air movement. Controls on convective and advective transport are strongly influenced by WRD construction. For example, constructing WRDs in lifts allows the opportunities of highly trafficked surfaces, where compaction introduces a horizontal layer with a textural discontinuity; in this case compacted layers with a lower permeability. Segregation of waste material also serves to create textural discontinuity within a WRD, and the question is whether this typical characteristic of WRDs can be taken advantage of, and/or enhanced, to help manage and/or control oxygen availability within a WRD.

Given the appropriate climate and *in situ* conditions, a layer of waste rock within the WRD, and/or within a cover system, that is saturated (or near-saturated) represents a potential opportunity to limit the ingress of oxygen as a result of diffusion. Table 1 presents calculated oxygen flux and acidity load changes for a lower permeability layer (higher moisture retention characteristics) when the degree of saturation in a layer is changed (whether it be part of a cover system or a layer within the WRD). In the calculation, oxygen concentration in the pore-space within the material below the

saturated layer is assumed to be 3%; the thickness of the layer is 2 m, and the area over which the calculation is applied is ~150 ha. Acidity is calculated assuming all oxygen reacts with pyrite as per Eqn. [1].

According to Table 1, the acidity load is ~350 kg acid/day at an average degree of saturation of 85% in the engineered layer, while the acidity load is ~100 kg acid/day at 90% saturation; implying that acidity load can be significantly decreased by a small increase in the degree of saturation.

Table 1. Changes of oxygen influx and acidity load against selected lower permeability layer saturation conditions.

Average LPL Saturation (%)	Oxygen Influx		Acidity Load	
	(g/m ² /yr)	(mol O ₂)/m ² /yr	mol H ₂ SO ₄ /m ² /yr	(t CaCO ₃ acid/yr)
75	230	7.2	3.8	580
80	116	3.6	1.9	290
85	50	1.6	0.83	130
90	14	0.45	0.24	36
95	1.8	0.055	0.029	4.5
100	0.1	0.002	0.001	0.2

5.2 Controls on Net Percolation

Similar to oxygen controls, understanding of both climatic conditions and WSF geometry are required to determine NP rates. Within different climate regimes there exists a continuum of performance for a cover system in regards to net percolation rates. Indeed, even within a single climatic regime, the same continuum may exist as a result of the influence of differing abilities of a cover system to evapotranspire and/or cause water to run off the landform.

Cover systems that act to reduce NP also serve to attenuate hydrograph peak discharge. By utilising water for internal cover processes such as transpiration, evaporation, and storage the burden on water treatment facilities may be lessened following large magnitude storm events. Although large magnitude storm events may still contribute to NP and subsequent treatment requirements due to remobilisation of 'slug events', the overall volume of water to be treated will be decreased. More specifically, less storage capacity and less water treatment infrastructure may be required as a result of attenuating peak discharge events by utilising well designed cover systems.

6.0 DEVELOPMENT OF SPECIFIC ACIDITY LOAD CURVE

It is clear that almost every WSF, with and/or without a cover system, possesses a unique acidity load generation and net percolation relationship (model curve). Hence, there is an opportunity to optimise the design of the cover system such that the 'true' benefits, in regards to predicting acidity loading from the WSF, can be better understood. Achieving this objective, would then allow for an appropriate evaluation of closure alternatives from a cost-benefit perspective. Questions that arise include:

- 1) How to develop this unique model curve so that the cover system design can be optimised and cost-benefits of all cover design alternatives can be demonstrated;
- 2) Is it cost effective to develop this unique model curve in terms of the studies and/or research that are required; and
- 3) What is the best way to ensure the model curve developed is applicable to a field (commercial scale) cover system design?

To answer these questions, a site-specific technology evaluation and transfer program is required. In general, a technology transfer program should identify knowledge gaps, conduct research and/or studies at various scales, and finally implement the technology at a commercial scale. The program obviously involves the mine owner, but can also involve researchers from universities,

consultants, and sometimes a technical advisory panel consisting of multi-disciplinary experts. A stage by stage method is used in all aspects including problem identification and optimisation, technology screening, research and development planning, and implementation. Furthermore, the technology transfer program helps the mine site develop sufficient information and timely interpretation of research results that allow operators to make timely and informed management decisions. O’Kane et al. (2014) discuss the above noted technology transfer approach in greater detail.

The technology transfer program provides a tool to assist mine sites in approaching the development of a specific acidity load curve for the specific WSF and cover system design alternatives, with minimum or affordable investment, which is balanced against developing the optimum cover system design alternative based on the specific acidity load generation curve.

It should never be forgotten that a cover system is the final stage in the construction of an engineered landform. A well-constructed WSF in combination with an appropriate cover system can have significant costs savings, both components leading to reduced long term costs associated with the management of acidity.

7.0 NET PRESENT COST EVALUATION FOR COVER SYSTEM DESIGN ALTERNATIVES

From an economic perspective, the potential benefits of a cover system for prevention / mitigation of AMD versus collection and treatment can be demonstrated using a net present cost (NPC) evaluation. The costs associated with the mine closure plan, including costs for pre-closure, closure, and post-closure, should all be included in conducting the NPC evaluation. The method seems straight forward and simple; however, uncertainties embedded in each specific closure activity, and the assumptions required in applying the technology, can make NPC comparisons challenging. In addition, something as simple as the discount rate used for an NPC evaluation, a value that can be the subject of substantial debate amongst stakeholders, can substantially influence the NPC used to compare alternatives. A generic case study is summarised in Tables 2 and 3 to illustrate this relatively simplistic, but key aspect, when conducting an NPC evaluation.

The generic case study includes AMD management options for closure of a WRD; namely, direct water collection and treatment (Base Case), a ‘simple’ soil cover, a ‘complex’ soil cover, and a cover system that includes a geomembrane. In each cover option, any AMD still needs to be treated before reaching the receiving environment; however, the treatment scale is decreased depending on cover system options. This reduction is based on an assumption in regards to the influence of the cover system on net percolation and oxygen ingress, and therefore the influence of this on internal conditions / controls in the WRD; hence, sulphide oxidation and acidity loading. The process calls for one to assume that these controls and key mechanisms (i.e. controls on the key physical, chemical, and biological mechanisms present at the site) are ‘valid’, so as to evaluate, from an NPC perspective, whether there is merit in evaluating to a greater extent the technical validity of the control and mechanism.

To compare the combined closure and post-closure costs, it is necessary to calculate the NPC for each option. Those calculations require an assumed discount rate and require all costs to be located in time. It was assumed that the WRD would be closed in Year 20, and that all closure costs would be borne in that year. The pre-closure costs were assumed to occur in Year 10 and post-closure costs in Year 30, although post-closure costs would commence following the dump closure and, for most cases, continue in perpetuity. The discount rate used in the NPC calculations is 6% in Table 2 and 2% in Table 3, both of which are assumed to include a 2% inflation rate.

Table 2. Summary of net present costs for selected technologies (values in \$ Million) with a discount rate of 6%

	Direct Water Treatment (Base Case)	Simple Soil Cover	Complex Soil Cover	Geomembrane Cover
Pre-Closure Cost (Year 10)	-	\$15	\$15	\$25
Closure Cost (Year 20)	\$100	\$110	\$110	\$160
Post-Closure Cost (Year 30)	\$150	\$135	\$115	\$40
Discounted Total Cost (Year 20)	\$185	\$215	\$200	\$230
Δ from Base Case		\$30	\$15	\$45

Table 3. Summary of net present costs for selected technologies (values in \$ Million) with a discount rate of 2%

	Direct Water Treatment (Base Case)	Simple Soil Cover	Complex Soil Cover	Geomembrane Cover
Pre-Closure Cost (Year 10)	-	\$15	\$15	\$25
Closure Cost (Year 20)	\$100	\$110	\$110	\$160
Post-Closure Cost (Year 30)	\$150	\$135	\$115	\$40
Discounted Total Cost (Year 20)	\$225	\$240	\$225	\$225
Δ from Base Case		\$15	\$0	\$0

Table 2 indicates that at a discount rate of 6% none of the cover systems appear to “pay for themselves” in terms of reduced water treatment costs, compared to the base case. Of course, the cost calculations are very dependent on assumptions about the cover designs, their effectiveness in reducing infiltration, the assumed influence on acid loading reduction, and specifics of the unit cost assumptions. Table 2 also shows that the complex soil cover option costs less compared to simple soil cover and geomembrane cover options, and the geomembrane cover costs \$45M more than the base case. However, when the discount rate was changed to 2% (Table 3), both the complex soil cover and geomembrane cover show promise to “pay for themselves” compared to the base case (direct water treatment).

The NPC calculations in Table 2 and Table 3 illustrate that the discount rate can significantly change the decision on whether an alternative merits further study, and hence the selection of a certain cover system design option. In turn, the assumptions that are inherent to the internal conditions of the WRD as a result of the changes in oxygen ingress and net percolation offered by the different cover systems would also merit further study, depending on the discount rate applied. Nevertheless, NPC comparisons can provide clear messages and help focus whether a technology is worth pursuing in terms of understanding its application to a particular mine site.

One aspect, often not considered by NPC, is allocation of risk for the treatment after closure option. Treatment in perpetuity presents a risk to stakeholders that the treatment will continue until the effects of AMD are appropriate for the receiving environment. This can be hundreds of years plus in some cases and is reliant on company commitment and solvency. Inclusion of this risk into NPC can be difficult, although adequate bonding by regulators can enable better NPC analysis of options, with the gap between prevention / mitigation and treatment closing. In short, NPC calculation are a useful tool to determine whether there is economic merit in developing a site-specific and/or WRF specific acid loading curve for closure.

8.0 OTHER BENEFITS OF COVER SYSTEMS

Optimising costs is an important aspect for consideration in the selection of cover system design alternatives for managing AMD compared to other AMD management options. However, cost saving should not be considered as the only factor when choosing a cover system to manage AMD. Cover systems also offer significant additional benefits, including:

- Growth medium for vegetation and bio-diversity;
- Wildlife habitat;
- Land reclamation;
- Ecosystem restoration;
- Aesthetic landscapes;
- Potentially additional safety to humans and wildlife; and
- Reduced long term risk

A costing analysis may not show that there is this additional benefit for constructing a certain cover system for managing net percolation rates and thus acid loading. However, the above noted additional considerations must also be addressed when determining a cover system design for closure.

9.0 SUMMARY

Cover system technology has advanced significantly in the past 10 to 20 years. There is a greater understanding for the need to develop site-specific holistic designs influenced by site climate conditions, hydrogeology, geochemistry, materials, etc. While advancements are still required in terms of design methodology, construction quality control, and performance monitoring, the mining industry is gaining confidence with its ability to properly design and implement cover systems.

This paper attempts to put into perspective the need to now take the application of cover systems to another level in terms of developing a realistic 'model' of the benefits a cover system offers in regards to acid loading, when net percolation and oxygen ingress rates into a WSF are altered by the cover system. Current models assume generic relationships between net percolation and acid loading. This paper illustrates that a unique acidity load generation and net percolation relationship must exist for each waste storage facility and cover system design alternative, and more importantly that acknowledging this provides an opportunity to optimise AMD management in a cost effective manner.

This paper also offers a framework in which to evaluate whether there is merit in determining the unique acid load generation and net percolation relationship, or more specifically, whether there is merit in research and study on specific aspects of the relationship, and if so on which aspect to focus limited resources. The author acknowledges that developing a site specific, or facility specific, acid load against net percolation curve requires additional investment. However, it must also be acknowledged that with this relationship 'in hand', or at least developed for a site with a greater degree of confidence, clear and concise information can then be used to make cost effective operational and closure management decisions.

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CHARACTERISATION OF 26 YEAR OLD WASTE ROCK STOCKPILES AT THE DETOUR LAKE PROJECT

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ABSTRACT

Existing waste rock stockpiles at the Detour Lake Mine (DLM) in Canada present a unique opportunity to characterise the geochemical and hydrological behaviour of decommissioned waste rock under field conditions. The stockpiles have a 16 year operating and 10 year post-closure history, which can be likened to large kinetic test plots. This research program is in progress, and the present paper details a forensic excavation study on two stockpiles to investigate structure, matric suction, and construction techniques in situ.

A total of 100 physical samples were collected over three sampling campaigns to characterise the waste rock material. Subsequent laboratory analyses include particle size distribution, water content, paste pH, Munsell soil colour, saturated hydraulic conductivity, and soil-water characteristic curves, which are ongoing. The observations presented in this paper discuss the field testing program, which quantified the compositional and structural characteristics of two stockpiles. To characterise large scale waste rock, photographs were taken for digital image processing (DIP) analyses to detect particles >75 mm. The DIP analysis was successful in quantifying material up to 0.5 m. Structural features within the stockpiles and segregation of waste rock were observed. No significant thermal activity was observed, and neutral paste pH results suggest that no significant ARD production has yet occurred.

1.0 INTRODUCTION

The Detour Lake Mine (DLM) is a gold deposit property operated by Detour Gold Corporation. The site is located approximately 300 km northeast of Timmins, Ontario, Canada. The climate is moist mid-latitude continental with an average annual temperature of 0.6°C, with summer highs averaging 24°C and winter lows of approximately -25°C. The average annual precipitation is 880 mm, with average annual rain and snow of 583 mm and 297 cm, respectfully (Environment Canada 2013). The deposit was originally mined from 1983 to 1999 as a combined open pit and underground operation. The mine was closed in the early 2000's (Robertson et al. 2012). In 2006, the DLM was reassessed and has subsequently been redeveloped. Production for the new open pit mine commenced in early 2013.

Existing waste rock piles on site from the early operations presented a valuable opportunity to evaluate the behaviour of waste rock under site conditions. The piles represent full kinetic test plots. The waste rock piles are currently under study to assess and understand the potential for production of Acid Rock Drainage (ARD). A joint investigation by the University of Waterloo and University of Alberta was initiated in 2011 and is currently underway to evaluate the physical and geochemical properties of the existing waste rock. The research program involves the assessment of both physical structure and geotechnical properties as

well as geochemical and gas transport mechanics. The objectives of this study program are to:

- assess the influence of pile structure and configuration on oxygen diffusion, oxidation, and movement of water within pile and its relation to water chemistry;
- characterise the hydrologic characteristics and behaviour of the waste rock;
- understand the geochemistry and pathways for both acid- generating and neutralizing components as well as their evolution with time; and;
- assess the potential for scale-up between laboratory, the existing waste rock and future operational waste rock piles to assist in future design and closure.

This research program is a work in progress, and the paper will primarily address the early milestones of the program including the physical characterisation of the waste rock and some preliminary results.

2.0 WASTE ROCK AND STOCKPILE HISTORY

Numerous waste rock investigations have been conducted to understand the physical and structural properties and their relation to water flow. Past excavation projects in waste rock include Golden Sunlight Mine (Herasymuik 1996), a sponsored study by the International Network for Acid Prevention (INAP) of two waste rock dumps (Fines et al. 2003) as well at the Grasberg Mine (Andrina et al. 2003, 2006, 2012). Others currently underway include the Diavik project in Canada (Smith et al. 2012) and the Antamina waste rock project in Peru (Harrison et al. 2012).

Consistent evidence of structural features has been found in waste rock studies. The upper portion consists of alternating layers of fine and coarse waste rock sloped at the angle of repose, which are formed from end or push dumping of the material. Larger waste rock material travels the farthest down the tip face and forms a rubble zone of larger particles at the base of the pile or bench. These characteristics are intimately related to the method of construction, typically a result of end-dumping or push dumping during construction (Wilson 2011; Aubertin 2013). This highly heterogeneous structure results in fluid flow that is difficult to characterise, as sections of the waste rock profile can behave as both a soil-like and rock-like medium (Smith and Beckie 2003). Water flow is controlled by the infiltration rate, where low infiltration rates cause fine material to conduct fluid flow, and as infiltration rates increase, local flow systems or macropores can create channelized flow paths (Wilson 2011). Due to this complexity, the hydrologic behaviour as cited by many authors is very difficult to predict; however, understanding the internal systems is essential to the prediction and management of ARD (Aubertin 2013).

The research currently underway at DLM emphasizes the necessity to understand the historic waste rock dumps to better predict ARD and construct future waste rock structures (Robertson et al. 2012). The historic waste rock piles have an average 16 year operating history and a 10 year post-closure history. Closure and rehabilitation of the DLM site in 2000 resulted in re-sloping of the waste rock and overburden stockpiles on site and placement of a 0.5 to 1.0 m overburden cover. Four waste rock stockpiles were present after mine closure covering areas up to 28 hectares. ARD has not developed at the site to date, and seepage and metal leaching rates are low (Robertson et al. 2012). The development of a new mine plan involved the creation of a large open pit where two of the four existing waste dumps would be located inside the proposed footprint. This necessitated the relocation of the two waste dumps, and provided an opportunity for an excavation study during their deconstruction. The two other existing stockpiles did not require relocation and were

instrumented to assess in situ conditions. This paper will focus on two of the waste rock piles requiring relocation, specifically, waste rock Stockpiles 1 and 2.

3.0 WASTE ROCK EXCAVATION PROGRAM

The first stage of the research program involved the forensic excavation of Stockpiles 1 and 2 prior to and during their relocation. The excavation aimed to investigate the current conditions of waste rock within these stockpiles to assess the structural, weathering and hydrological conditions. Stockpiles 1 and 2 are approximately 15 to 20 m high above the surrounding topography of low rolling hills of the glacial lowlands. The investigation was conducted in two stages. First, a surficial test pitting program was used to investigate material near to the surface prior to excavation, and second, an investigation was conducted to study the pile interior during excavation and relocation of the material.

3.1 Test Pit Sampling Program

A detailed test pitting program was conducted in August 2011 on a grid pattern across the surface of Stockpile 1 and Stockpile 2. Test pits were constructed using a backhoe, with a footprint of approximately 10 m by 10 m. Each test pit was excavated downward in benches of approximately 1 m to create an overall 1:1 slope to prevent sloughing of material and to provide safe ingress and egress (Figure 1). Parameters of interest in each test pit included major rock types, evidence of oxidation, visible sulphide minerals, waste rock structure, texture, angularity, and degree of weathering. *In situ* matric suction and temperature were measured and representative samples were collected for detailed analysis. Photos were taken to document each sampling location and for use in digital image processing. A total of 13 test pits were completed on Stockpile 1 and 15 test pits on Stockpile 2.



Fig. 1. a) Typical test pit on the surface of Stockpile 2, b) Tensiometer installation on Stockpile 1 in fine waste rock

Three 2725ARL Jet Fill Tensiometers (Soilmoisture Equipment Corp.) were installed in each test pit, two in waste rock and one in the cover overburden material to measure soil suction. After equilibration, small 1-2 kg bag and larger 20-L pail samples were taken around the tensiometer ceramic cup. On Stockpiles 1 and 2, 39 and 28 samples were collected, respectively, during this sampling campaign.

3.2 Profile Sampling Program

The second stage of the field campaign was completed while waste rock from the stockpiles was relocated. Excavation of the waste rock resulted in exposed vertical profiles of waste rock exposing the inner structural characteristics. A similar logging procedure to the test pit program was conducted and samples were collected with an excavator and collected in 20-L buckets (Figure 2). Limited access to the slope face prevented matric suction measurements, and moisture content was not measured due to the length of time the slope was exposed.



Fig. 2. a) Profile sampling using an excavator on Stockpile 1, b) Typical samples collected during the profile sampling program in 20-L pails

A total of 21 samples on Stockpile 1 and 12 on Stockpile 2 were collected during the second sampling program. A total of 100 samples were collected from the test pit and profile sampling programs, 16 samples of the cover and 84 samples of waste rock material.

3.3 Laboratory Testing Program

Post-processing of samples taken during the field programs was conducted at the University of Alberta. Samples from the test pit program were first oven-dried at ? prior to any testing to limit further oxidation. Testing for all samples included sieve analysis for determination of particle size distribution, paste pH and Munsell soil colour classification. Moisture content was determined only for samples collected during the test pit program.

Particle size distribution utilized sieves up to 75 mm, while larger particles were sized manually. The coarse fraction was processed in one batch in a TS-2 Gilson Testing Screen for 10 minutes. Standard brass sieves were used to process the fine fraction with a standard mechanical sieve shaker. Paste pH was conducted to indicate the presence of readily available neutralising potential (NP) and stored acidity from surface oxidation. Munsell soil colour was determined to investigate if the waste rock colour correlated with other characteristics.

Detailed laboratory testing is in progress on 13 representative samples, three cover samples and ten waste rock samples. Detailed testing includes the determination of saturated hydraulic conductivity and the soil-water characteristic curve (SWCC) to characterise the unsaturated soil behaviour. In addition to laboratory tests, photos from each of the 13 select

samples were subjected to a desktop analysis using Digital Image Processing (DIP). The physical sample size collected prevented characterisation of large-sized waste rock as screens and sieves for material larger than 10 cm are uncommon and expensive. Consequently, alternative methods such as DIP analysis have been employed to assist in evaluation of grain size distribution and fragmentation. Split Desktop (Split Engineering LLC 2011) is DIP software that allows the measurement of grain size utilizing a photo. Large particles were delineated manually and areas of fine material were grouped together (Figure 3). Split-Desktop utilizes four steps for image analysis: i) determining an image scale, performing automatic delineation of the photograph, ii) manual user editing of the delineations, iii) calculation of the particle size distribution, and iv) providing the graphical output of the results in the form of a particle size curve. After particle delineation, the program creates a best-fit ellipse for the particle and statistically based correction factors for fragment overlap and fines (Split Engineering LLC 2011).

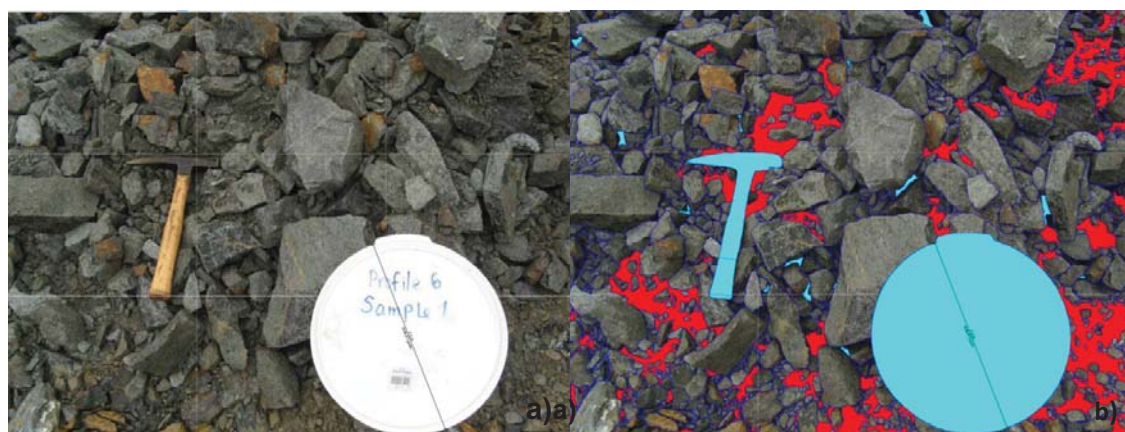


Fig. 3. a) Photo of sampling location before DIP analysis, b) Photo with delineation of particles outlined in dark blue and fines represented by red shaded zones. Masked zones are light blue and not measured.

4.0 FIELD PROGRAM RESULTS AND DISCUSSION

The test pit program was effective at characterizing the upper portion of the undisturbed waste rock as well as the cover. The cover ranges typically from 0.3 to 1.0 m in thickness, and consists of silty sand till material with some glacial cobbles. Waste rock composition and weathering were found to be highly varied throughout the waste rock pile. Upper portions of the waste rock are matrix supported with clasts, and large waste rock material formed a clast supported structure with open void space or infilled with fine matric material.

The specific method of construction for the DLM stockpiles was not known prior to the excavation study. Structural features identified during the excavation study indicate that the pile was likely constructed as a combined push dump and paddock dump. Elevated areas of the piles exhibit a coarsening downward structure with defined traffic surfaces and angle of repose layering. Figure 4 shows a profile view of Stockpile 1, with identifiable structural features.

Weathering of waste rock was found throughout the entire pile. Figure 4 further illustrates that oxidation occurred throughout each bench. A distinct oxidation front was not evident,

and oxidation was variable in degree and spatial location and staining was observed on the majority of particle surfaces. Pyrrhotite is the dominant sulfide mineral with lesser amounts of pyrite. Cementation due to mineral precipitation and potassic alteration were also observed. The presence of oxidation appears to be related to the waste rock type and amount of sulphide minerals rather than the location in the pile. The waste rock moisture contents indicated that the cover did not prevent water ingress as the waste rock was moist. Investigation into the effectiveness of the cover for limiting oxygen and water ingress is ongoing.

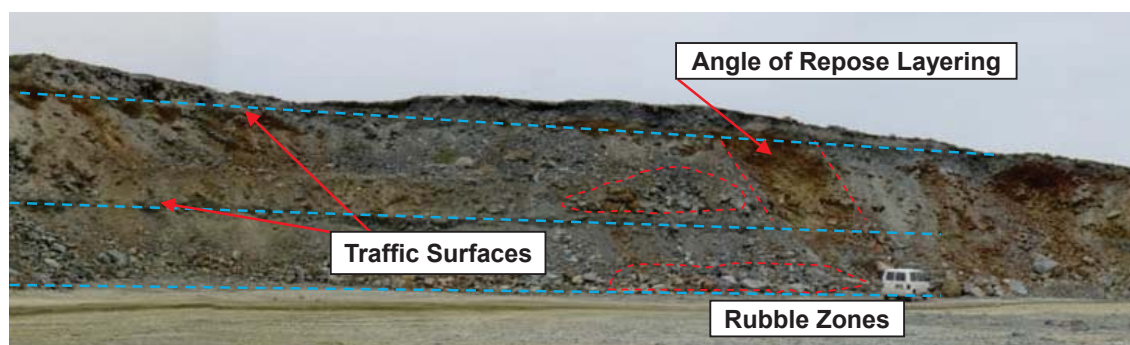


Fig. 4. Excavated face of Stockpile 1 showing evidence of layered structure, traffic surfaces and rubble zones

A total of 51 readings were taken in the test pits, with matric suction values for the cover material typically between 10 kPa and 50 kPa, and waste rock matric suction between 3 kPa and 30 kPa. Gravimetric moisture contents of material surrounding the tensiometer cups were typically between 3% and 6%. The larger 20-L pail samples typically had lower gravimetric moisture content range due to the larger particles (>4 mm), which do not hold water in tension. Moisture contents were typically within the range of 1% and 6%.

Temperature profiles within Stockpile 2 decreased with depth, and no hot spots or thermal zones were observed. All recorded temperatures were typically between 5°C and 20°C and the mean ambient air temperature during the sampling period, August, was 15.9°C.

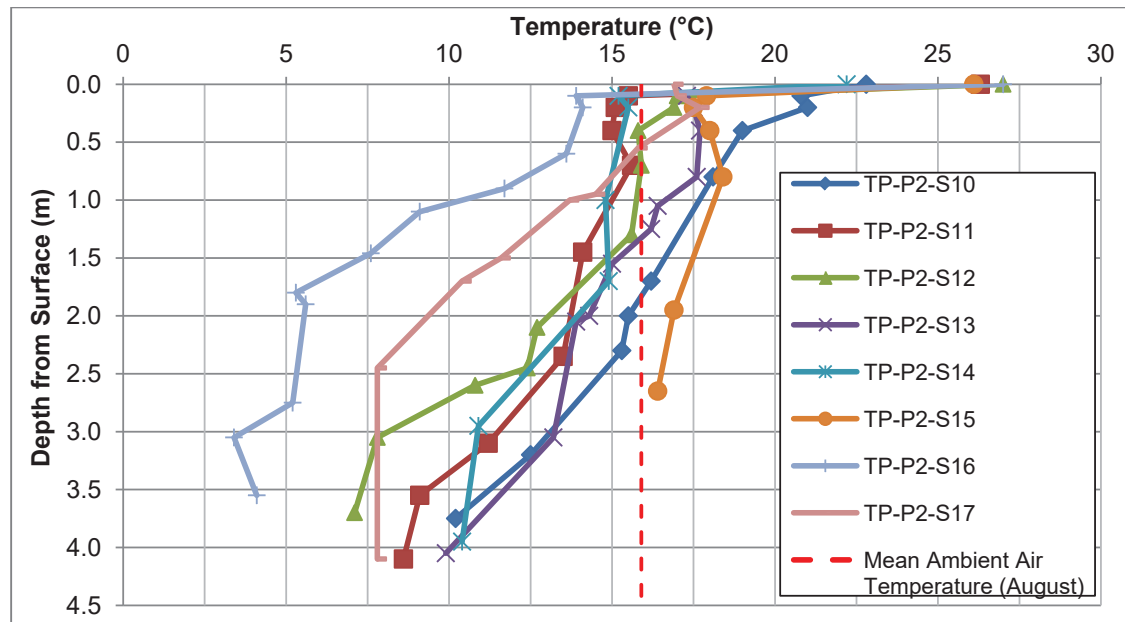


Fig. 5. Temperature profile recorded in test pit walls in August for sampling locations on Stockpile 2

5.0 LABORATORY RESULTS AND DISCUSSION

All samples collected were mechanically split and sieved, with subsamples utilized for particle size distribution and to determine paste pH. Preliminary results of particle size distribution have been compiled in the following section.

Paste pH was conducted on <10 mm material. All cover samples were within the neutral pH range of 6 to 8, and waste rock samples had a bimodal distribution as shown in Figure 6. Approximately 20% of samples have a pH of less than 4, and most samples fall within the neutral pH range. Similar distributions were found on Stockpile 1 and 2.

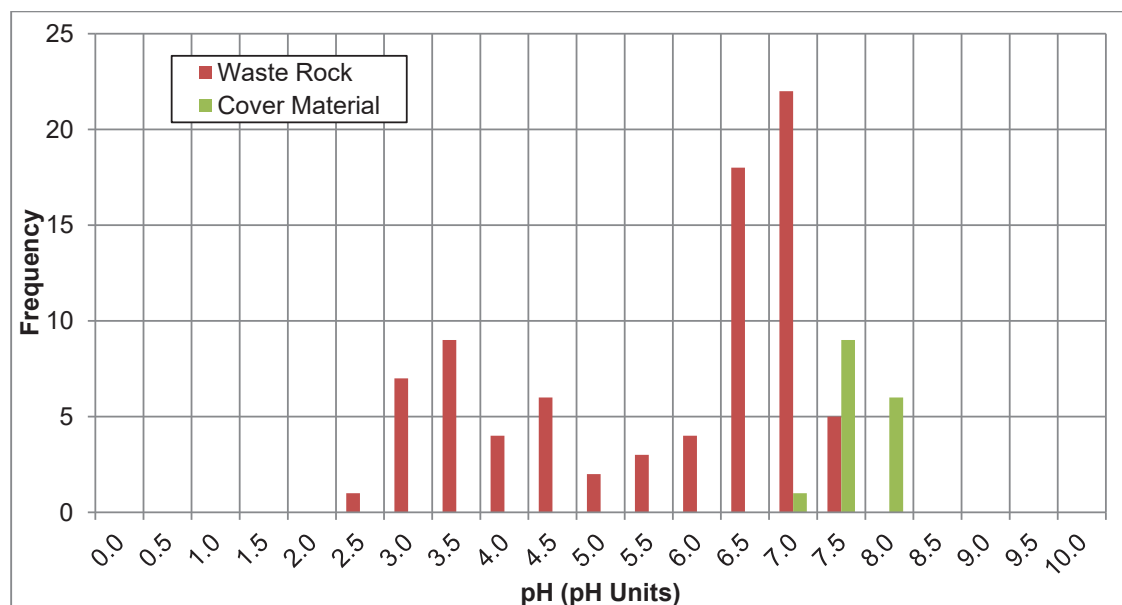


Fig. 6. Frequency distribution of paste pH results for waste rock and cover samples

Due to limits on sample sizes collected, grain size distribution determined using sieve analysis best defines the fine fraction endpoint for the waste rock. To accurately characterise material up to 150 mm, samples in excess of 500 kg would be required (Herasymuik 1996), and samples sieved during this study were typically limited to 4 kg and 7 kg. Cover samples are presented in Figure 7. Cover material was typically found to be well graded, and are divided among three main material types. The stockpile cover material was primarily sand sized with varying amounts of silt and clay and some glacial highly rounded cobbles or waste rock particles. Similar materials were found in adjacent locations, suggesting progressive reclamation occurred with material that was available at the time of construction of the cover. No organic matter was placed over the cover material; however, vegetation is present on most areas with roots extending into the upper 10 cm. Particle size distribution curves were similar for cover material on Stockpile 1 and Stockpile 2, indicating similar source material was used during closure activities on both piles. For detailed testing, three cover samples were selected to represent the three materials distributions found during particle size analysis (Figure 7).

Figure 8 illustrates 84 grain size distributions for all waste rock samples, along with a particle size distribution envelope. Samples collected during the test pit program have a higher proportion of fines as samples were collected in the near surface areas of the waste rock pile. Profile samples were taken from a variety of locations including those with higher proportion of coarse material, and consequently the fines content is lower. Particle size distribution data from physical samples was effective in characterizing the fine grained portion of waste rock at the DLM. Due to the limited sample size that was collected, the upper range of particle sizes was not characterised. The typical maximum size of particles sampled was 75 mm, and the measured curves illustrate a well graded fine fraction of material less than 10 mm. Material larger than this was less abundant, and the particle size curves are gap graded for material larger than 80 mm, due to the limited pail sample size. During sampling, very large particles were observed up to 2 m to 3 m in diameter.

Thirteen select samples, three samples of cover and 10 of waste rock, are presented in bold and black in the particle size distributions in Figures 7 and 8, and were selected for detailed sample testing to represent the range of properties encountered from the sampling program. Saturated hydraulic conductivity, SWCC are ongoing to assist in the characterisation of unsaturated properties. These data will be utilized in unsaturated seepage modelling to characterise the total water flux through waste rock, which in turn can be applied to simulations using reactive transport models to estimate time and spatially dependent behaviour.

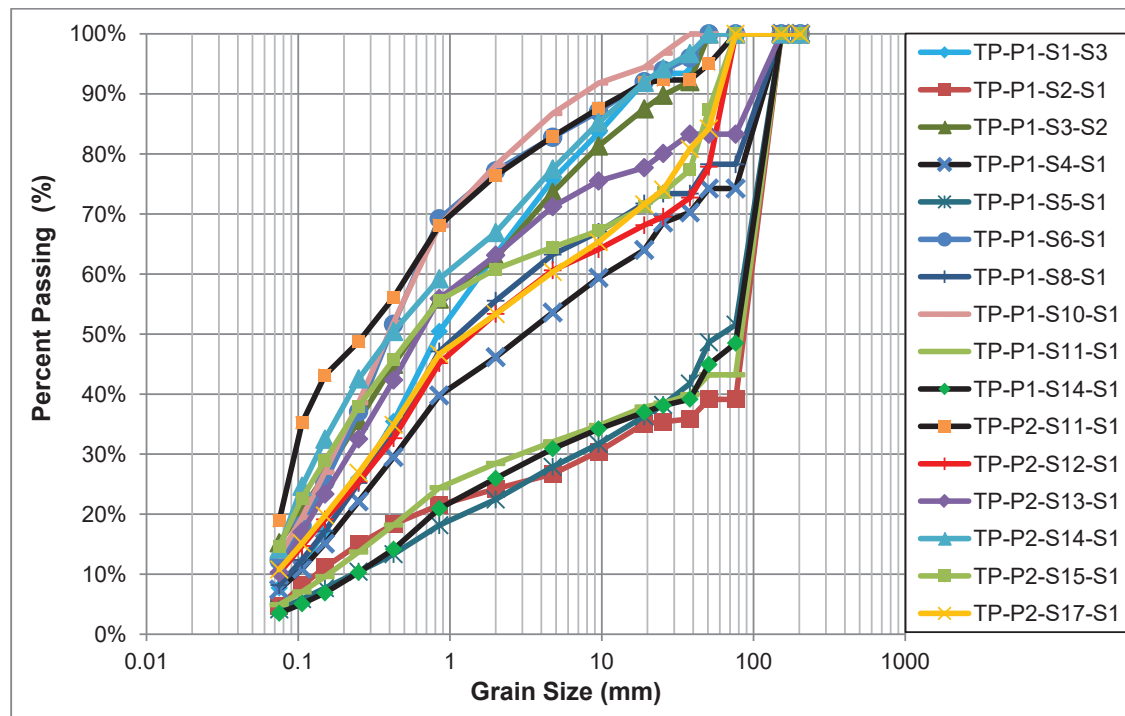


Fig. 7. Particle size distribution curves for stockpile cover samples. Black highlighted curves with coloured data points represent cover samples selected for additional detailed testing (TP-P1-S4-S1, TP-P1-S14-S1 and TP-P2-S11-S1)

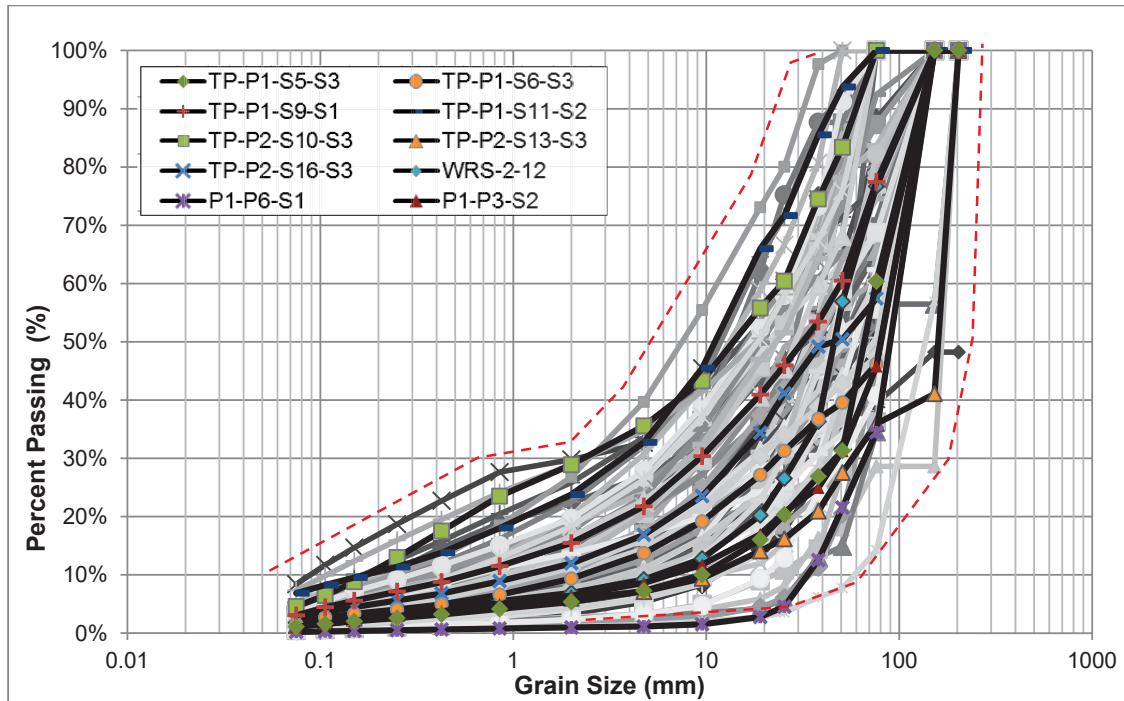


Fig. 8. Particle size distribution envelope of curves for waste rock samples for Stockpile 1 and 2. Black highlighted curves with coloured data points represent samples selected for detailed testing to represent grain size range

In order to quantify the upper range of waste rock particles for the numerical model, digital image processing was employed to characterise large scale particles. Photos taken at the sampling location of the ten select waste rock samples and their computed particle size distributions are provided in Figure 9. The envelope of curves from the physical samples is also overlain for comparison.

The DIP software sharpens particle boundaries and shadows during processing, and will also assess particle boundaries in accordance with the resolution of the photograph. The fine fraction portion of the curve is statistically calculated, as pixel size limits the size of detectable particles. The DIP analyses output from this investigation provides a reasonable characterisation of fine material, with a similar envelope to the measured laboratory curves (*Cash In Progress*). The distribution of the upper particle size range (>50 mm) is less distributed as the proportion of larger particles in a given photo is less varied. The analysis was successful in delineation of particles up to 0.5 m in diameter, which cannot realistically be sampled in a field investigation. DIP therefore is a valuable tool to represent the portion of waste rock where sampling is not practical. Particles larger than 0.5 m were observed in the field and may not have been characterised in this analysis due to the size of the area photographed as well as the overlapping of particles in photos or infilling of fines which can mask the entire boundary of particles. Larger photographs of cross sections may be required to ensure even greater accuracy in capturing large scale material.

Typical field programs collect samples that overestimate the amount of fine material due to the limited sampling volume. Understanding the true proportion of waste rock that is large

clasts or fines is important, as water flow occurs primarily in the fine fraction, which can hold water in tension in unsaturated conditions. The fine fraction volume is displaced within the waste rock pile among larger particles that do not transmit significant flow. Therefore, it is essential to understand the proportion of fines that are present in the total waste rock profile to determine the proportion of material which contributes to water flux. Employing DIP techniques to better understand grain size could result in more accurate determination of water flux.

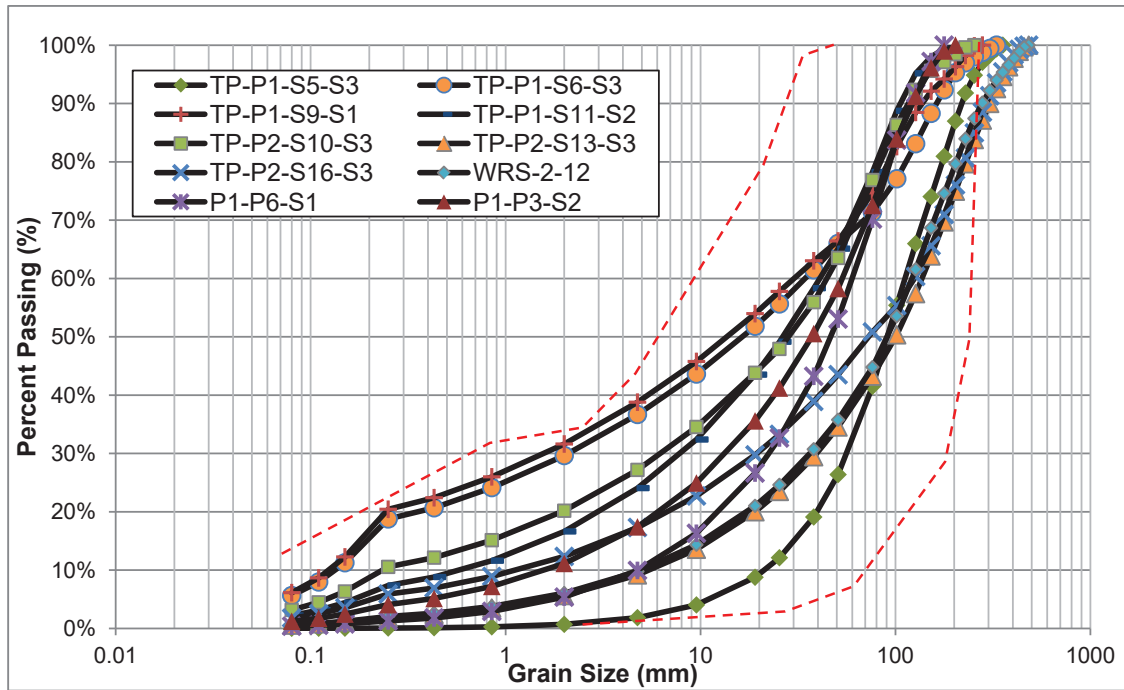


Fig. 9. Grain size distribution results from Split Desktop digital image processing of select waste rock samples

Further analysis of the DIP data is ongoing, and integration of detailed laboratory studies will also be incorporated. Completion of the laboratory program will facilitate the estimation of a rock corrected SWCC to understand the influence of large clasts of waste rock on water flow in unsaturated conditions. Continuing analysis of DIP data will also include the evaluation of air filled porosity, solid phase content and fines to determine an *in-situ* bulk density. These data will be assembled into a numerical model using commercial seepage software in 2014 to depict hydrologic behaviour. The accompanying geochemical and gas transport study results are expected throughout 2014.

6.0 CONCLUSIONS

While there is still considerable analysis to complete on the Detour Lake Mine waste rock project data, preliminary conclusions from the stockpile sampling program include:

- The stockpiles show evidence of segregation of material and structure consistent with push dump construction techniques, similar to characteristics observed in other waste rock excavation studies;

- After a 26 year history, evidence of oxidation is present throughout the entirety of the stockpiles; however the degree of weathering and oxidation is varied which appears to be attenuated within the rock materials. The presence of a cover has not prevented oxidation, but may limit advective gas transport into the waste pile. It also may have limited the flow of water as there is little evidence of seepage escaping from most of the stockpiles;
- The majority of paste pH values are with a neutral pH range which supports the observation that ARD has not yet developed on site;
- There are no distinct thermal zones or hot spots evident in the waste rock, however measured temperatures are greater than the mean average annual temperature suggesting some thermal activity;
- A wide range of grain sizes were found within the waste rock pile. The grain size distribution curves illustrate the fine grained endpoint for the waste rock material as sample sizes greater than 75 mm cannot realistically be sampled in a representative way. Digital image processing was effective at accurately representing the fine fraction as well as delineating material up to 0.5 m; and;
- DIP analysis could reduce the number of physical samples required to accurately depict the range of grain sizes within a large scale waste pile, as outputted results correlate well with measured results.

The observations compiled here will form the basis for ongoing testing and will support the development of models to understand water flow and reactive transport mechanisms in the future. This research will assist in development of knowledge for the investigation and key areas for characterisation of waste rock for full scale waste rock dumps on site. Furthermore, information of how to predict, mitigate and prevent ARD will assist in more efficient and informed efforts for mine closure of waste rock structures.

7.0 ACKNOWLEDGEMENTS

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LANDFORM DESIGN FOR GEITA MINE: COMPARISON OF SIBERIA AND CAESAR-LISFLOOD

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ABSTRACT

Geita Mine is situated in the Mwanza Region of north-western Tanzania. Previous characterisation has shown up to 38% of the waste rock materials are potentially acid forming, with elevated SO_4^{2-} and associated elevated concentrations of F and As in the regional groundwater. The tailings materials were enriched with As and Sb and slightly enriched with Hg and Se. Tailings pore-water has shown elevated concentrations of elements, primarily As and Sb and the tailings are PAF due to the presence of sulfides. Geita's remediation objective is to minimize the infiltration of water into the waste materials and thus reduce the contaminant loading to downstream receptors. A cover and landform to meet these objectives were designed and evaluated for long-term performance. One aspect of long-term performance was the erosion potential of the landforms. Two commercially available 3-D software packages were used to simulate long-term erosion rates: SIBERIA and CAESAR-Lisflood as well as a 2-D model, WEPP. A brief description of each model and the results of long-term simulations on landforms at Geita using these modelling packages are presented.

1.0 BACKGROUND AND SITE DESCRIPTION

The modern Geita Gold Mine (GGM) is an open-pit mine situated in the Mwanza Region of north-western Tanzania, approximately 90 kilometres from the regional capital of Mwanza and 20 kilometres south of Lake Victoria, in an area known as the Lake Victoria Goldfields. The main Geita Mine operations are located within the Mtakuju River catchment that drains to Lake Victoria 25 km to the northwest.

The GGM operation comprises two mining areas: one centred on the old Geita Hill area comprising several pits: Nyankanga, Geita Hill, Lone Cone and their associated waste rock dumps: WD1, WD5, WD6, WD14, WD15, Star & Comet, and the other located ~20km away comprising the Kukuluma and Matandani pits and waste rock dumps WD7, WD8 and WD9. Tailings have been deposited in a historic TSF referred herein as the Old TSF and a modern storage referred to as the New TSF. Currently gold ore is mined from Nyankanga, Geita Hill and Star & Comet Pits. Kukuluma and Matandani pits and the Lone Cone pits are not operational. The current life of mine is 2030.

GGM's broad reclamation objective is to leave the site in a condition that is safe, stable, and minimises long term environmental impacts as well as providing opportunities for alternative forms of business development. GGM currently rehabilitates their waste rock dumps concurrently with active mining as detailed in Sibilski and Stephen (2006). The dumps are

battered to a slope <20 degrees and then topsoil of 20 cm depth is spread on top and cross-ripped to form 50 cm deep ridges. GGM uses indigenous species for revegetation of the reclaimed WRDs to provide a sustainable ecosystem and maintain the appearance of the areas that existed prior to disturbance (Sibilski and Stephen, 2006).

The GGM waste rock contains sulphides, the majority being pyrite with minor pyrrhotite and trace arsenopyrite, chalcopyrite, galena, and sphalerite. Carbonate minerals are also present in the rock, consisting of mainly calcite with some dolomite, ankerite, and siderite (Sibilski and Stephen, 2006). Previous characterisation has shown that up to 38% of the waste rock materials are potentially acid forming (PAF), with elevated SO_4^{2-} and associated elevated concentrations of F and As in the regional groundwater. The tailings materials were enriched with As and Sb and slightly enriched with Hg and Se. Tailings pore-water has shown elevated concentrations of elements, primarily As and Sb and the tailings are PAF due to the presence of sulfides. GGM categorizes the waste rock into PAF and NAF streams. A mine waste block model was developed to encapsulate the PAF waste within each dump. Current management of ARD is by waste rock encapsulation, progressive reclamation to reduce net percolation, and treatment of ARD seepage.

GGM is refining their current conceptual closure plan with detailed studies to characterise the waste and potential cover materials from a geotechnical and geochemical basis, as well as to improve their understanding of the transport of contaminants of concern from the waste dumps to the receiving environment. Details on this project, including additional information on the geochemical characteristics of the GGM materials, can be found in Scott *et al.* (2013) in these proceedings. The objectives of these studies are to refine the closure criteria and to improve the closure cover and landform design to ensure that the closure criteria are met.

2.0 GEITA MINE CONCEPTUAL LANDFORM DESIGN

To control ARD, one of GGM's remediation objectives is to reduce the net percolation of water into the waste materials and thus reduce the contaminant loading to downstream receptors. Reduction of net percolation requires that other components of the landform water balance be increased, such as runoff, interflow, and evapotranspiration. Increasing these components then results in less water available to percolate into the waste material below, where it can react with sulphide minerals. The cover and landform design must, therefore, enhance runoff and evapotranspiration.

The average altitude for the Geita mine site is 1,180 m to 1,350 m ASL and up to 1,620 m for the Kukuluma-Matandani mine site. The Geita District has a highland equatorial wet-dry weather pattern with a bimodal wet season (generally October to December and February to May) with a mean annual rainfall of about 1000 mm. A distinct dry season extends from June to August/September with annual potential evapotranspiration of about 1,300 mm. The annual minimum and maximum temperatures for the GGM area are between 14°C and 32°C.

To enhance runoff and evapotranspiration, a landform must be able to move water without significant erosion and must retain sufficient moisture throughout the growing season to support vegetation. The philosophy behind the landform design for GGM is about meeting these objectives for water management and creating a landform that has a similar aesthetic to natural landforms, thereby also meeting other GGM reclamation objectives such as re-creating a land capability similar to that pre-disturbance and re-establishing local self-sustaining vegetation.

2.1 Landform Design Philosophy and Methodology

Waste rock dumps often require extensive re-grading prior to site closure because the final landform for reclamation was not addressed during the mining/construction phase. During reclamation effort, angle-of-repose slopes are generally flattened due to long-term geotechnical stability, erosion and safety concerns, and in many cases, to facilitate placement of a topsoil / growth medium layer or an engineered cover system (Ayres et al. 2006). Historically, rehabilitated stockpile landforms possess uniform slopes conforming to neat lines and grades. This lends itself to uniformity of design and construction, but does not necessarily achieve the mine closure objectives of minimum erosion and long-term sustainability (Sawatsky et al., 2000).

Uniformity represents an immature landform at a high energy state that will evolve to lower energy states by shallow slope failures or accelerated erosion (climate and material properties dictate how quickly this will occur). Figure 1 illustrates a landform at a high energy state (rice terraces in Longji, China). Here, without daily maintenance to repair eroded channels and small slope failure, the landform will revert back to a smooth landform with bamboo forest in just a few years.



Fig. 1. Example of landform at a high energy state.

In contrast, the development of a sustainable landscape for mine closure involves the development of landforms that replicate natural landscapes (Ayres et al, 2006). The replication of mature and relatively stable natural systems reduces the rate and risk of accelerated erosion. It also encourages replication of the self-healing erosion control systems that help preserve the stability of the natural analogue (Sawatsky et al., 2000).

Most slopes in nature are characterized by a variety of shapes including convex and concave forms interspersed with ridges (spur ends) and swales (hollows). Carson and Kirkby (1972) compiled data on a large number of hillslopes, and determined that the majority possess rounded convex summits and have shallow concave elements at the base. Drainage systems generally follow natural drop lines in the slope, with catchment sizes defined by undulating relief on the slope.

Vegetation on natural slopes grows in discrete vegetation units that are adjusted to hillside hydrogeology, incident solar radiation, and other microclimate effects. Trees and shrubs are

concentrated in concave areas, where moisture conditions are higher, while grasses / legumes generally dominate the drier convex portions.

2.2 Landform Erosion Assessment

The landform erosion assessment conducted for GGM focused on the WD1 waste dump, owing to available information and the ability to verify model predictions with actual survey data. The conclusions from the WD1 assessment can be used to provide design principles and recommendations for other landforms across the site.

The objectives of the assessment were to:

- Provide an estimate of erosion rates given the site material and current landform profile, and to provide some interpretation and discussion of these rates in the context of GGM's long term landform stability objectives;
- Use, where possible, measured materials characteristics. OKC performed a material characterisation program in 2012/13 to determine the geotechnical and geochemical characteristics of the waste and topsoil materials. Results of the geotechnical characterisation were used to determine the structural and hydraulic behaviors of the materials and for use as input parameters into erosion models and assess potential for use in final landform and cover system design.
- Provide a comparison of predicted erosion rates for a range of possible outer slope configurations, and in doing so provide practical advice as to the profiles which can be expected to perform well given approximate crest and toe constraints; and
- Provide surface water management recommendations aimed to improve the erosion performance of the site landforms.

2.2.1 Landform Assessment Methodology

The methodology used to assess landform stability performance and to provide recommendations for improvements has been developed using a number of commercial tools and supporting site specific information. The main tools and information used include LiDAR survey of the landforms (provided by GGM), aerial imagery (Google Earth), the SIBERIA landform evolution model, the Water Erosion Prediction Project (WEPP) model, the CAESAR-Lisflood model, and material laboratory analysis results. The tools and their application in this assessment are discussed in the following sections.

2.2.1.1 Volumetric Erosion Assessment of WD1

WD1 waste landform has eroded significantly since its construction in 1999/2000. The outer dump rehabilitation work commenced in 2002-2003; therefore, most of the outer dump final land-surface is less than 10 years old. Erosion of the surface material in the (approximate) 10 years has been severe (Figure 2) and is unlikely to satisfy GGM's long term landform stability objectives. A LiDAR topographic survey was provided by AngloGold Ashanti and interrogated in specialist terrain modelling software (Autodesk Civil3D) to determine the approximate volume of erosion that has occurred over the (approximate) 10 years in a selected 500 m x 500 m square area (the assessment area) of the landform (Figure 3).

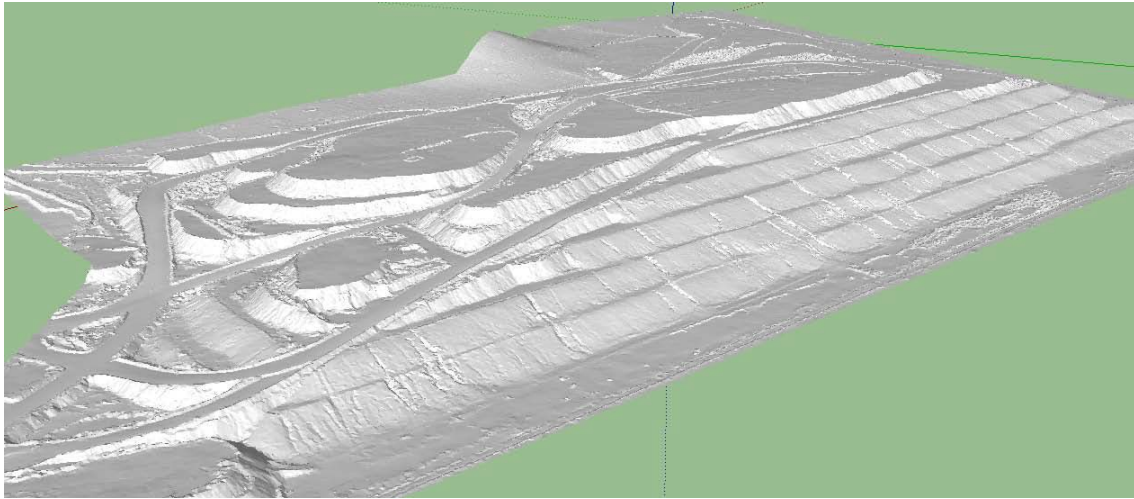


Fig. 2. Digital Terrain Model illustrating significant gully erosion on the outer embankments of WD1 over (approximate) 10 years since construction.

WD1 was selected for this analysis owing to:

- Its susceptibility to erosion;
- The high resolution of the LiDAR-surveyed gullies; and
- Its 'regular' shape. This allows the analysis to be less affected by other features potentially contributing to the landform erosion (e.g. the curvature of the landform).



Fig. 3. WD1 assessment area.

The erosion volume was calculated from the incised gullies and a nominal surface erosion across the assessment area. The incised gullies were clearly observed on the survey. Twelve significant gullies were identified (Figure 4) and the total volume of these formed the

bulk of this volume-based erosion assessment. The eroded volumes were calculated by removing (effectively filling) the surveyed gullies in Civil3D and then comparing the pre-eroded and post-eroded surfaces. Figure 4 illustrates the eroded surface contours (left) and the edited surface contours (right).

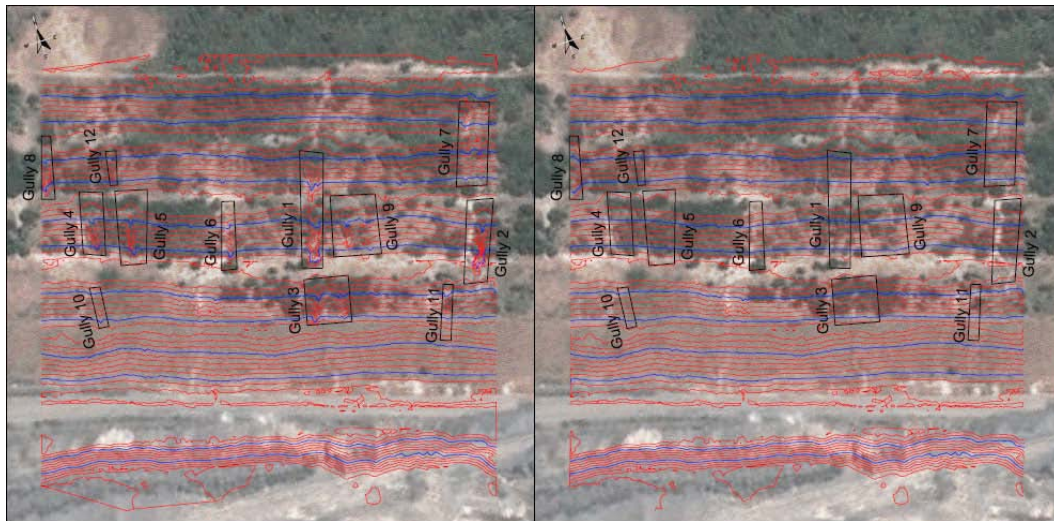


Fig. 4. 500 m x 500 m section of WD1 landform with identified gullies in contours (left) and gullies removed from contours (right).

The total eroded volume of the gullies was found to be approximately $9,569 \text{ m}^3$, occurring over the 25 ha area. In addition, a nominal 10 mm erosion depth (over the entire assessment area), ($2,500 \text{ m}^3$), has been added to the account for small erosion rills and interill erosion not picked up in the gully volume calculations. The total erosion volume is therefore estimated at $12,069 \text{ m}^3$, equating to $48 \text{ m}^3/\text{ha}/\text{year}$ ($83 \text{ t}/\text{ha}/\text{year}$). This forms the 'best estimate' erosion rate referred to in the following sections of this report.

2.2.1.2 Conceptual WD1 Landforms

Three conceptual waste rock dump landforms were developed to compare for erosional stability using the modelling tools. The conceptual landforms were based on the 500 m x 100 m section of WD1 shown in Figure 3. The first landform developed was based on the current reclamation plan, with uniform benches and slopes. The second landform was a linear slope of 2.75H:1V, representing the maximum slope angle for safe operation on the contour by a dozer. The third landform was a concave-style landform with a steeper section at the top, and a flatter section at the bottom. The concave waste rock dump (WRD) landforms are designed to be similar to the surrounding natural landforms. The three conceptual landforms are illustrated Figure 5.

3.0 MODELLING TOOLS

3.1 SIBERIA Model

SIBERIA is a physically-based 3-D model for simulating the evolution of landforms over geomorphic timescales (Willgoose et al. 1991). SIBERIA predicts the long-term evolution of channels and hillslopes in a catchment on the basis of runoff and erosion. The location and

speed with which gullies develop are controlled by a channelization function that is related to runoff and soil erodibility (Willgoose et al., 1991). The model solves for two variables; elevation, from which slope geometries are determined, and an indicator function that determines where channels exist. An activation threshold governs channel growth: a surface may commence with no gullies, but when the activation threshold, which depends on discharge and slope gradient, is exceeded, a channel develops. Hancock et al. (2000) demonstrated that SIBERIA is an appropriate model for assessment of erosional stability of rehabilitated mine sites over time spans of around 50 years.

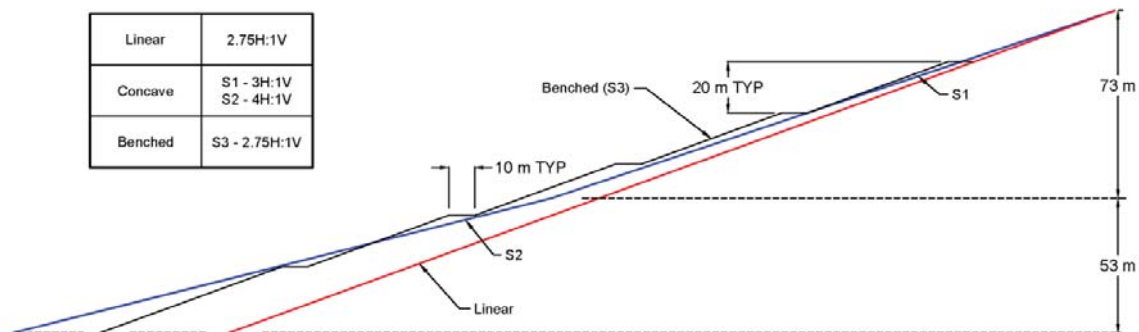


Fig. 5. Conceptual landform designs (linear, concave, and benched) applied to 500 m x 100 m section of WD1 for comparison of erosional stability using the modelling tools.

The SIBERIA model requires as input erodibility parameters for the surface soil, which is related to the soil and vegetation properties of the placed surficial materials. SIBERIA does not utilize recorded rainfall, rather the model applies the 'average' rainfall conditions, for every year of the simulation, over the predicted time.

The SIBERIA model needs to be calibrated before evaluating whether it correctly models the observed evolution of rehabilitated mine landforms. The following methods can be used for obtaining erosion parameters required for input to the SIBERIA model:

- Collect erosion data from rainfall and runoff testing using rainfall simulators as described by Loch et al. (2001), and subsequently use a model such as WEPP (see below) to determine erosion rates for each soil type;
- Measure controlled flow through a series of flumes constructed on the hillslope; this method does not simulate rainfall / runoff but allows assessment of the impact of high flow rates on a range of armoring methods; and
- Determine erosion rates from digital mapping of slopes actively being eroded.

3.2 WEPP Model

The WEPP model is a process-based program that was developed in the late 1980's and early 1990's by the United States Department of Agriculture (USDA). It is best suited for detailed considerations of short-term (up to 100 years) impacts of slope length, gradient, and management on erosion rates. The appropriate scales for application are tens of meters for hillslope profiles, and up to hundreds of meters for small watersheds (Flanagan and Livingston, 1995). The model explicitly considers rill and interrill erosion and is therefore better able to consider interactions of slope length and gradient than other models. WEPP

estimates net soil loss for an entire hillslope or for each point on a slope profile on a daily, monthly, or average annual basis. Basic inputs required for the WEPP model include climate data, slope configuration, soil properties, and soil management (vegetation) properties.

3.3 CAESAR-Lisflood MODEL

The numerical model CAESAR-Lisflood v1.6a was used to examine the geomorphological / landscape evolution of the GGM conceptual landform designs. This model combines the Lisflood-FP 2d hydrodynamic flow model (Bates et al., 2010) with the CAESAR geomorphic model (Coulthard et al., 2012). The CAESAR model simulates landscape development by routing water over a regular grid of cells and altering elevations according to erosion and deposition from fluvial and slope processes. The Lisflood-FP model is a one-dimensional inertial model that is applied in the x and y directions to simulate two dimensional flow over a raster grid (Coulthard et al, 2013). The integration of LISFLOOD-FP with CAESAR into the model CAESAR-LISFLOOD has resulted in a landform evolution model that is faster, simpler, and has a stronger physical basis than CAESAR (Coulthard et al, 2013). Key inputs to this model are elevation data, grain sizes of surficial materials, hourly rainfall data, and a vegetation parameter.

4.0 MODELLING RESULTS

4.1 SIBERIA Calibration to WD1 10-Year Performance

For this part of the erosion assessment, SIBERIA was used to predict the landform evolution and erosion rates shown to be occurring on WD1 over the ten-year time period. This was done in an attempt to calibrate the SIBERIA erodibility parameters so that long term performance of the landform and different design concepts could be compared. While SIBERIA is better known for its ability to predict erosion over centennial to millennial time scales it can also predict erosion at annual time scales. SIBERIA does not utilize recorded rainfall, rather the model applies the 'average' rainfall conditions, for every year of the simulation, over the predicted time. For longer term predictions, errors in the predicted erosion rates are 'regulated' by the length of the analysis. Short term erosion predictions on the other hand have more chance of error (per annum), as the actual rainfall in the ten years of actual erosion may be much higher (or lower) than the average conditions modelled.

Firstly, the 'as constructed' landform surface for WD1 was prepared in Civil3D to create a starting landform for SIBERIA to model. The 'as-constructed' survey was not available so an idealized starting landform was created to indicate 'Year 0' of the model (i.e. 2002/03). The landform was created by taking a two-dimensional landform transect from the LiDAR survey of WD1, modifying the transect profile so that eroded features were removed and then 'stretching' it laterally (in AutoCAD) to form a three-dimensional surface (see benched profile in Figure 5). SIBERIA was then used in an attempt to re-create the observed erosion of the landform over ten years. It was found that SIBERIA did not adequately produce the actual erosion profiles observed on the surface of WD1 (i.e. SIBERIA did not predict the highly incised gullies, rather a more consistent rate of smaller gullies across the profile (in the 10 years modelled)). Further to this, SIBERIA modelling indicated an average erosion rate over ten years of approximately 23 m³/ha/y (i.e. 48% of the total erosion calculated as part of the volumetric assessment).

The calibration assessment underestimated the total rate of erosion and did not predict the form of the observed (in-situ) gullying. This indicates that additional factors contribute to the gullying. It is thought that landform-plateau surface water may be contributing to the incident flows on the outer embankments and therefore increasing the rate of erosion. It is reasoned

that the plateau surface water flows have accumulated in areas at the crest and overtopped crest embankments, causing a series of cascading-type failure of the benches. This, in combination with a high fines, low strength material has led to deep incisions of the bench material. This theory appears to be supported by an examination of the WD1 DTM, which indicates areas of bund failure along the crest bund. As the Yr 0 SIBERIA model was constructed from an idealized surface (in the absence of an 'as constructed' survey), the catchment extents draining to the assessment area is unknown, impacting the ability to calibrate SIBERIA material properties using this method.

It is acknowledged that this failure mechanism is somewhat intuitive, and it is widely recognized that landform plateau surface water disposal is an ongoing challenge for many sites. This issue is particularly prevalent in high rainfall climates, where landforms contain AMD-forming material and where a lack of suitable surface materials is available. Notwithstanding this, for SIBERIA to be an effective model for long-term landform evolution, an accurate calibration of material properties is needed. This is commonly undertaken on-site with a rainfall simulator or in a laboratory using bulk samples and specialized erosion measurement equipment.

It is thought that in the absence of an accurate calibration for the material properties, long term erosion prediction using SIBERIA would be speculative. The WEPP erosion model was instead used for modelling design slopes, as certain materials property inputs (particle size distribution, rill and interill erodibility) required for WEPP modelling were calculated as part of the 2012/13 materials characterization programme. The WEPP modelling is discussed in the following section.

4.2 WEPP Modelling

4.2.1 WD1 Existing Profile

WEPP was used to provide materials specific assessment of erosion for WD1. WEPP was selected for the assessment as it allows for the rapid evaluation of different embankments without knowing the exact details of the plateau catchment and its influence on the outer embankments (i.e. overtopping). Testwork in the 2012/2013 material characterization programme included the calculation of the WEPP material erodibility parameters which allowed the model to be checked against the volume-based erosion assessment for WD1 (Section 2.2.1.1). Furthermore, the meteorological data used in WEPP allows for the introduction of variability in rainfall depths and the seasonal nature of the climate. Data used in this assessment was generated for a 100-year time period.

The performance of the WD1 (two dimensional) embankment profile was modelled using WEPP to check the erosion rates for the input parameters. A 100-year climate sequence was used in WEPP along with dump-specific material erodibility parameters calculated during the 2012/13 testwork. The 100-year climate database was developed based on monthly data from a dataset developed by the University of East Anglia Climate Research Unit (CRU, 2013) which was used to generate a complete dataset using CLIGEN (USDA, 2004) and adjusted to match the known local temperature and windspeed trends.

Visual results of the modelling of the existing WD1 profile are presented in Figure 6. The modelled profile is shown, indicating the soil loss identified by the intensity of the red coloration of the central layer. The soil loss graph, which provides the rates of erosion along the profile is provided in the bottom left window.



Fig. 6. Results of WEPP modelling for the existing WD1 embankment profile.

The WEPP modelling predicts an average (embankment area) erosion rate of 53t/ha/year over the profile length. This estimation is based on no external catchment draining down the outer profile, as did the SIBERIA modelling discussed in the previous section. Recalling that the volume-based assessment predicts erosion at 83 t/ha/y (approximately 1.6 times greater than the WEPP model and reflects erosion from all contributing catchments to the embankment over the lifetime of the landform). The results indicate that the plateau surface water contributes approximately 30t/ha/year erosion, or 36% of the total erosion in the WD1 assessment area.

No attempt has been made to quantify additional catchment areas contributing to erosion of the assessment area; however, it is suspected that surface water and/or seepage from local subcatchments may be contributing to flows and erosion in the assessment area.

4.3 SIBERIA & WEPP COMPARISON (WD1)

In this part of the assessment, the predicted long-term erosion provided by the WEPP profiles was used in an attempt to validate the SIBERIA model (in the absence of proper calibration). Similarly, this assessment was conducted to provide further confidence in the erosion rates predicted by the WEPP modelling. As discussed in the SIBERIA calibration section, the SIBERIA input parameters were not calibrated for the material types and therefore for this assessment the input parameters were selected from a database as an estimate of the material properties.

The profiles adopted for this assessment were those shown in Figure 5:

1. Current reclamation plan--final profiles of each lift wall are to be constructed at individual batter angles of 20 degrees, interspersed with 5 m to 10 m wide berms at 20 m vertical increments,
2. A single linear slope of 1V:2.75H; and
3. A dual gradient concave (comprising 3:1 and 4:1 grades).

For the current reclamation plan (benched profile):

- Results indicate that the average erosion rate of the current reclamation plan predicted by SIBERIA over 100 years was 52 m³/ha/y (94 t/ha/y) (shown in Figure 7). WEPP erosion rates derived for the current WD1 landform, (which are assumed in accordance with the current reclamation plan) are 52 t/ha/y.
- For the 1000 year SIBERIA simulation, the average predicted erosion rate was 58 m³/ha/y (104 t/ha/y), WEPP was not modelled for 1000 years owing to limitations of the climate file.

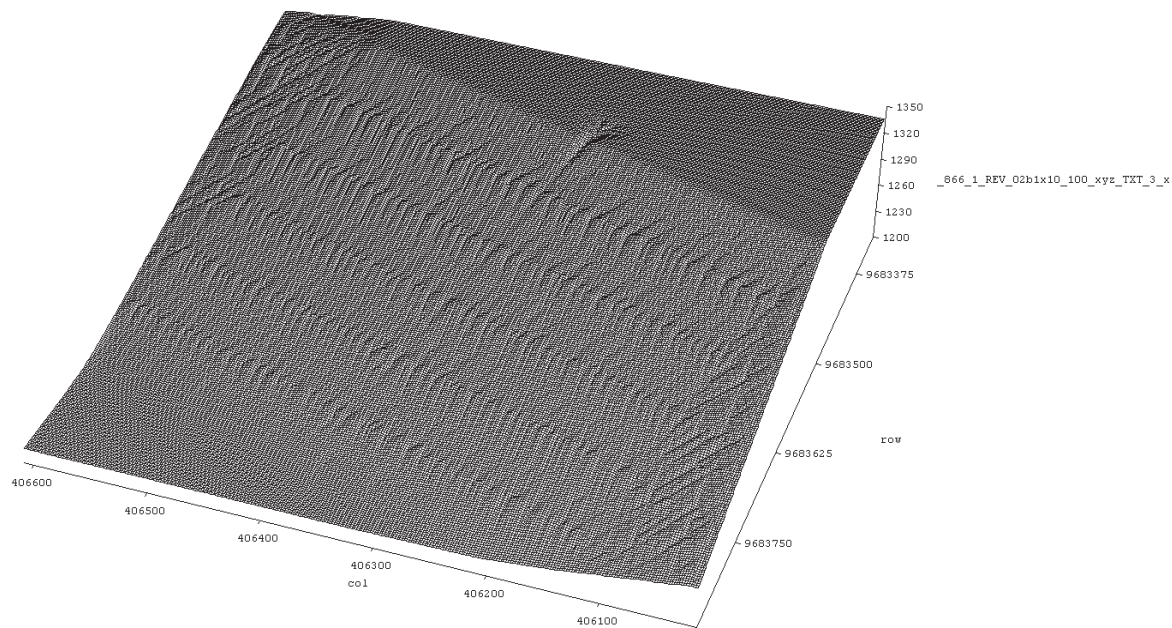


Fig. 7. Current reclamation plan surface – SIBERIA output year 100.

The average erosion rate of the 1V:2.75H linear profile by SIBERIA over 100 years was 74 m³/ha/y (133 t/ha/y). Erosion rates derived for the current WD1 landform in WEPP, (which are approximately in accordance with the current reclamation plan) were 53 t/ha/y. For the 1000 year SIBERIA simulation, the average predicted erosion rate was 58 m³/ha/y (104 t/ha/y).

The average erosion rate of the 1V:3H to 1V:4H dual concave slope calculated by SIBERIA over 100 years was 39 m³/ha/y (70 t/ha/y). Erosion rates derived for the same landform in WEPP were 44 t/ha/y. For the 1000 year SIBERIA simulation, the average predicted erosion rate was 48 m³/ha/y (86 t/ha/y).

4.4 CAESAR-Lisflood

Modelling using the CAESAR-Lisflood model was conducted on the three landforms shown in Figure 5. CAESAR-Lisflood requires a climate database with hourly rainfall rates. The daily climate database described in Section 4.2.1 was modified based on typical intensity rates for the area. Other inputs for CAESAR-Lisflood are particle sizes of the surface materials and a vegetation parameter. Based on the material characterisation programme

conducted in 2012/13, particle size distributions of both fine and coarse-textured topsoil materials were available. CAESAR-Lisflood uses a maximum of nine particle sizes; therefore the original gradation was adjusted to fit this criteria. The adjustment was made by selecting nine representative sizes and all of the sizes that are not used are divided up equally and distributed amongst the nine selected sizes. This typically results in a slightly higher curve, which correlates to a slightly finer-textured material, leading to a slightly conservative estimation. For this modelling, vegetation was considered minor based on visual estimation of the GGM reclamation areas.

As with the SIBERIA modelling, there was no direct way to calibrate the CAESAR-Lisflood model at this time without a DTM of WD1 at Year 0 or direct erosion measurements on the soils. Similar to the process described in Section 4.1, an attempt was made to calibrate the CAESAR-Lisflood model to the estimated total volume of erosion that was observed/estimated for the first 10-year period of WD1 evolution. The model consistently under-estimated the erosion rates and gully depth compared to those shown in the LiDAR surface. Therefore, as with SIBERIA, the CAESAR-Lisflood modelling was used for comparison purposes rather than for quantitative estimates of the erosion on the various conceptual landform designs.

The three conceptual landform designs shown in Figure 5 were simulated in CAESAR-Lisflood over a 100-year time period using hourly rainfall rates. Simulations were considered for both the coarse and fine-textured topsoil materials. The results of the modelling are presented in Table 1.

Table 1. CAESAR-Lisflood average erosion rates for GGM conceptual landform designs evaluated over a 100-year period.

Slope Type	Erosion Rate (t/ha/yr)	Erosion Rate (m ³ /ha/yr)
<i>Coarser-Textured Topsoil</i>		
Current Reclamation Plan	2.0	0.9
Linear Slope 2.75:1	0.9	0.4
Concave Slope 3:1 to 4:1	0.5	0.3
<i>Finer-Textured Topsoil</i>		
Linear Slope 2.75:1	1.3	0.6
Concave Slope 3:1 to 4:1	0.5	0.3

5.0 CONCLUSIONS

Conceptual landform designs were developed for the waste rock dumps at Geita Gold Mine (GGM) in Tanzania using two 3-D commercially available software programs: SIBERIA and CAESAR-Lisflood and a 2-D software program (WEPP). The three conceptual landform designs represented a 500 m x 500 m square area of WD1, which has a similar embankment height and design to other waste rock dumps at GGM. The three conceptual designs evaluated were the current reclamation plan consisting of benches, a linear slope (2.75H:1V) and a dual concave slope (3H:1V transitioning to 4H:1V).

Average annual erosion rates evaluated by SIBERIA over 100 years for the three landforms were 94, 133, and 70 t/ha/yr for the benched, linear, and concave, respectively. The average annual erosion rates evaluated by WEPP over 100 years for the three landforms were 52,

53, and 44 t/ha/yr for the benched, linear, and concave, respectively. The results of the erosion modelling using the CAESAR-Lisflood model were significantly lower; however, the trends were similar. The average annual erosion rates calculated by the CAESAR-Lisflood model were 2.0, 0.9 and 0.5 t/ha/yr for the benched, linear, and concave, respectively.

Work is ongoing to compare the SIBERIA and CAESAR-Lisflood models. Major differences in model inputs are what is leading to the large differences in erosion rates predicted by the two models. SIBERIA uses average annual rainfall rates and storm events for climate inputs and erosion parameters as soil inputs, whereas CAESAR-Lisflood uses soil particle-size distribution and hourly rainfall as input. WEPP uses a different set of inputs, with daily climate and soil erodibility parameters (which can be measured in the lab).

Due to the uncertainty in the calibration of all the models, the results presented here can only be considered as a relative measure to evaluate various landform designs against each other. Based on these results, both the SIBERIA and WEPP results show that the linear slope shows an increase in erosion rates compared to the benched design (current reclamation plan). However, the CAESAR-Lisflood results show a decrease in erosion rate for the linear profile. The benched profiles modelled with SIBERIA and CAESAR were slightly different and it is hypothesized that these results will show similar trends once the same profile is modelled in each model. The ability of the benched design to transmit surface drainage effectively over the life of the landform is critical to erosion estimation. A “perfect” benched landform behaves well in simulation but with when minor variations in the surface are introduced (similar to what would occur due to actual construction constraints), the performance degrades quickly due to sedimentation of channels, leading to ponding and eventual overtopping of the benches. Modelling the same benched profile is part of ongoing work with GGM.

All three models showed the same trend for the concave landform design, which is as expected. All three models showed a decrease in erosion rates for the dual concave profile ranging from 15 to 75% compared to the benched design. Concave slopes represent the shape seen in natural slopes, which are the lower energy state of a landform and therefore more stable. As part of the ongoing work on the conceptual landform design for GGM, additional concave slopes will be evaluated, including those with three slopes, such as 2.75H:1V transitioning to 4H:1V transitioning to 5H:1V.

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DEFINING EFFECTIVE CLOSURE AND RECLAMATION MEASURES FOR TAILINGS AND WASTE ROCK STORAGE: AN AFRICAN CASE STUDY

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ABSTRACT

Open-cut metalliferous mines, where sulfides are prevalent require a detailed and comprehensive understanding of the distribution and volumes of waste rock types, particularly the potentially acid forming (PAF) waste as well as: acid neutralising waste, benign waste for erosion protection, oxidised low sulfur waste that can be used for compacted layers, and soil that can be used for growth medium in the final remediation cover system design. A geological model of the ore and host rocks was developed for this site prior to the commencement of mining using the drillhole database compiled from the exploration and resource definition programs. The mining waste block model was developed from the geological model and the waste characterisation program to identify the volume and distribution of waste rock types within the pit area. Waste scheduling and dump design was developed for selective placement of potentially acid and metalliferous (AMD) generating waste. The effectiveness of covers, their performance over time and final landform design is very much dependent on the initial dump design and subsequent waste management to ensure correct placement of reactive waste.

The current life of mine is planned around 2030. The current tailings storage (TSF) footprint is about 300 ha, and there are several waste rock dumps (WRDs) covering more than 700 ha. The rock storages contain sulfidic material that has the potential to generate acid and metalliferous leachate. Initial assessment at mine commencement identified that the waste volume that will be mined over the life of the mine will comprise 38% PAF waste with elevated sulfate and elevated concentrations of metals in the regional groundwater. The tailings material is sulfidic and enriched with metallic elements of interest. An effective cover system design is required to prevent/limit contamination from AMD to local groundwater aquifers and surface water receptors upon mine closure and storage decommissioning.

This paper outlines the approach to evaluating and designing cover systems suitable for the mine waste storage facilities, including developing appropriate cover performance criteria and conducting a defensible soil-plant-atmosphere (infiltration) numerical modelling program. A detailed geochemical and geotechnical material characterisation program of the current waste storage landform, consisting of material sampling as well as field and laboratory testing, was undertaken at the mature Geita Gold Mine complex in Tanzania to provide design parameters for cover system options and final landform design.

1.0 INTRODUCTION

The modern Geita Gold Mine, owned by AngloGold Ashanti Ltd. and operated by Geita Gold Mining Ltd. (GGM), began to process ore mid-2000. The current life of mine is planned until 2030. There are currently ~300 ha of tailings within two tailings storage facilities (TSFs) (Old TSF and current or New TSF) and over 700ha of waste rock across eight waste rock dumps (WRDs), with four currently active. The materials' composition in these landforms has the potential to generate acidic drainage from natural oxidation of contained sulfidic minerals and result in metal leaching. Previous characterisation has shown up to 38% of waste rock materials as potentially acid forming (PAF), based on initial assessment in 2001.

A closure and reclamation requirement of the Mining Act 2010 Regulations 178 – Tailings Storages, and 201 - Waste Rock Storages is to ensure structural stability and for runoff to be controlled and meet discharge limits. This paper summarises the procedures adopted to define closure options for the GGM tailings and waste rock storages, including material characterisation, cover system design and erosion modelling.

1.1 Site Description

The Geita open-pit mine is situated in Mwanza Region of north-western Tanzania, approximately 90 kilometres from the regional capital of Mwanza and 20 kilometres south of Lake Victoria in an area known as the Lake Victoria Goldfields (Figure 1).



Fig. 1. Location Map

The main GGM operations are located within the Mtakuju River catchment that drains to Lake Victoria 25 km to the northwest. The average altitude at GGM is 1,180-1,350 m ASL for Nyankanga and Geita Hill-Lone Cone site, 1,550-1,620 m for Kukuluma-Matandani mine site and 1,450-1,500m ASL for Nyamulilima Star & Comet mine site to the west.

The Geita District has a highland equatorial wet-dry weather pattern with a bimodal wet season (generally October to December and February to May) with a mean annual rainfall of ~980 mm. A distinct dry season extends from June to August/September with an average annual pan evaporation of ~1,870 mm. The areas annual minimum and maximum temperatures are between 14°C and 32°C.

The GGM operation comprises three mining areas (refer to Figure 2): one centred on the old Geita Hill area comprising several pits: Nyankanga, Geita Hill, Lone Cone and their associated WRDs: WD1, WD5, WD6, WD14, WD15; the second is west of the Geita Hill area around Nyamulilima ridge comprising the Star & Comet pits and associated WRD: WD16; and the third is located ~20 km northeast of Geita Hill and comprises the Kukuluma and Matandani pits and WRDs WD7, WD8 and WD9. Tailings have been deposited in a historic TSF referred herein as the Old TSF and a modern storage facility referred to as the New TSF. Currently gold ore is mined from Nyankanga, Geita Hill and Star & Comet Pits. Operations at Kukuluma and Matandani pits were suspended during the second quarter of 2007. Reserves at Lone Cone pits have been exhausted.

Table 1: Design areas for waste storages

WASTE ROCK DUMPS						
Pit Name	WD Name	Design (ha)	Area	Actual Area (ha) (Dec 2011)	Remarks	
Nyankanga	WD1	376.9		297.4	Active	
Nyankanga	WD5	28.2		26.17	Active	
Lone Cone	WD6	43.2		43.2	Closed	
Kukuluma & Matandani	WD7&9	151.3		152.1	Temp closed	
Kukuluma	WD8	34.7		34.7	Closed	
Geita Hill	WD14	124.7		115.3	Active	
Geita Hill	WD15	31.6		31.6	Closed	
Nyamulilima -Star & Comet	WD16	57.6		9.3	Active	
TAILINGS STORAGEES						
New TSF		535.8		280.75	Active	
Old TSF		17.92		17.92	Closed	

2.0 GEOLOGY

GGM is located in the Geita Greenstones that form the northern arm of the regional Sukumaland Greenstone Belt of the Archaean Tanzanian Craton. Stratigraphically the Sukumaland Greenstone Belt belongs to the Neoarchaean Nyanzian Supergroup (Barth, 1990; Borg and Shackleton, 1997). The Geita deposits are hosted in Archaean-age rock Upper Nyanzian formations characterised by banded iron formation (BIF), felsic volcanics and andesite/diorite lithologies (Sibilski and Stephen, 2006). The BIF outcrops in all of the high ground in the area, while felsic volcanics occur in the lower flanks of the ridges and are either inter-bedded within the BIF or occur either side of it. The other main lithology in the Geita area is a trachyandesite, encompassing a suite of volcanic rock types, ranging from basalt to diorite in composition. The trachyandesite units are commonly interbedded and folded with the BIF. The volcanics, which host the gold mineralisation, have been

metamorphosed to lower greenschist facies. At Nyankanga the principal waste rocks are microdiorite and BIF. The Kukuluma host waste rock sequence is more variable and includes mafic to felsic volcanics, BIF and metasediments. The western section of the belt, in the GGM area, is cut by regional scale Proterozoic quartz-gabbro dykes with a strong north-east trend. The area is dominated by both NW and NE structural trends, with a weaker NNW trend occupied by Karoo dolerite dykes. The greenstone belts of the Tanzanian Craton are characterised by more localised and discontinuous structures.

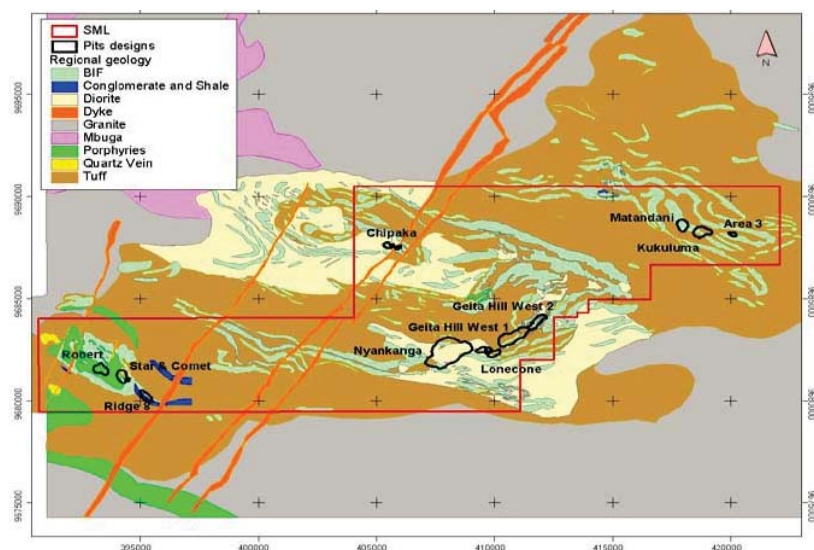


Fig. 2. Geology of the Geita Mine areas

2.1 Mineralogy

Gold mineralisation is typically associated with quartz veins (inner belts) and disseminated sulfides (outer belt) in quartz-carbonate-chlorite shear zones replacing and crosscutting magnetite-rich BIF. Gold mineralisation can also be associated with massive and disseminated sulfide bodies on contacts between felsic tuffs and BIF. The principal sulfide present in the shear and vein-hosted deposits is pyrite, with minor pyrrhotite and trace arsenopyrite, chalcopyrite, galena and sphalerite. Gold occurs as free native gold in quartz veins and as inclusions or surface adherents to iron sulfides. Carbonate minerals that are present in the mine sequence are dominated by calcite with accessory dolomite, ankerite and siderite. The carbonates occur as fine-grained pervasive mineralisation in the altered BIF matrix and as coarser grained calcite in veins. The supergene mineral assemblage is dominated by oxyhydrates of iron and aluminium.

2.2 Regolith

The regolith development is not uniform, with depth of weathering dependent on lithology, topography and elevation. At Nyankanga it is 15-80 m deep with dark red ferralitic soils overlying duricrust and a deep saprolite of mottled clays. At Kukuluma the depth of weathering is much deeper in the vicinity of the ore zones, probably related to oxidation of primary sulfides within and around the shear zones hosting the gold mineralisation.

3.0 WASTE ROCK MANAGEMENT

3.1 Construction of the Mine Waste Block Model and WRD Design

The presence of sulfides in the host rocks to the GGM gold mineralisation necessitated a set of waste management procedures be developed prior to commencement of mining to ensure effective WRD design and correct placement of reactive (PAF and metal leaching) waste to minimise and manage acid and metalliferous drainage (AMD). These procedures included:

- Waste rock and tailings characterisation using acid base accounting (ABA), metals analysis, kinetic testing of representative waste rock and ore;
- Construction of the geological model of the waste, based on drillhole database (including geology and pyrite distribution) and geochemical characterisation of the waste lithologies;
- Block modelling of waste rock and merging with the ore resource model, to create the life of mine (LOM) waste mining schedule;
- Validating the waste model using in-pit geological mapping;
- Selective handling and placement of waste in designated areas of the WRD and the New TSF embankments;
- Validating placement of waste within the WRD and tailings dam embankment;
- Monitoring for success of placement using piezometers within the WRD, tailings embankment and downstream of these facilities;
- Regular technical reviews monthly (internal), three to six monthly (external);
- Implementing these procedures for LOM.

Initial waste characterisation for construction of the waste block model was undertaken on selective samples of potential waste rock for each deposit. A total of 488 representative drillhole waste rock samples from Nyankanga, 127 for Geita Hill, and 1016 from Kukuluma/Matandani were selected for characterising *in situ* waste and to construct the waste geological model for each deposit respectively. Initially 60 metallurgy samples were tested to characterise the anticipated tailings composition. A similar program was adopted for the other deposits (e.g. Lone Cone, Star & Comet etc.).

Samples were analysed for Total sulfur (LECO), sulfate sulfur, sulfide sulfur (CRS) to estimate acid potential (AP), acid neutralising capacity (ANC), net acid generation (NAG), and metals: Sb, As, Bi, Cd, Ca, Cr, Co, Cu, Fe, K, Pb, Mg, Mn, Hg, Ni, Se, Ag, Na, Ti and Zn. Samples were classified as PAF or NAF (non-acid forming) based on the ABA results (Sobek *et al*, 1978; Coastech, 1989), NAG (Miller *et al*, 1997) and ANC:AP ratio where NAF was waste rock returning an ANC:AP of >3 and/or sulfide sulfur <0.1%.

The waste model for each deposit was defined from the exploration drillhole database utilising geological and ABA data on selected drill holes. The geological and geochemical data included:

- Lithology - major and minor (host rocks, cover rocks, construction material, soil, alluvium);
- Structural features - faults, shears, veins, fractures;
- Sulfide and carbonate mineralogy;
- Sulfide and carbonate texture and morphology;
- Spatial relationship of sulfides with acid consuming minerals in the host rocks;
- Depth of oxidation to include base of total oxidation (BOTO) and base of partial oxidation (BOPO);
- Depth of water table;

- Sulfide sulfur, acid neutralisation and metals analysis of the principal waste lithologies expected within the pit area proximal and distal to ore;
- Defining the transition waste areas to identify areas of partially oxidised PAF waste and flag these areas to be included in the waste model as reactive waste.

The waste rock distribution was defined for each waste type for the proposed pit areas using the following protocols:

- 10 metre bench plans (1:1000 scale) are constructed depicting the distribution of PAF and NAF waste blocks, ore blocks and pit boundaries are flitched for each Pit Cut.
- Waste blocks range in size from 20x20m external to the ore between 2.5x5.0m and 20x20m internal or adjacent to ore.
- Waste domains comprising large minable blocks of waste are defined according to 4 waste types and based on the percentage of PAF waste present in the Domain:
 - Type 1 Waste comprising 100% NAF and negligible PAF. NAF waste;
 - Type 2 Waste comprising up to 15% PAF. NAF waste;
 - Type 3 Waste comprising between 15 and 50% PAF. PAF waste; and
 - Type 4 Waste comprising >50% PAF. PAF waste.
- 10m flitches of the geology corresponding to the waste bench plans (1:1000 scale) are constructed; the flitched geology plans show the distribution of rock types within each Pit Cut.
- The geology is transferred onto each 10-metre bench plan for each Pit Cut.
- The bench plan depicting waste domains, geology and ore blocks, is the composite field map to be utilised by the pit geologist for validating waste domain classification and boundaries.
- Pit Geologist must undertake in-pit validation of the geological boundaries and the waste domain classification.

In-pit validation of waste by Pit Geologist involves:

- Checking the boundaries of the domains against exposed geology in the pit floor on the current mining bench; the pit geologist should utilise the geological information in the pit walls as well as the pit floor.
- Checking domain classification is correct.
- Correcting the boundaries and classification.

Detailed in-pit geological mapping is carried out by mine geologists on each production bench focusing mainly on the lithological contacts, structural features, alterations and sulfide mineralisation. A direct field-testing method to establish the reactivity of the rock using H_2O_2 is used when visual identification of PAF material from sulfide content is unable to be established. The method, designed by the authors, is based on the van Beers field oxidation pH test (Ahern *et al*, 2004) and is equivalent to a Field NAG pH test.

Following blasting, the waste materials are identified by the Mine Geologist and marked as ore or waste: PAF or NAF, for despatch to designated areas. Segregation of waste in the pit continues until their final destination on the WRDs, where PAF and NAF are dumped separately. WRD design facilitates PAF encapsulation, with a NAF base layer 10-20 m thick, PAF placed in the middle each dump lift, and an outer NAF layer between 80-100 m on each dump lift to complete the encapsulation. The basal layer is constructed by paddock dumping and then worked with a dozer to form the flattened base layer. Tipping of PAF material within the internal part of the dump is commonly by paddock dumping and worked continually by dozer to form the dump lift. Tipping of NAF material to form the outer layer is usually done at 35-37° batter angle depending on type of material (oxides to fresh), interspersed with 36 m wide berms at 20 m vertical increment lifts and the final batter angle is <20°.

Compaction of materials occurs as dumping progresses, therefore minimising infiltration of water into the dump.

4.0 FINAL COVER SYSTEM

4.1 Cover System Design

GGM closure requirements involve the installation of a mine waste cover system that will result in restoration of the waste deposits' surface to a stable, natural condition while minimising degradation of the surrounding environment, (Tanzania Mining Act 2010 Regulation 178 and 201; MEND, 2012). Figure 3 summarises the methodology adopted for waste facility closure, through developing site-specific performance criteria for a cover system designed to control metal leaching. The methodology links the predicted cover system performance to groundwater, surface water, and air quality impacts. The objective is to determine the appropriate level of control,- such as net percolation (NP), required by the cover system.

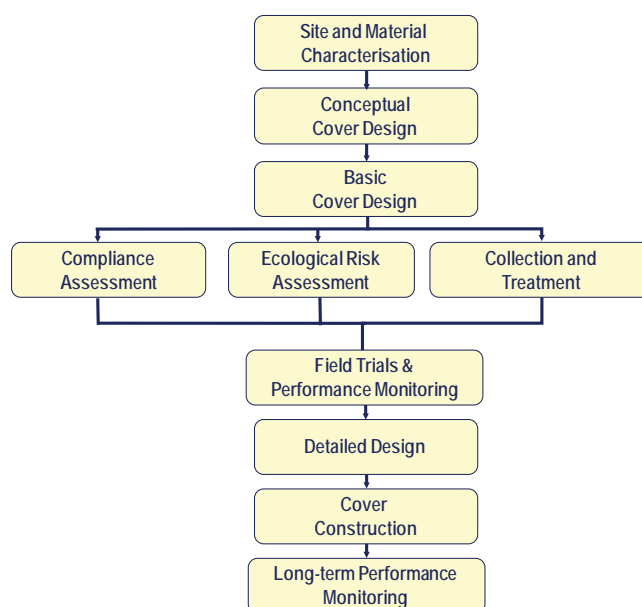


Fig. 3. Flow chart of the cover design process (adapted from O’Kane and Wels, 2003).

Cover systems over waste material can have numerous objectives, including the following that are relevant to GGM: isolation of waste; limiting influx of atmospheric oxygen; limiting influx of atmospheric water; controlling erosion of waste material; control upward movement of process-water constituents/oxidation products; and providing a medium for establishing sustainable vegetation.

One of the main purposes of placing cover systems over reactive waste material is to protect the downstream receiving environment following waste storage facility closure (O’Kane and Wels, 2003). This is achieved by reducing NP of meteoric water into the mine waste, which reduces effluent seepage volumes. This reduction in seepage volumes ideally limits peak concentrations of contaminants in receiving waters to levels that can be assimilated without

adverse impact to the aquatic ecosystem. In addition to controlling contaminant releases, cover systems can also provide chemical and physical stabilisation of waste material and a growth medium for establishment of a sustainable vegetation canopy.

The suitability of cover systems to control NP as well as run-off will depend on site-specific climate conditions; material availability; and the required performance criteria. Figure 4 illustrates generic cover system profiles. Water infiltrates during periods of high precipitation or spring melt in cold regions. The infiltrated water is stored within the cover until atmospheric and biotic demands are able to remove the water through evaporation and transpiration.

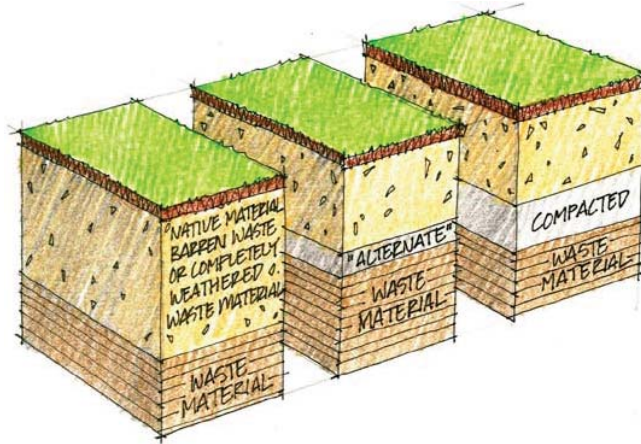


Fig. 4. Schematic of cover systems with store-and-release functionality: (left) basic store-and-release cover system; (middle and right) enhanced store-and-release cover systems showing additional lower hydraulic conductivity layers below the storage layer (from MEND, 2012).

The design philosophy for a cover system at GGM is one that integrates the waste material within its environmental context. This is in contrast to isolating the waste from the environment in order to completely prevent the production of contaminated seepage. A mine waste cover system, whether for a TSF or WRD, must be designed as an unsaturated system exposed to the atmosphere, with performance significantly influenced by seasonal, annual, and long-term site climate conditions. A key factor to consider during the design process is the anticipated climax vegetation species that will develop on the cover system. A vegetation cover is important not only for minimising soil erosion, aesthetics and creating wildlife habitat, but lower net infiltration rates can be realised with a mature vegetation canopy due to higher rates of interception and evapotranspiration.

The design for a mine waste cover system for GGM is being developed with respect to key factors that will control long-term performance. These key factors include: the climate regime at the site; the reactivity and texture of the mine waste material; the geotechnical, hydrologic, and durability properties of economically available cover materials; the hydrogeological setting of the waste storage facility; and long-term erosion, weathering, and evolution of the cover system.

While it is economically feasible in humid, tropical climates modified by altitude to construct a cover system that contains a highly saturated layer that limits gas transport across the cover system into the interior waste material, it is also necessary to include effective drainage to

prevent breakthrough under heavy rainfall conditions that can prevail at GGM. The cover system will be subjected to distinct dry periods as well as intense rain events, and therefore the effect of evapotranspiration can be significant. However, subjecting the cover system to evaporative demands can be beneficial in humid climates with distinct dry periods, such as the climate at GGM, and result in a reduction of infiltration to the underlying waste material. The objective is to control metal leaching and sulfate salinity as a result of controlling the transport mechanism (i.e. water) into the underlying waste material.

4.2 Cover System Construction Materials Characterisation

It is common to utilise NAF ROM and non-metal leaching material when constructing cover systems such as those that utilise the moisture store-and-release concept to manage mine waste for closure. As a result of this, there is a desire to directly place the ROM material used for the cover system as part of mining operations, with over haulage costs kept to a minimum. Ideally, this results in the cost of closure to be an incremental increase in the overall project costs, as opposed to being substantially higher if cover system construction was left to the end of operations. However, it is not yet fully appreciated by the mining industry that placement of the ROM materials, which do not meet the textural envelope for the ROM material used as part of design, has a substantial adverse impact on performance. The most common issue is placement of ROM material that is too coarse textured such that moisture retention within this cover layer is insufficient; in terms of providing a medium for surface infiltration to be stored near surface, and subsequently evapotranspire. The result is a failure of the cover system to store the required volume of moisture for any given rainfall event, and this infiltration then leads to higher NP rates. In many instances, the coarser textured material results in 'macro-pore' flow during higher intensity rainfall events, which leads to rapid infiltration to the base of the cover system, and ultimately higher than expected NP.

A material characterisation program for geochemical characterisation and rehabilitation cover design was undertaken. Samples of waste rock, tailings, and topsoil materials for detailed characterisation were collected to determine, with a high level of certainty, the geotechnical and geochemical characteristics of the waste and stockpile materials, and the fertility of the topsoil materials that potentially will be utilised in construction of final cover systems and landform design.

Sampling of the current near surface waste rock, old tailings, and new tailings was undertaken to understand the current and future pollution potential of ROM material that potentially will be used in cover design, especially potential AMD and metal leaching. Ninety-nine material samples were collected, comprising 28 of waste rock, 9 of tailings, 20 of soils, and 42 samples for geochemical analysis including soil fertility on topsoil samples. Samples were collected from eight WRDs (WD1, WD5, WD6, WD14, and WD15 at Geita Hill-Nyankanga; WD7, WD8, and WD9 at Kukuluma-Matandani; and WD16 at Star and Comet), two TSFs (Old TSF and New TSF), and from topsoil associated with rock and tailings storage facilities. Waste rock samples were collected from test pits excavated in the top plateau portions of each WRD using a tracked excavator fitted with a 1.5 m bucket. Test pits were excavated to a total depth of 4 m, compaction and depth limited further excavation. A full bucket sample was collected from varying depths between pits ranging from 1–4 m in depth based on composition and representativeness of the material exposed in the respective test pit. Samples screened through a 100mm "grizzly"; the oversize volume was estimated and recorded; the undersize was sampled for geotechnical properties testing.

4.2.2 Geotechnical material characterisation

Geotechnical material characterisation was undertaken to quantify the materials for geotechnical stability, hydrologic behaviour, and use in landform structures and as covers during rehabilitation. These characteristics are critical for all aspects of numerical modelling and design. The geotechnical characterisation testing included gravimetric water content (GWC), particle size distribution (PSD), Atterberg Limits, Standard Proctor, Emerson crumb tests, specific gravity (G_s), and moisture retention curves (MRC). Laboratory and field saturated hydraulic conductivities (K_{fs}) (Guelph permeameter tests) of soils were also measured. The tests were undertaken in accordance with Australian Standards.

4.2.3 Geochemical material characterisation

Forty samples of tailings and waste rock derived from seven WRDs and two TSFs were submitted for acid base accounting and metals analysis. Topsoil samples were assessed for soil fertility.

4.2.3.1 Acid base accounting

ABA data show that over half of the samples tested had S_{CR} (sulfide sulfur) concentrations $>0.1\%$ sulfur. Note that below 0.1% sulfide sulfur there is limited potential to generate acid drainage. The test data results were quite variable, with samples from Nyankanga, Geita Hill, and Star & Comet dumps WD1, WD5, WD14, and WD16 exhibiting elevated sulfide sulfur concentrations. The Kukuluma and Matandani dump material is oxidised and, as expected, has negligible sulfide sulfur and total sulfur. Also expected is the elevated sulfide sulfur concentrations in the New TSF and low to negligible sulfide sulfur present in the Old TSF material. ANC of the samples analysed follows a similar trend, with negligible ANC in the Kukuluma and Matandani dump cover materials, and elevated ANC in Nyankanga, Geita Hill, and Star & Comet materials. There is also some excess ANC present in samples tested in some of the dumps, identified by ANC:AP ratios >3 . In addition, there is some ANC in the tailings; however, in all samples tested the ANC:AP ratio is <2 .

4.2.3.2 Metals geochemistry

Metals were analysed by ICPOES/ICPMS method using a 4 acid near total digest. This included ICPOES for: Al, As, Ba, Be, Bi, Ca, Cd, Co, Cr, Cu, In, Fe, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, Sn, Sr, Ti, Tl, V, Zn. ICPMS was used to analyse for W, U, Ce, Cs, Ga, Hf, La, Nb, Rb, Re, Sc, Ta, Te, Th and Y. Mercury was analysed by Cold Vapour AAS from a nitric acid, hydrogen peroxide and hydrochloric acid digest. Element enrichments were identified using the Geochemical Abundance Index (GAI). Leachate analysis was undertaken using two methods: 1) Shake flask solubility test based on a modified ASLP test using DI water; 2) NAG leachate test. Leachates for both tests were analysed for: pH/EC, Ag, Al, As, Ba, Ca, Cd, Co, Cr, Cu, F, Fe, Hg, In, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, SO_4 , Se, Sr, Ti, V, Zn. Geochemical data was used in the contaminant flow modelling.

4.2.3.3 Soil fertility

The soil fertility assessment was undertaken to determine the potential for topsoil materials to be used as an effective growth medium. Vegetation is a critical aspect to reclamation success both for the ability of vegetation to remove and transpire water, to stabilise slopes and prevent erosion, for visual appeal, and to provide for final land-use. A total of 16 topsoil samples were tested for a range of soil fertility parameters including: sodium adsorption ratio (SAR); pH and EC – soil:water (1:5); organic carbon; available phosphorus – $0.5M NaHCO_3$;

exchangeable K, Ca, Mg – 1M NH_4OAc ; available sulfate – 0.01M $\text{Ca}(\text{H}_2\text{PO}_4)_2$; available Fe, Mn, Zn, Cu – 0.005M DTPA; available Boron – hot water or 0.01M CaCl_2 ; macronutrients: Total N, P, K, Ca, Mg, S; micronutrients: Total Cu, Zn, Fe, Mn, B, Mo, Cl; additional Micronutrients: Mo, Cl; toxic elements: Al, As, Cu, Cd, Cr, Hg, Mn, Mo, Ni, Pb, Se etc., and chloride -- soil:water (1:5)

4.3 Numerical Modelling

4.3.1 Soil-atmosphere modelling

Alternate cover system options were evaluated using a soil-plant-atmosphere (SPA) numerical model coupled to site climate and *in situ* conditions, suitable and available endemic vegetation species, as well as the laboratory and field material properties (e.g. moisture retention, hydraulic conductivity, and consolidation) compiled for the material characterisation program discussed in Section 4.1. VADOSE/W v2012 (Geo-Slope International, 2007) was utilised to conduct SPA cover design modelling required for cover system design optimisation. A probabilistic approach, in conjunction with a cost-benefit analysis, was used to determine the preferred cover system designs for the TSF and WRDs from the perspective of reducing NP. The probabilistic approach provides a means of estimating 'risk', or the 'probability of exceeding' a certain NP rate (as a percentage of annual precipitation), for the different cover system designs evaluated. In order for this approach to be representative, a relatively long climatic database must be simulated with each year of the database modelled, for each different cover system design. This probabilistic approach also provides the opportunity to evaluate the impact on cover system performance as a result of extreme climatic conditions and storm events (e.g. high intensity, short duration rainfall events, and/or extended dry periods).

4.3.2 3D-Groundwater seepage modelling

To advance the conceptual reclamation objectives into a detailed cover system design, it was necessary to define the cover system performance criteria with respect to impacts on downstream receptors. The following tasks were completed to develop these criteria:

- Development of source terms for contaminant of concern from both TSFs and WRDs; derived from surface and groundwater monitoring data, kinetic testing of waste rock, leachate analysis of waste rock and tailings;
- Development of a calibrated three-dimensional (3-D) groundwater model to characterise flow patterns of surface water, near-surface, and deep groundwater in the region.

The development of a site-wide seepage and groundwater flow model in order to optimise cover system designs for GGM WRDs and TSFs was undertaken. The finite element groundwater modelling system FEFLOW 5.4 was used to simulate 2-D or 3-D steady-state or transient flow, and mass and heat transport. Long-term NP rates predicted from SPA modelling were used as input for FEFLOW as the upper boundary conditions. The numerical model is being used to evaluate the effect of varying levels of cover performance on downstream receptors. 3-D groundwater modelling was coupled with geochemical source terms to develop a solute transport model. This key numerical model will allow determination of the required cover performance criteria. Along with defined compliance and water quality requirements, the interaction of the waste facilities and the surrounding regional watershed is being evaluated to determine the magnitude of contaminant loading reduction (and therefore cover system requirements).

4.4 Landform Design

To meet the reclamation objectives, a stable landform is required to ensure the long-term success of the TSF and WRD covers. The objective of the landform design is to determine respective final slope grades and surface water management systems.

The approach adopted to designing final landforms for the WRDs and TSFs is similar to that outlined in Ayres *et al.* (2006) and at a minimum, takes into consideration the following aspects:

- Long-term geotechnical and geomorphic stability;
- Aesthetics (i.e. visual blending with the surrounding landscape to the extent possible);
- Incorporation of adequate armouring in surface water drainage courses to withstand flow velocities associated with the design storm event; and
- Minimising long-term maintenance liabilities associated with the decommissioned waste storage facility.

Based on the authors' understanding of the site and experience with similar projects, the following approach was undertaken for the design of the TSF and WRD landforms at GGM:

- Determine the design storm event;
- Develop conceptual TSF and WRD landforms and surface water management systems;
- Evaluate and optimise the conceptual landform with numerical analysis (WEPP and SIBERIA);
- Evaluate tailings mass settlement with tailings consolidation analysis; and
- Finalise detailed design of surface water runoff channels to reduce surface water degradation and erosion.

5.0 CONCLUDING REMARKS

The paper outlines the waste management practices that were established from the commencement of mining at GGM to contain and manage reactive, acid- generating mining waste. It outlines the approach to evaluating and designing cover systems suitable for the mine waste storage facilities at GGM as part of the closure planning process, including developing appropriate cover performance criteria and conducting a defensible SPA (infiltration) numerical modelling program. The effectiveness of covers, their performance over time and final landform design is very much dependent on the initial dump design and subsequent waste management to ensure correct placement of reactive waste.

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EVALUATION OF COVER SYSTEMS UTILISING GEOSYNTHETIC LAYERS CONSTRUCTED IN A SEASONALLY HUMID GEOGRAPHIC LOCATION FOR THE CLOSURE OF COAL WASTE ROCK PILES

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ABSTRACT

Enterprise Cape Breton Corporation (ECBC) implemented a remediation program for the closure of historic coal mines located near Sydney, Nova Scotia. Waste rock piles (WRPs) at Langan, Scotchtown Summit (Summit) and Victoria Junction (VJ) were reclaimed with dry cover systems.

Summit and VJ were reclaimed with a cover system comprising of a 60 mil HDPE geomembrane and growth medium layer. VJ includes a granular drainage layer above the HDPE geomembrane. Langan was reclaimed with a growth medium layer placed directly on the waste rock and does not include a geomembrane. Instrumentation was installed to facilitate evaluation of their performance over time under site-specific climate conditions. The primary objective of the monitoring program is to develop a database of cover system moisture and thermal field responses, as well as internal WRP dynamics and groundwater conditions within the WRP footprint.

A brief overview of the WRP field performance monitoring systems is provided along with preliminary performance of the cover systems. The reclaimed waste rock piles display unique water balances and water dynamics. This study will provide an improved understanding for the performance of cover systems that utilise geosynthetic layers.

1.0 INTRODUCTION

In terms of composite covers, which include a geomembrane layer, water and oxygen can move through the geomembrane by diffusion, but the transmission rates are very low. In general the hydraulic conductivity corresponding to water diffusion is on the order of 10^{-12} to 10^{-15} cm/s (Giroud and Bonaparte, 1989). Considering that geomembranes are perceived as being essentially impervious when devoid of defects, performance monitoring programs are traditionally focused on water quality analyses of seepage discharged from the waste storage facility. However; this rational is somewhat flawed in that net percolation alone does not provide an understanding of cover system performance.

A cover system should provide for a stable landform and the establishment of sustainable vegetation. Landform instability is a common factor leading to failure of mine waste cover systems around the world (MEND, 2004). The greatest physical risk to reclaimed waste storage facilities is associated with surface water management. This is particularly important in composite cover systems where geosynthetics are placed within earthen layers. Cover systems with a geomembrane layer typically include a granular or geosynthetic drainage

layer to increase slope stability through limiting the buildup of pore-water pressures. Holding to the conceptual idea that the geosynthetic layers will perform as intended, the longevity of the geosynthetics and subsequent long-term performance of the cover system will be a function of the performance of cover layers placed above the geomembrane.

Given the aforementioned a monitoring program should be designed to measure the various components that influence the performance of a cover system. This would include components of the water and energy balances, as well as oxygen ingress rates, as shown schematically in Figure 1. MEND (2000) provides a detailed overview of field performance monitoring for cover systems. In terms of field performance monitoring for a full-scale cover system, a recommended minimum level of monitoring would include meteorological monitoring, such as determination of the potential evaporation (PE) and site-specific precipitation, cover material moisture storage changes, watershed or catchment area surface runoff, vegetation, and erosion (MEND, 2004).

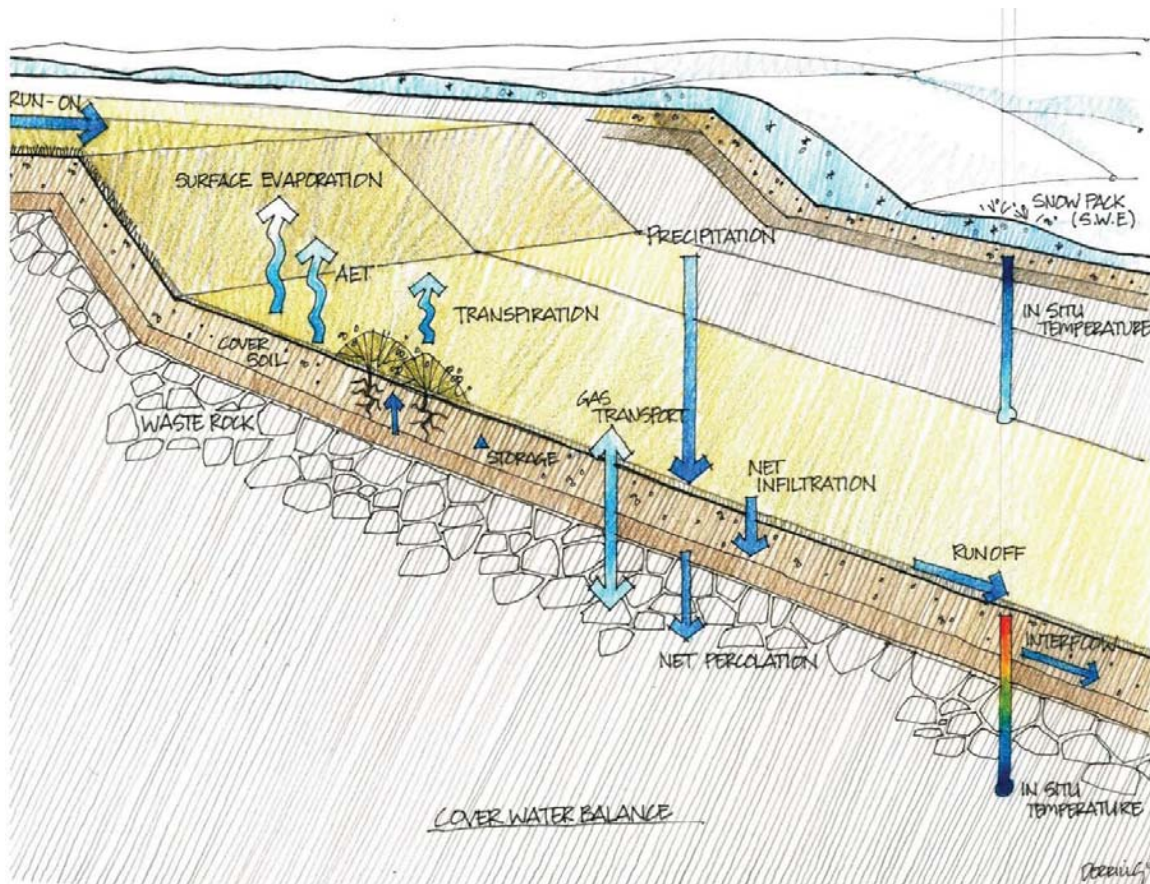


Fig. 1. Conceptual schematic of the components of a field performance monitoring system

2.0 BACKGROUND

Cape Breton Development Corporation (CBDC) was established as a Crown corporation in 1967 in order to reorganise and rehabilitate the coal industry on Cape Breton Island. In 2009, CBDC was dissolved and its assets and liabilities were transferred to Enterprise Cape

Breton Corporation (ECBC), a federal Crown corporation. Under the transfer arrangement, ECBC acquired stewardship obligations stemming from CBDC's past operations, including land holdings and environmental remediation.

Properties covered under the environmental remediation program stem from mining operations that began in 1685 and includes over 50 underground mines which produced over 500 million tonnes of coal. The history of coal mining in the Sydney Coal fields included 720 individual parcels of land on which there were 95 coal related operations covering over 1,000 km². Some of the properties required remediation of WRPs produced from the mining operations.

Dry cover systems were implemented as part of the closure plan for seven (7) WRPs. Public Works and Government Services Canada (PWGSC), under ECBC, provided project management for the remediation program. Local engineering consulting firms, through standing offer agreements with PWGSC, were engaged to develop detailed closure plans for the WRPs. In general, there were four cover system designs utilised for closure of the WRPs. ECBC implement field performance monitoring systems to develop an understanding for the performance of each of the cover system designs utilised for closure. This paper summarises the field performance of three of the reclaimed WRPs including Langan, Summit and VJ.

The waste rock within each WRP is currently acidic with the potential to continue to generate acid mine drainage (Phase, 2010). The Langan WRP is located at the former Langan Mine Colliery in New Waterford, approximately 16 km northeast of Sydney. The Langan WRP covers an area of 8.5 ha, is 15 m high and contains approximately 380,000 m³ of waste rock (Senes, 2009). The WRP has a plateau sloping at 3% from the center that transitions to 5:1 (H:V) side slopes. In 2010 the WRP was reclaimed with a 0.5 m nominally thick till growth medium layer placed directly on the waste rock. A perimeter ditch was constructed around the plateau which channels runoff waters to drop structures on the side slopes.

The Summit WRP is located in Scotchtown, approximately 15 km north of Sydney. The Summit WRP was used by the Dominion Coal Company for the placement of coal waste rock from nearby Collieries over the period 1911 to 1973. Between 1981 and 1987 the waste rock was reprocessed to recover residual coal, which resulted in an increase in the footprint. The WRP thickness varies from 1.5 to 10 m with the thickest deposits near the centre and covers an area of approximately 44 ha. The WRP plateau has a grade of approximately 2-3% and side slopes of 7:1. The WRP was reclaimed in 2011 with an engineered cover system that includes 60 mil high-density polyethylene (HDPE) layer placed below a growth medium layer constructed of processed till (35 mm minus).

The VJ WRP is located on the site of a historic coal preparation plant, approximately 3 km east of Sydney. The coal preparation plant was commissioned in 1976 and remained in operation until the Phalen Colliery closed in 2000 and then remained a blending facility until 2001. Waste rock was consolidated into a WRP with a footprint of approximately 26 ha and height of 40 m. The WRP plateau has a grade of 7% and 3:1 side slopes. Runoff and interflow on the plateau is collected within a perimeter ditch which channels these flows to drop-structures on the side slope to a collection ditch around the perimeter of the WRP. The WRP was reclaimed with an engineered cover system comprised of a till growth medium layer placed over a granular drainage layer (GRDL) and 60 mil HDPE.

Figure 2 provides a schematic of the engineered cover system profiles utilised for closure of the Langan, Summit and VJ WRPs.

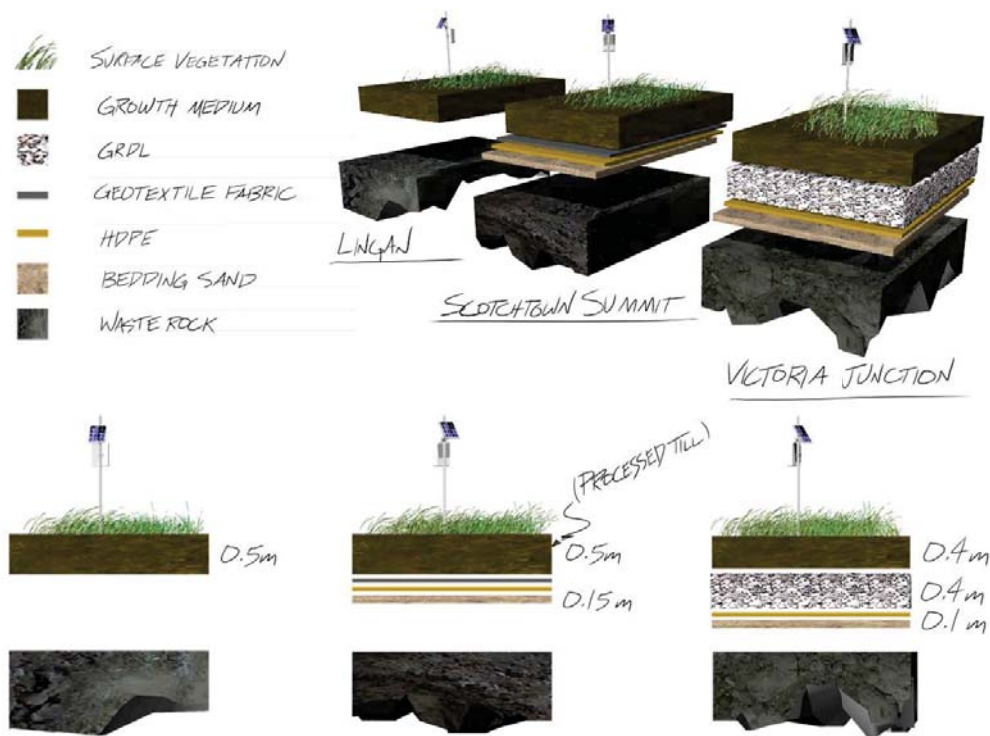


Fig. 2. Schematic of the reclaimed Lingan, Summit and VJ cover system profiles

3.0 COVER SYSTEM PERFORMANCE MONITORING

A multi-phase, multi-discipline monitoring program was implemented in support of evaluating the performance of the cover systems and to achieve the following objectives:

1. Obtain a water balance for the sites, and more specifically, for the cover systems;
2. Identify and characterise key mechanisms and processes that control performance;
3. Track the evolution of cover performance in response to site-specific physical, chemical and biological processes;
4. Obtain a representative set of field performance monitoring data to calibrate a soil-plant-atmosphere numerical model and ultimately predictions of long term cover performance;
5. Improve the design and selection processes of geosynthetics placed within cover systems; and
6. Develop confidence with all stakeholders with respect to closure performance of the WRP.

The monitoring system for each WRP includes a meteorological station, a v-notch weir for measuring runoff flows and interflow, four automated stations for measuring *in situ* moisture and pore-gas concentrations above and below the HDPE layer. Four systems were installed to allow for monitoring of the WRP internal conditions (pore-gas and pressure, ground water quality and levels, and temperature). Measurements of actual evapotranspiration are

collected with an Eddy Covariance system. In addition, a conservative tracer was applied to develop estimates of net percolation.

3.1 Climate

Cape Breton is in a seasonally humid region of Canada. Mean annual precipitation (PPT) and potential evaporation (PE) are approximately 1,500 mm and 650 mm, respectively. Slightly less PPT occurs during the summer (May to September) with approximately 97 mm of rainfall per month compared to 145 mm for the remainder of the year. In the winter (December to March) relatively equal proportions of rain and snow occur as precipitation. The atmospheric demand for moisture during the winter is low, typically less than 20 mm per month, which increases in the summer to greater than 100 mm. Figure 3 summarises the 2012 monthly average PPT and PE calculated utilising the Penman (1948) method for the Summit WRP. Given that actual evapotranspiration (AET) is always less than PE, PPT may exceed AET during the summer periods.

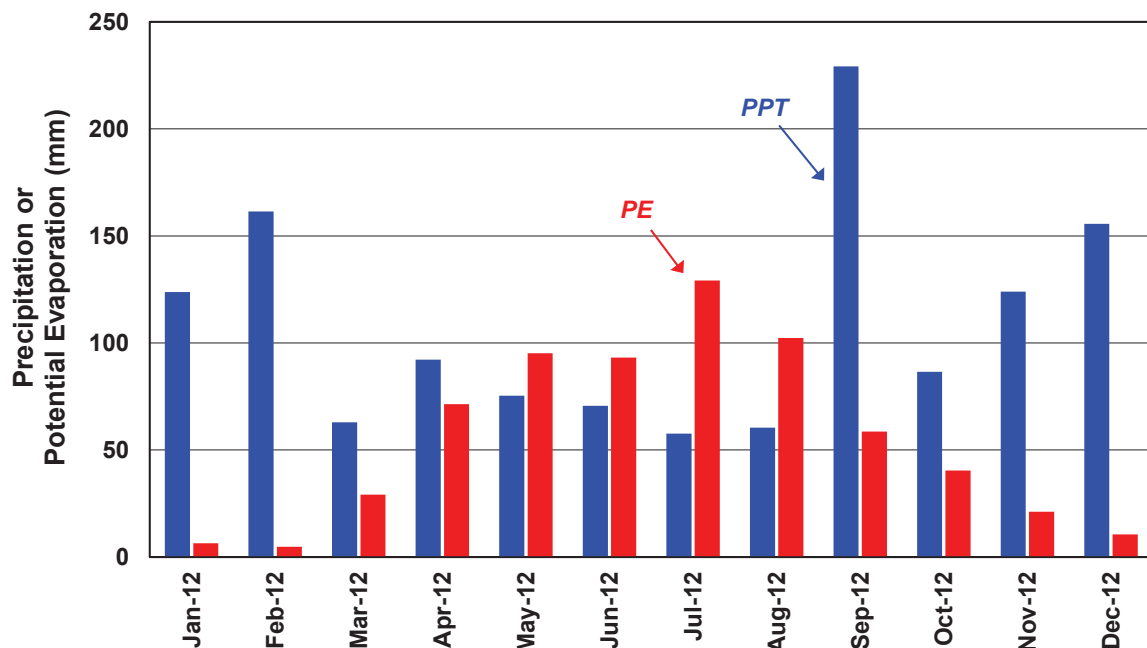


Fig. 3. Monthly PPT and PE in 2012 for Summit

Cover systems constructed within seasonally humid climates, which exhibit variation in atmospheric moisture demand, are required to manage a significant quantity of PPT through components of the water balance (i.e. runoff, interflow, net percolation and changes in cover water storage). Realising that net percolation rates are low for cover systems that utilise a geomembrane, demands are placed on the landform water management system to ensure the long-term performance / stability of the cover system is sustainable.

3.2 Pore Water Pressures

Stability and erosion issues can occur in cover systems with a geomembrane layer that do not include a drainage layer in the design. In general, geomembranes are impermeable limiting water ingress to the underlying waste material. The absence of a drainage layer coupled with a low flux barrier layer can lead to the development of positive pore-water pressures within the cover profile above the geomembrane. Positive pore-water pressures

reduce the effective normal stress and thus reduce the restraining friction along the geomembrane interface, resulting in a potential slope failure. The Summit WRP characteristic of shallow slopes and a large surface area does not include a drainage layer, except for a small section (~1.5 ha) which provided the steepest slope gradient. It is expected that the exclusion of a drainage layer for the remaining WRP surface area (~42.5 ha) was attributed to the shallow slopes.

While slope failures are generally restricted to steep slope angles where the restraining friction may be exceeded, seepage erosion can occur along shallow slopes. Seepage erosion occurs when particles are carried out of the soil mass under a hydraulic gradient. Particle movement is initiated as soon as the seepage force is greater than the particle self-weight and inter particle forces. A seepage face may occur within a cover system at the toe of the slope or when the height of ponded water within the cover profile exceeds the elevation at the base erosional features, which establishes a hydraulic gradient to flow. Once seepage erosion is initiated a positive feedback loop is established in that further deepening of the erosion feature leads to an increase in the hydraulic gradient. This is further intensified by high surface runoff volumes associated with the *in situ* water dynamics (i.e. limited water storage capacity in the cover profile).

Figure 4 shows the change in the total volume of water and pore-water pressure in the cover profile at Summit. In general, the volume of water fluctuates between approximately 155 mm and 180 mm early in the spring and then decreases to approximately 100 mm in the summer. The trend in water dynamics would suggest that the cover profile exceeds field capacity at water volumes greater than 160 mm; however, given the low slope gradient and absence of a drainage layer excess waters pond above the HDPE layer.

A pressure head of 0.6 m is attained on several occasions, indicating that the entire depth of the cover profile (cover profile 0.6 m thick monitoring location illustrated in Figure 4) is under positive pore-water pressure. Trends in pore-water pressure at other monitoring locations on the cover system are similar to that illustrated in Figure 4. Figure 5 is a photo of a seepage erosion feature observed on the Summit cover system, where the geotextile/HDPE layer was exposed.

In addition to potential erosion issues, vehicle restrictions are imposed on the cover system during periods of positive pore-water pressure due to the realised loss in shear strength. In essence the water dynamics dictate that the cover system layers may be susceptible to anthropogenic damage. Even though the cover system water dynamics will largely control surface erosion, it should be noted that the processed till (35 mm minus) limits the covers ability to self-arm and limit erosion.

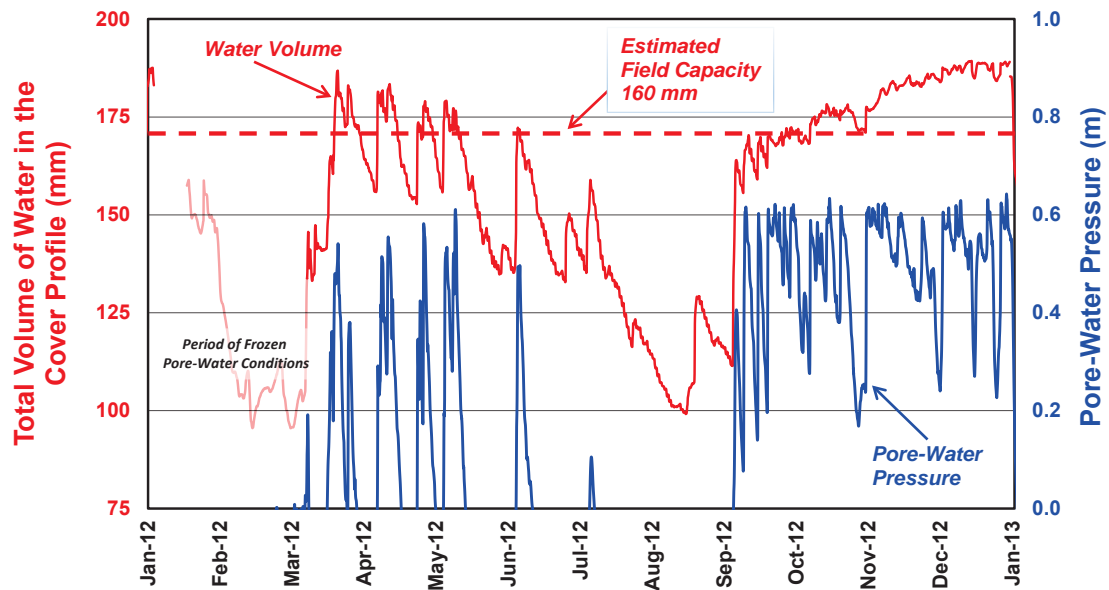


Fig. 4. Pore-water pressure and volume of water in the Summit cover profile



Fig. 5. Seepage erosion feature on the Summit cover system

Figure 6 is a one-dimensional (1-D) contour of the VJ cover profile volumetric water content. Water contents in the GRDL are lower than that of the growth medium. The contrast in water conditions between the growth medium and GRDL are attributed to textural differences, or more specifically, differences in the moisture retention characteristics. Low volumetric water contents observed in the GRDL indicate that the layer is free draining and that positive pore-water pressures have not occurred in the layer.

In general, flow to the GRDL occurs at the breakthrough pressure condition, which is the pressure condition corresponding to the matric suction at the residual water content of the GRDL. A slight increase in water contents in the GRDL occurs in the spring (April) and then consistently decrease until September suggesting the transmissivity of the GRDL is adequate to convey the realised interflow waters under the observed climatic conditions.

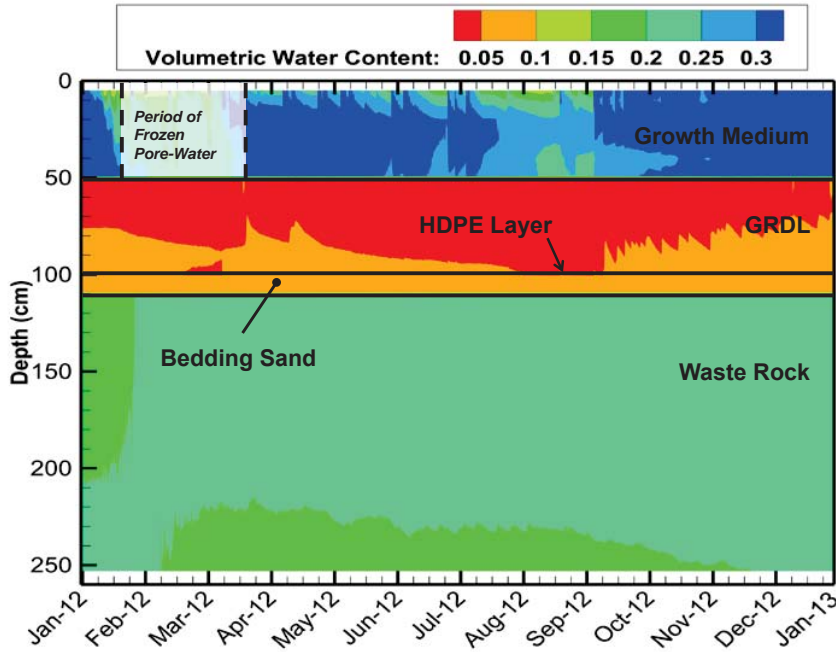


Fig. 6. 1-D contour of the volumetric water content measured at VJ

3.2.1 Estimates of Cover System Pressure Head

The height of ponded water within a drainage or growth medium layer is influenced by climate (surface infiltration), drainage length and slope, and saturated hydraulic conductivity of the layer. A comparative analysis was completed for Summit to determine the effect of potential changes in the saturated hydraulic conductivity of the growth medium layer under a range of steady state infiltration rates on the height of water ponded in the cover profile.

The following formula (Metry 1982) was used:

$$h_{\max} = \frac{L\sqrt{C}}{2} \left[\frac{\tan^2(a)}{C} + 1 - \frac{\tan(a)}{C} \sqrt{\tan^2(a) + C} \right] \quad [1]$$

where:

$$C = \frac{e}{K_s} \quad [2]$$

where: h_{\max} is the average daily head, L is the drainage distance, α is the slope, K_s is the saturated hydraulic conductivity of the layer, and e is the surface infiltration rate. Using Eqn. [1], in advance of numerical simulations, Table 1 summarises results of the comparative analysis. The color coded ranges were developed to provide a quick visual comparison of the estimated values. Red indicates that the height of ponded water exceeds cover profile thickness (i.e. 0.5 m), orange indicates that the height of ponded water remains below the cover profile surface but exceeds 0.25 m and green indicates that the height of ponded water remains below 0.25 m.

Table 1. Results of comparative analysis for a range of steady state infiltration rates and K_s of the growth medium on the reclaimed Summit WRP slope sections

K_s (cm/s)	Infiltration Rate (mm/day)				
	2.7	2.0	1.4	0.7	0.27
	Maximum Height of Water in Cover (m)				
1E-04	1.5	1.2	1.0	0.6	0.4
1E-03	0.4	0.3	0.3	0.2	0.1
1E-02	0.1	0.1	0.1	0.1	0.0
1E-01	0.0	0.0	0.0	0.0	0.0

Utilising the *in situ* hydraulic conductivity measured for the growth medium layer (approximately 1×10^{-4} cm/s) it is anticipated that the height of ponded water will exceed cover profile thickness even under low steady state infiltration rates, which is consistent with the observed performance. It is anticipated that an increase of approximately one order of magnitude (i.e. 1×10^{-3} cm/s) would be required to maintain water levels below the cover surface. Benson *et al.* (2007) and Meiers *et al.*, (2011) demonstrated that post construction increases in the hydraulic conductivity of a growth medium layer occur within three to four years and that materials with a high initial hydraulic conductivity (i.e. $\sim 1 \times 10^{-4}$ cm/s) are limited to marginal increases. Subsequently, the Summit cover system may be prone to periods of water ponding within the cover profile indefinitely based on the relatively high initial hydraulic conductivity and results of the comparative analysis.

In addition to the slope stability and erosion concerns, the drainage of water from above the geomembrane will minimise the head of water above the geomembrane layer and reduce the potential for infiltration through defects. Table 2 summarises the head of water estimated for the plateau over a range of K_s and steady state infiltration rates. Table 2 shows that the probability of water ponding on the HDPE layer would be high for almost all scenarios.

Table 2. Results of comparative analysis for a range of steady state infiltration rates and K_s of the growth medium on the reclaimed Summit WRP plateau

K_s (cm/s)	Infiltration Rate (mm/day)				
	2.7	2.0	1.4	0.7	0.27
	Maximum Height of Water in Cover (m)				
1E-04	17.5	14.7	11.5	7.5	4.2
1E-03	4.2	3.6	2.8	1.9	1.1
1E-02	1.1	1.0	0.8	0.6	0.4
1E-01	0.4	0.3	0.2	0.2	0.1

The Summit HDPE layer was constructed over 0.15 m of bedding sand; hence, leakage through defects would be free draining under ponded conditions. Transmissivity of the growth medium layer and spatial distribution of defects in the geomembrane would be the primary factors controlling infiltration. Benson (2000) reported that defects in geomembranes typically range from 2.5 to 25 defects/ha. Utilising the aforementioned defects and measured cover system water dynamics, numerical simulations would be utilised to develop an understanding for the anticipated range in leakage rates. Given the geochemical characteristics of the waste rock it is anticipated that leakage through the HDPE would not have a detrimental impact on the receiving environment.

3.3 Water Balance

A water balance was completed for the reclaimed WRPs using performance monitoring data collected at the sites in 2012. The water balance for a cover system consists of the following components:

$$PPT = R + AET + NP + \Delta S + IF \quad [3]$$

where: PPT is precipitation, R is surface runoff, ΔS is the change in water storage within the cover profile, AET is actual evapotranspiration, IF is interflow or lateral drainage, and NP is net percolation.

In the monitoring period runoff volumes were not available; hence, an analytical approach was utilised to estimate a water balance for the reclaimed WRPs. For the reclaimed Langan WRP, runoff and net percolation were estimated by back calculating the measured change in water storage from other components of the water balance.

Initially, utilising the water balance equation net percolation and runoff were assumed to be zero (i.e. $\Delta S = PPT - AET$) and the change in water storage was calculated. Figure 7 shows the measured and calculated change in water storage for Langan. The calculated change in water storage is similar during the first few months and then continually increases above the measured values in September. In July and August the hydraulic gradient across the waste rock / cover material interface was upward and adequate water storage capacity existed in the cover profile to store precipitation events. Under these conditions runoff and net percolation would have been negligible. Hence, the similar measured and calculated change in water storage during the summer would suggest that AET measured with the Eddy Covariance system is accurate and can be utilised for completing the water balance.

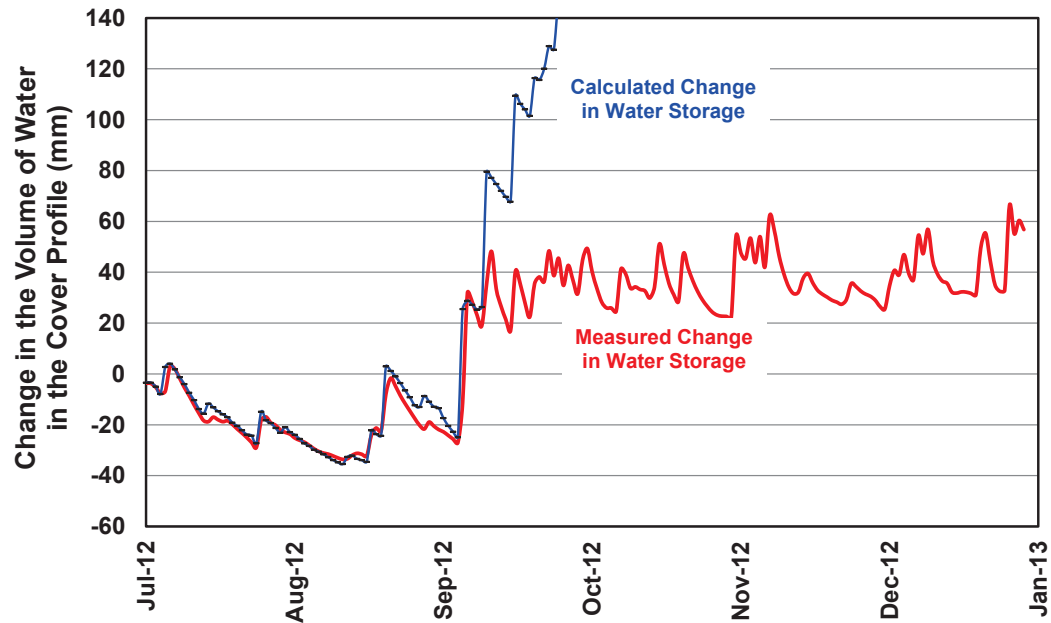


Fig. 7. Measured and calculated change in water storage using $\Delta S = \text{PPT} - \text{AET}$ for the Lingan cover system

Analytical estimates of net percolation and runoff were then developed. In order to match the calculated and measured change in water storage, water was removed from the water balance through runoff immediately following precipitation events and then through net percolation. Figure 8 shows the completed water balance for Lingan.

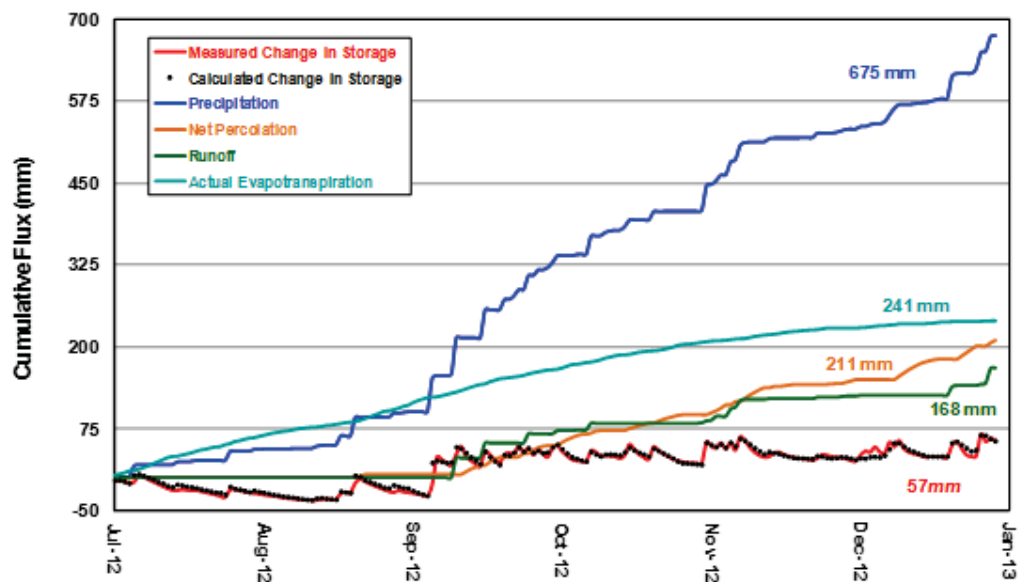


Fig. 8. Water balance for the reclaimed Lingan WRP

A slightly different approach was utilised for estimating the water balance for Summit and VJ. Net percolation was estimated through soil-plant-atmospheric numerical modeling completed for design of the monitoring system. An effective K_s of 2×10^{-9} cm/s was used for the HDPE layer based on findings by Benson (2000). Net percolation was estimated to be 8 mm and 20 mm for the reclaimed VJ and Summit WRP, respectively.

Figure 9 compares the water balance for the reclaimed Lingan, Summit and VJ WRPs. AET is similar for each WRP ranging from 40 to 43% of the water balance. There is a substantial difference realised in runoff volumes, which is directly attributed to differences in the cover system design. Summit has the highest estimated runoff at approximately 54% of the water balance. The lower runoff realised at VJ (40%) was primarily attributed to the GRDL allowing for the lateral percolation of water. In comparison, the absence of a geomembrane within the Lingan cover system offset greater runoff volumes through net percolation.

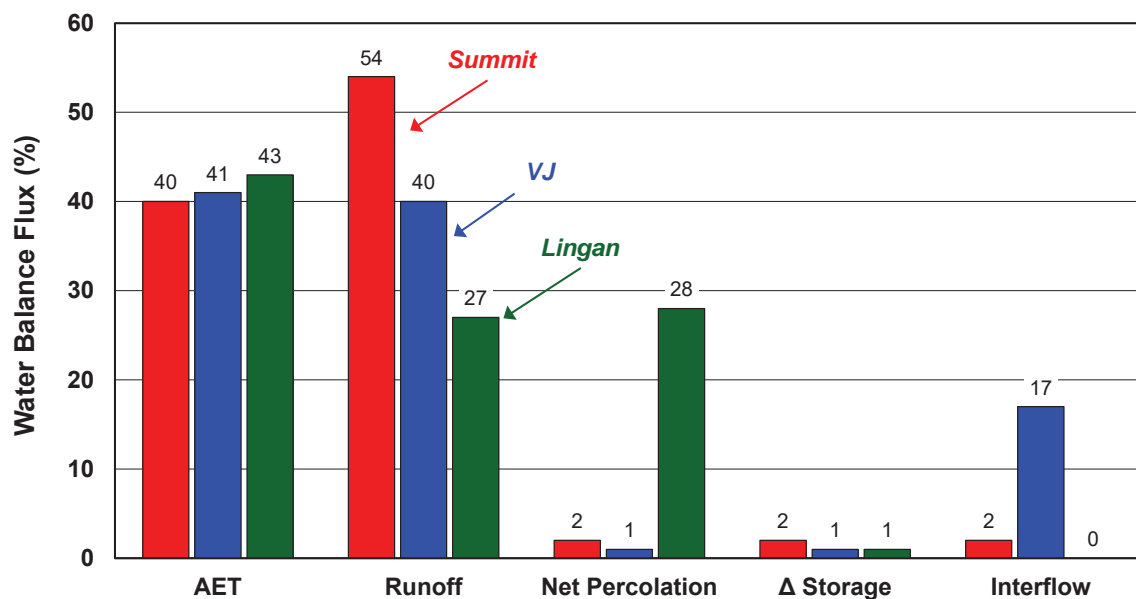


Fig. 9. Lingan, Summit and VJ water balance in 2012

Net percolation was a significant component of the water balance for the reclaimed Lingan WRP, indicating that cover systems which rely on the process of moisture store-and-release to mitigate net percolation may not be practical within seasonally humid climates. The results demonstrate that the inclusion of cover layers (i.e. HDPE and GRDL) below a similar growth medium thickness provide for a unique water balance.

4.0 SUMMARY

Three WRPs located near Sydney, Nova Scotia were reclaimed with dry cover systems. Cover system field performance monitoring data demonstrate that the inclusion of a drainage layer and / or geomembrane layer below a similar growth medium thickness provides for a unique water balance / dynamics. Absence of a drainage layer above the Summit geomembrane resulted in positive pore-water pressures, seepage erosion, high runoff volumes and an increase in the potential for leakage through the geomembrane layer. In

comparison, the transmissivity of the granular drainage layer at VJ was adequate to limit positive pore-water pressures, with the interflow volume resulting in a proportional reduction in runoff. Given that surface erosion is a common factor leading to the failure of mine waste cover systems, surface water management strategies should consider the influence of drainage layers on performance of geosynthetic cover systems.

The monitored water dynamics would suggest that the VJ cover system would provide for a more stable landform and minimise leakage through the HDPE layer. However, an understanding of performance under long-term average and extreme climatic conditions, through the use of numerical simulations, would be required to quantify any benefit in performance. On-going monitoring of the reclaimed WRPs will provide a unique dataset for the assessment and design of cover systems that utilise geosynthetic layers.

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ALKALINITY PRODUCING COVERS FOR MINIMISATION OF ACID MINE DRAINAGE GENERATION IN WASTE ROCK DUMPS

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ABSTRACT

Cover systems constructed over potentially acid forming (PAF) waste rock overburden dumps with a blend of low-permeability cement kiln dust (CKD) and weathered granite reduces atmospheric oxygen transfer into underlying acid-producing material, reducing the rate of pyrite oxidation, leading to a decrease in the severity of acid mine drainage (AMD). The focus of this paper is quantifying the secondary benefit of CKD-blended cover systems from an alkalinity generation perspective. CKD has an inherent alkalinity content of ~650 kg CaCO₃ eq. t⁻¹, which is steadily dissolved by the percolation of rainfall through the CKD/granite cover system, buffering acid generation in the underlying PAF material.

A small scale leaching trial was undertaken at the Stockton Mine on the West Coast of New Zealand, replicating generation of alkalinity in leachate through a 300 mm thick cover system, composed of 1 part CKD to 4 parts granite (by volume). Initially high leachate alkalinity rates (3000 – 4000 mg CaCO₃ eq. L⁻¹) were released that stabilised at approximately 500 mg CaCO₃ eq. L⁻¹ after 31 weeks. This corresponded to a long-term alkalinity yield of 10.9 t CaCO₃ eq. ha⁻¹ yr⁻¹. A net present value (NPV) calculation showed covering a waste rock dump with a CKD blended granite cover system and treating a reduced acid load had a breakeven of 8 years, compared to not covering and treating a higher acid load. Thallium is a major constituent of concern derived from CKD application as a cover system material, with the potential to adversely affect receiving ecosystems. The leachate thallium concentration rapidly decreases to less than 0.002 mg L⁻¹ following a first flush.

1.0 INTRODUCTION

Acid mine drainage (AMD) is one of the major environmental impacts of coal mines that disturb pyritic overburden on the West Coast of the South Island of New Zealand (Pope et al., 2010). Pyrite oxidation generates AMD through Eqn. [1].



The total sulphur content at the Stockton Mine varies, with some rock dumps having an average of up to 1.75 wt.% total sulphur (Elder et al. 2011), and this sulphur is predominantly present as pyrite. With a disturbed area of ~1000 ha at the mine, AMD management is a significant issue.

A key aspect of mine closure is the design of final engineered landforms (ELFs) and cover systems to minimise long-term maintenance liabilities (Bonstrom et al. 2012). Rehabilitation of waste rock overburden dumps at Stockton Mine is a three-step process. First, the surface of the dump is reshaped from the tip-head angle of repose (~ 38°) to a final landform slope of around 22°, with construction of benches at regular intervals to control surface water runoff

velocity and reduce erosion. Second, a 300 mm thick layer of blended cement kiln dust (CKD) and granite gravel (at a ratio of 1 part CKD to 4 parts granite by volume) is placed over the reshaped waste rock dump surface and compacted, producing a permeability of $\sim 10^{-4}$ to 10^{-5} cm s⁻¹. Finally, the dump is re-vegetated to provide a long-term stable cover system, either through vegetation direct transfer (Rogers et al. 2011) or by top-soiling (400 mm layer) and replanting.

The aim of engineered cover systems is to limit influx of atmospheric oxygen and water (Bonstrom 2012; O'Kane and Ayres 2012), impeding the oxidation of pyrite (Equation 1) and minimising the transport of oxidation products from the overburden. The primary reason for CKD addition is to increase the finer fraction of the CKD:granite material, filling voids in the relatively coarse, weathered granite and reducing rainfall net percolation. This has benefits for oxygen transfer rates as the diffusion coefficient decreases with lower porosity. The low water permeability also means the cover system holds moisture, further reducing oxygen transfer, as the effective oxygen diffusion coefficient decreases with an increasing degree of saturation (Aubertin et al. 2000).

Conceptually, alkalinity generating cover systems work by reducing oxygen transfer and percolation into a waste rock dump and ensuring water that does seep into underlying reactive materials is alkaline (Taylor et al. 2006). Miller et al. (2003) showed that placing an alkaline cover (with 2m of limestone) over a failed PAF/limestone blended dump resulted in leachate recovery from pH 3 to pH 4-6 after 2.5 years. After stabilisation at pH 6 in the field trial the leachate alkalinity stabilised at around 100 mg CaCO₃ eq. L⁻¹. These results were replicated in the laboratory using column leach tests (Smart 2010). By maintaining alkaline leachate, the oxidation of sulphides in ARD wastes was reduced by 90% (Smart et al. 2010). Taylor et al. (2006) has recommended using magnesium alkaline materials with superior solubility and dissolution kinetics for construction of alkaline covers to minimise short-, medium-, and long-term acid discharges, especially in environments with lower annual rainfall for which there would be an insufficient load of alkalinity produced by carbonate-based cover systems.

This paper focuses on the secondary benefit to be derived from using alkalinity generating cover systems from an acid-base accounting perspective, rather than from oxygen exclusion. This benefit had not yet been quantified in the field for the cover system investigated. The work was undertaken at the Stockton Mine where CKD is used as the alkaline amendment. The locally sourced CKD has a typical ANC of ~ 650 kg CaCO₃ eq. t⁻¹. The generation of alkalinity in water seeping through such covers adds alkalinity to the system, reducing the net amount of acid generated by a PAF waste rock dump. Thallium (Tl) was considered the major constituent of concern with the potential to leach from CKD at the Stockton mine. The Tl content of CKD used at the Stockton Mine is typically 10 mg kg⁻¹. Thus, leachate Tl concentrations were also measured to determine the potential effect of the use of CKD-based covers on the receiving environment as part of future work (e.g. ecotoxicity studies).

2.0 METHODS AND MATERIALS

A small scale leaching trial was set up at the Stockton Mine to determine the alkalinity yield from the standard CKD:granite cover system used for rehabilitation on site. The top was cut off two 200 L plastic drums, with a 600 mm diameter. Both drums were then filled with approximately 18 L (26 kg) of CKD and 72 L (130 kg) of granite gravel (blended in a wheelbarrow), resulting in a 300 mm deep layer across the base of both drums. This ratio of CKD to granite was used to replicate the 1 part CKD to 4 parts granite ratio (by vol.) used to cover engineered landforms at the mine. A further 114 L (180 kg) of topsoil was added to one of the drums (CKD cover + Topsoil) to simulate a 400 mm topsoil layer, while the other drum

contained only the 300 mm CKD:granite blend (CKD cover).

The drums were set up on racks about 800 mm off the ground at the mine site and left open to be naturally irrigated by rainfall. Five overflow holes were drilled in the side of each drum at the height of the topsoil or CKD:granite surface to prevent ponding of water within the drum. Thus, only water which would naturally infiltrate through the cover system reported as leachate. Leachate was collected through a hole in the base of the drum. The drainage hose was bent into a 'U' shape to create an air lock, preventing oxygen ingress into the drums from the outlet. The hose drained into collection buckets with an up to 43 L storage capacity for each drum.

The trial was monitored weekly over a period of 31 weeks. The pH and total volume of water collected in the bucket was measured in the field. An unfiltered and unpreserved 1 L sample was then taken from the collection buckets for each drum and sent to an external lab for alkalinity and TI analysis, as the analytes of interest to the project. The alkalinity was determined by titration using the APHA method 2320B. The TI concentration was determined by Inductively Coupled Plasma – Mass Spectrometry (ICP MS) using the APHA 3030B method.

3.0 RESULTS AND DISCUSSION

3.1 General

The weekly volume of leachate collected from each drum is shown on Figure 1.

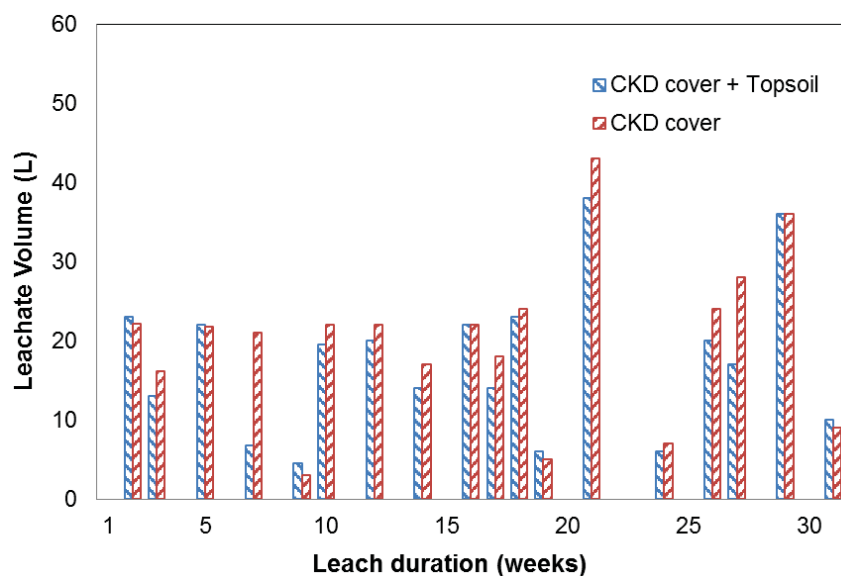


Fig. 1. Leachate volume (L) collected for CKD cover + Topsoil (blue bars) and CKD cover (red bars) drums over the 31 week trial

The weekly volume collected ranged from 0 L (weeks with no rain) to 43 L. The average leachate volume collected over the 31 week trial period was 10.3 and 11.7 L per week for the CKD cover + Topsoil and CKD cover drums, respectively. Water storage and release by

evaporation in the soil layer is expected to cause the difference in leachate production between the two drums. Over the 31 week trial period the site experienced rainfall at rates 8% lower than average, which was slightly lower than would normally be expected.

The leachate pH steadily decreased from pH ~13 to ~11 over the 31 week trial period for both trials, as shown in Figure 2.

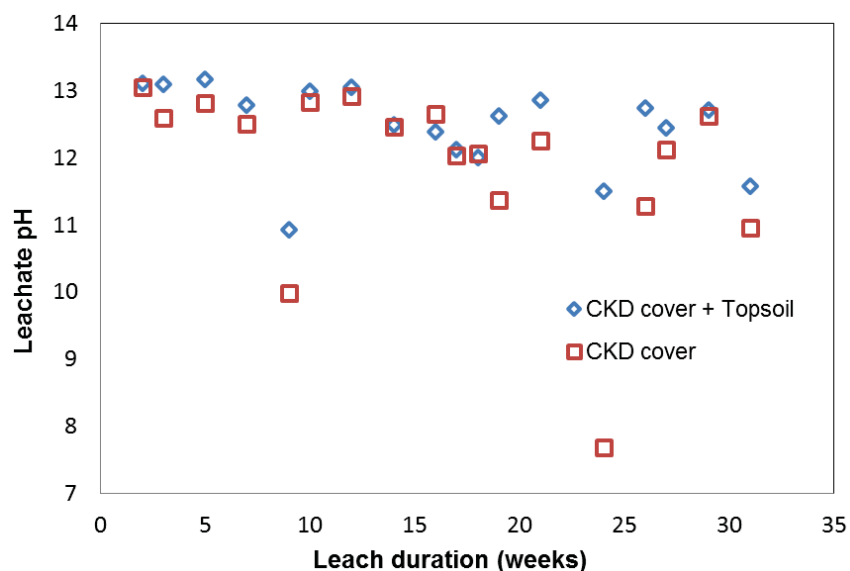


Fig. 2. Leachate pH for CKD cover + Topsoil (blue diamonds) and CKD cover (red squares) drums over the 31 week trial

X-ray diffraction testing has shown the ANC is predominantly from oxides (e.g. CaO, MgO, etc.), which explains its ability to produce leachate at pH 11-13.

3.2 Leaching Trial Alkalinity

The weekly leachate alkalinity is shown on Figure 3a. The leachate alkalinity from the CKD cover + Topsoil drum dropped from 4000 to ~500 mg CaCO₃ eq. L⁻¹ over the 31 week trial. The CKD cover alkalinity decreased from 3000 to ~500 mg CaCO₃ eq. L⁻¹. The alkalinity data from each drum were fitted by a power function (Table 1) and the continuous fitted lines produced by these functions (Fit. CKD cover + Topsoil and Fit. CKD cover) are overlayed on Fig 3a. The stabilised CKD leachate alkalinity of around 500 mg CaCO₃ eq. L⁻¹ is significantly higher than the 100 mg CaCO₃ eq. L⁻¹ expected from limestone (Smart et al. 2010). Thus, CKD is showing a higher alkalinity yield than limestone, which is likely to be due to the CKD alkalinity being present as more soluble oxides and hydroxides of calcium and magnesium as opposed to carbonates in limestone.

It is postulated, although not confirmed, that the difference between the two trials and elevated alkalinity from the CKD cover + Topsoil drum is due to the soil layer excluding atmospheric CO₂, thus limiting the formation of CaCO₃, and enabling higher, ongoing oxide and hydroxide dissolution when a soil cover is in place compared to the CKD cover drum.

The cumulative alkalinity loading (Figure 3b) was derived from the product of the weekly leachate volume and the corresponding alkalinity concentration derived from the fitting function in Table 1. The R^2 values of the fitting function are low, due to the variability of weekly alkalinity and leachate volumes collected. The objective of the fitting function is to allow further analysis of an idealised dataset without this variability. Figure 3b shows the CKD cover + Topsoil drum is continuing to release more alkalinity than the CKD cover drum. However, the fitted leachate alkalinity curves on Figure 3a show the alkalinity released from the two drums is converging at around 500 mg CaCO_3 eq. L^{-1} . Therefore, eventually the two cumulative alkalinity loading curves may become parallel.

The fitted cumulative alkalinity loading was determined as the product of the fitted weekly leachate alkalinity and the average weekly volume of leachate generated (10.3 and 11.7 L per week for the CKD cover + Topsoil and CKD cover drums, respectively), and extrapolated out to the end of the first year. The fitted cumulative alkalinity curve aligned with the actual data relatively well for both drums. The fitted data were used in further discussion as it evens out weekly fluctuations in alkaline load.

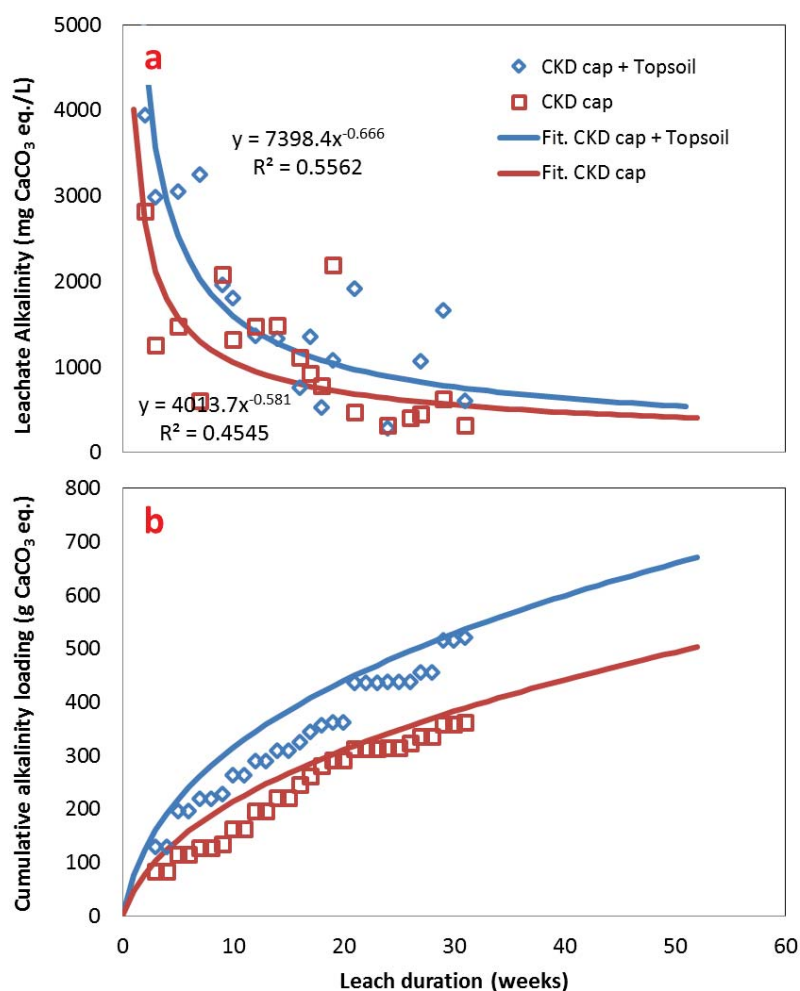


Fig. 3. Leachate alkalinity (mg CaCO_3 eq. L^{-1}) (a) and cumulative alkalinity loading (g CaCO_3 eq.) (b) for CKD cover + Topsoil (blue diamonds) and CKD cover (red squares) drums over 31 week trial

Table 1. Alkalinity decay rate fitting functions for CKD cover + Topsoil and CKD cover

Drum	Alkalinity decay (mg CaCO ₃ eq. L ⁻¹)	R ²
CKD cover + Topsoil	$Y=7398.4w^{-0.666}$	0.5562
CKD cover	$Y=4013.7w^{-0.581}$	0.4545

3.3 Theoretical acid treatment offset – CKD cover + Topsoil

Ongoing discussion has been restricted to the CKD cover + Topsoil scenario as this represents final cover systems at the Stockton Mine. The Mangatini catchment at the Stockton Mine has been significantly disturbed by both historic and more recent mining, with 232 ha of the total 319 ha catchment disturbed. The catchment produces 6600 t CaCO₃ eq. yr⁻¹ of acidity, which is currently neutralised by ultrafine limestone dosing (Elder et al. 2011). This value provides an estimated acid generation rate of 28.4 t CaCO₃ eq. ha⁻¹ over the disturbed area.

The surface area of the drums used in this trial was 0.25 m². The cumulative 671 g CaCO₃ eq. of alkalinity released from the CKD cover + Topsoil drum over the first year was extrapolated to 27.2 t CaCO₃ eq. ha⁻¹. Thus the alkalinity generation from the CKD cover system over the first year is approximately equal to the acid generation rate of the unamended catchment area.

Figure 3a shows the alkalinity generated from the CKD cover + Topsoil drum declines and then stabilises at around 500 mg CaCO₃ eq. L⁻¹ after one year. This corresponds to a lower alkalinity yield of 10.9 t CaCO₃ eq. yr⁻¹, which is just over a third of the pre-cover acid generation rate estimated from the catchment analysis above.

The acid neutralising capacity (ANC) of the CKD used at the Stockton Mine is approximately 650 kg CaCO₃ eq. t⁻¹, as determined using the method described in the AMIRA P387A Project ARD Test Handbook (IWRI and EGi 2002). Given 26 kg of CKD was used in each drum, the total alkalinity of each drum would be 16.9 kg CaCO₃ eq. The fitted cumulative alkalinity loading showed that for the first year, a cumulative 671 g CaCO₃ eq. of alkalinity was collected from the CKD cover + Topsoil drum, corresponding to leaching of 4.0 wt.% of the total alkalinity. Provided long term leachate alkalinity remains stable at 500 mg CaCO₃ eq. L⁻¹, the CKD cover + Topsoil drum would produce 268 g CaCO₃ eq. yr⁻¹, corresponding to leaching of 1.6% of the total ANC. Thus, at this rate it would take approximately 60 years for exhaustion of the alkalinity in the CKD.

Planning for mine closure requires consideration of the capital and operational costs associated with managing acid mine drainage in both the short- and long-term. Many mine operators choose to minimise capital expenditure early in a project and actively treat AMD drainage towards the end of mine operation and post closure in perpetuity, creating operational expenses over time frames in the order of 100 years or more. Such an approach will deliver significant savings from a net present value (NPV) perspective. While effective cover systems and rehabilitation of waste rock dumps is capital intensive, the decrease in acid production and subsequent treatment cost reduces longer term operational expenses and the ongoing risk of environmental harm.

Net present value (NPV) analysis methods essentially discount the future costs associated with treating acid mine drainage to a value in today's terms. The various parameters and assumptions used in the NPV analysis for this paper are shown on Table 2.

Table 2. NPV analysis parameters and assumptions

Parameter		Source
Cover Capital expenditure	\$55,000 ha ⁻¹	Pers. Comm. (Nathan Thompson)
Cover Operational expenditure	0	
Limestone treatment Capex	0	
Acidity decay rate after covering	40% over first 7 years 0.62% pa thereafter	Pers. Comm. (Phil Lindsay)
Acidity decay rate without covering	0.62% pa	Pers. Comm. (Phil Lindsay)
Inflation rate	2.03%	Pers. Comm. (Dave Thomas)
Discount rate	4.76%	Pers. Comm. (Dave Thomas)
Ultra-fine Limestone treatment cost	\$ 337.60 t ⁻¹ CaCO ₃ eq. neutralised	Pers. Comm. (Dave Thomas)
Alkalinity yield	27.2 t CaCO ₃ ha ⁻¹ in first year 10.9 t CaCO ₃ ha ⁻¹ yr. ⁻¹ thereafter	

The results of the NPV analysis are shown on Figure 4.

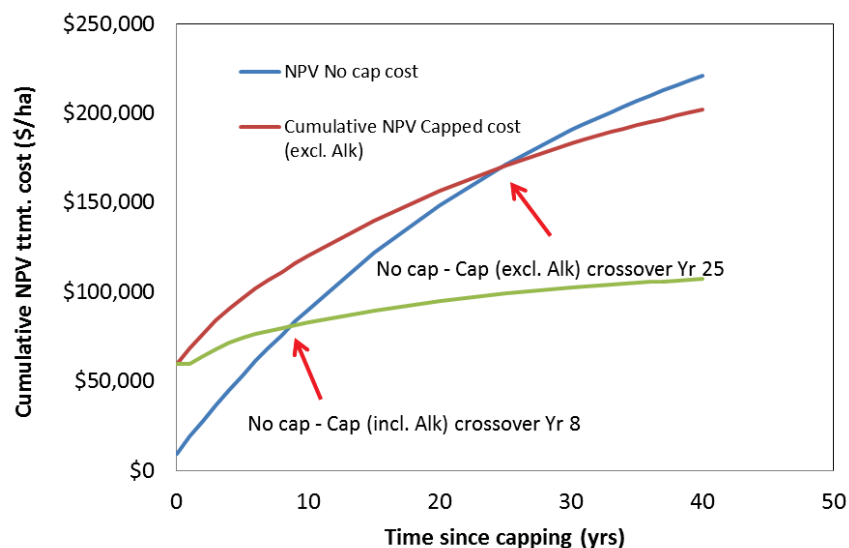


Fig. 4. Breakeven points resulting from the NPV analysis

Two breakeven points are shown on Figure 4. The first occurs at year 8 where the NPV of not covering and treating the full acid load of the catchment is equal to the cost of covering and treating the residual acid load of the catchment once the leachate alkalinity has been deducted from the total acid load. The second breakeven point occurs at year 25 where the NPV of not covering and treating the full acid load is equal to the NPV of covering and treating the residual acid load of the catchment without deducting the leachate alkalinity from the total acid load. The difference in the two breakeven points is due to the long-term

alkalinity generation rates of the CKD:granite cover system. If the long-term alkalinity generation rate is less than $500 \text{ mg CaCO}_3 \text{ eq. L}^{-1}$ the break-even point will be pushed out beyond 8 years, but will not reach 25 years unless alkalinity generation from the cover system ceases abruptly after year one.

3.4 Leaching Trial Thallium concentration

The weekly leachate thallium concentration is shown on Figure 5a.

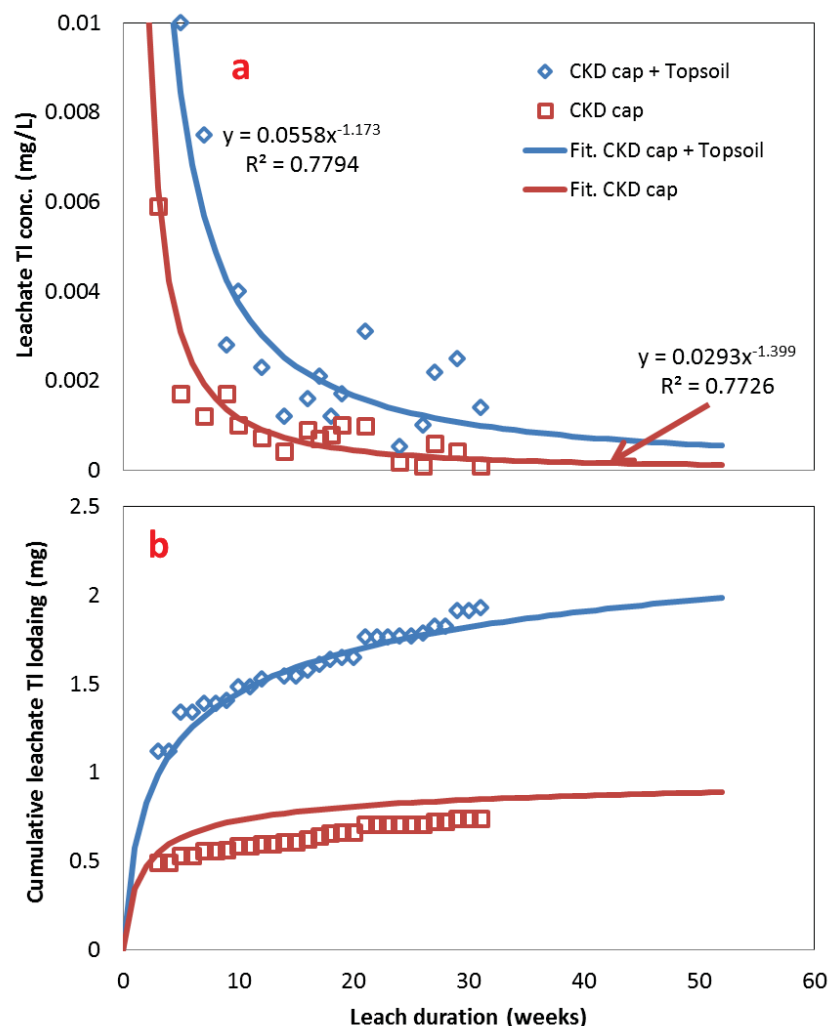


Fig. 5. Leachate thallium concentration (mg TI L^{-1}) (a) and cumulative thallium loading (mg TI) (b) for CKD cover + Topsoil (blue diamonds) and CKD cover (red squares) drums over 31 week trial

The leachate TI concentration from the CKD cover + Topsoil drum decreased from 0.04 to $\sim 0.002 \text{ mg L}^{-1}$ over the 31 week trial. The CKD cover leachate TI concentration decreased from 0.018 to $\sim 0.0005 \text{ mg L}^{-1}$. The TI concentration decay is consistent with a first flush type

scenario with TI concentration decreasing by over an order of magnitude before stabilising 7 to 12 weeks after cover system construction. A power trendline was fitted to the TI data from each drum with the equations shown on Table 3. These equations were used to generate the curves (Fit. CKD cover + Topsoil and Fit. CKD cover) shown on Figure 5a.

Table 3. Thallium concentration decay rate fitting functions for CKD cover + Topsoil and CKD cover

Drum	Thallium decay (mg L ⁻¹)	R ²
CKD cover + Topsoil	$Y=0.0558w^{-1.173}$	0.7794
CKD cover	$Y=0.0293w^{-1.399}$	0.7726

The cumulative TI loading (Figure 5b) was derived from the product of the weekly leachate volume and the corresponding TI concentration. Figure 5b shows the CKD cover + Topsoil drum is continuing to release more TI than the CKD cover drum.

The cumulative TI loading was determined as the product of the weekly leachate TI concentration produced by the fitting function and the average weekly volume of leachate generated, and extrapolated out to the end of the first year. The derived cumulative TI loading curve matched the actual data relatively well for both drums.

One year after construction the cumulative TI leached from the fitted CKD cover + Topsoil drum was 2.0 mg. Extrapolating from the 0.25 m² drum results in a per hectare yield of 80.7 g TI ha⁻¹ in the first year for the CKD cover + Topsoil scenario. Conservatively assuming the long term TI leaching rate is 0.002 mg L⁻¹ results in a long term TI yield of 43.5 g TI ha⁻¹ yr⁻¹. Evaluating whether this TI leaching rate is acceptable would require a site specific risk assessment to be done, including consideration of extant water quality guidelines and the extent of attenuation before relevant compliance monitoring points. There were insufficient data for the Australian and New Zealand Environmental Conservation Council (ANZECC) to recommend a threshold value for TI concentration for protection of aquatic ecosystems in its 2000 guidelines (ANZECC 2000). However, actual discharge TI concentrations from the site (when combined with surface water runoff) are likely to be below the current TI consent limit of 0.04 mg L⁻¹.

The CKD used for covers at the Stockton Mine generally has a total recoverable TI content (determined by acid digestion) of 10 mg TI kg⁻¹ CKD. 26 kg of CKD was used in each drum bringing the total recoverable TI per drum to 260 mg. Therefore, 0.8% of the total recoverable TI leached from the CKD over the first year. Beyond year 1, a long term TI yield of 0.5% per year would be expected. Thus, based on the current TI yield, it would be approximately 200 years for TI exhaustion of the CKD cover system to occur.

4.0 CONCLUSIONS

Construction of alkaline cover systems using alkalinity amendments such as CKD results in the generation of alkaline leachate with the potential to buffer acid generation from underlying PAF material. A small scale leaching trial using components of the cover system has shown the alkalinity generation rate stabilised at 500 mg CaCO₃ eq. L⁻¹ from a 1 part CKD to 4 parts granite blend. This alkaline leachate would offset the acid production of PAF

waste rock dumps from an acid-base accounting perspective, with alkalinity generated at approximately one third the current acid generation rate in the Mangatini catchment on an area basis. While cover systems are expensive from a net present value perspective, the reduction in AMD treatment costs for the Mangatini catchment with an alkaline cover system results in a breakeven point of 8 to 24 years after cover construction, depending on long-term alkalinity generation rates, which is a viable and achievable alternative to treatment in perpetuity. Initially high leachate Thallium concentrations rapidly decay. Determining whether these stable TI leachate concentrations are acceptable would require a site specific risk assessment to be done.

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ALKALINITY ADDITION FOR PYRITE PASSIVATION OF WASTE ROCKS FROM MT LYELL, WESTERN TASMANIA

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ABSTRACT

Alkalinity addition to pyritic mining wastes may be effective in causing armouring of pyrite with secondary minerals and inhibiting acid generation. This study investigates the applicability of this method to waste materials from Mt Lyell Copper Mine near Queenstown, Western Tasmania, using column leaching experiments: three open columns were used- a control, with a limestone cover, and with lime applied as a slurry. Two sealed oxygen consumption columns were used to measure the effect of alkalinity addition on pyrite oxidation in waste rock from the mine – a control with quartz gravel and a column with Mt Lyell waste rock to which doses of lime were added.

The initial results with open columns showed the lime slurry treatment resulted in greater reduction in acidity while the limestone cover addition resulted partially reduced acid generation. By estimating acidity generation from oxygen consumption measurements, and subtracting it from acidity measured by titration, it is possible to estimate the contribution of secondary mineral dissolution to acidity over time. Further monitoring will be aimed at determining whether these alkalinity additions to the Mt Lyell waste rocks will result in inhibition of pyrite oxidation, which would be indicated by reduced rates of oxygen consumption and reduced rates of sulphate production.

1.0 INTRODUCTION

Several studies have found that alkalinity addition to acid generating mining wastes appears to have potential for inducing microencapsulation of pyrite by secondary mineral precipitation (Huminicki and Rimstidt, 2009; Miller et al, 2009; Sahoo et al. 2013). Alkalinity addition can be applied by blending the alkaline material with the mine waste, so that it is in direct contact with acid metalliferous waters within the waste. This direct contact is advantageous since alkalinity-producing materials typically dissolve faster in acid conditions and provide more neutralizing power. However the effectiveness may be reduced by passivation of the alkalinity-producing material due to armoring by secondary mineral precipitation. Alternatively, the alkalinity- producing material can be added on top of the mine waste, thus protecting the material from armoring but at the cost of lower rates of alkalinity production (Taylor et al, 2009).

Acid mine drainage has been generated from Mt Lyell Copper Mine in Western Tasmania for most of the last century and has severely degraded the King and Queen Rivers and Macquarie Harbor, and it is predicted that AMD will continue to be a problem for hundreds of years (GHD,1994). A major part of the AMD generation at Mt Lyell is from historical waste rock dumps, and this study aims to test the potential of alkaline addition for preventing the AMD generation from these waste rock dumps.

2.0 METHODS

Both open leaching columns and sealed columns with oxygen probes were used to test the effect of alkaline addition on acidity generation from waste rocks obtained from the historical dumps. The three open columns, each containing 4.5kg of waste rock (a mixture of fine and coarse material), were leached with around 500mL of deionized water every three days for 23 days. A cover consisting of 81 g of finely crushed limestone in 225g of fine sand was applied to one column (column 2), and a slurry of 5g analytical grade calcium hydroxide was applied to another column (column 3) (equivalent to 1.5g CaCO_3 per kg waste rock). The columns were watered at a rate of 500mL deionized water every 3 days for 21 days after alkalinity application, and subsequently at 600mL deionized water per fortnight.

One sealed column (Column 4) had 3kg of quartz pebbles as a base layer, and a top layer of 1.5kg of waste rock fragments crushed to a gravel. The second column contained 4.5kg of quartz pebbles to serve as a control for measuring oxygen consumption by the oxygen gas probe. The oxygen consumption columns were watered with 750mL per fortnight over around 6 months. Two doses of 1.6g analytical grade calcium hydroxide were applied to the waste rock in the oxygen consumption column at 134 and 143 days respectively: these doses are collectively equivalent to 3 g CaCO_3 per kg waste rock. The oxygen consumption columns were opened for the purpose of watering and alkalinity addition, and then sealed to allow for measurement of oxygen consumption.

Leachate composition was analysed for metals, calcium, sulfate, alkalinity and acidity for all columns.

3.0 RESULTS

Open Columns

Prior to alkaline addition, the leachate from the three columns shows some variability, due to variation in the composition of the waste rock. Columns 2 and 3 had reasonably stable leachate composition, dominated by sulfate, iron, aluminium, magnesium and calcium (Figures 2a and 3a), although Column 2 has around half the total dissolved solids (TDS) as Column 3. Column 1 has a much higher TDS than 2 and 3 initially, but this declines rapidly to levels more similar to Column 2 (Figure 1a) as the soluble acidity was flushed from the column. The initial high rate of watering appeared to be preventing build-up of sufficient acidity to mobilise the iron in these columns, so from 44 days onwards, the watering regime was reduced in intensity.

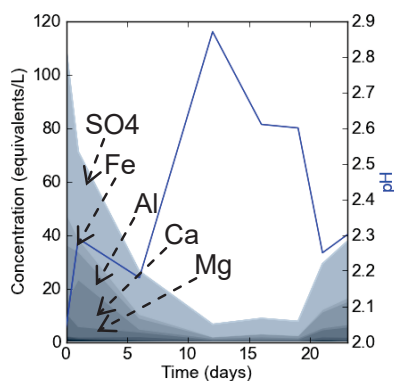


Fig. 1a. Column 1 (open) - control (0-23 days)

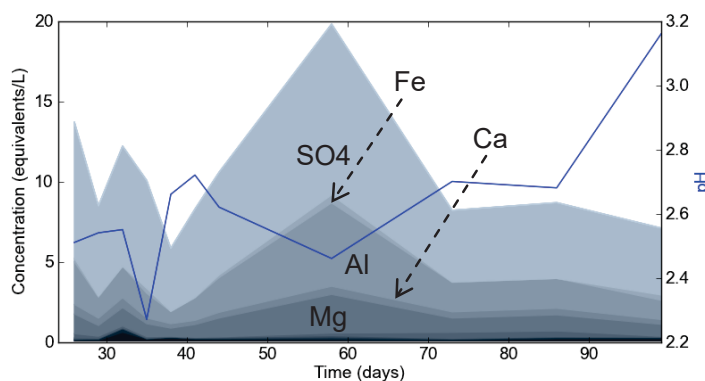


Fig. 1b. Column 1 (open) control (24-99 days)

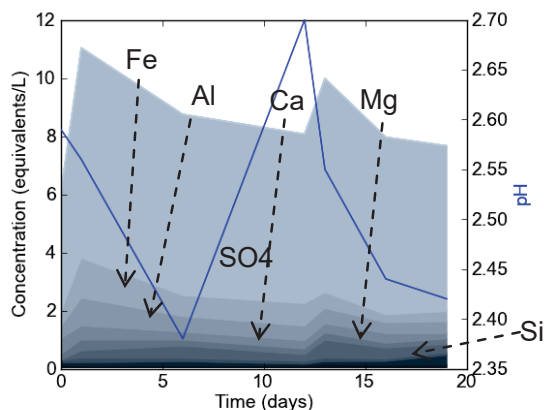


Fig. 2a. Column 2 (open) prior to limestone addition (0-20 days)

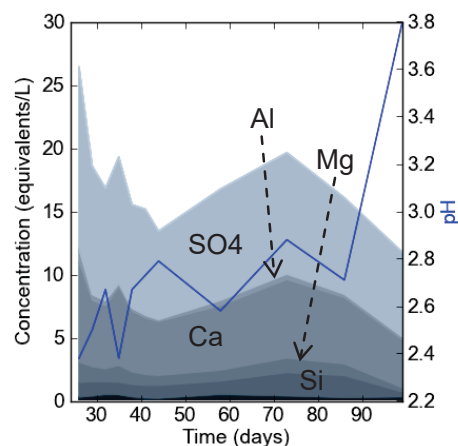


Fig. 2b. Column 2 after limestone addition (24-99 days)

Once limestone and lime were added to columns 2 and 3 respectively, iron was no longer leached, and aluminium levels were reduced (Figures 2b and 3b). Column 2 had lower pH (~3) and higher levels of aluminium than column 3, which has neutral pH (5-6). Calcium became the dominant cation in both columns 2 and 3, which suggests active neutralization of acidity in these two columns is taking place. The limestone cover in Column 2 is effective in creating neutral conditions only within a limited zone of influence within the waste rock materials, which precipitated aluminium and reduced leaching the aluminium from this column. In Column 3, the lime dose was effective at sustaining neutral conditions throughout the column, which resulted in more complete removal of aluminium from the leachate.

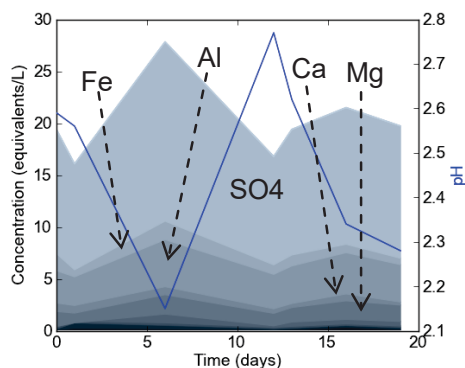


Fig. 3a. Column 3 (open) before lime addition (0-20 days)

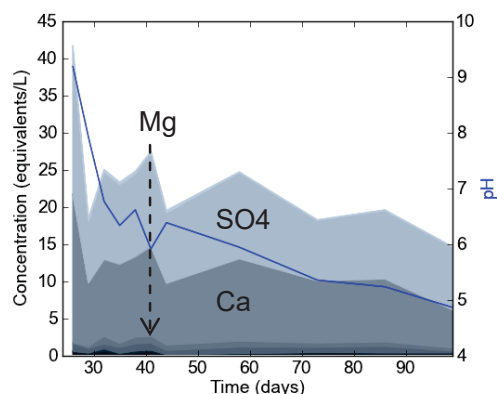


Fig. 3b. Column 3 after lime addition (24-99 days)

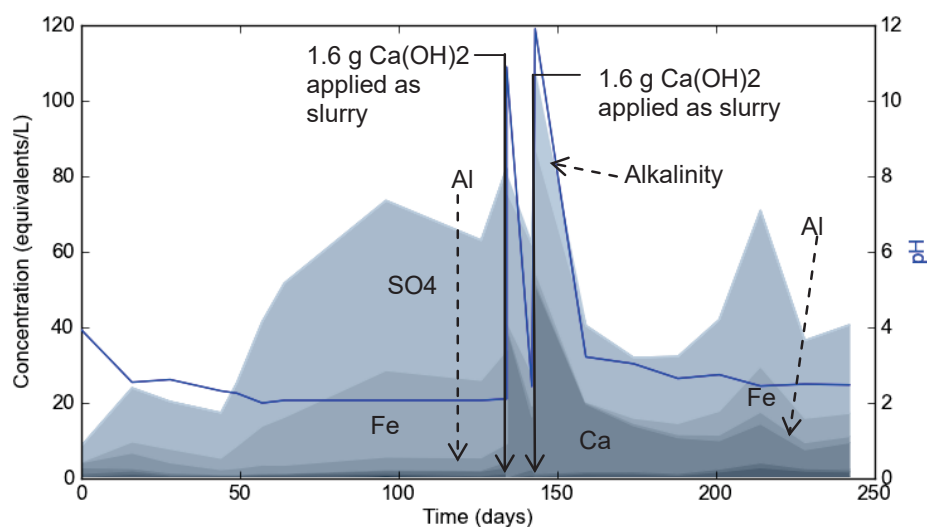


Fig. 4. Column 4 (sealed) with two doses of lime-leachate composition and pH

Sealed Columns

The leachate from waste rock in the sealed column has much higher levels of iron than the waste rock in the open columns. This is likely due partly to the use of a sample with higher pyrite content and the lower intensity watering regime. When doses of lime slurry were applied to this column, calcium largely replaced iron and aluminium as the dominant cation in the leachate. However, unlike the open column with lime application, the effect was not sustained. Whereas in the open column 3 (waste rock with lime), one dose of lime equivalent to 1.5g/kg CaCO_3 was sufficient to sustain neutral conditions for at least 2 months, in the sealed column two doses of the same magnitude are not sufficient to sustain neutral conditions for even one leach. In the sealed column after lime dosing, pH of the leachate returned to around 3 and in the absence of further dosing, subsequent leaches showed decreasing pH and increasing acidity (Figure 5).

The acidity calculated using oxygen consumption measurements is less than the acidity measured by titration, suggesting that prior to alkaline addition, acidity in the leachate had two sources, secondary mineral dissolution and pyrite oxidation. Over time, pyrite oxidation became the dominant source of acidity (Figure 6). Immediately after alkaline addition, acidity in the leachate (measured by titration) is only a fraction of acidity calculated from oxygen consumption measurements, due to neutralization reactions. However in subsequent leaches, acidity increases. After alkaline addition the pattern of oxygen consumption is mixed, with lower than normal levels from the 16/11 to the 13/12, higher than normal levels from 13/12/13 - 10/01/14, and lower levels after the 10/01/14. Further testing is required to test whether the lime slurry application does reduce oxygen consumption and hence directly inhibit pyrite oxidation.

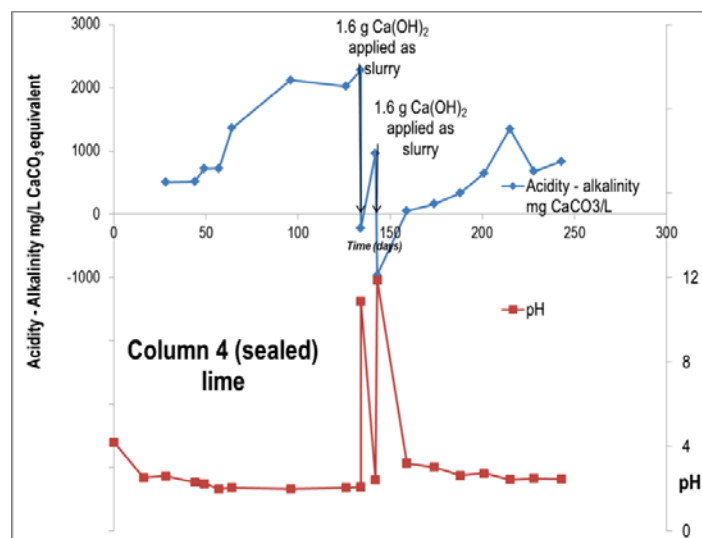


Fig. 5. Column 4- Acidity- Alkalinity mg/L CaCO₃ and pH

4.0 CONCLUSIONS

Both lime slurry and limestone cover applications were effective in reducing leaching of iron and aluminium from the waste rocks, and further monitoring will test whether this results in effective armoring of pyrite and inhibition of AMD generation. Estimations of acidity production based on oxygen measurements allow estimation of the relative contributions of secondary mineral dissolution and pyrite oxidation to acidity generation. Further testing will test whether alkalinity additions result in reduced oxygen consumption by pyrite passivation in the Mt Lyell waste rocks.

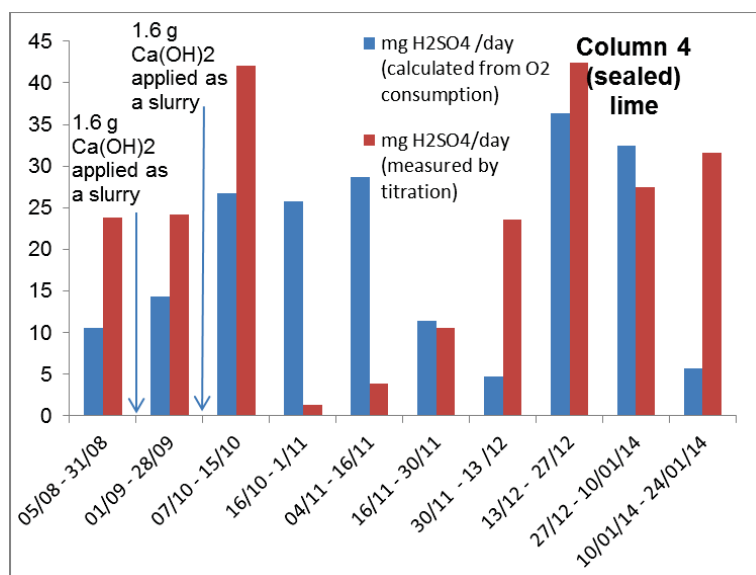


Fig. 6. Column 4 - acidity generation rates calculated from O₂ consumption compared with acidity generation rates measured by titration

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PERFORMANCE OF A COVER ON A BULK SAMPLE TAILINGS DAM AT CENTURY MINE

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ABSTRACT

In 2012 MMG Century mine staff excavated and removed the bulk sample tailings dam (BSTD) to the main tailings storage facility. The BSTD was rehabilitated in 1997. Rehabilitation consisted of a two meter thick store and release cover. The cover was constructed in horizontal layers consisting of a 0.5 m topsoil layer overlaying a reduced percolation layer (RPL). The RPL consisted of two 0.5m compacted clay layers separated by a 0.5 m capillary break. The main purpose of the capillary break was to prevent the vertical rise of salts and dissolved metals into the topsoil layer.

1.0 INTRODUCTION

1.1 Location

MMG Limited's Century mine is located 250 km north-north-west of Mount Isa, in Queensland, Australia. The topography of the site comprises of undulating hills and ridgelines. The site vegetation is low and sparse, as is typical of the semi-arid environment of northern Australia.

1.2 Rainfall

Rainfall data is available from the Bureau of Meteorology (BoM) weather station (Century mine: station 029167). The average annual rainfall is 544 mm, with a dry season from April to September and a wet season from October to March. January and February account for about 50% of total rainfall.

2.0 BULK SAMPLE TAILINGS

2.1 History

The bulk sample tailings dam (BSTD) was constructed in 1994 during the feasibility stage of the Century mine. It was designed for the containment of tailings produced by a bulk sample plant that was operated on site to produce a shipment of zinc (Zn) concentrate for smelter testing. The tailings within the BSTD were highly sulfidic.

In 1998 Environmental Geochemistry International (EGi) were commissioned to complete a feasibility study for closure strategies for the BSTD. The feasibility study was based on preliminary design concepts provided by Pasminco. An appropriate closure strategy was required because the sulfidic tailings were producing acid rock drainage (ARD).

It is believed that the cover was constructed before the wet season in 1998.

The BSTD was excavated and removed in 2012. The tailings and cover material were deposited into the main TSF. It was removed because the cover had failed. The deposited tailings continued to produce ARD after the closure strategy was implemented. The BSTD cover was approximately 15 years old in the year the BSTD was excavated and removed.

2.2 Conceptual Closure Strategy

The intent of the EGi (1998) conceptual closure strategy was threefold:

- develop a cover capable of storing infiltrating rainfall in a infiltration storage layer for later release as evaporation and vegetation transpiration;
- remove excess infiltration as lateral flow before it could percolate into the tailings; and
- prevent capillary rise of salt and metals back into the infiltration storage layer.

The BSTD cover was a store and release cover (EGi 1998). Infiltration into the PAF tailings was to be reduced by maintaining a compacted reduced permeability layer (0.3 m) (RPL) at near-saturated conditions. This was to be achieved by overlaying the RPL with soil/rock mulch (0.5 m) and a thin veneer of topsoil (0.2 m) (Figure 1). Capillary rise of sulfate salts and metals, by-products of ARD, was to be prevented by incorporating a capillary break (0.5 m thick) (Figure 1). The capillary break was to be constructed below the RPL (Figure 1).

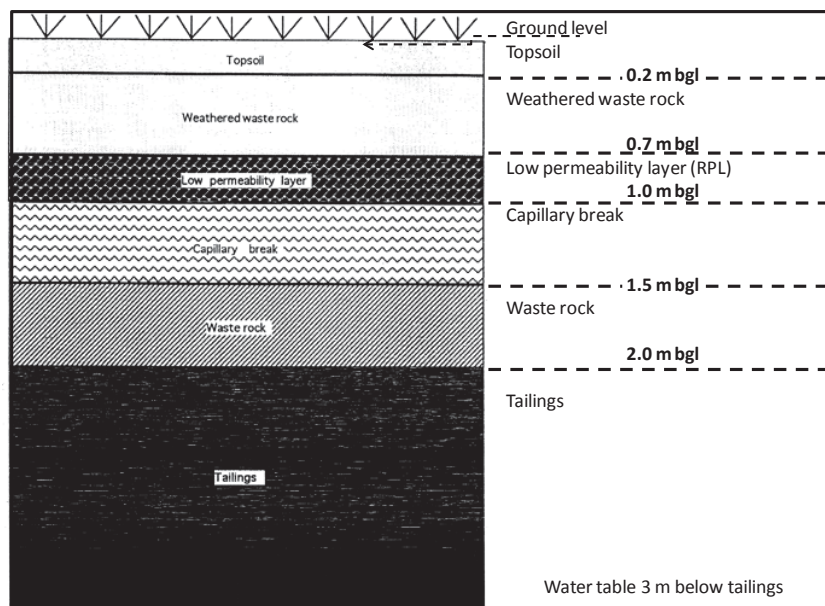


Fig. 1. Conceptual store and release cover

Prior to constructing the store and release cover a running surface of waste rock (2 m) was to be constructed above the tailings so that vehicles could safely access the BSTD.

2.3 As-built Closure Strategy

During the excavation and removal of the BSTD the store and release cover was described and cover layer thicknesses recorded (Figure 2).

The as-built cover was also a store and release cover and had the same design intention as the conceptual closure strategy. Infiltration into the PAF tailings was to be reduced by maintaining two RPLs (0.5 m) at near-saturated conditions. This was to be achieved by overlaying the RPLs with topsoil (0.5 m) (Figure 2). Capillary rise of sulfate salts and metals was to be prevented by incorporating a capillary break (0.5 m thick) (Figure 1). The capillary break was constructed between the two RPLs (Figure 2).

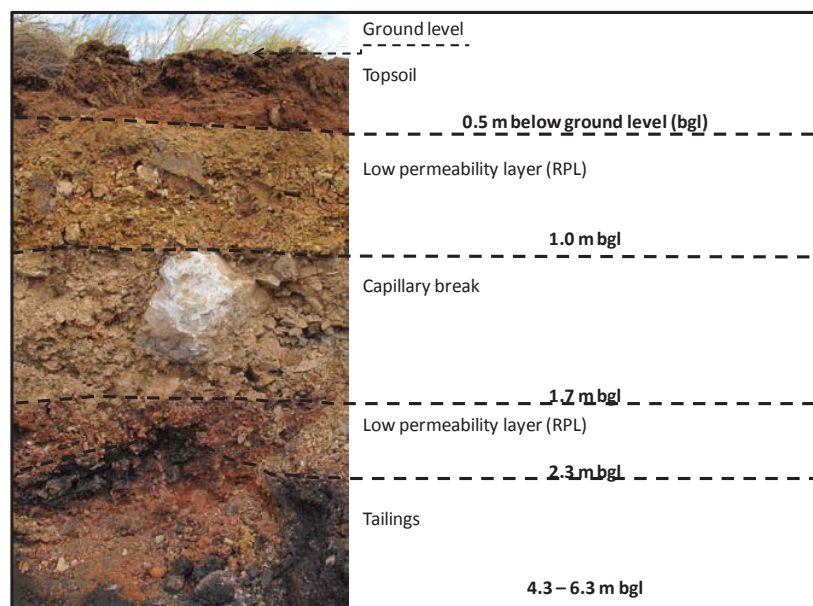


Fig.2. As-built store and release cover with double RPL and

2.4 Differences Between Conceptual and As-built Closure Strategy

The as-built store and release cover was different to the conceptual strategy. The structural differences between the conceptual strategy and the as-built store and release cover are summarized in Table 1.

Table 1. Structural differences between the conceptual and as-built closure strategies

Cover element	Conceptual	As-built
Vegetation	✓	✓
Topsoil	✓ 0.2 m thick	✓ 0.3 m thick. This is thicker than the conceptual cover.
Weathered rock	✓	✗ This layer is not included in the constructed cover.
Low permeability layer or reduced	✓	✓ This is the upper RPL in the constructed cover.

Cover element	Conceptual	As-built
permeability layer (RPL)		
Capillary break	✓	✓
Waste rock	✓	✗ This layer is not included in the constructed cover It was substituted with: An RPL. This is the lower RPL boundary in the constructed cover.
Tailings	✓	✓

The as-built store and release cover did not (from Table 1):

- contain a weathered rock layer immediately below the topsoil layer. According to the EGI (1998) report the purpose of this layer was the same as subsoil in a natural soil profile. That is the weathered oxide layers purpose was to supply additional storage for infiltrating rain water; and
- contain a waste rock layer immediately below the capillary break. According to the EGI (1998) report the purpose of this layer was to provide a bridging layer to prevent the soft tailings from mixing with the coarse capillary break and to also provide a running surface for equipment movement during construction. Mixing of fine tailings and coarse capillary break rock will impact on its function and performance.

It is unclear why the as-built store and release cover was different to the conceptual strategy. It is inferred that this is the result of insufficient quality assurance testing and construction supervision.

2.5 Performance of the As-Built Closure Strategy

It is clear that the as-built store and release cover design had failed because the BSTD continued to produce ARD. The as-built store and release cover had failed to (in reference to Section 2.2):

- store infiltrating rainfall for later release as evaporation and vegetation transpiration; and
- remove excess infiltration as lateral flow before it could percolate into the tailings.

The unsaturated cover performance modeling (using Vadose/w) is not discussed in this paper however the model showed that the cover failed to limit infiltration because of three reasons.

The topsoil layer was too thin to store rainfall and contained a large depression (1,000 m² Figure 3) that allowed water to pond on the store and release cover. Once the topsoil storage layer became saturated it passed at least 315 mm/yr (equivalent to a saturated permeability of 10⁻⁸ m/s), or 50% of annual rainfall until such time as the topsoil storage layer desaturated. Desaturation occurred as either percolation (deep drainage through the cover) or evaporation and transpiration by plants.



Fig. 3. Surface depression in cover

The capillary break was incorrectly placed within the cover sequence. Sandwiching the capillary break between an upper and lower RPL allowed it to saturate. Once saturated the capillary break created a 'conduit' between the topsoil infiltration storage layer and the tailings allowing unrestricted movement of sulfate salts, dissolved metals and pore water. Vegetation roots had not penetrated the capillary break. Once saturated, desaturation of the capillary break was near impossible with the upper and lower RPLs holding the pore water in this layer.

Finally the capillary break was not gap graded and contained fines (Figure 4), likely from internal migration from the RPLs. Uniform pore size and distribution between the RPLs and capillary break required less infiltrating rainfall to recharge fine capillaries (conduits) with each successive rainfall. The internal migration of fines further compounded the failed as-built cover performance described in this section.

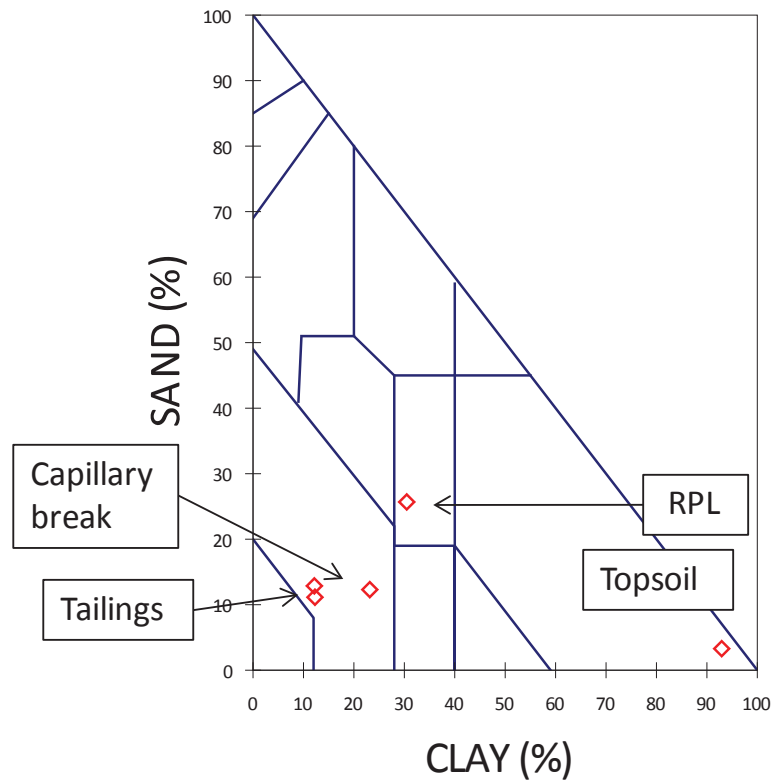


Fig. 4. General particle size distribution for clay and sand sized particles in the as-built store and release cover

The following method and results describe whether the as-built store and release cover had met its third design intention (in reference to Section 2.2):

- prevent capillary rise of salt and metals back into the infiltration storage layer.

3.0 METHOD

3.1 Sampling

In mid-2012 the BSTD was excavated and removed by Century. The excavation process involved sub-sampling of the full thickness of the BSTD at nine locations (Figure 4 and Table 2). Two back-ground samples were also collected for comparison (Table 2).



Fig. 4. Sampling locations

Table 2. Samples submitted for laboratory analysis

EP03A	EP03B	EP01B	Background samples (BG)
EP03A(0.2-0.3)	EP03B(3.3-3.4)	EP01B(3.3-3.5)	BG (0.1-0.2)
EP03A(0.6-0.9)	EP03B(4.5-4.6)	EP01B(4.5-4.7)	BG (0.1-0.2)
EP03A(1.0-1.2)	EP03B(6.2-6.3)	EP01B(5.2-5.3)	
EP03A(1.7-1.9)	EP03B(6.3-6.4)		
EP03A(3.0-3.1)			
EP03A(4.1-4.2)			
EP03A(4.5-4.6)			
EP03A(4.6-4.7)			

Notes: 1. bracketed numbers are the meters below ground level (bgl) that samples were collected
 2. samples labelled as BG were collected from the natural soil profile on the ridge above the BSTD

3.2 Laboratory Analysis

Selected samples (Table 2) from location EP03A, EP03B and two background sites were sent to a NATA accredited laboratory where they were tested for the constituents described in Table 3.

Table 3. Laboratory analysis

Test	Description
Total S	This is a measure of the maximum potential acidity (MPA). The test includes both sulfide and sulphate species;
Chromium	This is a measure of the inorganic S and is an indication of acidity formed from both inorganic

Test	Description
Reducible Sulphur	(ARD) and organic (organic matter) S forms;
Soluble Sulphate	This is a measure of oxidised sulfide, it is not responsible for ARD and can be subtracted from the sulfide/sulphate balance;
NAGpH	This is a cross-check of ARD potential to be compared to the sulfide/sulphate balance. If the sample after reaction has a pH of 4.5 or less (i.e. NAGpH≤4.5) then it is considered to be acid forming;
pH and EC (1:5)	Gives an indication of the inherent acidity and salinity of waste material when exposed. A material categorised as non-acid-forming (NAF) may still have existing salinity and acidity risks that make it unsuitable for surface or uncontrolled placement; and
Multi Element Scan	ARD typically produces pH values ≤4.5. Under acidic conditions metals become soluble and transportable in seepage. Analysis will include Al, As, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, Sn, Th, U, Zn.

4.0 RESULTS AND DISCUSSION

4.1 Acid Producing Potential of Tailings

The EGi (1998) study calculated the net acid producing potential at 43–143 kg H₂SO₄/t. During the autopsy net acid producing potential was calculated as 81.68–203.73 kg H₂SO₄/t (Table 4). The acid rock drainage potential of the BSTD deposited tailings had not reduced since the facility was rehabilitated in 1998.

Table 4. Summary of acid producing potential from tailings

Parameter	Unit	Tailings Samples										
Solids acid-base account												
		EP03A (3.0-3.1)	EP03A(4.1-4.2)	EP03A (4.5-4.6)	EP03A (4.6-4.7)	EP03B (3.3-3.4)	EP03B (4.5-4.6)	EP03B (6.2-6.3)	EP03B (6.3-6.4)	EP01B (3.3-3.5)	EP01B (4.5-4.7)	EP01B (5.2-5.3)
Total S	%S	7.54	6.87	7.48		7.51	7.98	6.26		5.48	7.71	8.02
Sulfate-sulfur	mg/kg	23200	23400	16000	740	14300	24900	3370	1420	28100	20700	13600
	%S	2.32	2.34	1.6	0.074	1.43	2.49	0.337	0.142	2.81	2.07	1.36
Sulfide-sulphur	%S	5.22	4.53	5.88		6.08	5.49	5.923		2.67	5.64	6.66
Maximum potential acidity	kg H ₂ SO ₄ /t	159.68	138.57	179.87	-	185.99	167.94	181.18	-	81.68	172.53	203.73
Acid neutralising potential	kg H ₂ SO ₄ /t	-	-	-	-	-	-	-	-	-	-	-
Net acid producing potential	kg H ₂ SO ₄ /t	159.68	138.57	179.87	-	185.99	167.94	181.18	-	81.68	172.53	203.73
Solids NAG test												
Final NAG pH		2.3	2.8	3.9	-	3	3.7	2.9	-	2.7	3.8	2.8
NAG – titration to pH 4.5	kg H ₂ SO ₄ /t	81.1	4.9	4.2	-	23.1	3.1	25.1	-	35.2	4	30.3

Parameter	Unit	Tailings Samples								
NAG – titration to pH 7.0	kg H ₂ SO ₄ /t	129	83	99.4	115	106	110	91.2	118	129

Net acid generation testing results indicated (with reference to Table 4):

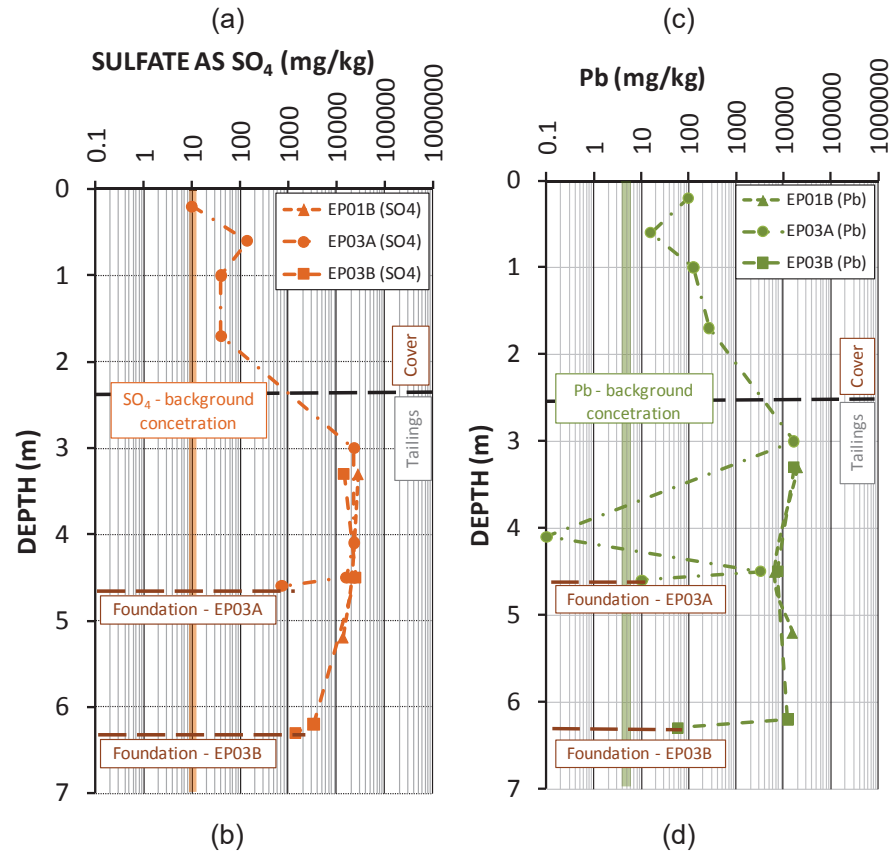
- sulfides in samples were reactive, with net acid generation pH decreasing to approximately pH <3;
- titration of the net acid generation liquor to pH 7 indicated that the amount of acidity was approximately two-thirds of the net acid producing potential value. The difference between net acid producing potential and net acid generation was likely due to the presence of galena, which does not produce acidity and sphalerite which may not generate as much free acid as pyrite;
- approximately 60-70% of acidity was due to free acidic ions; and
- the titration value from pH 4.5 to 7 included other metals that form precipitates such as Zinc (Zn), Lead (Pb) and cadmium (Cd). Within the BSTD it is expected that more than half of the acidity produced would have been from soluble metals.

4.2 Mobility of Sulfate and Metals

Sulfate (SO₄), Pb and Zn and to a lesser extent Cd are all elevated in the as-built cover indicating that the capillary break has been ineffective and that capillary rise of salts and metals from the tailings has entered the as-built cover (Figure 5a, b, c and d). Pb, Zn and Cd concentrations are all higher at the base of the as-built cover where contact with the tailings is the greatest. The concentration profiles generally all show a decreasing trend back towards the as-built cover surface except for the top 0.5m. This may have been due the concentration of roots within this layer. The concentration profiles are all above the background range.

The National Environment Protection (Assessment of Site Contamination) Measure (NEPM) soil investigation levels (2013) have recommended that threshold values (C – human exposure levels based on public open space land use) be set at 60 mg/kg for Cd, 600 mg/kg for Pb, and 30,000 mg/kg for Zn for soil:

- Cd was above background concentration, but did not exceed NEPM guideline in the foundation and as-built cover;
- Pb was above background level but did not exceed NEPM guidelines in the foundation and as-built cover; and
- Zn was above background concentration but did not exceed NEPM guidelines in the foundation and as-built cover.



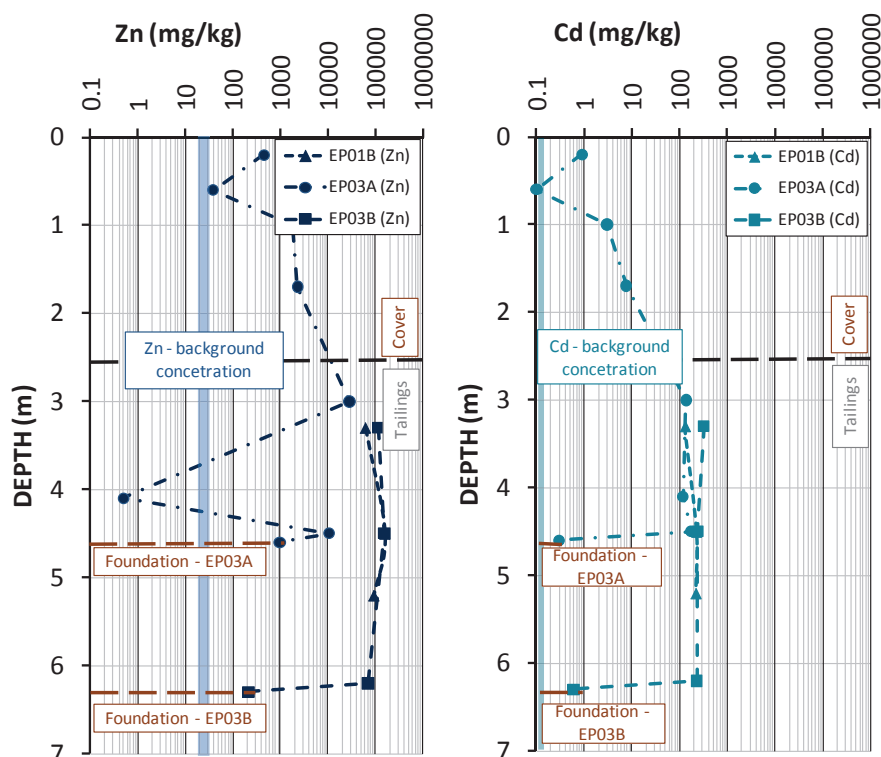


Fig. 5. (a) Sulfate (SO_4) profile; (b) Zinc (Zn) profile; (c) Lead (Pb) profile; (d) Cadmium (Cd) profile

4.0 CONCLUSIONS

The BSTD was constructed as part of development of Century mine during the 1990s. The engineered cover system employed to decommission the BSTD eventually failed, allowing recharge of the tailings, acid mine drainage and capillary rise of contained metals.

It is likely that this occurred because the topsoil storage layer was too thin. This rainwater was able to infiltrate the soil layer during very wet periods. Once infiltration infiltrated the topsoil, it ponded over the RPL. However, established vegetation failed to remove excess infiltration. As a result the upper RPL was slowly wet-up by infiltrating rainfall that passed through the topsoil layer.

Once infiltration breakthrough of the upper RPL occurred infiltration slowly wet up the capillary break. Field observation during excavation noted that roots did not penetrate into this layer therefore Infiltration that reached this point in the cover could no longer be removed by transpiration. Breakthrough of the lower RPL occurred and percolation into the underlying tailings followed. The capillary break contained a high proportion of clay and sand sized particles/fines which promoted capillary rise of dissolved metals throughout the cover. This is because the fines create uniform and contiguous capillaries that salts and metals can rise through.

Differential settlement of the BTSD compounded the above described process. The depression allowed ponding of water at the surface of the BTSD which pushed water into the

constructed cover at a faster rate than might have occurred on flat sections of the BTSD constructed cover.

The design process for the BSTD mistakenly characterized site material and then went straight from concept to cover construction. As a result, they missed the opportunity to use the critical design data needed to increase rehabilitation success and ultimately contributed to the failure of the system.

This example demonstrates that the complexity of engineered cover systems requires a design process to be followed such as that in the GARD Guide (2009).

3.0 REFERENCES

- NEPM (1999) National Environment Protection (Assessment of Site Contamination) Measure.
- Environmental Geochemistry International Pty Ltd (EGi) (1998a) Report to Pasminco Century Project Evaluation of cover design options for the bulk sample tailings dam. Dated September 1998.
- Environmental Geochemistry International Pty Ltd (EGi) (1998) Report to Pasminco Century Project, Assessment of acid forming characteristics and metals leaching behavior. Dated February 1998.
- GARD Guide (2009) Global Acid Rock Drainage Guide.

USING KINETIC GEOCHEMICAL TESTWORK TO ASSIST WITH MINE PLANNING, OPERATIONS AND POST CLOSURE

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ABSTRACT

Kinetic geochemical testwork is routinely used to assist with the acid and metalliferous drainage (AMD) assessment component of environmental and social impact studies during the pre-feasibility and feasibility stages of mine project development. Such studies are generally completed to fulfil regulatory requirements, but kinetic testwork data can also be relevant to mine planning, operations and the development of closure strategies.

The three main types of kinetic geochemical test procedures are column or test pile leach, humidity cell and oxygen consumption. The primary aim of these tests is to quantify acidity generation rates, estimate the likely lag time prior to the onset of acid drainage and to provide an indication of the likely quality of leachate from mine materials. In essence, these tests routinely identify the need for AMD / NMD management and provide focus on problematic materials and potential soluble pollutants. The three common procedures offer different capabilities for assessing the geochemical behaviour of mine materials and the strengths and limitations of each approach is discussed.

Recent advances in kinetic test procedures have extended their capabilities beyond routine assessment of material behaviour under wetting and drying scenarios, to the evaluation of closure scenarios of increasing complexity. Kinetic procedures now permit accurate determination of acidity generation rates from waste rock, wallrock in pits, underground voids, tailings, ore and concentrate as a function of particle size distribution, sulfide type, sulfide concentration, moisture content (analogue for oxygen diffusion), humidity, temperature, oxygen concentration and pore moisture pH. With independent control over these test variables, mine plans, closure strategies, remediation options and water treatment requirements can be more accurately designed and evaluated.

Kinetic test procedures make it possible to quantify the likely performance of management strategies such as: oxygen limiting cover systems; waste rock encapsulation, waste rock blending or waste rock-tailings co-disposal; optimising waste rock compaction and moisture contents to limit acidity generation; limiting moisture contents in waste rock in arid settings; paste tailings, paste rock or tailings-cement mixtures; alkalinity producing covers to decrease the pyrite oxidation rate in mine wastes; variable depth water covers over tailings; and inert atmospheres in mine voids. The procedures can also assist with pit lake water quality modelling and dewatering water quality predictions.

As their predictive capability improves, kinetic geochemical test procedures are becoming increasingly relevant to mine planning, operational activities, closure planning and site rehabilitation.

1.0 INTRODUCTION

Kinetic geochemical testwork is routinely used to assist with the assessment of acid and metalliferous drainage (AMD, also called acid mine drainage or acid rock drainage) as a component of environmental and social impact studies during the pre-feasibility and feasibility stages of mine project development. Such studies are generally completed

primarily to fulfil regulatory requirements and are rarely subject to further interpretation. However, kinetic geochemical testwork can in fact provide a range of important and highly useful information that can greatly assist mine planning, operations and the development of effective closure strategies. This technical review presents the types of information that can be obtained using the latest kinetic test methods, and the range of potential management strategies that can be both developed and directly evaluated using kinetic testwork.

The main objectives of effective AMD management are to:

- Prevent sulfide oxidation where possible;
- If sulfide oxidation cannot be avoided:
 - minimise sulfide oxidation rates;
 - maximise the potential for the neutralisation of generated acidity;
 - maximise the formation of insoluble acid sulfate salts;
 - lower net acidity release rates to levels that can be handled naturally by the receiving environment or treated by passive measures.
- If acidity release rates cannot be sufficiently minimised, then appropriate water treatment is required.

The purpose of kinetic geochemical testwork is to identify the likely rate of acidity or metals release over time due to the oxidation of sulfide minerals (primarily pyrite), and assess likely leachate chemistry. Kinetic testwork is needed because sulfide oxidation rates vary with the specific type of reactive sulfides, including mineralogy (e.g. pyrite, pyrrhotite, chalcopyrite, etc), thermo-barometric and structural evolution (e.g. metamorphic grade and deformation history / dislocation density), the surface area (e.g. framboidal or coarsely crystalline materials), and crystal chemistry (i.e. element substitutions, impurities) of sulfides. As sulfide oxidation rates can vary by orders of magnitude between different samples and cannot be reliably estimated from static geochemical data or mineralogy alone, kinetic geochemical testwork is needed to determine the sulfide oxidation rate, or pollution generation rate, of any given material.

2.0 KINETIC GEOCHEMICAL TEST METHODS

Three different kinetic geochemical test procedures are currently in widespread use, each with differing capabilities, advantages and disadvantages: column leach tests (or test piles), humidity cell tests and oxygen consumption tests. The primary aim of these tests is to quantify acidity generation rates, estimate the likely lag time prior to the onset of acid drainage, and to provide an indication of the likely quality of leachate from mine materials. In essence, these tests are routinely used to identify the need for management of AMD, neutral metalliferous drainage (NMD) and salinity, and provide focus on problematic materials and potential soluble pollutants.

2.1 Column Leach and Humidity Cell Tests

Column leach tests, most commonly conducted as a free-draining leach test, involve measurement of the sulfate flux from a periodically wetted column of test material (such as waste rock or tailings) normally over a period of 12-24 months. The rate of sulfide oxidation is then estimated from the measured sulfate flux. Humidity cell tests are conducted on a similar principle but aim to accelerate sulfide oxidation by applying elevated temperature and humidity.

These tests are used worldwide, but are subject to a number of important limitations and potential inaccuracies with respect to determining the rate of sulfide oxidation and hence acidity generation. Most significantly, the measured sulfate flux often does not accurately reflect actual sulfide oxidation rates for the following reasons:

- In the majority of natural materials, sulfur released by the oxidation of sulfides will at least in part be re-mineralised as secondary sulfate salts, such as jarosite, in the presence of aluminosilicates and hence may not be present in leachate.
- The dissolution of existing jarosite (which is sparingly soluble) in test materials, particularly from significantly oxidised materials, during periodic flushing will contribute to the measured sulfate flux, even if no sulfide is present.
- The dissolution of gypsum (which has relatively high solubility) during wetting and flushing will contribute to measured sulfate flux, even if no sulfide is present. Gypsum can be present as a primary mineral in some cases, but is often present in oxidised sulfidic materials containing reactive acid neutralising capacity (ANC).
- The conditions of the test (temperature, moisture, flushing frequency) can differ considerably from actual conditions in the field.
- Repeated wetting and drying is generally inconsistent with climatic conditions at most mines and can result in unrepresentatively short acid residence times that interrupt slower acid neutralisation processes such as aluminosilicate dissolution.

Column leach tests do, however, provide good approximations of likely sulfate release and metal leach rates, and are therefore valuable for leachate characterisation tests. Sulfide oxidation rates estimated from column leach tests, however, are generally considered to be unrepresentative for the reasons above, sometimes significantly so (by an order of magnitude or more, both under- and overestimated).

2.2 Oxygen Consumption Tests

Oxygen consumption cell tests involve direct measurement of oxygen consumption due to sulfide oxidation in a hermetically sealed cell, providing a direct and robust measurement of the sulfide oxidation rate for a given test material, independent of gypsum/jarosite formation and dissolution. Oxygen consumption-based tests for the determination of sulfide oxidation rates have been used in the field (ANSTO 1997; Cook et al. 2004; Kempton, 2009; Ward et al. 2004a, 2004b; Scarek et al. 2006), and laboratory (Anderson et al. 1999; Bennett et al. 2005; Eidsa et al. 1997; Hollings et al. 2001; Bourgeot et al. 2011 and Schmieder et al. 2012) for over a decade, but the utility of the method for AMD management, not just measurement, has yet to be widely recognised. In some oxygen consumption apparatus, the test also permits measurement of carbon dioxide generation rates derived from carbonate dissolution (i.e. carbonate neutralisation rates). Such measurements also permit assessment of the impact of carbon dioxide on dilution of oxygen measurements to be accounted for in the test method.

The key advantages of oxygen consumption tests compared with leach tests are discussed in Bourgeot et al. 2011 and Schmieder et al. 2012. These include:

- Better accuracy through direct measurement of sulfide oxidation rates via oxygen consumption (no sulfate interferences).
- Rapid determination of sulfide oxidation rates – in most cases accurate oxidation rates can be obtained within 1-4 weeks, compared with 12-24 months for a column leach test.
- Tests can be performed at considerably lower cost due to shorter test durations and fewer analyses.
- Test configurations permit assessment of sulfide oxidation rate as a function of moisture content.
- Tests can be configured to investigate any number of specific parameters or material management strategies, such as water covers, low permeability covers, low oxygen atmospheres, neutralisation agent blending etc.
- Due to the relatively low cost and short duration, multiple tests can be performed to investigate a particular pyrite oxidation control variable of interest, the effectiveness

- of material management strategies or to refine material management strategy parameters.
- Carbon dioxide generation can be measured independently to improve measurement accuracy and in some cases to quantify neutralisation rates.

Where the results of column leach tests provide limited accuracy and utility in mine planning, the data that can be obtained through oxygen consumption tests present some valuable opportunities for testing and refinement of proposed geochemical engineering management strategies.

3.0 KEY KINETIC GEOCHEMICAL PROCESSES RELEVANT TO MINE PLANNING

Using oxygen consumption tests, it has been possible to reveal and explore a number of important geochemical processes and relationships relevant to the generation of AMD and NMD, as outlined below. Each of these processes has the potential to be applied to AMD management.

3.1 Pyrite Oxidation Rate and Pyrite Decomposition Over Time

The results of kinetic geochemical tests are typically reported in units of kilograms of sulfur oxidised per tonne of material per year (kg S/t/yr), or kilograms of oxygen consumed per tonne of material per second (kg O₂/t/s). Both of these units effectively describe only the intrinsic rate of acidity generation for the sample under test (i.e. for a specific sulfide content). That is, these values cannot be readily applied to materials with different sulfide contents, or even the same sample after a period of oxidation. Hence the values provide little information on acidity generation rate 2, 10 or 100 years in the future.

Another way to represent sulfide oxidation rates is by normalising the observed oxidation rate against the sulfide content of the sample. This is typically performed using the pyrite equivalent content, which assumes all sulfide is present as pyrite (the most common sulfide), and can be expressed as a pyrite oxidation rate (POR) in units of weight percent of remaining pyrite (equivalent) that oxidises to form sulfuric acid (equivalent) acidity per year (wt% Py/yr).

In-house oxygen consumption cell tests and those conducted by ANSTO and others have shown that the oxygen consumption rate is proportional to sulfide sulfur content for materials having the same type of pyrite (i.e. rocks with the same geological / thermo-barometric history). An example of such data is obtained through in-house oxygen consumption cell testwork is shown in Figure 1(a).

When expressed in units of wt% Py/yr, the POR remains effectively constant over a wide range of sulfide concentrations for comparable pyritic 'lithologies', as shown conceptually in Figure 1(b). An example of data obtained using oxygen consumption tests for different samples of the same lithology (but differing sulfide content) is shown in Figure 1(c). In this example, the POR determined by oxygen consumption cell tests varies between ~3 and 4 wt% Py/yr, whereas column leach tests on the same materials gave widely varying POR values of 2.5-44 wt% Py/yr, in this case reflecting the dissolution of gypsum and release of non-pyrite-related sulfate.

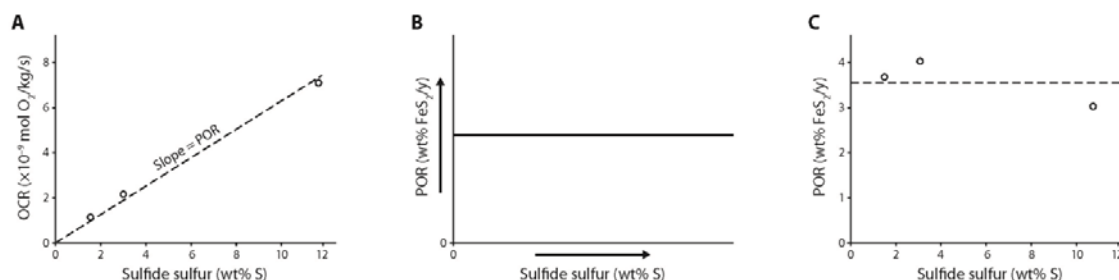


Fig. 1: (a) Example of linear relationship between oxygen consumption rate and sulfide content for materials of comparable lithology. (b) Conceptual relationship between sulfide content and the pyrite oxidation rate (POR) in units of wt% pyrite per year, as determined from measurements of oxygen consumption in an oxygen consumption cell test. (c) Example of data obtained in oxygen consumption cell tests on samples of comparable lithology with different absolute sulfide contents.

The presentation of the POR as a normalised value in units of wt% Py/yr has two important implications:

- The POR value can be applied to all materials of similar geological history as a function of sulfide sulfur grade, which permits estimation of field-scale acidity generation rates (in units of kilograms of sulfuric acid equivalent acidity per year) using average sulfide sulfur grades.
- The POR remains constant over time, but the acidity generation rate calculated from it is proportional to the amount of sulfide remaining, decreasing over time according to a decay curve consistent with the progressive decomposition of sulfide.

3.2 Net Acidity Generation Rate

Using normalised POR values, it is possible to predict the long-term evolution of the acidity generation rate consistent with the progressive decomposition of sulfide, as shown in Figure 2. This prediction provides an estimation of the longevity of sulfide oxidation (and hence acidity generation), as well as a prediction of the maximum potential acidity generation at any time point in the future.

As the POR can be considered constant for materials with comparable geological histories, the acidity generation rate and its evolution over time can be recalculated for any sulfide content, such as a lithological average for example, permitting estimation of field-scale acidity generation rates for waste rock piles or reactive portions of tailings accumulations.

The acidity generation rate over time can also be used to evaluate acidity loads on mine closure and beyond, providing key information for assessing potential short- and long-term AMD management requirements.

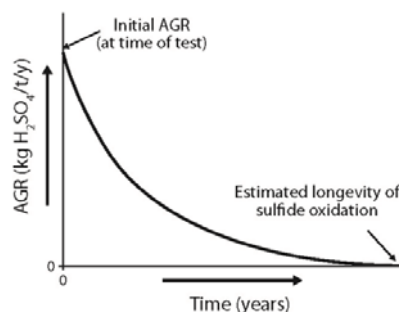


Fig. 2: Conceptual evolution of acidity generation rate (AGR) over time based on POR (in units of wt% pyrite per year).

By drawing on ANC data (a static geochemical parameter), it is possible using the POR to estimate the lag time to onset of acid conditions for any given material. However, the standard ANC test effectively treats all potentially neutralising mineral content as 100% available and efficient for acid neutralisation. This is rarely achieved under real-world conditions and results in a significant overestimation of ANC in most cases¹. Hence, a better estimate of lag time can be established by using a combination of a sample's POR and the acidity consumed to pH 4.5 on its acid buffering characteristic curve (ABCC) test. An example of this is shown in Figure 3. The ABCC test is a quasi-kinetic geochemical test that involves incremental addition of acid to a test material. As with kinetic testwork, the ABCC test is under-utilised in the interpretation of geochemical characteristics, but essentially can be considered analogous in many ways to the progressive oxidation of sulfide and associated acid generation.

That is, the amount of acid required to lower the rock material-water mixture pH to 4.5 can be considered a good approximation of the ANC that is effective in maintaining non-acid conditions over time. In Figure 3, two rock samples with the same measured ANC yield significantly different $ANC_{4.5}$ values, due to the different reactivity and/or availability of the acid neutralising minerals within the sample. $ANC_{4.5}$ values can be a factor of 10 or more less than measured ANC for a given sample.

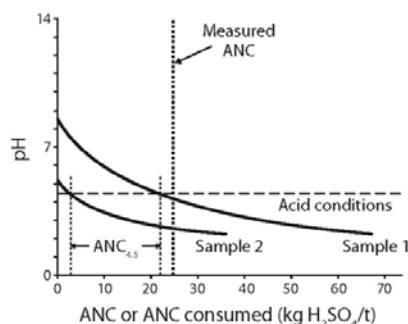


Fig. 3: Example acid buffering characteristic curves (ABCCs) for two different rock samples with similar measured ANC.

Using the $ANC_{4.5}$ value and the POR, the evolution of the net acid generation rate (i.e. net acid generation after the effect of acid neutralisation) over time can be predicted, as shown in Figure 4.

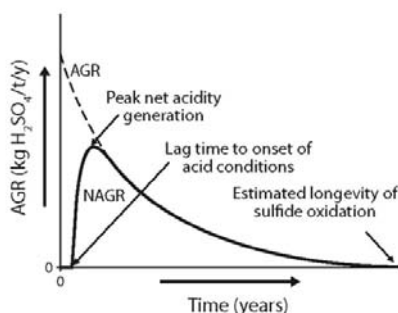


Fig. 4: Conceptual evolution of net acidity generation rate (NAGR) over time based on POR (in units of wt% pyrite per year), $ANC_{4.5}$ and total ANC.

The NAGR curve shows the minimum lag time for a material (based on $ANC_{4.5}$), providing a quantitative measure of the maximum handling time for potentially acid forming materials including ore and concentrate. The utility of the NAGR curve in mine planning is clear and is

¹ pH values achieved in the standard ANC test are routinely below pH 2, which may result in overestimation of the measured ANC with respect to real-world conditions, due to the enhanced dissolution of some alumina-silicate minerals in the sample.

both readily understandable and directly applicable to materials scheduling and management.

3.3 Temperature

Temperature has a pronounced effect on sulfide oxidation rates even over the range of typical natural conditions. The trend observed from oxygen consumption cell tests is shown in Figure 5. Oxygen consumption cell data for multiple samples of comparable material (as reported in Anderson et al. 1999) suggests that from 20°C to 30°C, the POR for the same sample increases by 6-8 times.

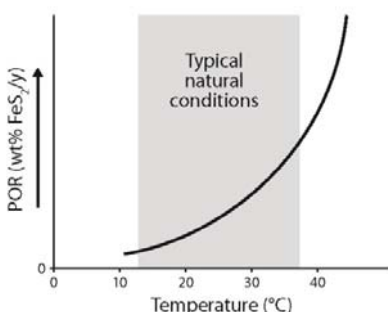


Fig. 5: Conceptual relationship between ambient temperature and POR.

Understanding temperature effects and accounting for them in laboratory kinetic geochemical testwork is crucial, as laboratory results can differ markedly from those under field conditions for both cold (Elberling et al. 2005) and hot regions.

Managing temperature effects can be crucial for AMD mitigation. In some sulfidic materials, pyrite oxidation (an exothermic reaction) can locally increase temperatures to a point where pyrite oxidation becomes instantaneous (i.e. spontaneous combustion occurs). Detailed dump construction methods specifically designed to control waste rock heating rates have been developed for highly reactive waste materials. Strategic placement of limited tonnages of reactive waste between layers of non-sulfidic materials with sufficient thermal mass to dissipate heat, lower oxidation rates and ultimately to prevent spontaneous combustion, is an example waste management to limit pyrite oxidation rates.

3.4 Particle Size

In-house oxygen consumption cell testwork indicates that the POR is proportional to the particle size of the sample. The conceptual relationship is shown in Figure 6(a). The shape of the curve closely follows the theoretical surface area per volume curve, suggesting that pyrite surface area provides a fundamental control on pyrite oxidation rate. The exact position and shape of the particle size - oxidation rate curve varies between samples with different geological histories. Figure 6(b) shows an example of typical data obtained in the course of in-house oxygen consumption cell testwork aimed at testing the influence of particle size on POR.

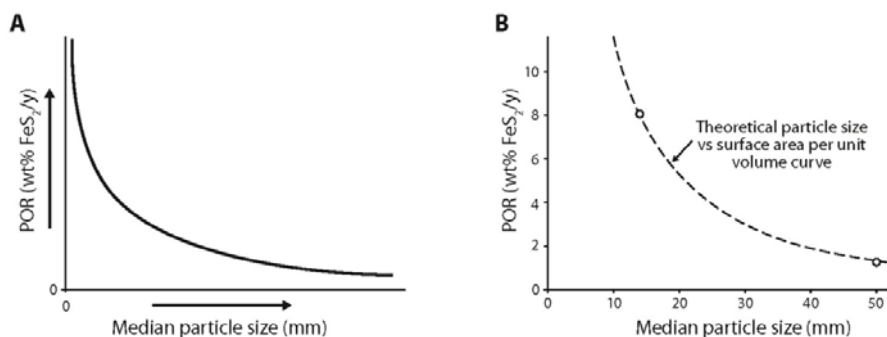


Fig. 6: (a) Conceptual relationship between particle size and POR. (b) Examples of data obtained in oxygen consumption cell tests for the same sample at different particle sizes.

The importance of understanding the reactivity of mine wastes as a function of particle size is important for mine operations as careful use of explosives can influence the particle size distribution of waste materials.

3.5 Moisture Content

The typical relationship between POR and gravimetric moisture content (GMC) of geologic materials as obtained through oxygen consumption cell testwork is shown in Figure 7(a). The POR typically peaks at a GMC equivalent to approximately 33-66% of saturation, but notably does not decrease to zero at either 0% GMC or complete saturation. A ten-fold variation in POR across the full moisture range is common, and at least a two-fold variation across the range of intermediate GMC conditions (20-80% of saturation) can be expected. Figure 7(b) shows an example of the variation in POR for the same sample at different GMCs. These data are taken from oxygen consumption cell testwork aimed at characterising the effect of moisture content on POR.

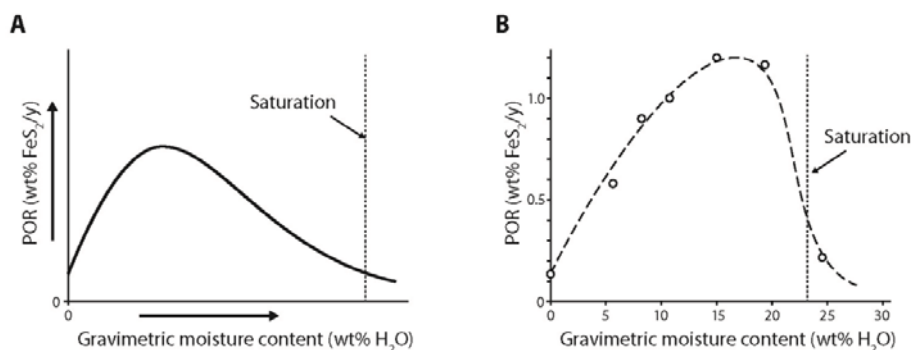


Fig. 7: (a) Conceptual relationship between moisture content and POR. (b) Example of data obtained in oxygen consumption cell tests for the same sample at different gravimetric moisture contents (GMCs).

This GMC vs POR relationship is one of the most important relationships that can be demonstrated and explored using oxygen consumption cell tests. Such data cannot be obtained by column leach methods. The typical relationship shows that pyrite oxidation can be significantly lowered at the moisture extremes (dry and saturated), but apparently cannot be easily halted.

At the very low moisture end, it is considered likely that even trace water levels, or atmospheric moisture concentrations only, are sufficient to allow pyrite oxidation to continue,

albeit at a diminished rate that reflects the role of water in the oxidation process (Jerz and Rimstidt 2004). At the initial point of saturation, sulfide oxidation can continue at a lower rate via the diffusion of oxygen through pore water.

Oxygen consumption cell tests can be used to quantify the effect of moisture on sulfide oxidation rates either to identify target moisture contents for AMD management, or to estimate the likely variation in sulfide oxidation rate across a profile of variable moisture content. For example, the variation in sulfide oxidation rates in de-saturated wall rock above a groundwater cone of depression can be quantified across the moisture profile in order to estimate acidity release rates. The effectiveness or otherwise of irrigating ore stockpiles to suppress sulfide oxidation can also be quantified using oxygen consumption cell techniques.

3.6 Water Cover Depth

As mentioned above, achieving initial saturation alone is insufficient to halt all sulfide oxidation. A water cover of some depth is required to halt the process. Oxygen consumption cell tests provide a direct and accurate means of investigating the effect of water cover depth on sulfide oxidation rates. The typical relationship obtained in such tests is shown in Figure 8(a).

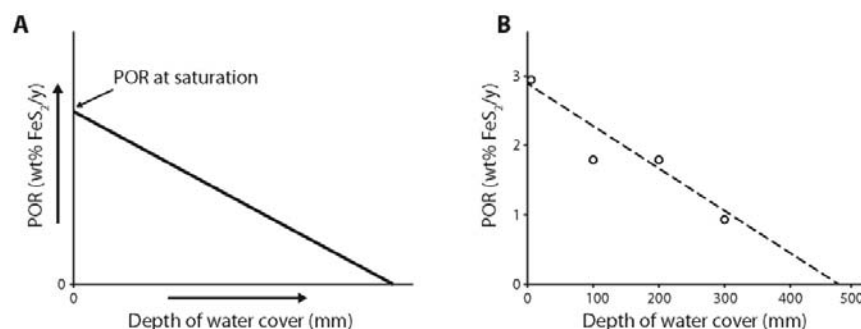


Fig. 8: (a) Conceptual relationship between water cover depth and POR. (b) Example of data obtained in oxygen consumption cell tests for the same sample at different water cover depths.

Over a range of shallow water depths, the rate of sulfide oxidation decreases approximately linearly with increasing water depth, under the static conditions of an oxygen consumption cell test. This reflects the fact that the rate of oxidation through a water cover is limited to the rate of oxygen diffusion through the water column.

Data obtained using oxygen consumption cells tests examining the effect of different water cover depths for the same material is shown in Figure 8(b).

In addition, the effectiveness of water cover enhancement strategies such as sand covers to prevent sulfide re-suspension and aggregate addition to reduce water surface area (and lower oxygen diffusion rates) can also be tested and refined using oxygen consumption cell tests.

3.7 Oxygen Concentration

In most circumstances, the rate of sulfide oxidation is proportional to atmospheric oxygen concentrations. The conceptual relationship based on in-house oxygen consumption cell tests is illustrated in Figure 9(a). The relationship is sublinear in the natural region (0-20.9% O₂). An example of a data series obtained in an oxygen consumption cell test for sulfidic mine waste is shown in Figure 9(b).

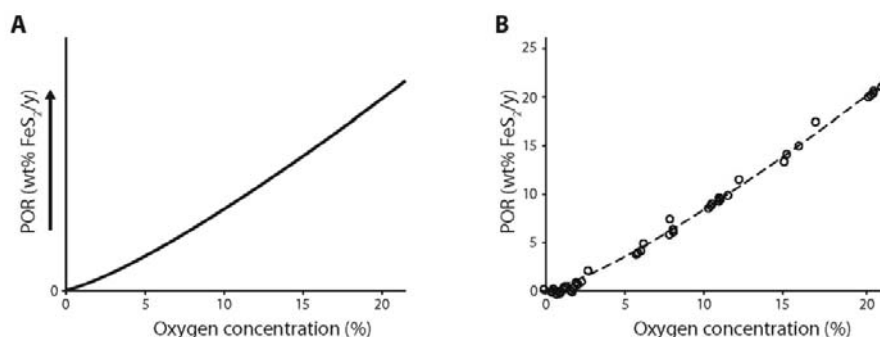


Fig. 9: (a) Conceptual relationship between atmospheric oxygen concentration and POR. (b) Example of data series obtained in an oxygen consumption cell test.

This relationship demonstrates that in order to achieve significant suppression of sulfide oxidation, atmospheric oxygen concentrations need to be lowered to approximately 3 vol.% or lower. These findings are of particular relevance to mine closure planning for waste storage facilities relying on limiting oxygen ingress to sulfidic materials.

4.0 USE OF KINETIC GEOCHEMICAL TESTWORK FOR GEOCHEMICAL ENGINEERING

The key kinetic parameters outlined above provide the basis for a wide range of potential geochemical engineering strategies that can be investigated using kinetic geochemical testwork and in particular, oxygen consumption cell testwork. This approach makes it possible to validate and refine closure scenarios of increasing complexity and sophistication.

4.1 Waste Rock Encapsulation / Oxygen-Limiting Cover Systems

The encapsulation of potentially acid forming waste rock within a layer of low-permeability material such as clay can lower sulfide oxidation rates by limiting oxygen supply to sulfides to diffusion related processes. The merits of various encapsulation strategies can be quantified at both laboratory and pilot scales using oxygen consumption cell techniques at the design stage.

4.2 Waste Rock Blending and Waste Rock/Tailings Co-Disposal

Using the sulfide oxidation rate obtained using kinetic geochemical testwork and the net acid generation rate (NAGR) (e.g. see Fig. 4), it is possible to simulate the effect of blending and co-disposal strategies for the management of mine waste. An example of blending potentially acid forming (PAF) waste rock with non-acid forming (NAF) material containing reactive ANC (e.g. enhancing the significant ANC_{4.5} of the blended material) at a 1:1 mass ratio is shown in Figure 10. The resultant material has half the AGR at any time point (due to sulfide 'dilution'), and the lag time is extended by the addition of reactive ANC, resulting in a considerably more manageable NAGR profile. The aim of such geochemical engineering is to develop a blending strategy that produces a material that is not expected to release acidity over the entire period of sulfide oxidation.

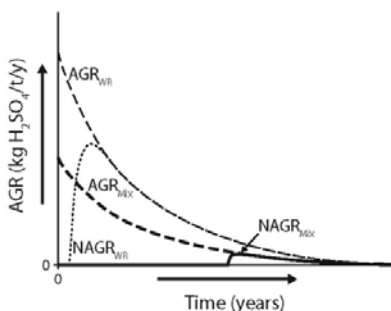


Fig. 10: Example of simulated evolution of NAGR for blended material (denoted 'mix') based on the AGR curve and $ANC_{4.5}$ value for PAF waste rock ('WR') and NAF material with additional $ANC_{4.5}$ assuming a 1:1 blend (see details above).

Such simulations can be performed to identify minimum blending and ANC requirements, or to characterise the NAGR profile of an existing or pre-defined blend.

4.3 Optimisation of Waste Rock Compaction and Enhancing Moisture Content

Limiting oxygen ingress and maximising acid neutralisation potential are key AMD management strategies for waste rock materials. Encapsulation and cover systems are important aspects of this strategy, but the PAF waste rock pile itself can also be optimised through base-up waste rock dump construction to form thin lifts that are compacted at optimal moisture content. The effectiveness of this strategy for lowering sulfide oxidation rates at a site can be quantified directly using oxygen consumption cell tests at laboratory and pilot scales, during the design phase. Carbon dioxide generation can also be monitored at laboratory and pilot scales to quantify acid neutralisation rates using carbonates, a factor that may be relevant to the geotechnical stability of the waste rock dump, should significant quantities of mass loss (via CO_2 emissions) be expected.

4.4 Limiting Moisture Content in Arid Settings

The general relationship between moisture content and sulfide oxidation rate (e.g. see Fig. 7) indicates that sulfide oxidation can be suppressed by limiting pore water content and atmospheric moisture. However, this strategy is only likely to be effective in arid environments with low moisture and low humidity, and for materials with good drying properties (e.g. coarse waste rock rather than clays). The degree of sulfide oxidation suppression provided by this strategy can be readily quantified at both laboratory and field scales using oxygen consumption techniques.

4.5 Waste Rock and Tailings Co-Disposal

The benefits of waste rock and tailings co-disposal on lowering sulfide oxidation rates can be readily measured at laboratory or field scale using oxygen consumption techniques. Quantifying the optimum mix ratios, moisture content and compaction for retarding sulfide oxidation can be measured directly. The drying rates and associated increases in sulfide oxidation rates can also be measured.

4.6 Alkalinity-Producing Covers

The introduction of alkalinity into mine waste covers can assist with the passivation of reactive sulfide minerals (e.g. precipitated calcium carbonate, caustic magnesia, kiln dust or fly ash; Taylor et al. 2009). The soluble alkalinity raises pore water pH in the underlying waste pile. This can lower sulfide oxidation rates by passivating sulphide minerals through

the precipitation of secondary minerals on sulfide surfaces. Oxygen consumption testwork can be used to quantify the effect of sulfide passivation and neutral pore water pH values on sulfide oxidation rates in comparison to unmodified systems. Such data can permit the refinement of AMD management strategies, including the type and mass of alkalinity addition, the optimal pore water pH and cover design aspects.

4.7 Inert Atmospheres in Mine Voids

Some underground mines offer the possibility of using an atmospheric oxygen-limitation strategy to suppress sulfide oxidation to manageable or environmentally acceptable rates. The void can be filled with inert gas (e.g. CO₂ and/or N₂) under slight positive pressure to significantly lower the rate of sulfide oxidation and hence acidity generation (eg. Taylor and Waring 2001). Oxygen consumption cell techniques offer an easy way to identify target oxygen levels and test the effect of atmospheric gas compositions on effective sulfide oxidation rates. The oxidation rates determined by oxygen consumption cell tests can also be used, for example, to estimate the time required to deplete atmospheric oxygen as a result of sulfide oxidation in an air-sealed (but not water sealed) underground mine.

4.8 Variable Depth Water Covers

In situations where the water cover is expected to fluctuate, it is possible using oxygen consumption cell tests to quantify the change in sulfide oxidation rate with water cover depth (see Fig 8(b) for an example of such an investigation). Specific scenarios can be tested at large and pilot scales using oxygen consumption techniques using HDPE tanks, bags and large columns.

5.0 CONCLUSIONS AND FUTURE DIRECTIONS

Kinetic geochemical testwork methods have advanced significantly in the last decade and the latest techniques are highly flexible and capable of quantitatively exploring the effects of a wide range of parameters on sulfide oxidation rates with unprecedented accuracy, precision and speed, and at low cost. This has significantly raised the utility of kinetic geochemical data for geochemical engineering and AMD management. Oxygen consumption cell tests now permit the fast, accurate and inexpensive determination of acidity generation rates from waste rock, wallrock in pits, underground voids, tailings, ore and concentrate as a function of particle size distribution, sulfide type, sulfide concentration, moisture content, humidity, temperature, atmospheric oxygen concentration and pore water pH. With independent control over these test variables, mine plans, closure strategies, remediation options and water treatment requirements can be more accurately evaluated. Such information can also assist with void dewatering water quality predictions and pit lake water quality modelling.

Kinetic geochemical testwork can be used to design, confirm and refine geochemical engineering strategies at the design stage, providing a wider range of AMD management tools and confidence than previously available. Kinetic geochemical testwork is no longer just a regulatory requirement. Oxygen consumption cell tests represent a valuable tool that can be used to quantify the benefits and water quality outcomes of AMD management strategies during operations and post closure.

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HETEROGENEITY PROFILING: A TECHNIQUE TO IMPROVE GEOCHEMICAL SAMPLING AND ANALYSIS FOR AMD ASSESSMENTS

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ABSTRACT

Heterogeneity is the measure of the degree of compositional variability of a material. For typical waste geochemistry assessments heterogeneity affects range from macroscale (inter-sample) to microscale (intra-sample) heterogeneity. Or, put another way, heterogeneity effects are scalar as well as spatial in nature. In the majority of cases therefore, the degree to which the scalar effects of heterogeneity are understood is likely to be a key factor in determining how successful a characterisation program is likely to be.

As part of routine geochemical characterisation programs, materials are generally classified based on the analysis of a finite (often small) number of samples. Although these effects of heterogeneity are acknowledged it is not common for detailed assessment of heterogeneity to be routinely carried out as part of acid mine drainage (AMD) assessments.

As part of a large scale materials assessment programme, O'Kane Consultants (OKC) has taken the opportunity to study the effects of material heterogeneity on geochemical assessment in detail. As part of the assessment, OKC has developed and trialled a profiling tool to allow the quantitative assessment of the scalar and temporal effects of heterogeneity on geochemical classification. The tool has been developed and used to gain an understanding of the following

- *The influence of intra-sample particle size*
- *The influence of inter- and intra-sample compositing*
- *The influence of intra-sample mineralogy*
- *The influence of sample preparation (crushing, grinding, sieving, splitting)*

1.0 INTRODUCTION

The geochemical characterisation of materials forms the fundamental basis of acid and metalliferous drainage (AMD) studies. Because of the need for a scientific and rational approach to materials characterisation, internationally accepted testing standards (e.g. ASTM), and guideline documents have been developed to cover the sampling and characterisation process. These documents have been developed by technical experts and specialist groups (e.g. INAP) and, are generally used as standard reference documents throughout the industry. The vast majority of these documents cover the topic of sampling accuracy, precision and correctness and most have a strong focus on the fundamental principles of analytical precision given in the context of laboratory analysis. Much of the focus is given to analytical precision and measurements of this concept. However, when dealing with the kind of heterogeneous materials encountered during AMD studies, the term analytical precision must clearly be distinguished from attempts to determine sampling accuracy and, indeed, sampling error in relation to materials characterisation. Previous studies and decades of research within the resources industry have highlighted that sampling

errors relating to insufficient sampling frequency and sampling methodology, rather than analytical precision, are in many cases the controlling factor on data quality. The discussion of how to improve data quality and therefore how to best characterise the heterogeneous material should therefore move away from the current central focus of analytical precision to be re-focused on how to improve sampling method and frequency as the most effective means to address what and how materials are sampled and characterised.

It is interesting to note that much of the technical developments of sampling theory and practice has been developed by the mining industry over many years. The establishment of Joint Ore Reserves Committee (JORC) 2012 reporting standards for example are a case in point as they have very comprehensive guidelines with respect to reporting of sampling methods, measurements of sampling error and correctness. Because AMD assessments are typically carried by environmental geoscientists and not resource geologists, these theories, methods and practices developed by the mining industry have not generally been integrated into AMD assessments.

The lack of integration of the “theory of sampling” (TOS) used by the mining industry into AMD studies, a number of problems are commonly and repeatedly raised during AMD assessments for which solutions have not been consistently developed or adopted. A selection of these questions include:

- What size should the samples be?
- What preparation method should be carried out prior to analysis?
- How much analysis is required?
- How should data quality objectives (DQO)s be defined and assessed?

Many published AMD related guidance documents offer little in the way of specific guidance on how to tackle some of these questions.

This paper has been prepared in an attempt to integrate some of the principles of TOS and practice from the resource characterisation industry to AMD studies, and to provide a useful tool for answering the questions given above.

2.0 SAMPLING THEORY

A comprehensive publication of the theory of sampling of particulate materials was published in 1979 by Dr Pierre Gy, the last publication being 1993 by means of a translated overview of his work by Francis F. Pitard (Pitard 1993). The theory covers all facets of sampling, how errors are generated, how errors can be reduced or eliminated, and how the residual error can be estimated. The theory of sampling (TOS) has been widely adopted in the resources industry. In contrast, it is not common for TOS to be familiar to many in the environmental geoscience field which includes many practitioners of AMD management. Indeed to the authors’ knowledge, TOS is not routinely covered in university courses such as environmental science, and it is very uncommon to see AMD-based research papers produced that make reference to TOS.

TOS indicates that total sampling variance is the preferred measure of the total sampling error, but determining the variance into its component parts is very challenging. Ten sources of sampling error (Pitard, 1993) that contribute to non-representativeness of samples have been identified and include:

1. In situ Nugget Effect (NE)
2. Fundamental sampling error (FE)
3. Grouping and segregation error (GE)
4. Long-range heterogeneity (quality) fluctuation error (QE1)

5. Long-range periodic heterogeneity (quality) fluctuation error (QE2)
6. Increment delimitation error (DE)
7. Incremental extraction error (EE)
8. Weighing error (WE)
9. Preparation error (PE)
10. Analytical error (AE)

Errors 1-5 are random and are related to the intrinsic properties (heterogeneity) of the materials we are analysing, errors 6-10 relate to laboratory analytical techniques.

Consideration of all ten of these errors is far beyond the scope of this paper, and in some cases is not relevant to many AMD sampling programs, however for many waste material characterisation assessments, it is FE that is of primary importance. Given this fact a large proportion of the paper is given to the concept of FE.

Through numerical calculation and testing programs, it is possible to estimate, measure and influence FE by determination of a materials heterogeneity and interrogation/optimisation of the sampling process used as part of analysis.

It is relevant to note that FE is the only error that can be reasonably estimated before sampling is carried out (Minnitt et al 2007).

As previously stated, industry standard AMD publications tend to focus on AE as the key measure of sampling representativeness as this is one of the most widely quoted means of assessing sampling accuracy. As can be seen, this is only one of the ten possible errors however and with modern equipment AE is commonly of much less importance than FE. Analytical errors for modern equipment can be expected to be less than 5% in many instances based on quoted values by commercial laboratories for standard analysis like ICP (metals). Errors related to FE however have been found to be >40% in many case studies.

It is relevant to note that AE values of >5% are in fact commonly reported in many material characterisation studies so the reader may question the 'discounting' of AE as part of this assessment but, upon even a cursory review of how laboratories/consultants/industry report AE, it is likely that it is FE that is being incorrectly reported as AE. This 'accusation' can be proven simply by looking at commercial laboratory reports that quote QA/QC data that quantify AE but make no attempt to quantify FE.

Generally, the error introduced by modern laboratory-based analytical instrumentation as analytical bias is likely to be relatively low; therefore the precision of laboratory analytical test data is high. A study carried out in Europe (CLAIRE, March 2004) indicates that for a given case study the effects of sample heterogeneity on sampling method (FE) was by far the greatest cause of uncertainty rather than analytical bias (AE). Precision bias was estimated at $\pm 83\%$ of the concentration value for the sampling method (due to effects of heterogeneity), but was much lower at $\pm 7.5\%$ for analytical method. The overall random component of uncertainty (FE) was estimated as being $\pm 83.6\%$. The majority of the overall variability was therefore related to the effects of intra sample heterogeneity on sampling (FE) method rather than analytical factors (AE).

From a practical point of view the principles of TOS would indicate that the standard approach to estimating AE by undertaking laboratory duplicate sampling is a process that unless done with care is at the mercy of FE effects.

To understand FE, heterogeneity must first be understood and defined.

3.0 HETEROGENEITY

Heterogeneity is the measure of the degree of compositional variability of a material and is a scalar concept (i.e. it is evident over micro- to macro-scales). For the purposes of a waste materials assessment, two broad scales can be considered over which heterogeneity will operate:

1. Inter-sample (macro-scale)
2. Intra-sample (micro-scale) heterogeneity.

Inter-sample heterogeneity: This defines large scale variability differences between sample locations, i.e. the variability in composition between 2 samples taken at different spatial locations m to 100m apart. For example, mineralisation may be concentrated within a particular material (e.g. in a vein of primary sulfide mineralisation), at a particular location (near a fault zone), or at a particular depth (possibly a layer of material in a waste dump). The degree of inter-sample heterogeneity will therefore have a potentially significant impact on the ability of the sampling programme to characterise the geochemistry of the material on site.

Intra-sample heterogeneity: This defines small scale heterogeneity of a material with respect to the differences in geochemical composition within a 2kg RC sample for example. At these smaller scales, various mineral phases may be present and unequally distributed (e.g. isolated macro pyrite crystals, variable surface crusts of secondary minerals), and various sources of material may be present (e.g. overburden, ore, varying grade of alteration minerals).

The importance of determining between inter- and intra-sample variability effects cannot be overstated when it comes to interpretation of sampling data for AMD assessments. Spatial trends for example are entirely dependent on the determination of inter-sample variability. However if the intra-sample variability has not been well established prior to analysing inter-sample variations, then the data points used for the spatial analysis and variographic analysis may be not be considered representative.

Because advances in variography are beyond the scope of this paper, inter-sample variability will not be considered further. It is intra-sample heterogeneity that will instead be the focus of discussion.

4.0 DETERMINATION OF HETEROGENEITY

Each type of material has specific characteristics in regard to the manner in which the mineral of interest is distributed in the host rock.

Two major characteristics of heterogeneity were identified by Gy (1982):

- Compositional heterogeneity (CH): difference in composition between individual fragments of a sample due to the inherent natural compositional variability. The greater the difference in composition between each fragment, the greater the compositional heterogeneity
- Distribution heterogeneity (DH): difference in average composition of the lot (sample) from one place to another. This is reflected in irregular distribution in grade in groups of fragments. Distribution heterogeneity can be influenced by particle density and size.

It should be noted that the term 'fragment' for the purpose of geochemical sampling assessments can include a single fragment (single gravel fragment) but it can also include groups of very small material fragments of the same weight. For example a 1kg lot split into

50 separate but equal 20g pulp samples is given to comprise 50 fragments of the lot for the purposes of assessment.

Factors that influence CH include size, shape, density, mineralogical composition of fragments within a sample. Because these factors are inherent to geological materials it is apparent that all samples assessed for AMD will be by nature heterogeneous and will have positive values of CH.

As a result of CH and DH, no matter how well a sample is extracted and prepared, it is likely that any analysis carried out will yield different results to the “true” value.

Detailed methods for estimating CH and DH are given in Pitard (1993) but these methods rely on some simple assumptions about the materials being assessed that cannot be generally applied to waste rock (although may be applied to in situ material prior to mining).

The assumptions include:

- The mineral phase being assessed has a defined density (i.e. restricted to one mineral phase for example pyrite)
- The mineral phase is present in gangue rocks that have a defined density (i.e. assumes for example pyrite present in monolithic quartz gangue)
- The liberation of the mineral phase of interest occurs at an optimum grain size that can be defined
- Shape of the mineral phase can be defined (flakes such as biotite for example compared to cubes of pyrite)

5.0 DETERMINATION OF CONSTITUTION HETEROGENEITY (CH_L)

For the purposes of determining the heterogeneity of a sample and to provide useful information about sampling procedure, it is the heterogeneity between (not within) fragments that we are interested in determining. The objective therefore is to characterise the heterogeneity of a lot (L) with respect to a critical component of a fragment (A_i). The total mass of the critical component (A) in each fragment (i) making up a lot (L) is defined by the fragment mass (M_i) multiplied by the critical content (a_i) in the relation:

$$A_i = a_i M_i \quad [1]$$

The constitution heterogeneity CH of a Lot L is defined as the relative and dimensionless variance of the heterogeneities (CH_L) with respect to the values for A_i carried by the number of fragments (N_F) making up the lot. This is shown in EQ2 where M_i is the fragment mass, M_L is the mass of the entire lot, a_i is the critical component (concentration) of a fragment and a_L is the average value (concentration) of the critical component in the entire lot (L):

$$CH_L = N_F \sum_i \frac{(a_i - a_L)^2}{a_L^2} \cdot \frac{M_i^2}{M_L^2} \quad [2]$$

The CH_L is difficult to calculate in completeness for a given lot because N_F can be very large (for example the number of 3mm gravel fragments in a 5kg crushed sample for example is very large and would require a prohibitive amount of data analysis). To calculate the characteristic of the material making up the lot independently from the size of the lot, CH_L can be multiplied by the average weight of a fragment. This results in determination of the constant factor of constitution heterogeneity IH_L .

$$IH_L = CH_L M_i \quad [3]$$

IH_L has the dimensions of a weight (g). IH_L can always be calculated regardless of the large number of fragments and is used in the calculation of FE.

6.0 DETERMINATION OF FE

The fundamental error variance identified by Gy (1982) is the smallest achievable residual average error, or a loss of precision inherent in the sample due to compositional variability (including particle size, density, chemical composition etc). FE is therefore directly related to CH and DH, and therefore IH_L . According to Gy's work FE is proportional to the cube of the nominal particle size and inversely proportional to the mass of the sample.

Knowing these parameters of FE steps to reduce FE can be deduced as such:

1. Increase the mass of the sample M_S
2. Reduce the particle size (by crushing for example)

Point 1 can be addressed at the sampling stage as larger samples to be taken in the field. Point 2 is addressed as part of sample preparation where particle size reduction can be carried out (i.e. crushing/grinding). On this basis reducing the particle size provides a practical means to reduce FE.

Estimation of the variance of the fundamental error σ_{FE}^2 is given by Gy as follows where M_S is the sample weight and M_L is the weight of the lot:

$$\sigma_{FE}^2 = \left(\frac{1}{M_S} - \frac{1}{M_L} \right) IH_L \quad [4]$$

If a lot has a very large weight compared to M_S then a maximum variance of FE can be calculated:

$$\sigma^2(FE)_{max} = \left(\frac{1}{M_S} - 0 \right) IH_L \quad [5]$$

If the sample M_S has come from a splitting operation (e.g. riffle splitter) then a minimum value for the variance of FE can be calculated:

$$\sigma^2(FE)_{min} = \left(\frac{1}{M_S} - \frac{1}{2M_S} \right) IH_L \quad [6]$$

Using these equations therefore we can determine the appropriateness of the sampling program and give meaningful advice about optimising the sample size.

7.0 SAMPLING APPROACH

The approach to sampling and analysis of heterogeneous materials then becomes very important when FE is considered. Sample sizes for standard AMD tests are generally small. Typically, for ICP analysis a laboratory will extract a small 1-10g sub sample of a parent sample on which to complete analysis. For a NAG test approximately 2g of material is analysed. For a TCLP leach test 100g of material is analysed. For paste pH 10g of material is analysed.

In addition the selection of the sub sample may take place after sieving or crushing of the parent sample which may introduce preparation biases. This is an important consideration as different tests require different particle sizes of material.

Therefore if FE is significant then clearly it is unlikely that a single 1-10g sub sample of the parent will be representative (chemically or mineralogically) of the sample as a whole.

To illustrate this point an estimation of the sample size required for a toxicity characteristic leaching procedure (TCLP) test is given in Pitard 1993. For a waste sample with maximum particle size of 9.5mm using 100g for the leach test results in a maximum standard deviation

of fundamental error FE of 46%. To be able to reduce this error to $\pm 5\%$ a sample size of 8,574g is required; clearly much larger sample than laboratories can routinely cater for.

There is significant potential for FE to be very large for standard AMD testing given commercial labs use such small sample volumes. Preparation methods may aid reduction in FE, however given the best way to reduce FE is to reduce particle grain size, this is not possible for leach tests which are carried out on material of specific grain size fractions due to increased mobilisation of mineral phases at smaller grain sizes.

8.0 A TYPICAL HETEROGENEITY PROBLEM FOR WASTE ROCK SAMPLING

Figure 1 shows the heterogeneity problem faced when trying to determine how to sample waste material. This 90cm long core from drilling of a waste rock dump weighs approximately 10kg and contains at least 4 material types (based on colour alone). Some common questions would be:

- Should the whole core be retained or sub samples taken in the field?
- Is a single composite adequate?
- If a composite is adequate how large should the composite be?
- Should this core be sub sampled into 4 separate samples?
- Should screening of particle sizes be carried out?
- Should the whole sample be crushed?



Fig. 1. Typical drill core from waste rock dump investigation

Although at first glance the simple answer is that the best method would be to sub sample the entire fraction of each of the 4 main material types, however:

- This may not be feasible due to project costs of completing extremely detailed sampling in this manner.
- It may be that with good laboratory preparation a composite sample may provide adequate results for parameters such as total sulfur
- It may be that the colour does not have strong correlation with the mineral phase of interest (for example sulfide content)

In addition various sampling methods require different sampling preparation.

- Total sulfur analysis requires pulverised material for example ($<100\mu\text{m}$)
- Leach column may use $<6\text{mm}$ material
- Paste pH may use $<2.3\text{mm}$ material

There are therefore increasing amounts of processing work the more the sample is subsampled, and sample volumes are becoming increasingly small (which as we have established can increase FE)

To best tackle this problem one should determine:

- The heterogeneity of the material with respect to the mineral species that we are studying (CHL)
- The fundamental sampling error (FE) for the proposed sampling and preparation method

A simplified heterogeneity analysis tool has been developed for this task to aid with answering these questions that arise nearly every time waste rock characterisation projects are started.

9.0 SIMPLIFIED HETEROGENEITY ANALYSIS TOOL

A simplified tool has been developed to determine the effects of heterogeneity, to produce heterogeneity profiles for AMD characterisation assessments.

To allow for the large amounts of analysis suggested, a benchtop XRF unit is suggested as a cost effective analysis tool. As has been outlined in this paper it is the FE not the AE that will have the largest impact on producing correct results; therefore the use of analysis tools with lower than laboratory-grade precision can be statistically justified.

A suggested procedure is outlined as follows for the heterogeneity assessment and sampling procedure assessment for a project where a reasonable number (>20 samples) of large (>5kg) samples of waste rock material are collected (Table 1).

It is suggested that at least 5-10 sample lots be subjected to heterogeneity analysis. Sample lots should be selected that are broadly representative of the types of materials gathered from site. This can be based on colour, rock type, density etc.

Table 1. Sampling process for each lot subjected to heterogeneity study

Sample selection	Sample processing	Analysis	Purpose
Sample set 1: 10 small grab samples from the lot, approx. 20-200 g per sample	<i>Crush sample, and pulverise entire sample</i>		Assesses heterogeneity profile for grab samples
Sample set 2: 30-50% material crushed to <6mm	<i>Using splitter, break entire sample down into a large number of sub samples representative of the size of typical laboratory sub samples (50-100g each), pulverise individual samples in full</i>	<i>FPXRF</i>	Estimate FE for sampling method and heterogeneity profile for crushed material
Sample set 3:	<i>Crush each fraction to <6mm. Use splitter to break</i>		

30-50% of material screened through as many sieve sizes as possible, minimum of 3 (2.3mm, 6.7mm, 13.2mm)	<i>each sieved fraction into a large number of sub samples (50-100g each), pulverise individual samples</i>	Assess influence of particle size distribution on heterogeneity profile
Sample set 4: Select at least one of the pulp samples from set 2 and 3	<i>Split pulp sample into a large number of smaller samples approx 5g in weight</i>	Assess effectiveness of pulverising. Use samples obtained to assess AE

10.0 CASE STUDY

A case study is presented based on the following analysis work:

- 2 Drill core samples (approx 90cm long, 8kg in weight) from AMD waste rock dump characterisation investigation were subject to detailed analysis
 - Core A contains material that is visually heterogeneous, and contained visual evidence of sulfides/sulfates
 - Core B contains material that is visually homogeneous and does not have evidence of sulfides/sulfates
- Approximately 3kg of material from each core was run through 15 stage sieve sorting with standard sieve sizes ranging from 0.075mm to 26mm. Each fraction was then crushed in a rock crusher to <6mm, and 50g (where enough material was present) sub sample was obtained using a riffle splitter. The 50g sample was then pulverised.
- Approximately 1.5kg material from each core was crushed to <6mm in a rock crusher. The crushed material was then entirely split using a rifle splitter into approximately 65 samples of 20g each. Each of these samples was pulverised.
- 10 grab samples (25g) were taken from each core before processing occurred at 10cm intervals.
- One pulp sample from Core A and Core B was subject to micro-splitting to approximately seven, 6g sub samples.

The aim of the study was to allow detailed heterogeneity assessment of the two samples of waste materials as part of an assessment of the suitability of the current sampling and analysis program.

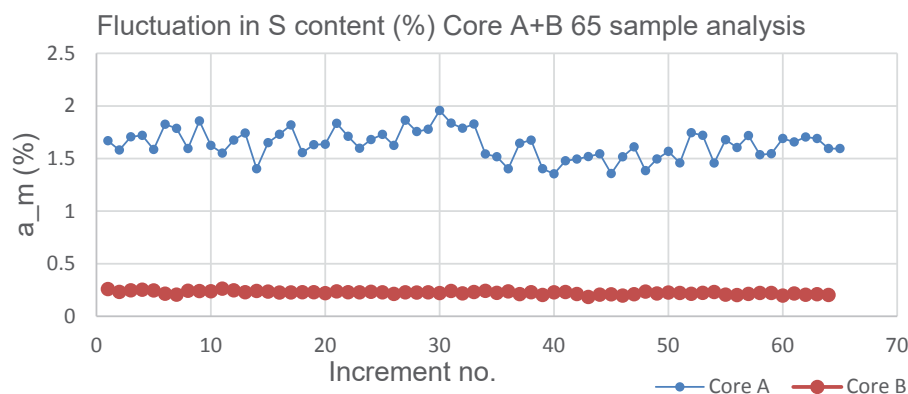


Fig. 2. Sulfur content (a_m %) in 65 sub samples (20g each) from complete analysis of 1.5kg of crushed material from Core A and Core B.

Figure 2 shows the results of the 65 sample analysis of the <6mm crushed material samples of Core A and B. It is clear that Core A has higher sulfur content and appears to have higher variability with respect to intra-sample heterogeneity given the range of fluctuations in S content appears higher. In general the results indicate that the process of riffle splitting appears to produce samples that are reasonably consistent for both cores.

Figure 3 shows the same data plotted as a heterogeneity chart (where h_m is the dimensionless value for variance in heterogeneity which is equal to the variance of the sample critical content from the weighted average critical content for the lot). It is clear that although Figure 2 indicates Core A to have higher intra-sample variability, when the concentration data is plotted on a relative scale the variance in heterogeneity is actually very similar between the two cores.

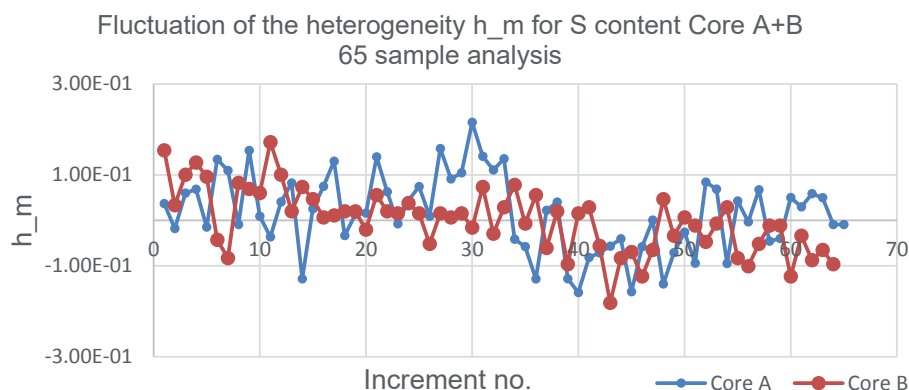


Fig. 3. Heterogeneity profile for 65 sample analysis from Core A and Core B

Table 2. 65 sample heterogeneity analysis Core A+B

	Core A	Core B
M_L (g)	1300	1280
A_L (mg)	2092	286
a_L %	1.609597	0.223578
M_m(avg) (g)	20	20
CH_L	0.007258	0.004752
IH_L (g)	0.145165	0.095047
$\sigma^2(FE)_{min}$	+8.6%	+6.8%

Table 2 shows that

- As expected the calculated values for weighted average concentration (a_L) of S is much higher for Core A.
- The value A_L is given because this indicates the total amount of S in the sample which can be more useful than a concentration to calculate acid loading.
- M_m is the average weight of an individual sample.
- Values for CH_L and IH_L are quoted, although this is not the CH_L of the raw material as samples have been processed, the value is useful as a metric to gauge the relative heterogeneity of the sub samples obtained. Core A can be seen to have a higher intra-sample heterogeneity given CH_L is higher than Core A.
- The minimum variance of fundamental error can be seen to be close between the 2 cores and less than 10% in both instances indicating that taking a 20g sub sample by riffle splitting crushed material (<6mm) for S analysis would produce an adequate result for characterizing S concentration in either material lot.

The minimum variance of the fundamental error related to obtaining a 20g sample from the whole lot was calculated using Eqn. [6], the working for Core A is shown below:

$$\sigma^2(FE)_{min} = \left(\frac{1}{20} - \frac{1}{1300} \right) 0.15 = 0.007 \quad [7]$$

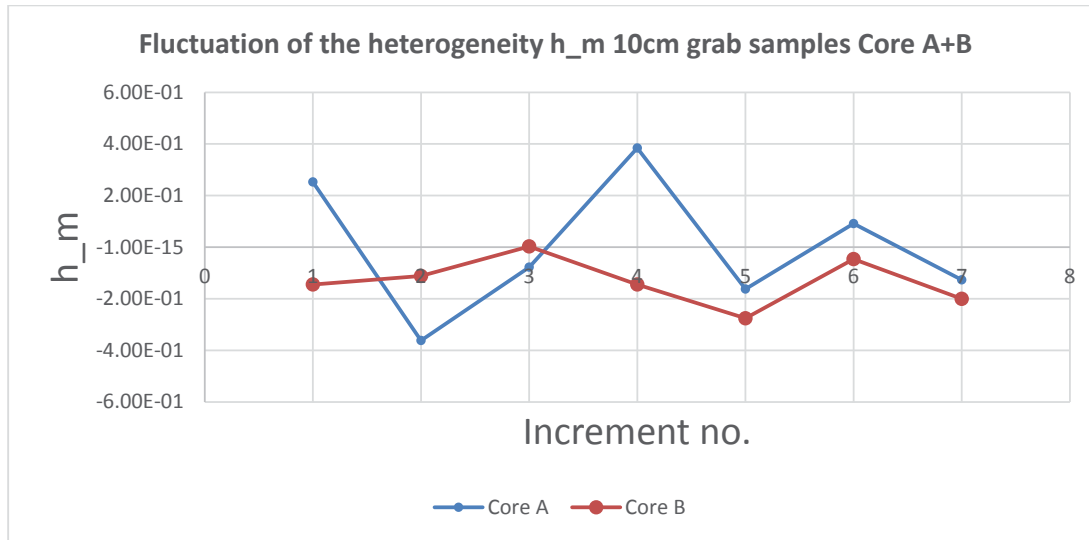


Fig. 4. Heterogeneity profile for 10cm grab sample analysis from Core A and Core B

Figure 4 shows the heterogeneity profile for the 25g grab samples taken at 10cm intervals along each core. It is clear that samples from Core A have a higher degree of heterogeneity than Core B.

Table 3. 10cm grab sample analysis

	Core A	Core B
M_L	175	175
A_L	292.4753	46.375
a_L %	1.671287	0.265
M_m(avg) (g)	25	21.875
CH_L	0.056998	0.011164
IH_L (g)	1.424945	0.244215
$\sigma (FE)_{min}$	24%	9.8%

Table 3 shows that:

- The values for a_L are close for both cores relative to the values given in Table 2 from the 65 sample analysis, indicating that (perhaps surprisingly) taking 9 grab samples produces a reasonably accurate estimation of the mean S grade.
- The value for CH_L in this instance reflects spatial variation I heterogeneity. S content within Core A has a high degree of variability with depth (related to layering of materials with different S content)
- The minimum variance of the fundamental error is significantly higher in Core A at 24% than Core B 9.8% which is indicated from Figure 4.
- The minimum variance of the fundamental error for Core A is much higher than that given in Table 2 which indicates that taking small numbers of grab samples is likely to

be a poor sampling technique for this material. The value for Core B is lower at 9.8% but this is still significantly higher than that given in Table 2 so that, although the material has a lower degree of heterogeneity, grab sampling is still not recommended as good sampling technique.

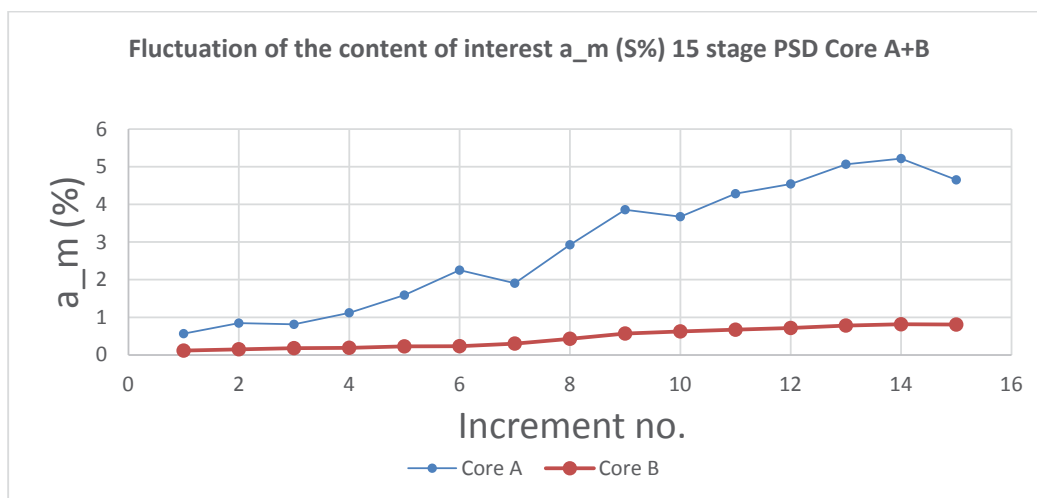


Fig. 5. Concentration profile for S% in 15 stage PSD analysis from Core A and Core B

Figure 5 shows the fluctuation of S concentration relative to particle size class. Increment 1 is the largest (26mm) particle size and 15 is the smallest (0.075mm). A very strong relationship is noted between particle size and S concentration with concentrations being more than 5 times higher at smaller particle sizes. This is an important observation as it shows that the materials have strong partitioning effects with respect to S and particle sizing (analysis was completed on pulverized samples which negates the influence particle size can have on XRF analysis).

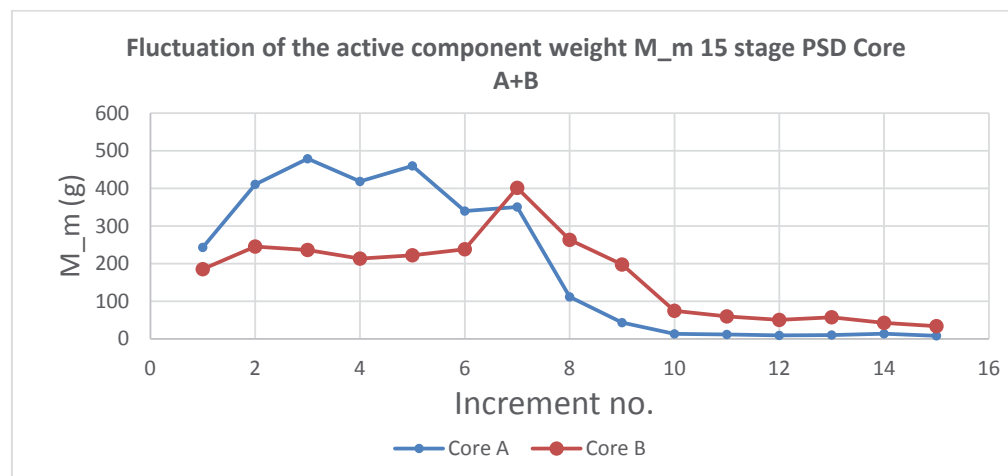


Fig. 6. Sample weight profile for 15 stage PSD analysis from Core A and Core B

Figure 6 shows the weights of each of the particle size classes. It is clear that, as with S concentration, there is strong influence of particle size with smaller particles having much smaller mass contributions in both cores.

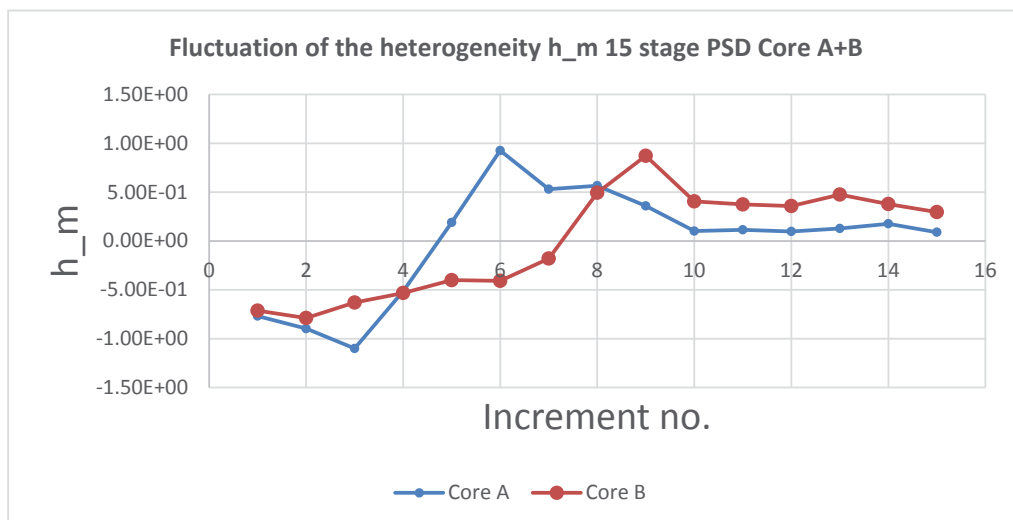


Fig. 7. Heterogeneity profile for S% in 15 stage PSD analysis from Core A and Core B

Figure 7 shows the heterogeneity profile for both cores with respect to the 15 stage PSD. Both cores have a similar profile. It should be noted that the values for h_m are significantly higher for this graph than others presented indicating heterogeneity is very high with respect to S concentration and particle sizing. It is interesting to note that Figure 5 appeared to show Core A had the higher degree of heterogeneity but re-scaled both cores appear to show similar levels of heterogeneity related to particle size.

Table 4. 15 stage PSD fraction analysis Core A+B

	Core A	Core B
M_L	2921.6	2521.4
A_L	4298.311	823.3702
a_L %	1.471218	0.326553
M_m(avg)		
(g)	194.7733	168.0933
CH_L	0.305581	0.270901
IH_L (g)	59.51895	45.53662
$\sigma (FE)_{min}$	117%	89.4%

Values for the minimum variance of the fundamental error were calculated related to obtaining a 50g sample (which is the smallest sub sample that can be obtained selecting material at the largest size fraction).

Table 4 shows that:

- A_L values are close to those given in table 2, this is not surprising as it may seem though as although heterogeneity is high in this experiment, the average S content of the lot can be expected to be consistent given the weighted method of calculating S content using A_L and M_L.
- CH_L and IH_L values are significantly higher for both cores confirming the extreme heterogeneity distribution of S within particle size class
- Values for the minimum variance of the fundamental error are similar between the two cores indicating the partitioning of S is similar between the two materials. Values of over 80% for Core B and over 100% for Core A indicate that selectively sampling a particle size fraction for S analysis will have extreme influence on the representativeness of the results.

Table 5 shows the distribution of S as weighted percentages within the size fractions. The proportion of the S content derived from the <2.36mm size classes is 27% of the total S content derived from only 17.3% of the sample weight. The top size fractions 13.2-26mm in comparison contribute only 20% of S for 39% of the sample weight.

Table 5. 15 stage PSD fraction analysis Core A, S distribution

PSD fraction	S% of total weight	Sample weight fraction % total
26	3.197507	8.320783
19	8.075447	14.05737
13.2	9.0513	16.38828
9.5	10.89784	14.32092
6.7	17.00343	15.73795
4.75	17.79972	11.62719
2.36	15.53777	12.00027
1.18	7.596266	3.819825
0.6	3.868454	1.475219
0.425	1.127871	0.451807
0.3	1.165889	0.400465
0.212	0.961663	0.311473
0.15	1.202046	0.349124
0.075	1.638368	0.462076
-0.075	0.876426	0.277245

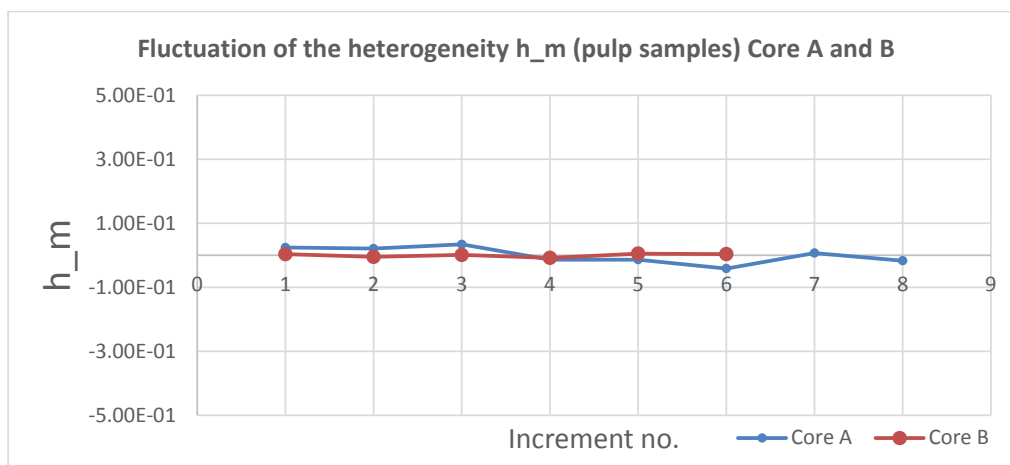


Figure 8: Heterogeneity profile for S% in pulp sample analysis from Core A and Core B

Table 6. Pulp analysis Core A+B

	Core B	Core A
M_L	56	36
A_L	20.32073	51.17106
a_L %	0.36287	1.421418
M_m(avg)		
(g)	7	6
CH_L	0.000575	2.41E-05
IH_L (g)	0.004023	0.000145
$\sigma (FE)_{min}$	0.08%	0.08%

Figure 8 shows the heterogeneity profile for the micro-analysis of pulp samples. For both cores heterogeneity is very low which is expected given that pulverizing samples is known to significantly reduce $\sigma (FE)_{min}$ as a result of size reduction. From table 6 the variance of the fundamental error is very low for both cores and indicates:

- The pulverizing step carried out introduces a very small amount of error
- The XRF analysis must have a high level of precision to be able to produce this extremely low value for $\sigma (FE)_{min}$

11.0 CONCLUSIONS

The results presented in this paper are a small fraction of the data that OKC has gathered using this profiling tool in conjunction with utilization of bench top XRF analysis.

From the research carried out, heterogeneity has been demonstrated to be a significant factor controlling the fundamental sampling error, and thus the ability to produce analytical results that are representative when dealing with waste rock materials. A profiling tool has been presented which allows heterogeneity to be studied both in the context of the impact on the fundamental error during the sampling process, and with respect to the properties of the material. General findings from utilization of the profiling tool include:

- Heterogeneity is shown to be very high with respect to 'grab' sampling from different depths of core material and from different particle sizes. Extreme care is therefore required when interpreting the results of either:
 - Grab sampling of small samples
 - Samples that have been screened for particle size fractions
- Crushing samples of around 1-3kg of material to a grain size of <6mm and using riffle splitting to take small sub samples (20g) for analysis appears to result in acceptable levels of fundamental sampling error. However it is recommended for any sampling program that as large a sub sample as possible be taken as the larger the sample the lower the fundamental error will become.
- The lowest value for the variance of the fundamental sampling error for any of the sampling techniques has been found to be around 6.8% for crushing to <6mm and riffle splitting, this is a minimum so the actual errors are likely to be higher.
- Selecting a 50g sample based on particle size has been shown to have the highest value for the minimum fundamental sampling error of 117%.
- To achieve the high volumes of analysis required for these types of intensive analysis studies, low cost analysis techniques are required. Benchtop XRF has been demonstrated to be a very useful tool for this purpose.
- As is commonly known, pulverizing material produces samples with extremely low levels of heterogeneity and therefore the larger the sample that is pulverized the lower the fundamental error will become.
- Selecting very small samples (<10g) of material that has not been pulverized for analysis, and has been size selected as is the case for commercial analysis like paste pH (which is done on <2.3mm fraction), is very likely to introduce very large fundamental sampling errors

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MINERALOGY OF MINE SITE NEUTRALISING MATERIALS: A MISSING LINK IN AMD CONTROL PLANNING

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ABSTRACT

*Most mine sites have non-value minerals capable of providing some neutralisation of acid mine (rock or tailings) drainage. They are often not considered or even surveyed in primary site ore assessment for mining value. Where they are considered, standard assessment methods define only total potential acidity or alkalinity of these materials, e.g. ANC/MPA ratio. This does not consider the **rates** at which acid generating and neutralising reactions may take place, although it is these relative rates in disposal of rock and tailings wastes that determine whether acid and metalliferous drainage occurs. Methods are now available to assess the rates at which neutralisation can be supplied from reactive silicate minerals (additional to carbonates) in on-site waste rock types. Knowledge of both the amount and the rates of acid generation and neutralisation can be used to assess future acid rock drainage liabilities but, more importantly, to plan greenfield or operating dumping to make maximum use of these on-site materials. The complete definition of geochemistry and mineralogy of site materials can provide more effective and reduced-cost management of these mining wastes. Examples where neutralising waste rocks have been identified and assessed at BHP Billiton Mt. Whaleback and Hidden Valley (PNG) are discussed. At Savage River (Tasmania), a chlorite calcite schist waste rock has been used successfully to alleviate serious AMD.*

1.0 INTRODUCTION

1.1 Sustainable Management of AMD for Closure

In principle, a sustainable approach to acid mine drainage (AMD) management in pre-planning or in operation should be to use, where possible, resources available at the mine site to reduce the acid generation rate (AGR) sufficiently that the acid neutralisation rate (ANR) can match the AGR in rock and tailings disposal before final rehabilitation. This matching of acid and neutralisation rates in kinetically controlled processes is, in principle, the only sustainable option for long-term closure. This has not yet been fully designed or achieved, but this paper will describe this approach with case studies illustrating some parts of the overall strategy that have been implemented to date. There are real opportunities for improved practice using full geochemical and mineralogical assessment of site materials.

1.2 Geochemical Matching of AGR and ANR

AMD management involves strategies, at any level from molecular to site storage scales, to minimise the interaction of reactive sulfide surfaces with air and/or water. At the site engineering scale, the selection of minimisation and control strategies will be influenced by a range of factors including climate, topography, hydrology, mine geology, geochemistry and mineralogy of waste rock, tailings and available neutralising materials. A critical requirement remains, however, to understand, measure and control relative **rates** of acid generation and neutralisation, with the goal of matching them in long-term management. In the standard procedures (e.g. AMIRA/EGi, 2002, GARD, 2012), kinetic tests of AGR and ANR are normally limited to kinetic leach columns where the AGR and ANR are combined, and the tests can take years to yield reliable information on likely reaction evolution of waste dumps and tailings. There remains a need for separate less time consuming estimation of AGR and ANR, not only in the acid waste material, but also in identifying suitable neutralising mineralogy on site that may be available from materials not currently included in mined and processed wastes. This has been partly addressed in current research (e.g., Smart et al. 2006; Schumann et al. 2008), but much remains to be done for this to be used effectively in mine planning and operation.

The principle of reducing AGR by addition of alkalinity generating cover materials to passivate pyrite was used at the legacy Brukunga mine site (Taylor et al. 2009). These authors also noted that this AGR reduction may be the sustainable approach to long-term AMD control. This trial used externally-introduced rapidly-dissolving enhanced caustic magnesia in the soil capping material which was successful in controlling AMD for the first year but was less successful after this in the absence of continuing alkalinity release to match the reduced AGR, since no carbonates or reactive silicates were available at this small site. Where they are available in site rocks, they have the potential to provide sufficient on-going ANR to match or exceed the reduced AGR.

The importance of carbonate minerals, particularly non-iron carbonates, in neutralising acidity generated through sulfide oxidation has long been understood, but the role that silicate minerals can play in providing significant acid buffering has only been recognised more recently (MEND 2009). At the same time, and given the abundance of silicate minerals in mine wastes, the widespread occurrence of AMD shows that at some sites, silicates in the absence of carbonates do not provide neutralising capacity at a sufficiently fast rate to neutralise acidity resulting from sulfide oxidation. In other cases, they have (e.g. Jambor et al. 1999, Miller et al. 2010). Hence, two additional requirements are needed to match AGR and ANR namely: the AGR (sulfide oxidation) has to be reduced; and the ANR, both carbonate and non-carbonate (mainly silicate) measured to ensure that the rates can be matched.

Can this matching be done with site silicate minerals alone? Blodau (2006) reviewed AMD from the view point of rates of acid generation and neutralisation. He noted that at pH 5 and 25°C, pyrite oxidation was three orders of magnitude slower than carbonate (calcite and dolomite) dissolution. Hence pyrite oxidation in the presence of excess carbonates results in neutral pH pore waters. Dissolution rates of silicates are typically two to three orders of magnitude slower than pyrite oxidation. However, even if the available neutralising capacity of the silicates can match the amount of acid generated by the pyrite, the pH of the resulting waters can often be in the range of 3–6 because of hydrolysis of dissolved metals (e.g., Al, Fe) which control the final pH of the solution. Silicates can offer significant buffering in this pH range as demonstrated by Mattson (2009), who showed that in a granodiorite from which carbonates had been previously removed, hornblende and epidote buffered the leachate to between pH 4.5 and 6.

In the longer-term evolution of a dump or tailings, the pyrite content is reduced sufficiently for matching of AGR and ANR to occur naturally. Miller et al. (2010), examining the leachate chemistry of a number of long-term (>10 years) kinetic leach column tests, in which carbonate content was exhausted early in the tests, found that the silicate ANR appeared to match the AGR when the latter decreased to around 10 to 15 mg CaCO₃/kg/week. When the ANR exceeded the AGR, leachate pH was always above 3 and generally above 4. Based on these results they concluded that silicates are likely to buffer in the region of pH 4.5 when ANR is greater than or equal to the AGR measured in the waste effluent. While they concede that pH 4.5 may still be too low for environmental regulations, acidity loads, both proton and iron particularly, will be significantly reduced. They therefore concluded that if the AGR can be controlled to below 10 to 15 mg CaCO₃/kg/week, then silicates present in wastes are likely to offer long-term mitigation to acidity generated in the waste. This is promising if the AGR can be sufficiently reduced but different silicate minerals have very different neutralisation rates (Palandri and Kharaka, 2004), so that a method of measurement of the ANR of a waste material is essential to this strategy. The current stage of this development is discussed below, but first strategies for reduction of AGR are considered.

As an example of AGR control, Miller et al. (2010) also examined the design of the degree of saturation and thickness of covers to inhibit oxygen diffusion and reduce AGR to around 1 mg CaCO₃/kg/week as a conservative design target. They showed that by manipulation of the cover thickness at site saturation levels that can be maintained, oxygen diffusion can be limited to reduce the AGR to the design target of 1 mg CaCO₃/kg/week at which silicate neutralisation rates are likely to control seepage pH to 4 or above. Given appropriate climatic conditions and waste emplacement construction techniques, limiting oxygen diffusion to achieve the necessary reduction in AGR has been realised in practice (Miller et al. 2012).

The addition of carbonates, where available on site, in covers and layers, is recognised as the preferred method of AMD control in acid rock dumps (GARD Guide). While carbonates, such as limestone and dolomite, are often thought of as simply neutralising acid that has already been generated through sulfide oxidation, both laboratory (Schumann et al. 2009; Huminicki and Rimstidt 2009; Nicholson et al. 1990) and field (Li et al., 2012; Li et al., 2011; Miller et al., 2009) studies have shown that under neutral pH conditions with added carbonates, pyrite oxidation rates can be reduced by one to two orders of magnitude. The mechanism for the reduction in sulfide oxidation rate is found to be through formation of continuous, coherent iron oxyhydroxide coatings which develop on the surface of the reacting sulfide and reduce oxygen diffusion rates to the surface (Huminicki and Rimstidt 2009; Nicholson et al. 1990). The nature of these surface passivating layers has been extensively investigated (Smart et al. 2010; Schumann et al. 2009). Under circum-neutral pH conditions, multi-layered coatings of iron oxyhydroxide develop which incorporate silicate, sulfate and carbonate. These thin, continuous and conformal coatings are amorphous and often overlaid with a thicker (1-2 µm) coating of (semi)-crystalline goethite-like material. Where clay minerals are present in the waste, strongly adhering particles with micaceous texture can form a further armouring layer (Miller et al. 2009).

Recent studies have investigated the role pH and dissolved silicate play in stabilising these passivating layers (Zeng et al. 2013; Fig. 1). Results indicate that pyrite oxidation rates are reduced most significantly when the pH is maintained above 6 and when the surrounding solution contains dissolved silicate (10–20 mg/L as Si). It appears that in the presence of dissolved silicate, the iron oxyhydroxide layer formed during pyrite oxidation retains its amorphous structure, while in the absence of silicate, conversion of this layer to the more-crystalline and less-passivating goethite-like material occurs. If initial alkaline amendments can be made so that these passivating layers can be formed and maintained, then for wastes with low to moderate AGR containing some reactive silicates, acid generation rates may be

sufficiently reduced (more than 90%) to be matched by silicate ANR in the long-term. Under these conditions further amendment with carbonates may not be required. This sulfide passivation, with the use of reactive silicates from site rocks, forms the basis of the geochemical strategy for sustainable AMD management.

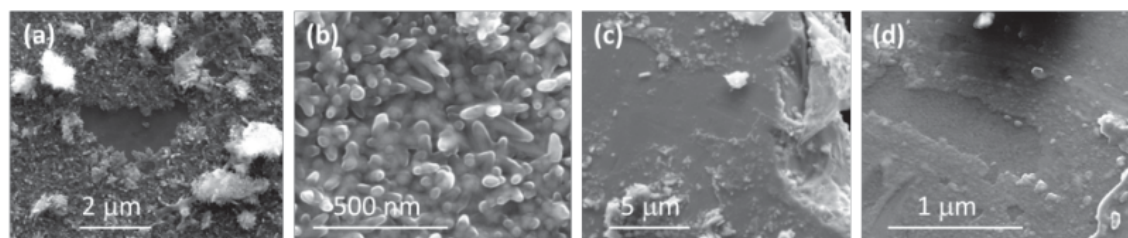


Fig. 1. Evidence of the effect of the presence of silicate on pyritic surface layers at circumneutral pH. (a) and (b) pyrite surfaces immersed in calcite saturated water for 160 days showing clear crystalline overgrowths; (c) and (d) pyrite immersed under the same conditions with added silicate (20 mg dm^{-3}) showing clear difference in surface layer morphology and significantly reduced pyrite leaching (Zeng et al., 2013).

1.3 Assessment of Silicate Neutralisation and Sulfide Oxidation Rates

Both silicate acid neutralisation rates (ANR) and sulfide oxidation rates can be estimated from kinetic tests such as humidity cell tests or kinetic leach column tests; however, these tests can be very time-consuming (lasting from months to years) leading to obvious time and resource constraints for application of this approach. In addition, it can be difficult to separate out the neutralisation of carbonates and silicates when both types of neutralising materials are present, while precipitation reactions can also infer lower oxidation and neutralisation rates than actually occur *in situ* during weathering.

Several studies have attempted to determine silicate ANR at time scales significantly shorter than the traditional kinetic methods. Most of these are based around combining mineralogical assessment and dissolution rate data of individual minerals. Eary and Williamson (2006) combined mineral dissolution rate data from Palandri and Kharaka (2004) with solution speciation modelling to examine dissolution of a number of theoretical rocks, with silicate mineralogy ranging from silicic through to mafic and containing 0–3 wt.% pyrite. Their results suggested that anorthite feldspar and some mafic minerals such as forsterite, augite and hornblende may dissolve quickly enough to neutralise acidity from pyrite oxidation, but felsic minerals such as K-feldspar and albite will not provide sufficiently fast neutralisation to avoid low pH as a result of pyrite oxidation. A similar mineralogical approach was used by Miller et al. (2010), wherein silicate ANR derived from mineral dissolution rates were compared with rates calculated from leachate analysis of solutions obtained from long-term kinetic leach column tests in which any carbonates had been previously exhausted. They found good agreement between the rates derived from mineralogy and those obtained from leach column tests when the pH was below 3, but at greater pH (>4), ANR calculated from mineralogy generally exceeded those calculated from leach column testing.

In a similar approach, Morin and Hutt (2011) compared silicate ANR derived from mineralogy with that obtained from humidity cell tests. They found that the rates calculated from mineralogy were similar to, or substantially less than, the measured humidity cell values. There are several possible reasons for this including unrecognised amorphous material, but it is important to consider that the measured ANR will initially be provided by relatively fast

ion exchange. This mechanism will eventually slow to match the pH-controlled dissolution rates of the bulk silicate mineralogy. For this reason, for instance, biotite is an effective neutralising mineral in ion (Mg) exchange but a slow ANR provider in the longer term. Hence, ANR values from fine reactive silicates within the dump at or very near the reacting pyrite surfaces are likely to be more than the estimates from rates based on dissolution rates of the mineralogy alone.

Ciccarelli et al. (2009) added the kinetic dissolution cell method to the assessment suite for silicate ANR. Samples containing carbonate must first be treated to remove these minerals, so that the measured neutralisation originates only from silicates. They compared the silicate ANR values obtained from the kinetic dissolution cell method with those calculated from mineralogy and kinetic leach column tests. Their data show the initial effect on pH and ANR of ion exchange followed by the steady long-term dissolution rate of the minerals. At low pH (2), there was close agreement between the three methods, but in other cases, where the measured rates were smaller because the pH was higher (>4), agreement between the methods was not as close. Recent investigations (Ciccarelli, 2012) have shown that the silicate ANR measured by the kinetic flow cell method is flow rate-dependent, and this constraint must also be taken into consideration when comparing rates determined using different approaches. Both the kinetic flow cell method and the mineralogical estimates provide ANR values that, with consideration of errors, can be used to compare with AGR for waste rock dump planning. Refinement of the methods and site testing is still a requirement for this part of the matching approach and sustainable AMD management.

As for rates of silicate neutralisation, acid generation rates (AGR) from sulfide oxidation can be determined from kinetic methods such as the humidity cell and kinetic leach column tests. However, as with the determination of ANR, the assessment of AGR using these methods requires extended time and can be significantly influenced by precipitation of secondary sulfate minerals. A laboratory-based method, the sulfide oxidation cell (SOC) or oxygen consumption method (OCM), which can be performed in both a relatively short time and at low cost, involves the measurement of oxygen depletion within a sealed chamber in which the waste rock sample has been placed. The mass of oxygen consumed over a defined time period allows the rate of sulfide oxidation to be calculated based on the stoichiometry of the reaction between the particular sulfide and oxygen.

Hollings et al. (2001) used the OCM to evaluate the pyrite oxidation rate in waste rock samples containing low sulfur (<0.5 wt.%) content. They compared the results with AGR determined from sulfate leached from the sample immediately following the OCM analysis and with rates calculated from sulfate measured during humidity cell tests. There was good agreement between the AGR calculated from the oxygen consumption and sulfate leached from the OCM sample, suggesting that oxygen consumption is a good indicator of pyrite oxidation rates. The more recent OxCon method (Schmieder et al., 2012) is a variation on the OCM with humidity and CO₂ generation and absorption considerations.

In another study, Sracek et al. (2006) measured pyrite oxidation rates in a mine waste rock dump based on oxygen and temperature profiles measured *in situ* and from pyrite content profiles measured in drill core samples. These results were compared with oxygen consumption measurements made in the laboratory. They found good agreement between all three measurements when the oxygen consumption tests were conducted on weathered samples. The authors concluded that small-scale test methods such as OCM may provide oxidation rates comparable to field measurements in regions where oxygen transport is not limited, *i.e.* close to the dump slope, or where gas transport is dominated by convection. They noted that other laboratory-based measurements, such as humidity cell tests or meso-scale column tests, generally overestimate sulfide oxidation rates in comparison to field-

based measurements. Again, estimates provide AGR values that, with consideration of error ranges, can be used to compare with ANR for waste rock dump planning and refinement of the methods with site testing required for sustainable AMD management.

2.0 IMPLEMENTATION: CAN IT WORK?; WHAT IS NEEDED?

A specific example of the use of on-site alkaline materials in amelioration of AMD from previous rock dumps was provided by Cook et al. (2008). Re-mining of the Buffalo Creek coal seam site (Virginia, USA) to obtain coal from lower in the geologic column exposed new overburden, including an alkaline sandstone, that was used to add covers to the existing valley-fill AMD rock dumps. Drainage areas from the fills started with high acidity/high metal drainage (pH 3.5–4.5 with 100–200 mg/L CaCO_3 acidity) in the sub-watersheds. Addition of the sandstone covers improved water in the main stream of each drainage area from the fills. The water quality in the tributaries of Laurel Creek has been dramatically improved to pH 6.2 and acidities of <1 mg/L. Placement of alkaline material, recognised in the new mineralogy, into the extended valley fills eliminated the need for chemical treatment. With more than 10 years of data, it is apparent that there is high probability for the long-term success of the amendments.

The successful implementation and testing of on-site amendments to AMD from the historic B-Dump in the Savage River Rehabilitation Program (SRRP), a cooperative arrangement between the Tasmanian Government and current operators Grange Resources, has been fully documented (Hughes et al., 2009, Hutchison et al. 2009, Li et al., 2011, Li et al., 2012). This used a water-shedding compacted top cover (from graded material) with over-dumped side covers and a base-dumped flow-through barrier comprised of chlorite-calcite schist waste material from the site. The ANR of this schist is near 2000 mg CaCO_3 /kg/week at pH 2, near 500 mg CaCO_3 /kg/week at pH 3.5 and near 90 mg CaCO_3 /kg/week at pH 5. The measured AGR from the B dump seep is now near 7 mg CaCO_3 /kg/week and from the flow-through near 0.2 mg CaCO_3 /kg/week. The covers have effectively reduced the sulfide oxidation rate within the dump by about 43% over the past five years. Alkalinity from the covers has been migrating down into the acid-forming waste and forming passivating layers of silicate-stabilised iron oxy-hydroxide on pyrite grains (Li et al. 2011). Sulfate, Al and Cu flux in Main Creek have been decreased by 50% after B-dump was capped. The discharge from Main Creek to the river has neutral pH, low sulfate (218 mg/L) and metal concentrations, e.g. Al 200 $\mu\text{g/L}$ and Cu 30 $\mu\text{g/L}$, that are below the SRRP fish target allowing biota 20 km back up the river (Hughes et al. 2009).

The significance of this work is that it resulted from a comprehensive report (Thornett, 1999) on the rock types and mineralogy of the site for the previous owners Australian Bulk Minerals. This identified a large variety of potentially neutralising mineralogy including the abundant chlorite-calcite schists with some dolomite, magnesite, talc-carbonate schist, dolerites and reactive silicates in mafic rocks, amphibole-chlorite-albite schists, serpentinites and possibly tonalite. This was recognised by site personnel and Rumble (2005) in planning of the amendments and provided the basis for the successful remediation.

Hence, there are two requirements that can potentially improve current practice in AMD control. The first is the need for detailed mineralogy, additional to standard AMD classification testing, of all site lithology, preferably in greenfield assessment but also in operating and legacy sites. This will allow a first-level estimate of ANR for different lithologies based on the mineralogy alone (Ciccarelli et al., 2009, Miller et al. 2010). The AGR from the waste rock in passivated form at pH 6 can also be estimated from the OCM and sulfate release as discussed previously. Associated with this should be ANR non-carbonate (ANRnc) testing of potentially neutralising rock types using the dissolution cell method.

These estimates can provide a reasonable basis to assess whether the ANR can match the reduced AGR at source. The second requirement is to plan to use these materials in layer dumping with AMD wastes and in cover design together with the engineering considerations. Short-term costs of this approach are likely to be higher than current practice but with the very high costs of closure and on-going management and liability in retained capital (e.g. several hundred million dollars at Newmont, Dowd, 2005), it may ultimately be more cost-effective. This comparison has not yet been fully evaluated but is clearly worthy of further development for sustainable AMD control.

3.0 RECENT CASE STUDIES

3.1 BHP Billiton Iron Ore (BHPBIO)

The AMIRA P933A BHPBIO case study has investigated the strategy of pyrite passivation through the use of on-site ANC-containing wastes to minimise AMD in overburden storages at Mt. Whaleback. BHPBIO has noted that *"There currently is nowhere in the world that has achieved relinquishment with the use of store and release cover systems in an arid environment"* (B. Huys), so alternative approaches should be considered. Both the Jeerinah shale and the highly reactive Mt. McRae shales were tested in kinetic leach columns (KLC). Several ANC wastes were first evaluated with a Jeerinah dolerite waste performing better than other ANC material. The dolerite mineralogy (determined from Rietveld XRD, QEMSCAN, bulk assay and carbon analysis) contains low carbonates (calcite 2-3 wt.%, ankerite 0.5-1 wt.%) and reactive silicates including chlorite >30 wt.% (\approx 20 wt.% poorly crystalline seen in QEMSCAN but not in XRD), anorthite 5 wt.%, tremolite 9 wt.% with a large amorphous fraction, 54 wt.%, in XRD probably containing the chlorite-like material (from bulk assay reconciliation). The ANC of the carbonates from assay is around 30 kg CaCO_3/t but the measured ANC (modified Sobek test, AMIRA/EGi, 2002) of the dolerite ranges from 47 (pH 2.4) to 84 (pH 1.2) kg CaCO_3/t depending on the final pH after the acid aliquot choice. Analysis of the solutions from the ANC tests show that calcium concentrations were independent of the test method suggesting that calcite and ankerite were quantitatively dissolved under all conditions. However, as the pH of the test solution was reduced, the concentration of Al, Fe and Mg increased, indicative of increasing dissolution of chlorite. These results indicate that the remaining silicates (principally clinocllore) present in the dolerite contribute between 17 and 54 kg CaCO_3/t , depending on the *in situ* reaction pH. This initial characterisation is valuable for short-term neutralisation but, given the abundance of silicates in most mine waste materials, it is the rate of dissolution rather than total concentration which determines the ability of silicates to buffer acidity resulting from sulfide oxidation.

Blending 10 wt.% dolerite into the Jeerinah shale resulted in leachates characterised by neutral pH and a slight excess of alkalinity with low dissolved metal concentrations (not detectable in most cases). These results indicate that co-disposal of Jeerinah shale with Jeerinah dolerite, will substantially reduce the risk of drainage containing elevated levels of metals emanating from waste emplacements in which these materials are stored.

Initially, dolerite was evaluated as a capping material for Mt. McRae shale. Results demonstrated that Jeerinah dolerite failed to supply sufficient alkalinity in standard KLC tests (AMIRA/EGi, 2002) to keep the pH high enough to bring about pyrite passivation. Subsequently the use of dolerite, applied either as a flow-through base layer to treat acidic drainage from Mt. McRae shale, or as a vertical layer within the Mt. McRae shale to provide a neutralising preferential flow path for acidic drainage, was investigated.

Leachates from the KLC tests in which Jeerinah dolerite was added to Mt. McRae shale as a base layer or as a vertical interlayer were characterised by: pH>7; no acidity (slight excess of alkalinity); low salinity (conductivity $\approx 100\text{--}200\ \mu\text{S}/\text{cm}$); low dissolved metal concentrations (non-detectable in most cases) and unexpectedly low sulfate, calcium and magnesium concentrations. It appears that, under KLC test conditions, dolerite provides a preferential flow path for water in the presence of Mt. McRae shale and could potentially be used in waste rock emplacements to channel acidic water flowing from PAF waste, and treat the water by removing both acidity and salinity.

Using the analyses for AGR and ANR, the evolution of W39 Terrace Dump at Mt. Whaleback has been characterised using water monitoring data collected at the AMD dam. The water samples had an average pH of 4.2 with high Mg (273 - 1210 mg/L), Al (up to 98 mg/L) and Si (up to 96 mg/L) concentrations suggesting that dissolution of (Mg, Fe) containing chlorite has been the major buffering mechanism within the dump. The calculated AGR/ANR_{nc} (ANR non-carbonate) data estimated from each of AMD dam water samples fall within the values measured for long-term column leach tests, suggesting that the estimates made for the dump are quite reasonable. The average ratio of AGR/ANR of 1.02 indicates that the AGR was fairly close to the ANR provided by silicate minerals within the dump in 2008-2009 and that any further reduction in the acid generation rate will likely result in less acidic drainage from the dump and subsequently increased water pH in the AMD dam.

3.2 Hidden Valley Joint Venture (PNG)

In this case study, an initial review of all previous reports on Hidden Valley (HV) waste rocks including waste classifications and KLC tests was completed. The lithologies of the main rock types at the site are granodiorite (GD) and metasediment (MS) (the latter hosting most of the ore). Both lithologies can contain significant pyrite, but the granodiorite also contains carbonate as calcite. The metasediment lithologies are all AMD-generating, but these reports identified ANC in granodiorite lithology from the Hidden Valley pit with more ANC compared with the granodiorite in the smaller, more weathered Hamata pit. An initial site visit was undertaken to identify lithologies in rock cores and collect relevant samples of both metasediments and granodiorites.

The HV granodiorite (HVGD) sample contained calcite (11 wt.%) with the majority as reactive plagioclase (anorthite with albite), chlorite, muscovite, orthoclase and quartz but also significant pyrite content (4 wt.%). The Hamata granodiorite (HamGD) is heavily weathered containing calcian albite as a major phase (48 wt.%), potentially supplying long-term ANC, with no pyrite or carbonate content.

To test whether passivation of pyrite and ANR matching can be achieved with these granodiorites, KLC tests of a composite metasediment sample with surface layers or in recycle through HVGD or HamGD sub-columns were set up. The ANR (from effluent analysis) of the HVGD surface-layered column is initially about 50% of the AGR but the ANR matched the decreasing AGR at around 70 weeks, as the fine pyrite was depleted but still with pH near 2.5. The ANR of the HamGD layer system was much lower (70%) than that of the HVGD layer system at 56 weeks and the HamGD layer did not contribute significantly to the suppression of AGR due to the lower ANR.

In the water recycle system, the AMD from the MS composite column to the sub-column containing HVGD maintained neutral drainage and provided alkalinity up to 56 weeks. In recycle of this effluent to the MS composite column, the pH of the main recycle MS composite column in this system was only slightly greater than that of the MS composite control, but the acidity of the effluent from the main recycle MS composite column was

overall significantly lower ($\approx 50\%$) than for the MS composite control column after 16 weeks. In recycle, the HVGD sub-column was not able to provide sufficient ANC to match the AGR of the MS composite main column. However, this recycle system has shown that HVGD is able to neutralise the acidity from the MS composite in low-flow conditions. This suggests that HVGD may be useful as a flow-through reactive barrier (similar to that used at the Savage River Mine) for low-flow seeps to reduce current lime additions.

To test this proposition, the effluent from a 2 kg batch of MS composite was used to flush a 1 kg HVGD sub-column in standard KLC flush conditions. The acidities (mg $\text{CaCO}_3/\text{kg}/\text{week}$) of the MS composite effluents varied significantly (e.g. <10 mg at 12 weeks; 45 mg at 16 weeks; 250 mg at 44 weeks) and then declined as the peak of the pyrite oxidation passed as found previously. The pH of the effluents from the HVGD sub-column maintained neutral to slightly alkaline pH across 60 weeks. Most of the toxic metal ions were precipitated and below detection in the HVGD effluent except manganese. The ANR (from effluent concentrations) of the HVGD sub-column was slightly greater than the AGR of the MS composite up to 16 weeks but became slightly less than the MS AGR beyond 16 weeks. However, HVGD ANR is underestimated due to precipitation of gypsum (i.e. loss of Ca) in the HVGD sub-column, so that it is likely that the actual ANR of the sub-column exceeds the AGR of the MS column across more than 80 weeks. SEM confirmed that the pyrite in the HVGD sub-column was passivated under these conditions.

This sample of HVGD had relatively high pyrite content (≈ 4 wt.%), twice as much as the MS composite material. However, the neutralisation rate of the calcite/anorthite/chlorite in the HVGD was sufficient to enable formation of passivated pyrite surfaces within the HVGD reducing the pyrite oxidation rate. These results suggest that testing to locate HVGD lithology with lower pyrite content may provide greater ANR for potentially matching AGR from MS wastes in co-disposal or layer dumping.

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ASSESSING METAL LEACHABILITY FROM LOW SULFUR WASTES - SEQUENTIAL EXTRACTION METHODS

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ABSTRACT

Some mineral wastes managed by Rio Tinto Iron Ore, although low in sulfur, may contain metals and metalloids such as As, Se, Sn and Mn. The mineralogical hosts for these elements will determine their long-term leachability. A sequential extraction scheme, comprising a series of leaching steps using different leachants, each more 'aggressive' than the last, has been trialled to understand the distribution of minor elements between different mineral components.

The results demonstrated that many trace elements are partly or entirely bound within relatively unreactive minerals such as crystalline silicates or iron oxides. In a typical waste rock storage facility, these minerals would remain stable and the release of the constituent trace elements to contacting waters is unlikely. The results did identify some trace element associations with more soluble mineral components: (i) soluble sulfates and carbonates, as sources of transition metals, alkaline earths, uranium and selenium, and (ii) manganese oxy-hydroxides as sources of barium, transition metals and thallium. Oxidisable components may also play a role, e.g. trace sulfides, carbonaceous material, and redox-sensitive elements adsorbed on mineral surfaces. These mineral components are likely to be the principle contributors to long-term leaching from low sulfur wastes. The trial suggested that only a minor proportion of the total trace element mass was involved, and that leachable masses were unlikely to be of environmental significance. When compared to high sulfur materials, low sulfur wastes therefore pose a negligible risk of acidic and metalliferous drainage.

Nevertheless, sequential extraction methods have provided valuable insight to trace element distribution amongst different mineral groups. Used in conjunction with routine waste characterisation testing, such results can be used to support prediction of long-term contact water quality. This paper presents the selective leaching approach that was developed and discusses the results and their application to field scale predictions and management.

1.0 INTRODUCTION

Mined material stored within a surface waste facility will be exposed to a geochemical environment different to that encountered in situ. This is especially true for materials originally sourced from below the pre-mining water table. For materials containing elevated sulfide mineralisation, it is relatively well understood that the main 'geochemical response' to disturbance will be oxidation of the sulfides. Well established laboratory protocols exist to measure sulfide oxidation rates, consequent acid generation, neutralisation and solute release rates. Higher risks of acidic and metalliferous drainage (AMD) are coincident with these lithologies and are managed accordingly.

In contrast, low sulphur materials generally are considered to have low risks of acidic and metalliferous drainage (AMD). Although low in sulfur content, many of these materials have high metal contents (e.g. Fe and As are often present at levels in excess of average crustal abundances). The mineralogy of these materials is generally dominated by aluminosilicates (typically clays and micas) and crystalline iron oxides. At ambient temperatures and neutral pH, these mineral groups are expected to be stable and thus long-term geochemical responses would be limited. Results from de-ionised water leach extract tests suggest that leaching from low sulfur materials is low, consistent with expectation. However, results from these tests reflect the condition of the sample at the time of testing and it is possible they do not capture the full range of leaching behaviour that could occur over the longer term.

To build confidence in the assertion that low sulfur wastes pose a low risk of metalliferous drainage, Rio Tinto Iron Ore (RTIO) recognised that there is a requirement to understand better the distribution of minor and trace elements amongst mineral components within the materials so that a more robust assessment can be made of metal mobility in the longer term.

This paper summarises key outcomes from a trial scheme of sequential extraction tests. The objectives of the programme were to:

- (i) quantify the minor/trace element content of different mineral components in the sample; and
- (ii) give insights to that proportion of the minor/trace element content that could be more vulnerable to mobilisation in the longer term.

2.0 REGIONAL AND GEOLOGICAL SETTING

The Pilbara region of Western Australia is located 1,200 km north of Perth. The arid climate typically has 2 seasons, a wet season in summer where cyclones are common followed by a dry winter. Annual rainfall is typically 395 mm/year and evaporation can be 1,200 mm/year.

Iron ore is mined from the mineralised Banded Iron Formation of the Hamersley Group (Brockman or Marra Mamba iron formations) or from eroded and redeposited material (e.g. Channel Iron Deposits). The Mount McRae Shale (MCS) hosts most black shale, and it is overlain by the Dales Gorge Member of the Brockman Iron Formation, which is often exposed in pits where the lower portion of Dales Gorge Member is mined. Most deposits at the Pilbara RTIO mine sites are ore bodies that exist under reducing conditions but the genesis is not related to sulfide mineralisation. Supergene enriched Banded Iron Formation (BIF) or Detritals that are above the water table and have been exposed to long term weathering are unlikely to contain sulfides within the ore. However sulfide bearing shale or lignite may be inter-bedded with or lie stratigraphically below the ore body.

3.0 METHODS

3.1 Sequential Extraction Approach

The approach trialled is summarised in Table 1 and is based on approaches presented by Hall et al., 1996; Piatak et al., 2007; and, Lavengren et al., 2009.

The approach differs from those published in two key ways:

- Intermediate steps involving phosphate-based leachants - designed to target reactive organics - have been excluded, due to concern that reaction of phosphate with iron oxide surfaces would affect reactivity of the iron oxides during subsequent steps. Any organic components present within the Pilbara materials are expected to

be relatively insoluble and un-reactive (e.g. the carbonaceous material present in black shale). Thus, there is little value in the inclusion of a leaching step targeting reactive organics.

- Inclusion of a peroxide-based step (Step 5) to target readily oxidisable components of the samples. Published schemes usually target this component using stronger oxidants such KClO_3 , often in a mixture including HNO_3 . In the current work one of the aims was to trial a method that could be readily adopted by commercial laboratories, i.e. using reagents in common use and avoiding the requirement to extend current health and safety procedures.

Table 1. Summary of sequential extraction approach adopted

Step	Leachant	Target Phase(s)
1	De-ionised water	Labile
2	Acetic Acid	Adsorbed/ exchangeable/ carbonates
3	Hydroxylamine Hydrochloride	Amorphous Fe/Al/Mn oxy-hydroxide / crystalline Mn oxides
4	Hydroxylamine Hydrochloride; Acetic Acid	Crystalline iron oxy-hydroxide
5	Peroxide (equivalent to a net acid generation, NAG, test)	Readily oxidisable sulfides/organics
6	Aqua regia	Less readily oxidisable / acid extractable components
7	4 acid digest	Residual

At each step, the leachate was collected and analysed. The residue was also rinsed (using de-ionised water for Step 2, 4 and 5, and acetic acid for Step 3) to remove residual reagent prior to next step in scheme. Rinsate was combined with the leachate prior to analysis. At selected stages, a sub-sample of the solid residue was submitted for additional characterisation (e.g. X-ray diffraction, total organic carbon and aqua regia digest).

Testing was conducted by Sydney Environmental and Soil Laboratory (SESL), Australia. Analytical support was provided by the National Measurement Institute (NMI), Australian Laboratory Services (ALS) and Queensland University of Technology (QUT).

3.2 Data Processing and Quality Assurance

The trial comprised three batches and involved twelve samples. Three samples were tested in duplicate. Examination of the results showed that reproducibility between the duplicates was good; generally duplicate results were within 10%.

Blanks were included in the programme of work so that trace contaminants introduced from reagents and/or equipment could be accounted for. If results for blanks suggested significant introduction of a solute, leachate compositions were adjusted accordingly.

All results were normalised to allow for mass loss during each of the leaching steps. In other words, all results are presented in terms of mass relative to the initial sample mass.

3.3 Sample Characteristics

Twelve samples selected for testing covered a range of lithologies, mineralogies, metal contents and acid-base accounting characteristics. Five samples were from Brockman iron

deposits, four from Marra Mamba deposits and three from Channel iron deposits. Two Mount McRae Shale samples and one lignite sample had sulphur concentrations greater than 0.1% but all other samples had sulphur concentrations less than 0.1%.

Mineralogical composition and sulfur content are illustrated in Figure 1. A significant portion of most samples was categorised as amorphous or 'unidentified'. Considering the geological setting, some of this unidentified component comprises amorphous Fe/Mn/Al oxy-hydroxides. In some samples this component could also include carbonaceous material, e.g. the lignite sample (EMD041) and some shale samples (ELD385, EPU952 and EAW096).

The measured concentrations of selected elements are provided in Table 2. There are many instances where elements were present at concentration in excess of crustal averages (highlighted). Low sulphur samples were selected from a larger batch of samples that were characterised by solids analyses and liquid extracts. The samples selected were chosen to further explore any significant enrichment or potential elemental leaching.

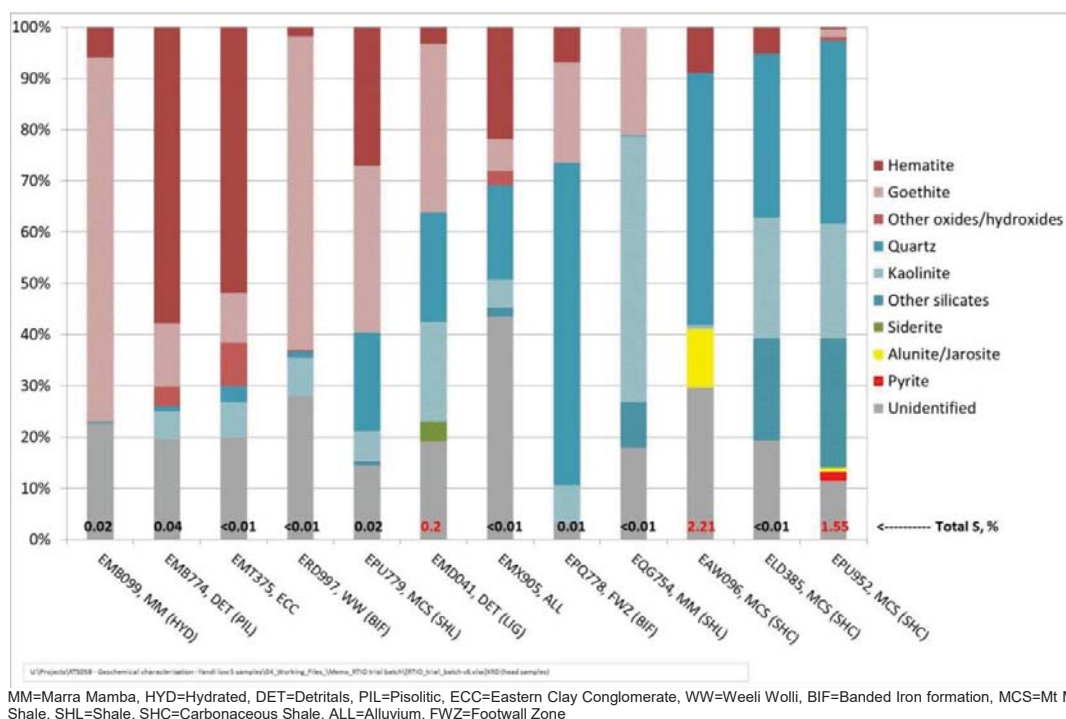


Fig. 1. Sample characteristics - mineralogy and total sulphur content

Table 2. Measured concentrations of selected elements (mg/kg)

Sample	Sample type	Sb	As	Cd	Co	Cu	Pb	Mn	Mo	Ni	Se	Sn	V	Zn
Crustal Abundance ^[1]		0.2	1.5	0.11	20	50	14	950	1.5	80	0.05	2.2	160	75
EPU952	MCS SHC	9.9	78	0.02	9.2	81	55	45	7.2	41	3	4.4	97	11
ELD385	MCS SHC	8.1	12	0.1	0.5	151	67	12	3.0	14	50	9.6	65	22
EAU096	MCS SHC	17.6	483	0.1	5.9	145	143	8	30.9	22	39	5.3	61	8
EQG754	MM SHL	5.9	31	0.1	55.4	69	41	21900	1.1	151	1	4.5	137	142
EPQ778	FWZ BIF	1.1	264	3.1	58.7	16	9	2300	4.9	118	<1	1.0	26	158
EMX905	ALL	4.0	11	0.03	9.2	48	17	307	0.7	24	1	1.3	205	34
EMD041	DET lignite	1.1	28	2.1	131.5	32	26	4800	3.1	133	50	2.2	118	318
EPU779	MCS SHL	16.7	896	0.2	7.6	28	23	232	16.9	31	1	12.7	33	107
ERD997	WW BIF	14.8	26	0.02	2.8	23	72	58	2.7	8	3	4.5	188	6
EMT375	ECC	6.2	21	0.03	13.5	45	40	189	1.9	25	2	3.3	562	14
EMB774	DET pisolitic	4.0	33	0.02	5.9	12	13	471	2.9	18	2	3.6	116	25
EMB099	MM HYD	0.5	7	0.3	9.9	12	2	174	0.7	15	<1	0.3	9	148

[1] Based on mean crustal averages given in Bowen, 1979.

Highlighted table entries correspond to values that result in calculated global abundance indices (GAIs) of 3 or more. GAI is calculated using the following formula (Förstner, 1993):

$$GAI = \ln \left(\log_2 \left(\frac{\text{Measured Concentration}}{1.5 \times \text{Average Abundance}} \right) \right)$$

Zero or positive GAI values indicate enrichment of the element in the sample when compared to average-crustal abundances. As a general rule, a GAI of 3 or higher signifies enrichment that warrants further evaluation.

4.0 RESULTS

4.1 Mineral Leachability

Mineralogical changes that occurred between sequential extraction steps were assessed by examining XRD analyses of residues, and making inferences based on major element leachate chemistries. Figure 2 shows the distribution of leachable Fe, Mn, S and Ca at each step; Table 3 summarises the corresponding mineral extraction.

There are many instances where target minerals were only partially leached. In these cases, undissolved mineral mass carried forward to the next steps. For example:

- Iron oxy-hydroxides. Step 4 was designed to target crystalline iron oxides such as goethite and hematite. Some of the samples contain up to 70% of these minerals (Figure 1). As shown in Figure 2, the mass of Fe leached during Step 4 equates to between 0 and 6% goethite or hematite. Significant goethite/hematite remained unreacted until Steps 6 and 7 were reached.
- Carbonates (Siderite). Step 2 was designed to target carbonate minerals. Siderite in the lignite sample, EMD041, was not completely digested during Step 2. XRD results indicate siderite remained present in the Step 2 residue, albeit in reduced quantities. High Ca leached during Step 2 for this sample and could suggest that Ca had substituted within the siderite, or that small, undetectable quantities of calcite were present alongside the siderite.

The effectiveness of the leachant is influenced by many factors – such as the mass of the target mineral present, the sample matrix and operational conditions (e.g. liquid:solid ratio, pH, etc).

The leach method also identified phases that could not otherwise be identified through XRD analysis. For example:

- Manganese oxy-hydroxides. Step 3 was designed to target manganese oxides and hydroxides. For some samples (Marra Mamba shale, EQG754; FWZ BIF, EPQ778; lignite, EMD041), significant quantities of Mn were leached in Step 3. Although such minerals were not identified by XRD, it is possible that they were present in small quantities, below the detection limit of the technique.

Only three samples contained significant concentrations of sulfur: EMD041 (lignite), EPU952 (pyritic MCS) and EAW096 (alunitic MCS). For the first two of these samples, sulfur leached most significantly during Steps 1 and 5, although some leachable sulfur was detected during other steps. Leaching during Step 1 is believed due to dissolution of sulfates and hydroxysulfates, e.g. jarosite in the case of EPU952. Step 5 is designed to target sulfides, explaining high sulfur leaching during this step. Sulfides were not identified in sample EMD041 (Figure 1), but may have been present at levels below XRD detection. In the case of the alunitic sample (EAW096), most sulfur remained unreacted until Step 7, indicating that alunite remained undissolved in earlier steps (confirmed by XRD analysis of the Step 6 residue).

In general, Al, Si and K leaching did not occur until Steps 6 and 7. XRD indicated that the main aluminosilicate minerals present were quartz (source of Si), kaolinite (source of Si, Al) and muscovite (source of Si, Al, and K). It was expected that these minerals would remain inert until the latter stages.

Four samples contained significant organic carbon: EMD041 (lignite), EPU952 (pyritic MCS), ELD385 (MCS) and EAW096 (alunitic MCS). With the exception of sample ELD385, the organic carbon content in residues was monitored. A marked decrease in organic carbon was noted at Step 6, believed due to reaction (oxidation) during Step 5.

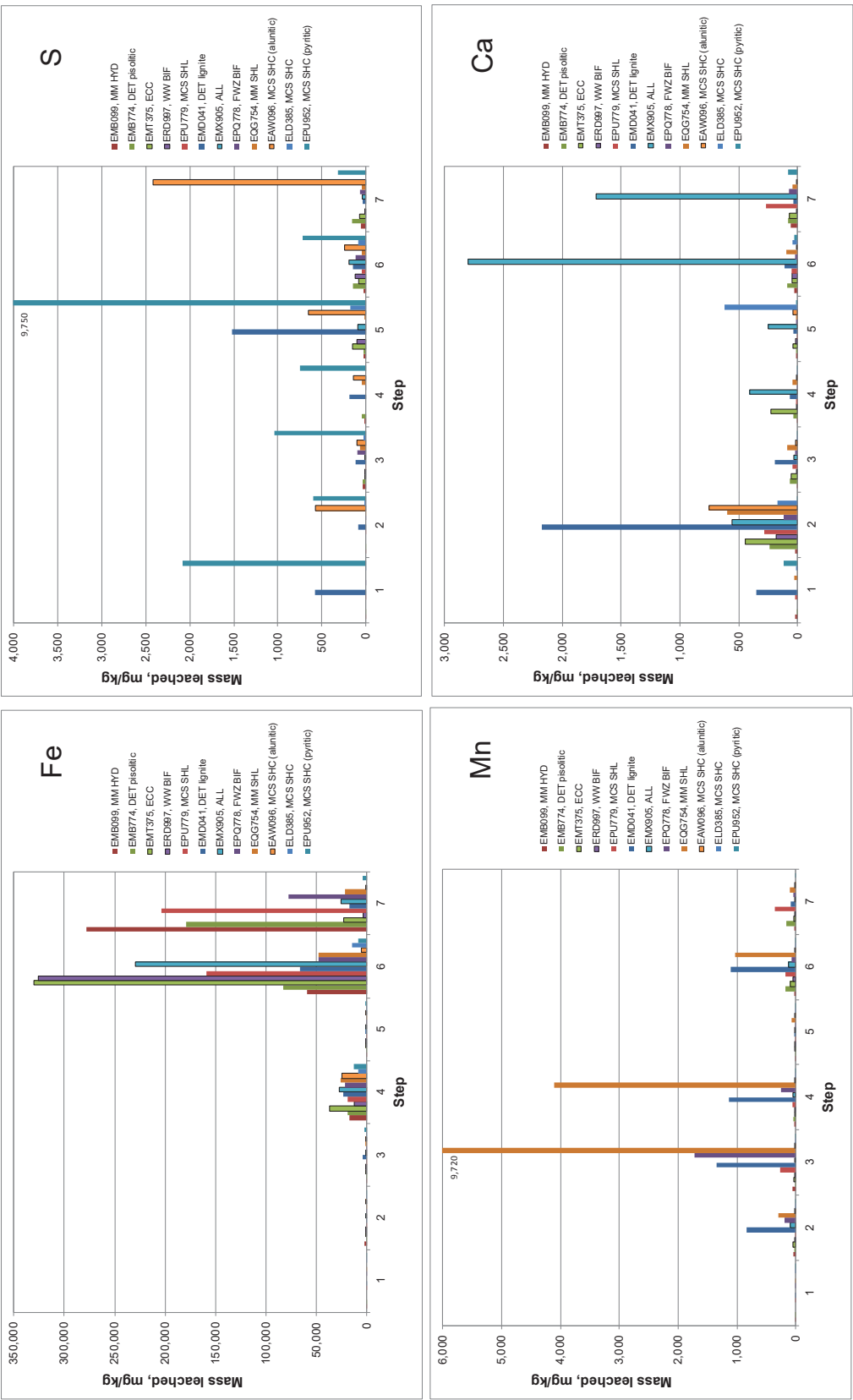


Fig. 2. Mass of Fe, S, Mn and Ca leached at each step

Table 3. Summary of mineral extraction achieved

Approximate Percentage of Mineral Leached ^[1]												
Step	EMB099 MM (HYD)	EMB774 DET (PIL)	EMT375 ECC	ERD997 WW(BIF)	EPU779 MCS(SHL)	EMD041 DET (LIG)	EMX905 ALL	EPQ778 FWZ (BIF)	EQG754 MM (SHL)	EAW096 MCS (SHC)	ELD385 MCS (SHC)	EPU952 MCS (SHC)
1												
2						40-50% (siderite)						<100% (calcite) ^[2]
3						30% Mn hydroxide or oxides		100% Mn hydroxide or oxides	60% Mn hydroxide or oxides			
4 ^[3]	<10% (<u>goethite</u> , hematite)	<10% (goethite, <u>hematite</u>)	<10% (goethite, <u>hematite</u>)	<10% (<u>goethite</u> , hematite)	<10% (goethite, hematite)	~20% (<u>goethite</u> , hematite)	~10% (goethite, <u>hematite</u>)	~15% (<u>goethite</u> , hematite)	~30% (goethite)	~80% (minor hematite)	~40% (minor hematite)	~40% (minor goethite, hematite)
5						>90% (TOC)				~80% (TOC)	? (TOC not quantified for this sample)	~100% (pyrite) <90% (TOC)
6												
7												

Significant reaction of silicates, and remaining hematite and goethite during Steps 6 and 7

Significant reaction of silicates, and remaining hematite and goethite during Steps 6 and 7

Notes

- [1] Based on measured changes in mineralogical composition of residues (XRD data), or observed distribution of major constituent elements in leachate solutions.
[2] Whilst XRD analysis did not identify calcite in the head sample, low levels of calcite (0.4-0.5%) were identified in the residues after Step 2 and 6 (but not Step 4). The trends cannot be interpreted with confidence due to proximity to XRD detection limits; it would appear however that calcite was not removed completely during Step 2.
[3] Dominant iron oxide mineral in initial sample underlined.

Target phases for each step

- Step 1: Labile forms
Step 2: Carbonates
Step 3: Amorphous Fe/Al/Mn oxy-hydroxide and crystalline Mn oxides
Step 4: Crystalline iron oxy-hydroxide
Step 5: Sulfides/ organics
Step 6: Reactive acid soluble silicates
Step 7: Remaining silicates

4.2 Trace Element Leaching

Correlations between trace and major element behaviour during the extraction scheme were used to give insights as to distribution of trace elements among the major phases of the samples.

For each step, Figure 3 shows the distribution of leachable As and Se (as examples of oxyanion elements), Cu and Co (as examples of transition metals). Table 4 summarises inferred host mineral forms for key trace elements.

4.2.1 Oxyanions

Arsenic (As) was present in appreciable quantities (often more than 100 ppm) in several samples. The majority of As leached during Steps 4, 6 and 7, showing behaviour very similar to Fe. The leachable As is likely present as a co-precipitate with iron oxy-hydroxides.

Antimony (Sb) and molybdenum (Mo) were present at low levels in the samples (close to analytical detection limits) but showed leaching trends, and thus associations, analogous to As.

Selenium (Se) showed some distinct trends. Two samples contained appreciable Se (> 50 ppm): ELD385 (MCS) and EMD041 (lignite). In both these samples, most Se leached during Step 5. The MCS sample contained negligible S. It is possible that Se release from this sample was related to:

- (i) oxidising carbonaceous material, or
- (ii) desorption related to redox transitions. Selenate (Se^{VI}) does not sorb as strongly as selenite (Se^{IV}) (Balistrieri and Chao, 1987) and so oxidising conditions may result in release of sorbed Se.

The lignite sample also contained significant sulfur, which was also released during Step 5, suggesting that Se could be present as an impurity in oxidisable sulfide, or may have been released as a result of the mechanisms described above.

4.2.2 Transition Metals

Copper (Cu) and cobalt (Co) leaching occurred in several steps suggesting multiple mineral hosts contributed to leaching behaviour. Cd, Cr, Ni and Zn showed analogous results.

High leaching during Steps 1 (pyritic MCS sample EPU952 only) and Step 2 was attributed to release from carbonates or sulfates. Significant leaching in Steps 4, 6 and 7 correlated positively with Fe behaviour and was considered indicative of the importance of iron oxides as mineral hosts. Significant Co leaching during Step 3 correlated positively with Mn behaviour, identifying manganese oxy-hydroxides as an additional mineral host for Co.

Significant leaching in Step 5 was observed for four samples: EPU952, EMD041, ELD385 and EAW096. Cu leached from all four, whilst Co leached from EMD041 and EPU952 only. All four samples contained oxidisable carbonaceous material, whilst EMD041 and EPU952 also contained sulfides. Reaction of either of these sources could be linked to release of transition metals.

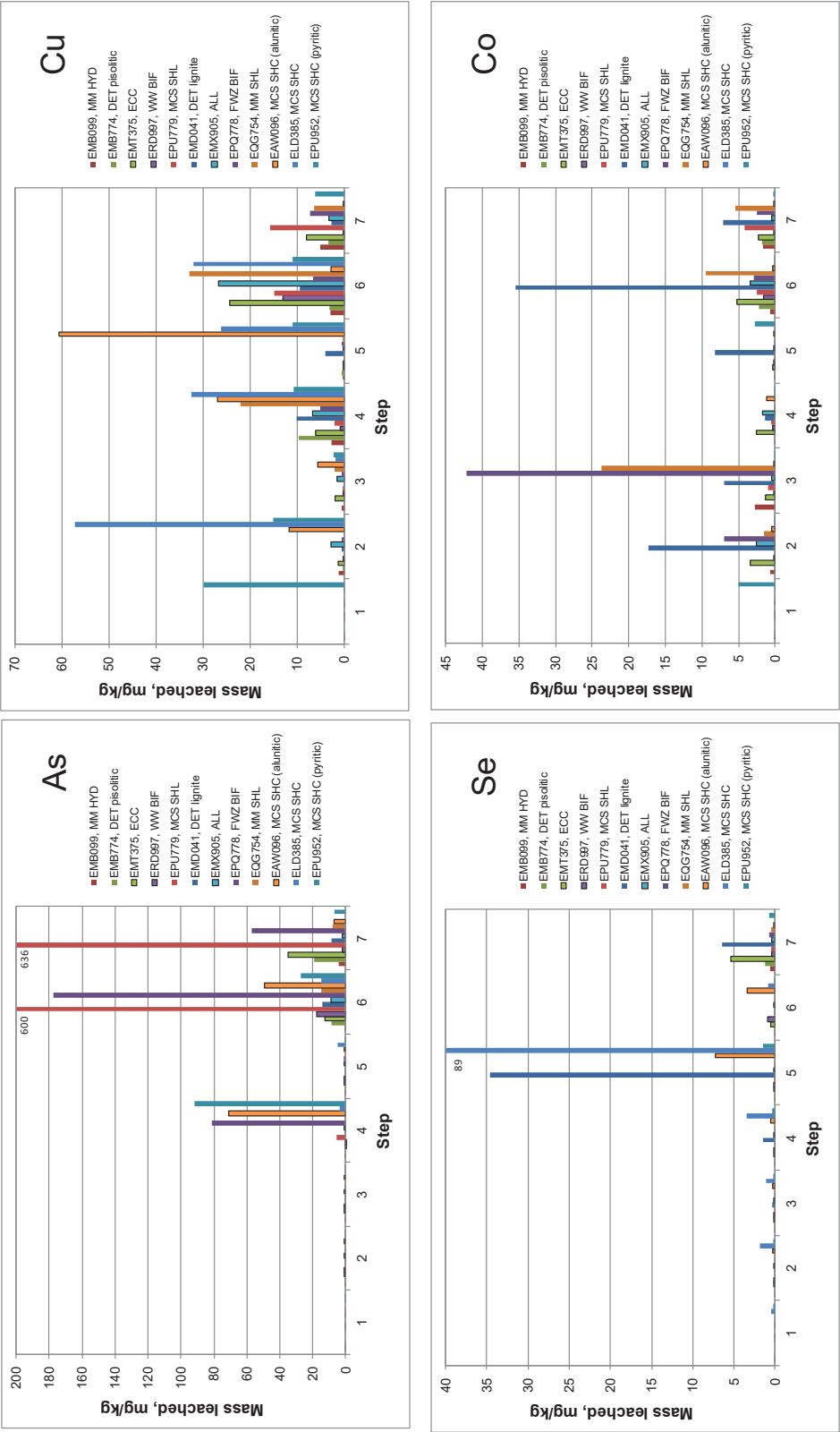


Fig. 4. Mass of As, Se, Cu and Co leached at each step

Table 4. Inferred Host Mineral Forms for Trace Elements

Step	Target Phase(s)	EMB099 MM (HYD)	EMB774 DET (PIL)	EMT375 ECC	ERD997 WW(BIF)	EPU779 MCS(SHL)	EMD041 DET (LIG)	EMX905 ALL	EPQ778 FWZ (BIF)	EQG754 MM (SHL)	EAW096 MCS (SHC)	ELD385 MCS (SHC)	EPU952 MCS (SHC)
1	Labile forms							Mn, Ba ^[1]			Zn ^[1]		Co, Cu, Ni
2	Carbonates	Ni	Sr, Ba	Sr ^[1]	[1]	Sr	Sr, Ba			Sr,		Cu	
3	Amorphous Fe/Al/Mn oxy- hydroxide and crystalline Mn oxides	Co, Mn, Pb				Mn, Ba	Ba, Mn		Ba, Cd, Co, Mn, Ni, Sr, Ti	Ba, Co, Mn, Sr, Ti	Ba		
4	Crystalline iron oxy- hydroxide	Pb, Th	Cu, Zn	Pb			Cd, Cu, Pb, U, V	Pb	Mo, Ni, V	Cu, Pb, Mo, Ni, V	As, Cd, Cr, Co, Pb, Mn, Mo, Ni, U, V	Sb, Pb, U, V	As
5	Sulfides/ organics					Sn	Se				Cu, Mn, Se	Ba, Mn, Ni, Se, Sr, Sn, Zn	Co, Se
6	Reactive acid soluble silicates	V	Sb, As, Cd, Co, Pb, Mo, Ni, U, V, Zn	Cd, Cr, Co, Cu, Pb, Mn, Mo, Ni, Th, V, Zn	Sb, As, Ba, Cr, Co, Cu, Pb, Mn, Mo, Ni, Sr, Th, Sn, U, V	As, Cd, Cu, Pb, Mo, Sn, V, Zn	As, Co, Cu, Pb, Mo, Ni, Sn, U, V, Zn	Sb, As, Cd, Cr, Co, Cu, Pb, Mn, Mo, Ni, Se, Th, Sn, U, V, V, Zn	As, Cu, Mo, V	Sb, As, Cu, Pb, Sn, V, Zn	As, Sr, Zn	Sb, As, Ba, Pb, Mo, Ni, Sr, Sn, Zn	Mo, Zn
7	Remaining silicates	Sb, As, Ba, Cd, Cu, Mo, Sn, U, V, Zn	Sb, As, Co, Mn, Mo, Ni, Se, Sn, U, V	Sb, As, Ba, Mo, Ni, Se, Sr, Th, Sn, U, Zn	Pb	Sb, As, Co, Cu, Pb, Mn, Mo, Ni, Se, Sr, Ti, U, V, Zn	Sb, As, Ni, Sn	Sb, Ba, Se, Sr, Sn	Sb, Cu, Pb, Sn, Ti, U	Sb, As, Sn, U	Sb, Cr, Pb, Th, Sn	n.d.	Sb, Ba, Pb, Mn, Mo, Sr, Ti, Sn, U, V, Zn

Notes

[1] Step 1 (de-ionised water) was omitted from the scheme

5.0 CONCLUSIONS

5.1 Minor and Trace Element Distribution

It has been demonstrated that many minor and trace elements are partly or entirely bound within relatively un-reactive minerals such as crystalline iron oxides or silicates. In a typical waste rock environment, these minerals would remain stable and release of the constituent trace elements to contacting waters is unlikely.

With respect to more reactive mineral components, the results to date support the following conclusions:

- Soluble and lesser soluble sulfates and carbonates may represent potential sources of transition metals (especially Cu), alkaline earths (Ba, Sr), U and Se. These mineral phases react readily, and should conditions become acidic could be expected to dissolve and result in significant concentrations in leachates.
- Copper oxides (as another possible host for copper) may be present and would be mobilised should conditions become acidic.
- Manganese oxy-hydroxides may contain trace elements that could be mobilised under conditions where the manganese phases are unstable (e.g. mildly acidic conditions). Within the current dataset, Ba, transition metals (especially Co) and Tl were identified as correlating particularly strongly with Mn behaviour.
- Sulfides may represent important sources of transition metals and oxyanions. This is not a new conclusion as the link between sulfide oxidation and contaminant release is well known. When exposed to oxidising conditions sulfides can break down releasing acidity as well as constituent elements.
- Alternative oxidisable components (not sulfides – but reactive during Step 5). Such components could include reactive carbonaceous material, or elements strongly adsorbed in a reduced oxidation state. Like sulfides, these components could be mobilised under more oxidising conditions. Such components could represent important sources of more readily mobilised Se and Sn. Note that the role of the carbonaceous material as a source of contaminants remains unverified; further data are required.

5.2 Application to Field Scale Predictions and Management

Results from sequential extraction testing can be used to infer the potential risk of long-term contaminant release from mined waste materials. Elements incorporated within non-reactive minerals are expected to be associated with a low risk of leaching in the long-term. Based on the sequential extraction testing to date, non-reactive minerals incorporate, on average:

- More than 90% of the arsenic (As), lead (Pb), thorium (Th) and vanadium (V). Some of these elements (e.g. As) would have been classed as 'high risk' based on comparisons with crustal abundances alone (Table 2).
- Between 30% and 70% of transition metals, cadmium (Cd) and selenium (Se).

Test conditions are not representative of those in waste facilities and therefore results cannot be used as a direct basis for quantifying leach rates or predicting longer term water quality. Other leach test protocols, combined with geochemical modelling, are necessary to place bounds on the possible range of dissolved concentrations that could be expected in contact waters. Insights gained from sequential extraction testing (i.e. identification of reactive minerals present and associated leachable elements) can be used to optimise the design of leach test programs and ensure that the results obtained are applicable to expected

conditions in waste facilities. Thus, sequential extraction methodologies should be considered a valuable complement to, rather than a replacement of, existing leach testing practices.

With respect to waste management, materials containing reactive sulfidic (and carbonaceous) mineral components are considered to represent a higher risk of AMD and RTIO manage these wastes accordingly. These materials are segregated and placed in facilities designed to control oxidation and to minimise contact with infiltrating water.

In low sulfur wastes, reactive mineral components mainly comprise carbonates, readily soluble oxy-hydroxides and adsorbed/exchangeable elemental loads. These components would usually form only a small proportion of the waste mass (in the materials studied in the current work, these components comprised less than 5 wt% of the sample mass). The potential for AMD from such materials is considered to be negligible. Even should geochemical conditions arise that maximise the reactivity of these components (e.g. exposure to acidic or saline conditions) the resulting contaminant release would be expected to be of short duration. Whilst no special management strategies need be adopted for such wastes, it is recommended that possible maximum (short-term) concentrations of key contaminants be quantified – using standard leach testing protocols, but extended to include a wider range of geochemical conditions.

6.0 ACKNOWLEDGEMENTS

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OXIDATION AND SOLUTE ACCUMULATION IN PIT WALL ROCK: LIMITING CHANGES TO PIT LAKE WATER QUALITY

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ABSTRACT

At many Rio Tinto Iron Ore sites in the Pilbara, current or planned mining activities will take place below the regional water table. Pit design and construction strategies used by Rio Tinto include limiting exposure of sulfides on the pit walls, thereby limiting exposure of the sulfides to oxygen. As a consequence, rates of sulfide oxidation and generation of oxidation products are reduced. Thus, the product load available for release to the pit and ground waters prior to water table rebound is reduced and changes to water quality due to mining are limited.

To estimate the benefits of avoiding sulfide exposure on pits walls, models were developed to describe oxidation of:

- *Exposed sulfide-bearing rock on pit walls; considering a number of factors including:
(i) exposed wall rock surface characteristics,
(ii) fractures resulting from blast damage and
(iii) talus accumulation along pit benches.*
- *Naturally fractured sulfide-bearing rock volumes located at depth behind the pit walls, beyond the blast damage zone.*

Calculations indicate the oxygen consumption rates in sulfidic rock located 5 m behind a layer of intact, non-sulfidic rock could be at less than 1/20th of that of sulfide bearing talus located on the pit wall.

This paper describes i) the relative magnitudes of contributions of sulfide oxidation from the various sources to pit lake water quality and ii) Rio Tinto's risk based approach to assess the benefits of avoiding exposure of sulfides on the pit walls at specific sites.

1.0 INTRODUCTION

Rio Tinto operates several open cut iron ore mines in the Pilbara region of Western Australia. The regional stratigraphy includes the Mount McRae Shale (MCS), which if unoxidised, is known to contain pyrite. MCS is mined as waste for those ore bodies within the lower Dales Gorge member of the Hamersley Group stratigraphy. The Mt Sylvia formation underlies the MCS and this lithology can also contain sulfides when unoxidised. The oxidation of unoxidised MCS (referred to in this paper as black shale) is known to increase the solute load and decrease the pH of the pit water quality during mining. Rio Tinto pit design and construction strategies include limiting the exposure of the unoxidised MCS on the pit wall. Pits incorporating these design strategies are yet to be completed and the benefits of these strategies to pit void water quality after mine closure are yet to be realised.

This paper investigates the benefits of limiting exposure of the MCS through modelling of sulfide oxidation. The modelling included representation of various features considered representative of pits in the Pilbara, e.g. talus, pit walls, fractures.

2.0 PIT CHARACTERISTICS

A model pit was developed to include the features present in a number of the Rio Tinto's mines in the Pilbara. Characteristics of the model pit were based on current construction practices and, where available, data was drawn from site-specific measurements.

Figure 1 shows a representative cross-section, including the pit shell and the distribution of different rock formations.

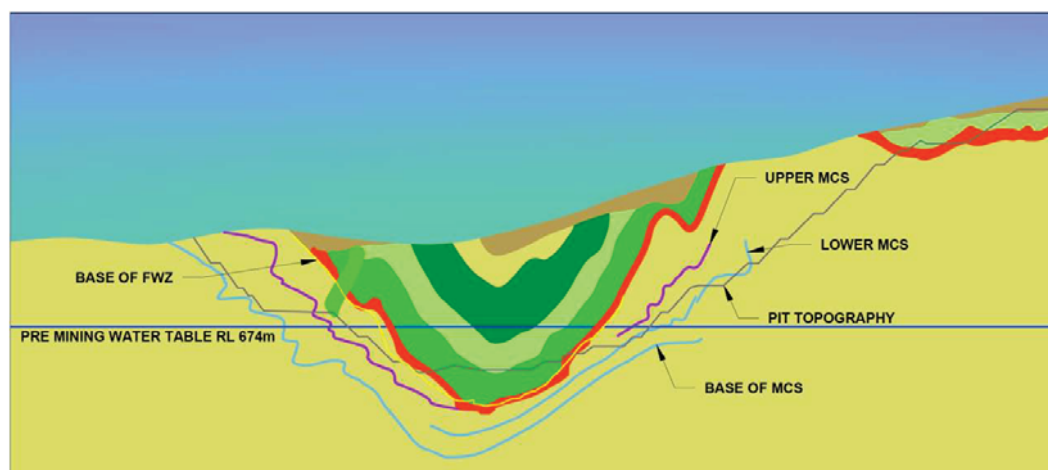


Fig. 1. Cross-section showing the pit shell and expected distribution of rock units

The pit was assumed to be excavated with 10 m wide and 10 m high benches with an overall angle of 63°. Fracturing adjacent to the pit wall due to blasting was assumed to extend 5 m from the surface with the fracture frequency and fracture aperture increased. Behind the blast-damaged rock zone, fracture frequency and aperture were assumed not to change as a result of mining and measured values were used to guide values used in the modelling. The thickness of the MCS was assumed to be 50 m.

Two conditions for MCS location were considered; one with the MCS exposed on the pit wall and one with the MCS 5 m behind the pit wall. These are illustrated in Figure 2. Following dewatering and consequent lowering of the groundwater table, the sulfide minerals in the MCS could be exposed to oxygen. The subsequent extent of oxidation depends on the time the groundwater table remains lowered and the rate of oxygen transport from the pit face to sulfidic minerals within the MCS layer.

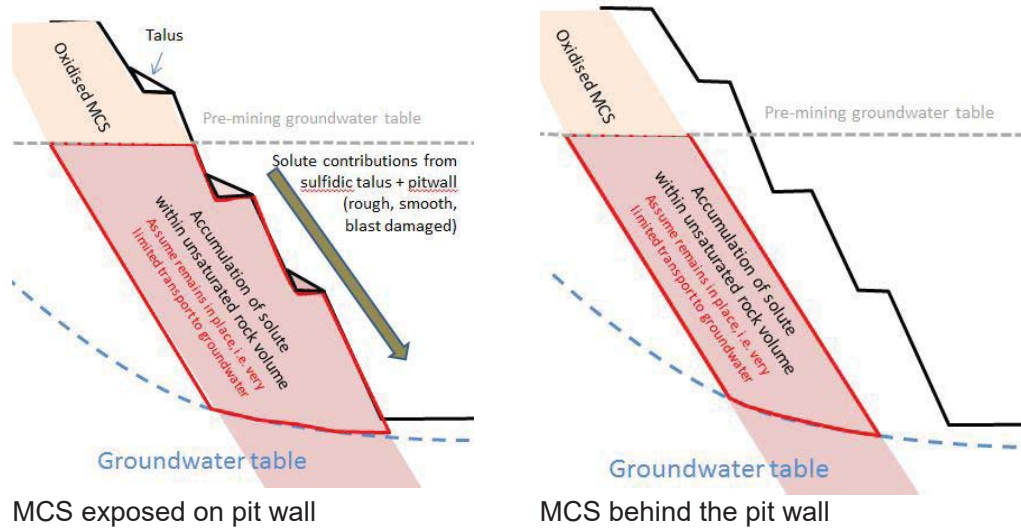


Fig. 2. Pit wall configurations showing two alternative MCS locations

Some transport of oxidation products due to movement of water within the unsaturated rock volume may occur. However, these movements are expected to be limited. Recharge in the Pilbara area is expected to be less than 1% of the average rainfall of about 200 mm to 350 mm/y. Therefore, at a recharge of about 2 to 3.5 mm per year, the average downward percolation rates would be about 400 to 700 mm/year (assuming porosity of about 1% and 50% saturation). For the purposes of the calculations described herein it has been assumed that transport of soluble salts down to the groundwater table within the timeframe of the modelling would be negligible.

As a pit is constructed, there may be periods when multiple benches are mined. Repeated mining of some benches may take place. Only a portion of each bench is likely to have MCS exposed on the pit wall. Figure 3 illustrates the modelled length of bench containing MCS, as a function of time, and is based on illustrative construction records provided by RTIO.

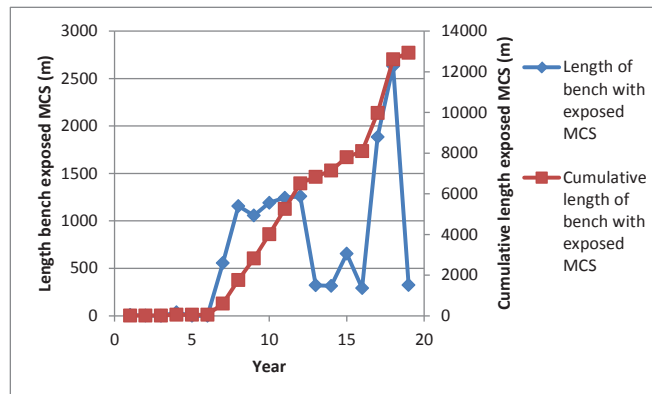


Fig. 3. Length of exposed MCS

3.0 OXYGEN TRANSPORT AND OXIDATION

Oxygen supply to the MCS was assumed to be by diffusion from the pit face or the top ground surface to the MCS. The oxygen concentration at the face of the MCS adjacent to or on the pit wall would in general be greater than that facing away from the pit wall (i.e. $C_1^b > C_1^a$ see Figure 4). For modelled cases where an overlying layer of non-sulfide bearing rock was present between the MCS and the pit wall, the thickness of the layer was taken as 5 m.

Talus was assumed to form immediately upon construction of a bench and to fill the bench at the angle of repose to the width of the bench. Further, oxygen was available to all talus present at concentrations that did not limit the oxidation rate of sulfides.

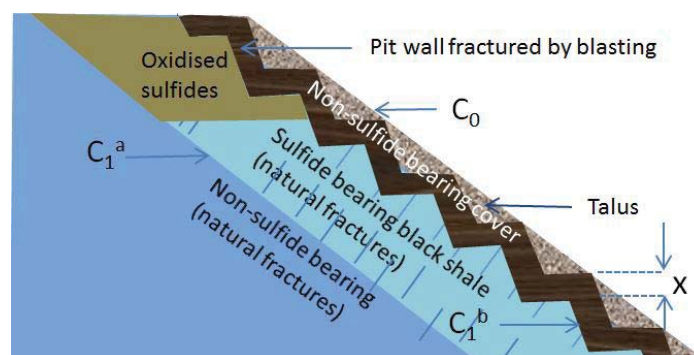


Fig. 4. Pit wall oxygen oxidation conceptual model

3.1 Oxidation Rates

When sulfides were available to oxidise, the oxidation rates of talus and fractured rock were assumed to be constant. In talus, oxidation continued until all sulfides had been consumed or until the talus was inundated. In low porosity fractured rock (porosity taken to be <1%) the availability of sulfides minerals was assumed to be infinite. Intrinsic oxidation rates measured in laboratory scale samples from Rio Tinto Pilbara sites ranged between 10^{-11} and 10^{-9} kg (O₂)/kg(rock)/s. As these rates were for small particles they were scaled by 0.2, based on estimated surface areas for the laboratory samples and talus. The estimated surface area of laboratory samples (2 m²/kg) and measured oxidation rates were also used to calculate an oxidation rate for fracture surfaces. The bulk density of the talus was assumed to be 1600 kg/m³.

3.2 Oxygen Penetration into Fractures

Water and gas transport in fractured rock are known to be different from those in porous media such as soils. However, methods of predicting fluid transport in fractured rock are not well established and those available rely on large quantities of appropriate field-measured site-specific data. In the absence of robust fractured flow theory and appropriate field measurements, models from other fields of investigation, and parameters measured for fluid transport in porous media, were used to estimate the degree of saturation and the effective oxygen diffusion coefficient within the non-sulfide bearing rock.

The distance that oxygen would penetrate into a fracture, z , was assumed to be:

$$z = \sqrt{\frac{2bD_{O_2}C_1}{G^R}} \quad [1]$$

Where z is the distance along the fracture from the opening, D_{O_2} is the effective oxygen diffusion coefficient, $2b$ is the fracture width, C_1 is the oxygen concentration at the entrance of the fracture and G^R is the rate of oxidation of the surface of sulfide bearing fracture exposed to oxygen

Fracture apertures were assumed to be 1 mm in non-blast damage black shale and a conservative value of 20 mm in blast damaged black shale.

A simple model of orthogonally oriented fractures that extend through the rock mass was assumed to represent the fractures as shown in Figure 5. Figure 6 indicates that the dependence of the fracture surface area per volume on the fracture interval is strongly non-linear. Thus, controlling blasting to limit or avoid the generation of fractures spaced at less than about 0.1 m within the blast damaged zone could greatly reduce the area of exposure due to fractures. Controlled blasting may further reduce the oxidation rate by reducing the fracture aperture ($2b$ in Eqn. [1]).

The fracture interval of the blast damaged rock was assumed to be 0.1 m in the three spatial directions. The intervals in the non-blast damaged rock parallel to the pit wall were taken as 1 m whilst that perpendicular to the pit wall was 0.1 m. The different spacing for the non-blast damaged rock was due to the anticipated preferential fracturing of the shale along planes parallel to the horizontal at the time the shales were deposited.

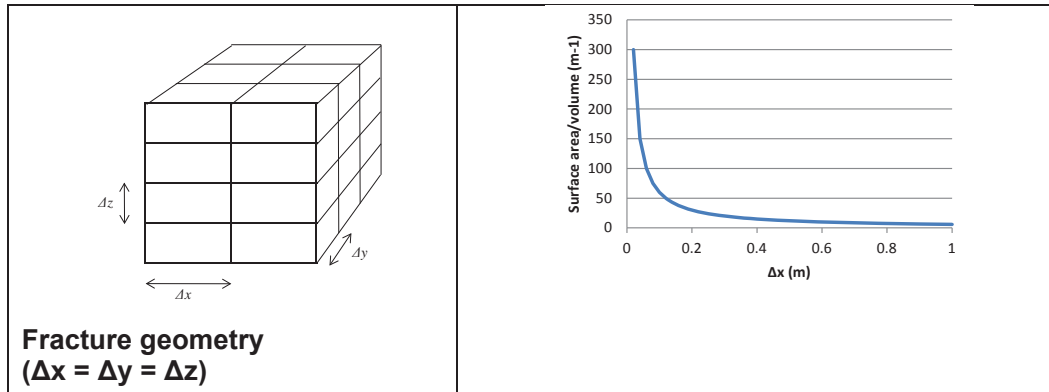


Fig. 5. Area of fractured surfaces per unit volume

Oxygen concentrations at fracture openings behind the pit wall were determined based on oxygen diffusion transport modelling.

Specifically:

- The degree of saturation was estimated using the HYDRUS 2D/3D software package. HYDRUS 2D/3D numerically solves Richards' equation for variably saturated water

flow. The soil water characteristic curves were represented in HYDRUS 2D/3D using the van Genuchten-Mualem representation and parameter values are given in Table 1.

- The oxygen diffusion coefficient was estimated based on the degree of saturation and correlations between saturation and measured oxygen diffusion coefficients for soils reported in the literature (Collin and Rasmuson, 1988).
- Oxygen concentrations C_1^a and C_1^b (Figure 4) away from the pit wall were estimated using the Constant Intrinsic Oxidation Rate (IOR) model (Ritchie, 1995). Parameter values used in estimation are given in Table 1.

Table 1. Hydraulic parameters of fracture rock units surrounding the pit

Material	Measured		Used in HYDRUS					
	K_s (m/d) ^(a)	Q_s-Q_r ^(b)	Q_r ^(c) /10	Q_s ^(c) /10	Alpha ^(c)	n ^(c)	K_s ^(c)	I ^(c)
Footwall Zone (Dales Gorge)	5 to 10		0.001	0.0038	2.7	1.23	0.0288	0.5
MCS	0.001	0.009	0.0007	0.0036	0.5	1.09	0.0048	0.5
Mt Sylvia Shale	0.01		0.001	0.0038	2.7	1.23	0.0288	0.5

Notes:

Use of larger K_s values for the Footwall zone prevented convergence in the numerical modelling.

(a) – Williams et al.

(b) – porosity was calculated from typical measured water contents and dry bulk densities, Lee, 2012b.

(c) – PC-Progress, 2006, HYDRUS version 1.11.

Table 2. Parameter values used to estimate the oxygen concentrations at the surfaces of the MCS

Parameter	Units	Value
Distance of MCS facing the pit wall to pit wall	m	5
Distance of MCS facing away from the pit wall (facing Mt Sylvia formation) to the top surface (nominal)	m	90
Intrinsic oxidation rates for MCS, S^*	kg(O ₂)/kg(rock)/s	10^{-11} , 10^{-10} to 10^{-9}
$D_{O_2}^e$ - diffusion coefficient of Foot wall zone	m ² /s	4×10^{-6}
$D_{O_2}^e$ – of diffusion coefficient of Mt Sylvia formation	m ² /s	4×10^{-9} , 4×10^{-8} , 4×10^{-7} , 4×10^{-6}

The pyrite oxidation reaction was taken as:



3.3 Contribution due to roughness

Roughness increases the surface area of the pit above that of a smooth surface. Therefore, the oxygen consumption rate by a rough sulfide bearing surface would be greater than that for a smooth pit wall. A simple model of roughness was used to illustrate the dependence. The roughness was represented by regular blocks exposed on the wall; the width of and separation between the blocks were assumed to be equal and of a length d and the height, h , of the blocks was also assumed to be uniform (Figure 7). The linear dependence of surface area on h/d is shown in Figure 8. The ratio of h/d was taken as 1 in the simulations, thereby doubling the surface area.

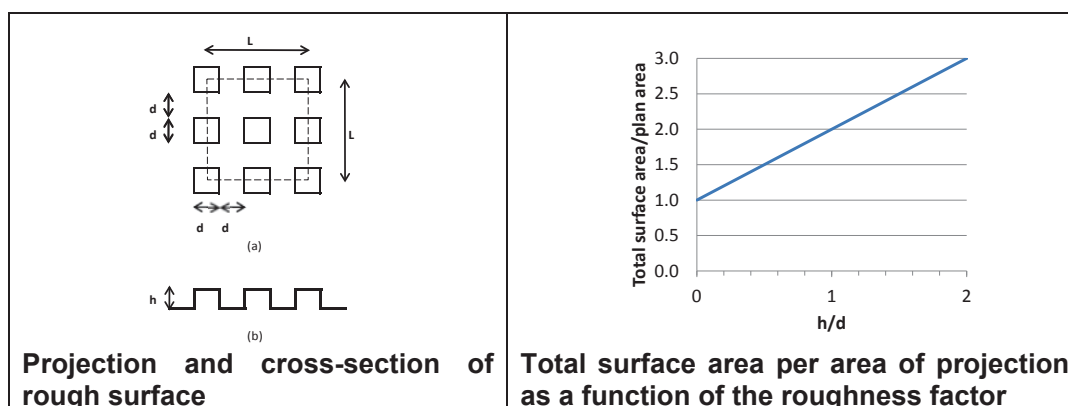


Fig. 6. Projection and cross section of rough surface and total surface area per area of projection as a function of roughness factor

4.0 PIT WATER AND GEOCHEMISTRY

4.1 Groundwater Rebound and Solute Mobilisation

At the end of mining the dewatering pumps will be stopped and the groundwater surface will rise in the vicinity of the pit (Figure 9).

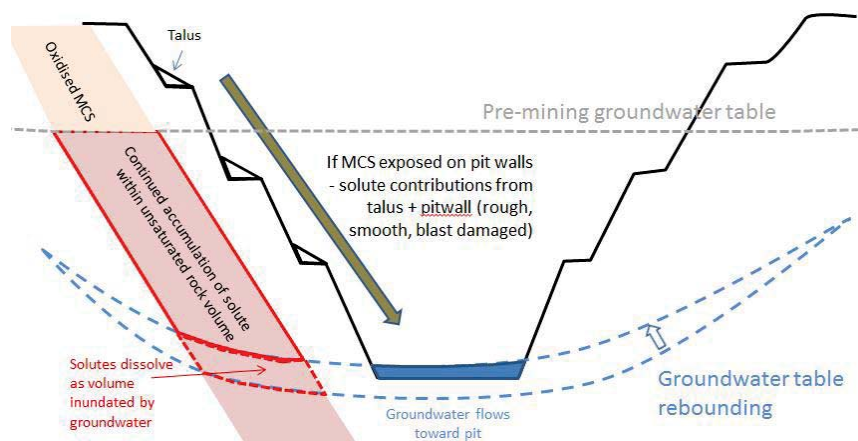


Fig. 7. MCS behind pit wall immediately after mining stops

As the groundwater table rebounds, it is expected that the oxidation products will dissolve in the inundating groundwater. Groundwater flow will result in mobilisation of dissolved solutes. The pit was assumed to be a permanent sink, which is common for pits in the area. Consequently, groundwater flow (and solute mobilisation) was towards the pit void; flows are expected to be most rapid immediately after cessation of mining and would decline with time.

For the calculations described herein the pit water balance (including pit lake elevation and groundwater in-flow rates as a function of time) were based on outputs from modelling of a pit with black shale dominating the final pit wall in the Pilbara

4.2 Geochemistry

Simplifying assumptions used in estimating pit water quality were:

- Prior to inundation all oxidation reaction products remained *in situ*;
- Inundation of a particular bench level occurred 'instantaneously';
- Once inundated, accumulated products dissolved and were transported to the pit void without solubility or transport constraints;
- The pit lake was assumed to be fully mixed, stratification was not considered.

At any time after cessation of mining, the total solute load in the pit void is the cumulative sum of (i) solute introduced by in-flowing groundwater, and (ii) reaction products mobilised and introduced to the pit void.

The assumed in-flow groundwater composition is shown in Table 3.

Table 2. Inflowing Groundwater Chemistry

Parameter	Groundwater value	Parameter	Groundwater value
pH	7.2	K	11
Al	0.05	Mg	97.5
Alkalinity	340	Na	122.5
Ca	95	SiO ₂	12
Cl	184.5	SO ₄	354.4
Fe	0.35		

Note: Units: mg/L, except pH.

The cumulative mass of sulfate due to oxidation was estimated from modelling oxidation (and comprises contributions from the pit wall and fracture surfaces – see discussion elsewhere). Based on data derived from MCS column studies (EGi, 2006, 2011), other solutes were added based on correlations that exist between those solutes and sulfate, see Table 4.

Table 3. Summary of the basis for calculated solute release rates

Element	Inferred Source Mineral	Mass fraction relative to mass of sulfate released ^[1]
Al	Aluminosilicate	0.068
Ca	Carbonate ^[2] , aluminosilicate	0.00458
Fe	Pyrite	0.421
K	Aluminosilicate	0.000425
Mg	Carbonate ^[2] , aluminosilicate	0.0244
Si	Aluminosilicate	0.0321

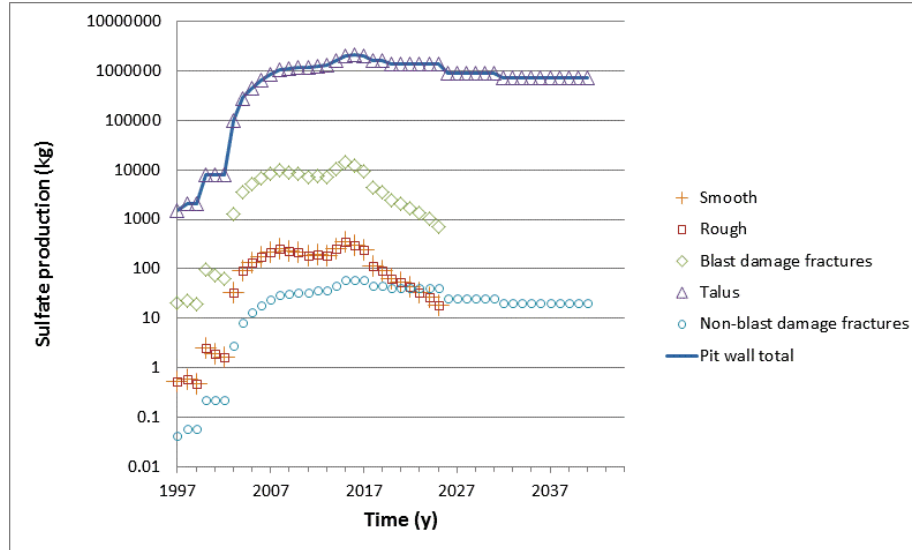
Notes:

- [1] Based on slopes of correlation lines between the element release and sulfate release. Slopes were only calculated for those columns that showed a strong correlation). The value adopted corresponds to the steepest slope that was calculated.
- [2] Minor source as it would largely be depleted at acidic pH conditions.

Pit lake chemistry was calculated by combining the solute load with the volume of water in the pit void during the time period of interest. Solute concentrations generated were processed using PHREEQC to investigate potential mineral solubility constraints. The following minerals were allowed to precipitate: alunite, jarosite, ferrihydrite, gibbsite, gypsum and amorphous silica.

5. MODEL OUTCOMES

For a range of assumed input parameters, e.g. oxidation rates, diffusion coefficients, modelling was performed to quantify the mass of sulfate produced. An example of modelled output for one case is presented in Figure 10 – individual contributors are shown separately. The intrinsic oxidation rate was 10^{10} kg(O₂)/kg(rock)/s. Talus makes the largest contribution and the blast damaged fractures make the second largest contribution. With a 5 m non-sulfide bearing layer on the surface contributions from the talus, smooth and rough surface and blast damaged fractures would be removed, leaving the non-blast damaged contribution as the largest contributor.



Note: In this case the rough and smooth contributions are identical, and the pit wall total and talus contributions are similar.

Fig. 8. Sulfate production

For comparison, three scenarios were modelled: a) no MCS present, b) MCS located behind the pit wall, and c) MCS exposed on the pit wall. Relative contributions to the solute load in the pit lake are presented in Table 5. With no MCS within the zone of influence of the pit lake, 100% of solutes would come from background groundwater. Where MCS is behind the pit wall, no solutes are generated by talus and the background groundwater provides the majority of the solute load. Where MCS is exposed on the pit wall, the solute load is dominated by inputs from talus.

Table 4. Relative solute contributions (%) to pit lake

Scenario	Solute Source	Time after cessation of mining	
		1 year	25 years
No MCS	Background Groundwater	100	100
	Oxidised MCS	0.3 - 1.3	0.8 - 25.8
MCS behind wall	Background Groundwater	98.7 - 99.7	74.2 - 99.2
	Talus	0	0
	Oxidised MCS	0.00004 - 0.003	0.01 - 0.3
MCS on wall	Background Groundwater	0.07 - 4.16	0.7 - 15.78
	Talus	95.84 - 99.93	83.95 - 99.29

Predicted pit lake sulfate concentrations are compared in Figure 11. With time after cessation of mining, sulfate concentrations decrease. This is largely due to dilution related to the higher volume of water within the pit void.

Where MCS is exposed on the pit wall and forms talus on the benches, sulfate concentrations are significantly higher. In this case, the sulfate concentrations could range from 560 to 380,000 mg/L (higher sulfate concentrations correspond to higher assumed oxidation rates).

The solute contribution from non-blast damaged MCS in the zone behind the pit wall is minimal, with estimated pit lake sulfate concentrations for the “MCS behind wall” scenario very similar to that calculated for the scenario where solutes come from background groundwater only.

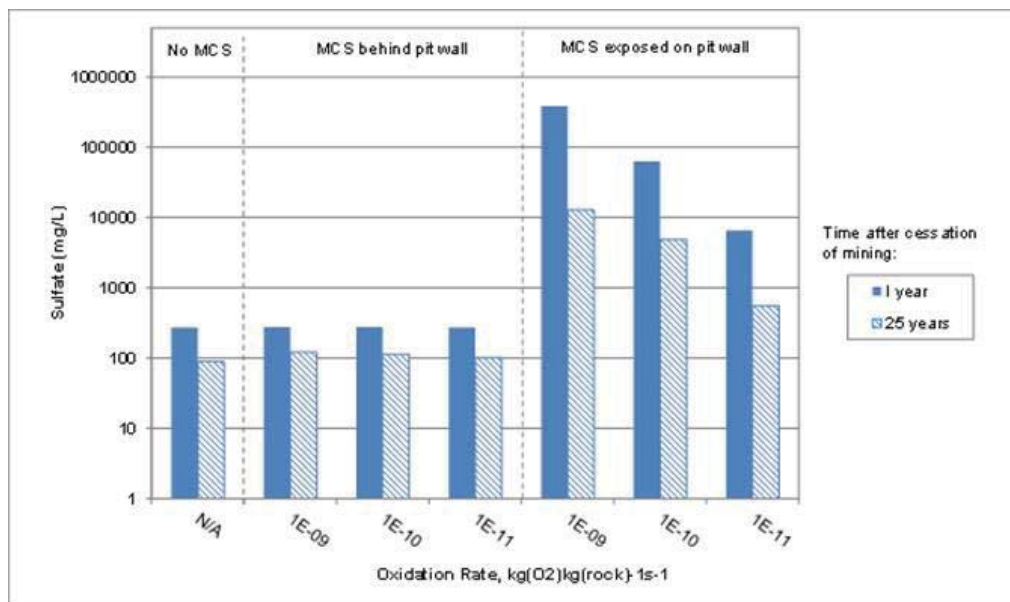


Fig. 9. Pit lake chemistry (sulfate concentrations) for different MCS exposure

6.0 CONCLUSIONS

Pit lake chemistry is dependent on:

- Solutes introduced due to the oxidation of sulfide minerals in talus, on exposed pit walls, on surfaces of fractures produced by blast damage, and natural fracture surfaces; and
- The chemistry of groundwater flowing into the pit.

Modelling of sulfide oxidation has shown that the relative contributions to sulfate production may vary by about five orders of magnitude. Indicative relative magnitudes are:

- Talus (1)
- Blast damaged fractures adjacent to the pit wall (0.01)
- Pit wall surface (smooth) (0.0001)
- Pit wall surface (rough surface contribution) (0.0001)
- Non-blast damage fractures behind the MCS (0.00001)

In practice the actual and relative values would depend on site-specific conditions.

Modelling assumed that the dominant oxygen transport mechanism was diffusion in the gas phase. The overall rate of sulfide oxidation in each material type depends on the rate of oxygen supply to the sulfide bearing materials and the intrinsic oxidation rate of the material. Therefore, limiting the quantity of sulfide minerals near the pit wall can significantly reduce the quantity of sulfates produced before groundwater inundates the sulfides after mining ceases.

Leaving a 5 m thick non-sulfide bearing layer over the sulfide bearing layer (MCS) would reduce the exposure of sulfides to oxygen and could result in significant benefits to expected pit water quality, particularly in the short-term when volumes of water in the pit are low and more sensitive to introduced solute loads.

Controlling blasting to limit or avoid the generation of sulfide bearing talus and fractures could greatly reduce the mass of sulfides exposed to oxygen, reduce the overall oxidation rate and the quantities of sulfate and acid produced prior to inundation of sulfides due to rebounding pit water.

7.0 REFERENCES

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AN ACID AND METALLIFEROUS PIT LAKE RISK ASSESSMENT TOOL

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ABSTRACT

Forecasting pit lake chemistry hundreds to thousands of years into the future can be challenging, especially in regions where comparable analogues are unavailable. The Rio Tinto Iron Ore (RTIO) Acid and Metalliferous Pit Lake (AMPL) Risk Assessment tool was developed as a transparent method to evaluate the potential for a contaminated pit lake to form at sites within the Pilbara iron ore region. The tool is primarily intended to be used pre-mining as a method to standardise evaluation of the major risks associated with potential pit lakes. The risks evaluated that can contribute to AMD generation are separated into categories related to physical pit lake properties, surface water contributions, potential groundwater contamination and possible constituents of concern. The overall likelihood of a contaminating pit lake forming for a given deposit is tabulated based upon the summation of weighted scores assigned to each parameter that could contribute to contamination. Results from the AMPL Risk Assessment tool can be used to determine the major risk factors for a given pit and guide the direction of potential additional investigations over life of mine. Whilst the AMPL risk assessment tool is specific to the Pilbara region, the concepts and method of assessment used are applicable at any site where a pit lake may form.

1.0 INTRODUCTION

Mine pit lakes have the potential to form in open cut mines that extend below the pre-mining water table (BWT). Mining BWT often requires extensive dewatering to access ore. Upon closure, dewatering ceases and the water table recovers to a post-mining level that could cause development of a pit lake. There are several characteristics of pit lakes that can influence their impact on the surrounding environment and potential to develop acidity. Final void water quality and its influence on the receiving environment are dependent upon climatic, hydrologic, biogeochemical, geologic and limnologic properties of the pit lake and the general region (Castendyk and Eary, 2009, INAP 2013). Evaluation of the risks associated with each of these categories and how they influence each other is a necessary step in predicting the behaviour of a potential pit lake.

An Acid and Metalliferous Pit Lake (AMPL) Risk Assessment tool has been developed by Rio Tinto Iron Ore (RTIO) as an initial assessment when evaluating a potential pit lake and as a tool for prioritising additional modelling and strategic steps. The AMPL tool has been designed to standardize pit lake risk evaluation across RTIO operations in the Pilbara and can be used at all stages of mining. The goal of pursuing this approach was to develop a tool that was easy to understand but considered all of the relevant risks associated with pit lake formation. The tool was developed in a spreadsheet, and a series of drop-down boxes are used to select options for which relevant scores are automatically assigned.

2.0 RISK ASSESSMENT DESCRIPTION

The AMPL Risk Assessment is based on the risk assessment framework previously developed to assess AMD risks at Rio Tinto Operations (Green and Borden 2011). The likelihood of forming an acidic or metalliferous pit lake is determined based upon weighted scoring of various parameters that contribute to the character of a given pit lake. AMPL is designed to be run throughout the Life of Mine, however it is particularly applicable pre-mining and during the study phase of an operation when pit designs can change rapidly and input data required for detailed pit lake models are unavailable. The tool is not designed to provide information with regards to water balance or conduct any geochemical calculations. It is intended to be a qualitative assessment of risk which can be used to prioritise what secondary quantitative modelling steps may be required. The parameters for the model were refined and optimised during a workshop that brought together both internal and external consultants and subject matter experts.

3.0 RISK ASSESSMENT TOOL PARAMETERS

The risks considered by the AMPL risk assessment tool as potential contributors to poor pit lake water quality are separated into three scored sections and one section regarding potential chemistry. The categories were chosen to encompass the major risk areas for pit lake closure that can be evaluated throughout the mine life. Each section is evaluated individually and weighted to determine an overall prediction of the potential for acidic lake formation. Physical parameters provide 20% of the overall score, and surface water and groundwater provide a combined 80% which is weighted based on the pit water balance (Figure 1).

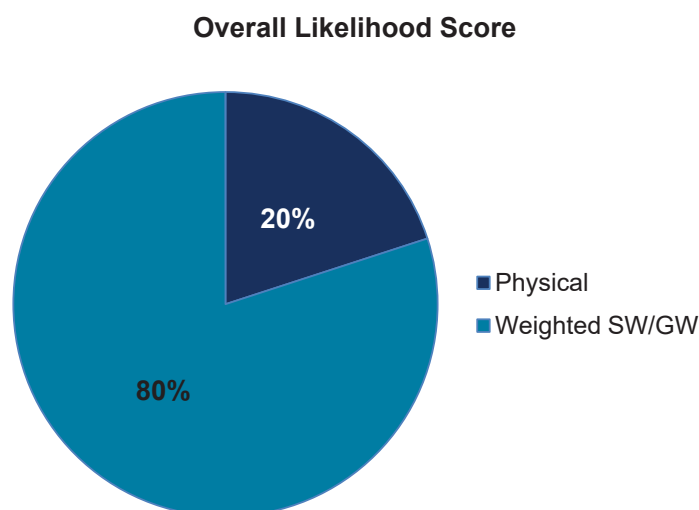


Fig. 1. Contribution of risk assessment tool parameters to the overall Likelihood score. Surface water (SW) and groundwater (GW) contributions total 80% of the score, and are weighted based on the percentage contribution to the water balance as selected in the AMPL risk assessment tool.

3.1 Section A: Physical Risks

The physical parameters associated with the potential formation of a contaminating pit lake, include pit dimensions and properties of backfill (if applicable). The planned pit depth and estimated pit lake surface area can give an indication of the potential for stratification. For user interpretation purposes, a range of relative depths are calculated based on surface area and depth to assist in assessment of stratification potential (Pieters and Lawrence 2012). A pit lake with a larger volume is generally considered to be a higher risk due to increased uncertainty in limnological and geochemical evolution. This section also considers options for pit backfilling including backfill volume, overall composition and neutralising capacity. The relative contribution of each variable to the 20% physical score is given in Figure 2.

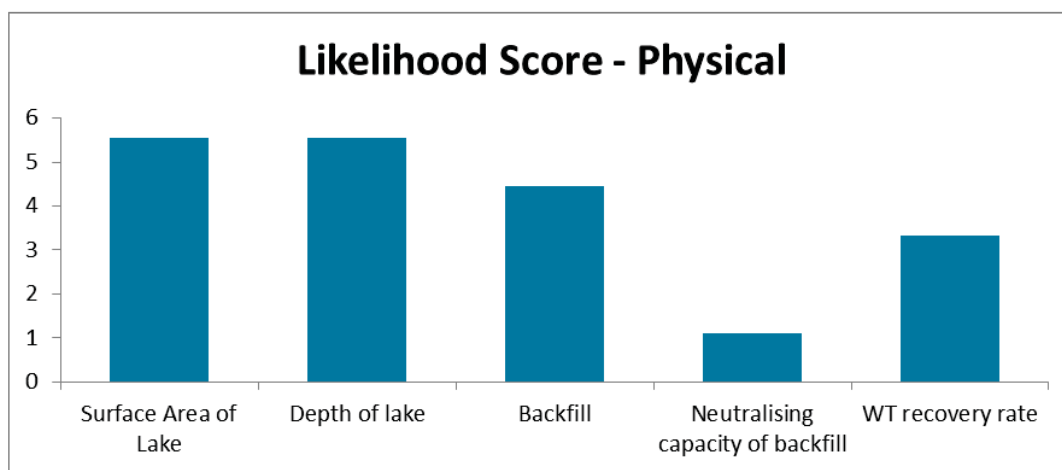


Fig. 2. Maximum potential contribution of physical variables to the likelihood of forming a contaminating pit lake.

3.2 Section B: Surface Water Risks

The risk evaluation in the Surface Water section is dominated by the effects of reactive material exposures on the final closure pit design. Surface water run-off over reactive rock exposed on the pit walls or accumulating as talus could contribute acid and metal loading to the pit lake. Three categories of reactive waste rock are identified as risks for Pilbara Iron Operations:

- Elevated Sulfur: material with sulfate minerals potentially capable of generating mildly acidic effluent when present in bulk with no neutralising material.
- Category S: Sulfidic material that is unlikely to pose a spontaneous combustion risk however may present a risk of acidic and/or metalliferous drainage (AMD). Waste typically has low sulfur concentrations and commonly less than 0.3%
- Category SR: Sulfidic material that has a risk for both spontaneous combustion and AMD.

Catchment properties and the associated risks are also captured in this section in a simplified manner. The geochemical, geotechnical and hydrological effects of whether a pit is isolated or intercepts creek flow are more complex than can be assessed by the AMPL tool.

The contribution of individual surface water parameters to the 80% weighted score is provided in Figure 3

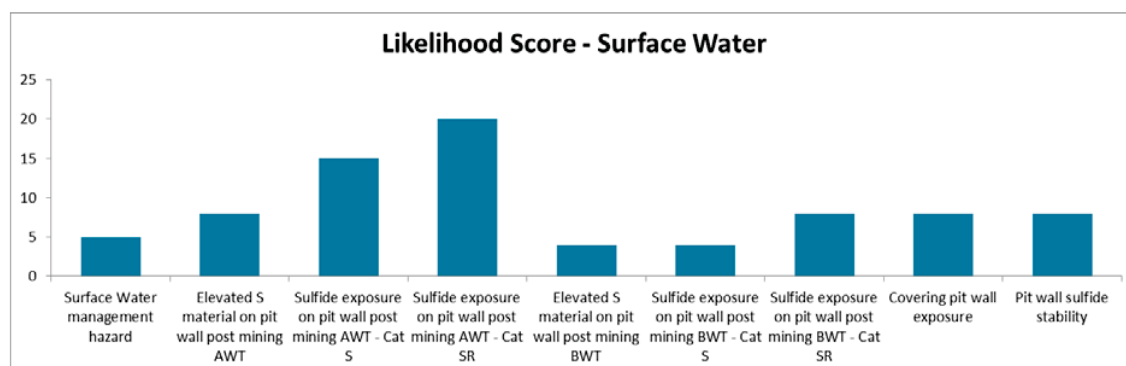


Fig. 3. Maximum potential contribution of individual surface water parameters to the 80% weighted likelihood score.

3.3 Section C: Groundwater Contamination

The potential contamination caused by pit lake development and contributions from groundwater to acid and metal loading within the pit lake are considered. Sulfidic material in the cone of depression created by dewatering the orebody, and the hydraulic conductivity of this material can have an effect on groundwater quality as the water table recovers. Quantifying this contribution is generally difficult due to lack of waste material drilling in the areas surrounding the pit, therefore a risk-based estimation is considered suitable in preliminary assessments. Groundwater alkalinity is also included in this section. Groundwater with high alkalinity can contribute to increasing the pH of a pit lake or buffering potential acidic drainage flowing from the lake. The contribution of groundwater parameters to the 80% weighted groundwater/surface water score is provided in Figure 4.

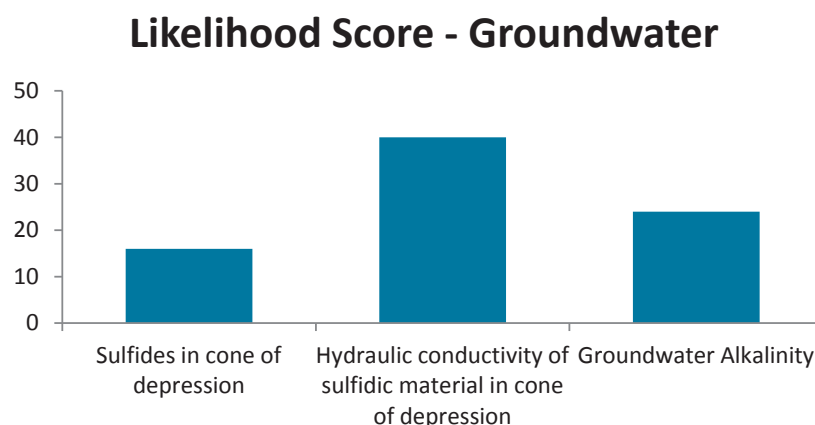


Fig. 4. Maximum potential contribution of individual groundwater parameters to the 80% weighted likelihood score.

3.4 Section E: Constituents of Concern

Reaction products resulting from the weathering of inert or sulfidic waste rock have different properties and mobility and could be classified as Constituents of Concern in a pit lake at sufficient concentrations. If a release mechanism is available, metals of concern such as Se and As will generally remain mobile at circum-neutral pH, however metals resulting primarily from sulfide oxidation (Fe, Cu, Zn, etc.) will be more mobile at lower pH. Determination of which metals require additional investigation is based on static testing of the lithologies expected to be exposed at a given deposit. Ideally, site specific testwork would be conducted, however if that is not available, testwork from a comparable geologic setting would be acceptable as an early substitute. Elements are considered to require additional investigation if > 50% of samples from 2:1 DI water leach extract testing exceed assessment triggers developed by RTIO. This section is not intended to provide a prediction of pit lake water quality. It is instead meant to highlight elements that may be of concern based upon the pit-specific geological setting. This thereby provides information on the elements that require regular monitoring within surface and groundwater at the site.

3.5 Overall Evaluation

To determine an overall prediction of water quality for a given pit void, the risk for each section described above is weighted by a percentage related to its significance and combined to provide a total risk score. Five overall assessment results are possible: Rare, Unlikely, Possible, Likely, and Almost Certain. Pit lakes deemed to be Almost Certain or Likely to form acidic or metalliferous water quality should be assessed for potential design changes such as limiting exposures of reactive material. Pits evaluated as Possible to form a contaminating lake should be prioritised for detailed numerical modelling early on in the mine life. This will allow for early management decisions to be made and implemented. Voids identified as rare or unlikely to become contaminating would not undergo detailed modelling to the extent of a possibly acidic pit lake.

4.0 AMPL RISK ASSESSMENT TOOL UNCERTAINTIES AND SENSITIVITY ANALYSIS

The AMPL risk assessment tool has been designed to be Pilbara-specific and evaluates parameters that are relevant to RTIO Pilbara operations. Due to the geologic and climatic environment of the region, the use of this tool to evaluate other deposit types would likely result in over- or underestimation of the associated risk and require site-specific modifications. For example, the presence of underground workings or the potential for lake freezing have not been accounted for in this tool and would need to be added to evaluate deposits for which they are applicable. The AMPL tool also cannot account for the potential acid mitigating effects of neutralising material present on the final pit design.

A sensitivity analysis was conducted using base-case example pit parameters in the AMPL tool to determine which factors had the greatest effect on the overall rating. The presence of category S and SR material above the post-mining water table has a significant effect on the risk. This material AWT has the potential to contribute significant acidity to the pit lake as it oxidises. Acidity and metals can be transported via surface water runoff and rainfall events.

The most important parameters that can affect the overall risk score are the options that can be managed throughout the mine life. When AMPL is applied early in the mining process, it can be used to show the benefit of pit designs that limit reactive pit wall exposures and are optimised for closure. During operations, prioritisation of backfill and segregation of optimal backfill material can be implemented in plan and lower the overall cost of closure during

decommissioning. The assessment of which management options are appropriate for a given deposit is one of the greatest benefits of using the AMPL tool.

4.1 Optimisation of Management Options

There are five parameters within the AMPL tool that can be modified to show the change in pit lake water quality that could be affected if risk management were incorporated into the life of mine planning process. These parameters are: Backfill Construction, Backfill Composition, Neutralising Capacity in Backfill, Time for Pit Lake to Reach Recovery WT, and Covering Pit Wall Exposures. The first four categories fall under Section A: Physical, and the last under Section B: Surface Water.

Pit backfilling can significantly decrease or eliminate the risk related to pit lake formation; however backfilling after production has finished is a very costly task. It is important to prioritise which pits critically require backfill and avoid over-committing to expensive backfilling operations where it may not be necessary. Assessment of the effect of full or partial backfill on the potential pit lake water quality using the AMPL tool can assist in determining these priority areas. The effects of backfill composition can also be evaluated to determine whether waste with additional neutralising capacity may be required to offset minor acid generation.

The fourth management option available is related to the length of time for water table recovery. Slow water table recovery in a pit lake results in increased oxidation of any potential sulfidic exposures on the final pit design. Acidity and metals are then progressively washed into the pit lake as the water level increases. If sulfides are rapidly covered in water, oxidation is limited and the metal and acid load can be decreased. In many operations, surplus water may be available from dewatering and can be directed into a completed pit, rapidly covering pit wall exposures. Covering pit wall exposures can also be accomplished using waste rock or potentially shotcrete. This option can be evaluated with AMPL, but given the uncertainty in its effectiveness, the influence of these parameters on the overall score is decreased.

5.0 CASE STUDY

A case study was developed to demonstrate the applicability of the AMPL tool to identification of appropriate closure options for a hypothetical “Deposit X”. “Deposit X” is assumed to have moderate exposures of pyritic black shale expected on the final pit design. Due to the sulfur content in this rock, acidic and metalliferous water would be expected if it were to remain a pit lake upon closure. “Deposit X” has been evaluated using the AMPL tool under various closure scenarios. In a “do-nothing” scenario, the pit lake that would form is expected to be acidic as it generates a “Likely” overall score (Figure 6). Several elements would have the potential to be elevated in the pit lake based on geochemical leach testing of the black shale lithologies as shown in the Constituents of Concern heat map (Figure 5).

Partial backfill of the pit to cover reactive exposures was considered as an alternative closure scenario. The AMPL assessment of the partial backfill closure scenario results in a decreased overall risk evaluation of Unlikely to form a contaminating pit lake (Figure 7). In this backfill scenario, black shale exposure would be covered by backfill with some neutralising potential. The resulting pit lake expected to form would be shallower with neutral pH and good water quality. A comparison of the results for both scenarios is given in Figure 8.

Analyte	Elevated sulfur (>0.1%)			Low sulfur (<0.1%)		
	n=	# > Trigger	% > Trigger	n=	# > Trigger	% > Trigger
TDS	0	0	0%	0	0	0%
SO ₄	20	4	20%	154	0	0%
HCO ₃	0	0	0%	0	0	0%
Al	20	17	85%	155	47	30%
As	20	18	90%	155	5	3%
Ba	20	0	0%	155	0	0%
B	20	12	60%	154	4	3%
Cd	20	11	55%	155	3	2%
Cl	10	0	0%	151	3	2%
Co	20	2	10%	155	0	0%
Cr	20	13	65%	155	3	2%
Cu	20	5	25%	155	0	0%
F	10	0	0%	134	6	4%
Fe	20	16	80%	155	37	24%
Hg	19	0	0%	155	1	1%
Mn	20	0	0%	155	7	5%
Mo	20	10	50%	155	2	1%
Ni	20	17	85%	155	29	19%
Pb	20	11	55%	155	6	4%
Sb	20	10	50%	155	3	2%
Se	20	18	90%	155	7	5%
Zn	20	0	0%	155	0	0%
NO ₃	20	0	0%	155	0	0%

Fig. 5. Example for “Deposit X” of a heat map of elements that were elevated above RTIO trigger values in 2:1 liquid extract tests of elevated and low sulfur waste rock.

Assessment Date Compiled by	Deposit X - "do nothing" scenario 10/12/2013 Stacey Hannam	Version Date: 18/02/2013 Version Number: 1
Likelihood of AMD Lake Development	LIKELY	Uncertainty Low

RTIO Acid and Metalliferous Pit Lake Risk Assessment
Preliminary Assessment

A. Physical

Flow mechanism	Select Relevant Option Below Sink
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	Select Relevant Option Below	Likelihood Score	Uncertainty
Surface Area of Lake	400, 000 - 600, 000 m2	3	Low
Depth of lake	60-80m	2	Low
Relative Depth	~7-11%		
Backfill	No backfill		Low
Backfill Material	No backfill	0.5	Low
Neutralising Capacity in Backfill	No backfill	1	Low
Time for Pit Lake to Reach Recovery WT	Fast water recovery (10-50 years)	1	Low

B. Surface Water

	Select Relevant Option Below	Likelihood Score	Uncertainty
Surface Water Management Hazard	Isolated pit	1	Moderate
Sulfur exposure on pit wall above post mining water table (AWT)	Elevated S (Alunite) Category S (Category 2) Category SR (Category 3)	<1% <1% <1%	0.1 0.1 0.1
Sulfur exposure on pit wall below post mining water table (BWT)	Elevated S (Alunite) Category S (Category 2) Category SR (Category 3)	<1% 3-10% 3-10%	0.1 2.4 6.4
Covering pit wall exposure	No cover	8	Low
Pit wall sulfide stability	Medium stability	4	Low

Surface water percentage contribution	70%	Low
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C. Groundwater Contamination

	Select Relevant Option Below	Likelihood Score	Uncertainty
Sulfides in Drawdown Cone	1-3%	4	Low
Hydraulic conductivity of sulfidic material in cone of depression	Medium (1-10 m/day)	24	Low
Groundwater Alkalinity	100-300 mg/L CaCO3	12	Low

Groundwater percentage contribution	30%	Low
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Environmental, social or heritage concerns that would immediately prevent approval if pit lake forms?	No
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Overall Likelihood of Acidic Lake Developing

Score **35.9**

Likelihood of Acidic Lake Developing **LIKELY**

Likely pH range **pH 4.0-4.3**

Concentrations requiring further investigation
Al, As, B, Cd, Cr, Fe, Ni, Pb, Se,

Fig. 6. "Deposit X" pit evaluation by AMPL under a "do-nothing" closure scenario.

Assessment Date Compiled by	Deposit X - Partial backfill scenario 10/12/2013 Stacey Hannam	Version Date: 18/02/2013 Version Number: 1
Likelihood of AMD Lake Development	UNLIKELY	Uncertainty Low

RTIO Acid and Metalliferous Pit Lake Risk Assessment Preliminary Assessment

A. Physical

	Select Relevant Option Below		
Flow mechanism	Sink		
	Select Relevant Option Below	Likelihood Score	Uncertainty
Surface Area of Lake	400, 000 - 600, 000 m2	3	Low
Depth of lake	20-60m	3	Low
Relative Depth	~2-8%		
Backfill	Backfill BWT		Low
Backfill Material	Inert backfill	0.2	Low
Neutralising Capacity in Backfill	NPR > 3	0.2	Low
Time for Pit Lake to Reach Recovery WT	Fast water recovery (10-50 years)	1	Low

B. Surface Water

	Select Relevant Option Below	Likelihood Score	Uncertainty
Surface Water Management Hazard	Isolated pit	1	Moderate
Sulfur exposure on pit wall above post mining water table (AWT)	Elevated S (Alunite) Category S (Category 2) Category SR (Category 3)	<1% 0.1 0.1	Low Moderate Moderate
Sulfur exposure on pit wall below post mining water table (BWT)	Elevated S (Alunite) Category S (Category 2) Category SR (Category 3)	<1% 0.1 0.1	Low Low Low
Covering pit wall exposure	N/A	0.1	Low
Pit wall sulfide stability	Medium stability	4	Low
Surface water percentage contribution	70%		Low

C. Groundwater Contamination

	Select Relevant Option Below	Likelihood Score	Uncertainty
Sulfides in Drawdown Cone	1-3%	4	Low
Hydraulic conductivity of sulfidic material in cone of depression	Medium (1-10 m/day)	24	Low
Groundwater Alkalinity	100-300 mg/L CaCO3	12	Low
Groundwater percentage contribution	30%		Low

Environmental, social or heritage concerns that would immediately prevent approval if pit lake forms?	No
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Overall Likelihood of Acidic Lake Developing

Score	24.2
Likelihood of Acidic Lake Developing	UNLIKELY
Likely pH range	pH 6.5-7.5
Concentrations requiring further investigation	

Fig. 7. “Deposit X” pit evaluation by AMPL under a partial backfill closure scenario.

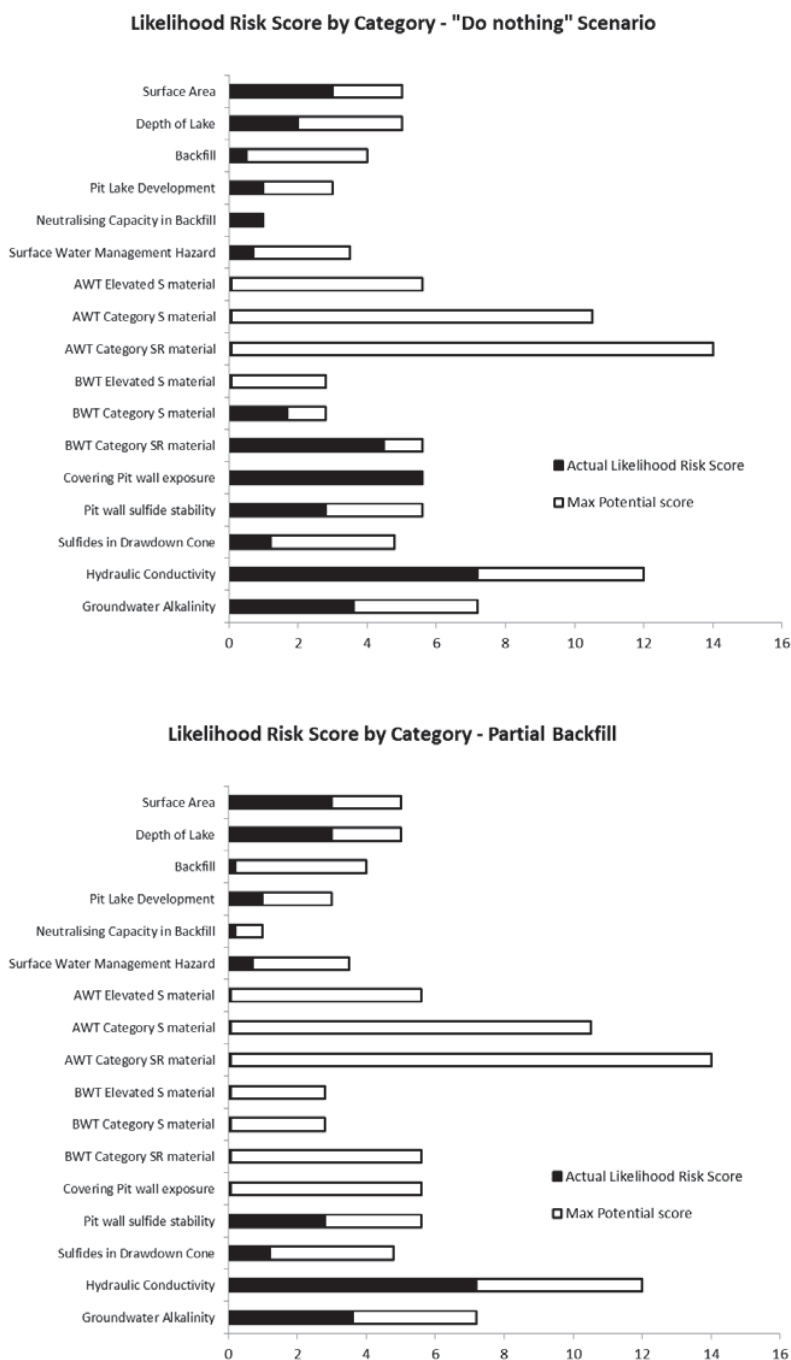


Fig. 8. Comparison of actual and potential scores for each parameter evaluated by AMPL in both the “do nothing” and partial backfill scenarios of “Deposit X”.

6.0 CONCLUSIONS

The AMPL tool was designed to be a simple and transparent risk-based method to standardise the evaluation of potential pit lake water quality across RTIO Pilbara operations. The tool can be used pre-mining to assist in prioritising pits for detailed geochemical modelling, and during operations to confirm and direct potential closure strategies. The AMPL tool can be used throughout the mine life to identify where gaps may be present in pit lake knowledge and direct data collection and studies. The transparency of the risk scaling system provides a distinct advantage over detailed geochemical models in the early stages of mine life. The AMPL tool provides a conceptual model of a given pit lake which is an essential step prior to embarking on more detailed modelling.

7.0 ACRONYMS

AMD	Acid and Metalliferous Drainage
AMPL	Acid and Metalliferous Pit Lake
AWT	Above Water Table
BWT	Below Water Table
RTIO	Rio Tinto Iron Ore

8.0 ACKNOWLEDGEMENTS

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PASSIVE TREATMENT OF ANTIMONY IN NEW ZEALAND, BY TWO METHODS: FORMATION OF STIBNITE IN A SULPHATE-REDUCING BIOREACTOR AND ADSORPTION ONTO AMD PRECIPITATES

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ABSTRACT

Two different passive water treatment methods to remove antimony from mine water were tested in field trials at a gold mine in New Zealand. In one method, antimony was removed through adsorption onto iron oxyhydroxides. The oxyhydroxides were collected from precipitate within an acid mine drainage discharge at a nearby coal mine. In the other treatment method, antimony was removed through precipitation as antimony sulphide (stibnite) in a sulphate reducing bioreactor. Each system consisted of a 1000 litre plastic tub, filled with the treatment media. The loading ratio of iron oxyhydroxide in the adsorption chamber was 253 grams of precipitate per litre of water. The bioreactor contained equal parts of post peel, bark and compost. Inlet antimony concentrations ranged from 0.3 to 2.3 mg L⁻¹. Hydraulic residence times ranged from three minutes to 14 hours (adsorption chamber) and two hours to 30 hours (bioreactor). Removal rates in the adsorption chamber were 14% at residence times of only three minutes and were up to 95% at residence times greater than 2.5 hours. As the experiment progressed, removal rates by the adsorption chamber decreased over time, suggesting that adsorption capacity was being approached. Removal rates in the bioreactor were greater than 80% at residence times above 15 hours. The greatest removal rate was 98% at a residence time of 30 hours. Quantitative analysis using a scanning electron microscope confirmed the presence of stibnite in the bioreactor. Bioreactor performance continued to improve over time when compared to the adsorption chamber. This difference is possibly due to multiplication of sulphate-reducing bacteria. These results show that passive treatment can be a successful treatment method for antimony at gold mine sites, and that treatment through sulphate reduction and precipitation as stibnite is a favourable long-term treatment method over a treatment method using adsorption onto iron oxyhydroxides.

1.0 INTRODUCTION

The metalloid antimony (Sb) is often associated with gold mines, typically in the form of the sulphide mineral stibnite (Sb₂S₃) (Nesbitt et al. 1989; Williams-Jones and Normand 1997). Removal techniques for Sb from solution are similar to those for arsenic (As) and include coagulation with ferric chloride, ion exchange, membrane filtration and adsorption onto ferric hydroxide (Kang 2003; Cumming et al. 2007; Klimko et al. 2008; Nokes 2008). Other workers have documented potential removal of Sb through sulphate reduction and formation of stibnite. Biswas et al. (2009) conducted an experiment in a test tube to show that Sb can be removed as a sulphide in the presence of sodium sulphate. Wilson and Webster-Brown (2009) suspected that precipitation of stibnite could be occurring naturally at the bottom of Lake Ohakuri along the Waikato River where concentrations of hydrogen sulphide and Sb approach stibnite saturation conditions.

Trumm and Rait (2011) conducted a laboratory column leaching experiment to test removal of Sb from mine water in New Zealand by two different passive treatment methods. In one

method, Sb was removed through adsorption onto iron oxyhydroxides. The oxyhydroxides were collected at two different coal mines where precipitate was forming within acid mine drainage streams. In the other treatment method, Sb was removed through sulphate reduction and precipitation as stibnite in a sulphate-reducing bioreactor. Both techniques were effective in reducing the concentration of Sb in the mine water and X-ray powder diffraction analysis of the bioreactor substrate showed that stibnite had formed.

In this work, we extend the results of Trumm and Rait (2011) by installing small-scale passive treatment systems at the Oceana Gold Globe Progress Gold Mine near Reefton on the West Coast of New Zealand to field test removal of Sb by two different techniques:

- (1) Adsorption onto AMD precipitates in an adsorption reaction chamber, and
- (2) Precipitation and formation of stibnite in a sulphate-reducing bioreactor.

2.0 METHODS

Two passive treatment systems were constructed using 1000 L plastic tubs (standard intermediate bulk containers) with PVC piping, alkathene piping and associated valves. The adsorption chamber was filled with a mixture of gravel and AMD precipitate from a nearby abandoned coal mine, composed primarily of goethite and ferrihydrite (Rait et al. 2010). The loading ratio of iron oxide in the adsorption chamber was 253 g precipitate per L of water. The bioreactor was filled with equal proportions of post peel, bark chips and compost. Each system was constructed in an upflow configuration, such that the inlet was at the base of each system and the water flowed upwards through the treatment media driven by sufficient hydraulic head (Figures 1 and 2).

The systems operated for a period of 71 d. Flow rates were varied to determine Sb removal for different hydraulic residence times (HRTs). Inlet and outlet samples were collected approximately three times per week from each system, or more often for very short HRTs, and analysed for dissolved As, dissolved Sb and sulphate by inductively coupled plasma mass spectrometry using APHA method 3125-B (for metalloids) and by ion chromatography using APHA method 4500-PG (for sulphate) (APHA 2005). Solid samples were collected from the top 10 cm of the bioreactor for mineralogical analysis by scanning electron microscopy (SEM) and energy dispersive X-ray spectrometer (EDS).

3.0 RESULTS AND DISCUSSION

3.1 Adsorption Chamber

Inlet concentrations to the adsorption chamber ranged from 0.3 to 2.3 mg L⁻¹. The system removed as much as 95% of the inlet Sb concentrations (to as low as 0.09 mg L⁻¹) (Figure 3). The greatest percentage removal was noted at the beginning of the experiment, with removal effectiveness decreasing over time. A drop in treatment effectiveness with time likely indicates that adsorption capacity of the treatment media was being approached. Since HRT varied, it is possible to analyse the effectiveness of similar HRTs over the duration of the experiment. In general, the effectiveness of each HRT dropped with time (Figure 4). This suggests that if a consistent removal rate is required, HRT must be increased over time.



Fig. 1. Adsorption chamber prior to being filled with treatment medium



Fig. 2. Adsorption chamber (on left) and bioreactor (on right) prior to being filled with water

At the very short HRT of 3 min, Sb removal in the adsorption chamber was 14%, lowering Sb concentrations from 1.4 to 1.2 mg L⁻¹ (Figure 5). Removal increased to 95% at an HRT of 2.4 h (lowering Sb concentrations from 1.9 to 0.09 mg L⁻¹). For HRTs longer than 10 h, however, Sb removal was only 65 to 75%. This is likely because Sb removal at these low flow rates were tested at the end of the experiment, after much of the adsorption capacity of the material had been exhausted and removal efficiency had declined.

3.2 Bioreactor

Antimony removal in the bioreactor was as low as 29% near the beginning of the experiment, increasing to 98% at the end of the experiment (Figure 6). Unlike the adsorption process where removal was initially good and decreased with time, Sb removal in the bioreactor was lower initially and improved with time, possibly due to expansion of the sulphate-reducing bacteria population.

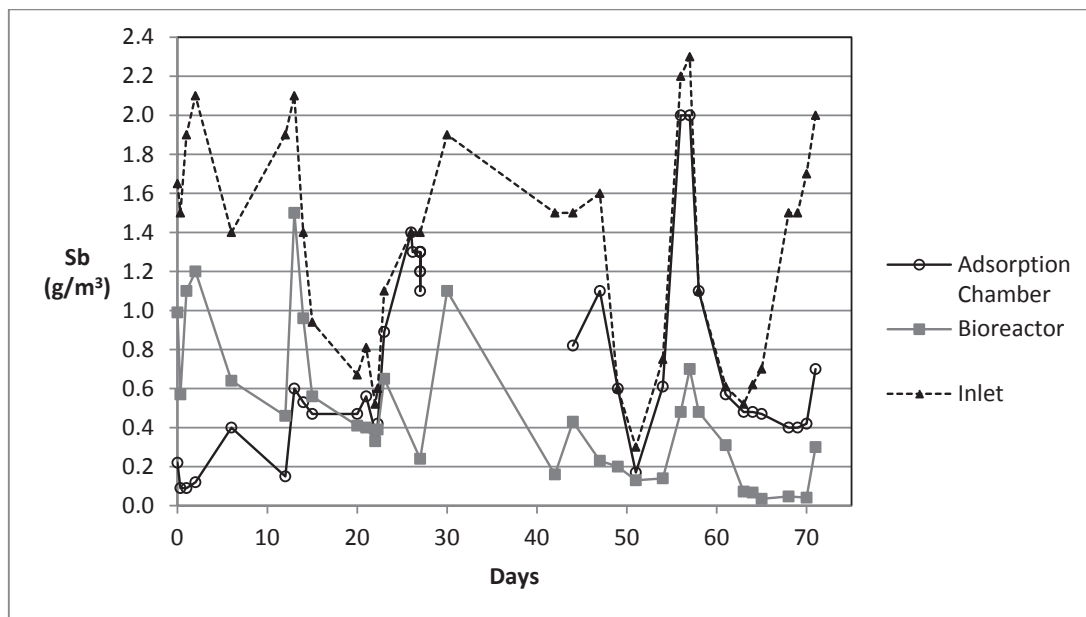


Fig. 3. Inlet and outlet Sb concentrations for adsorption chamber and bioreactor over the duration of the field trials

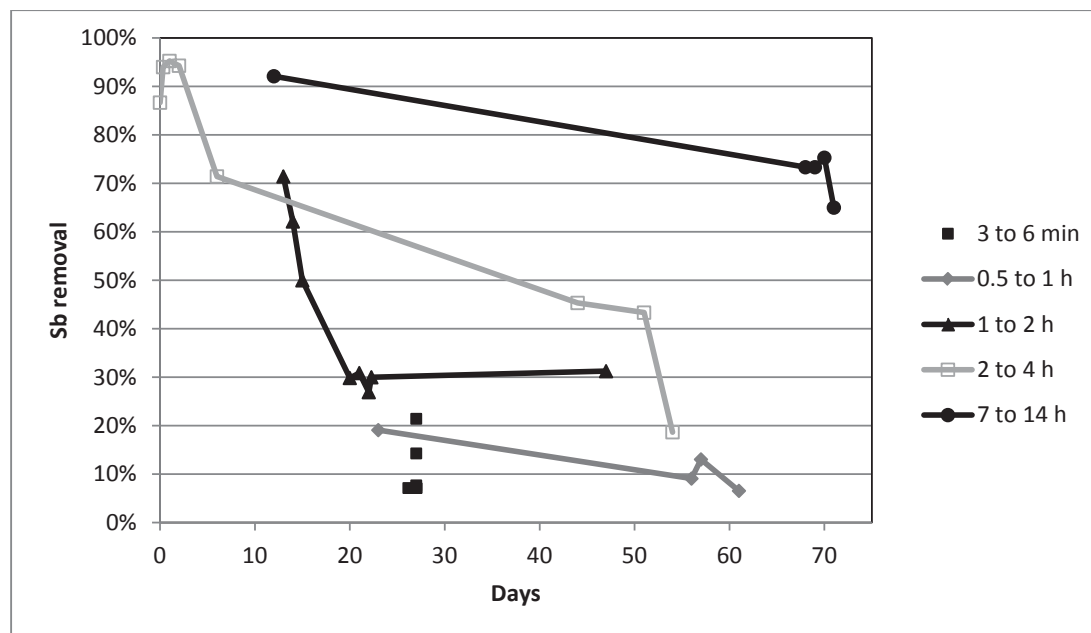


Fig. 4. Antimony removal by adsorption chamber over the duration of the field trials grouped according to hydraulic residence time in the system to show trends

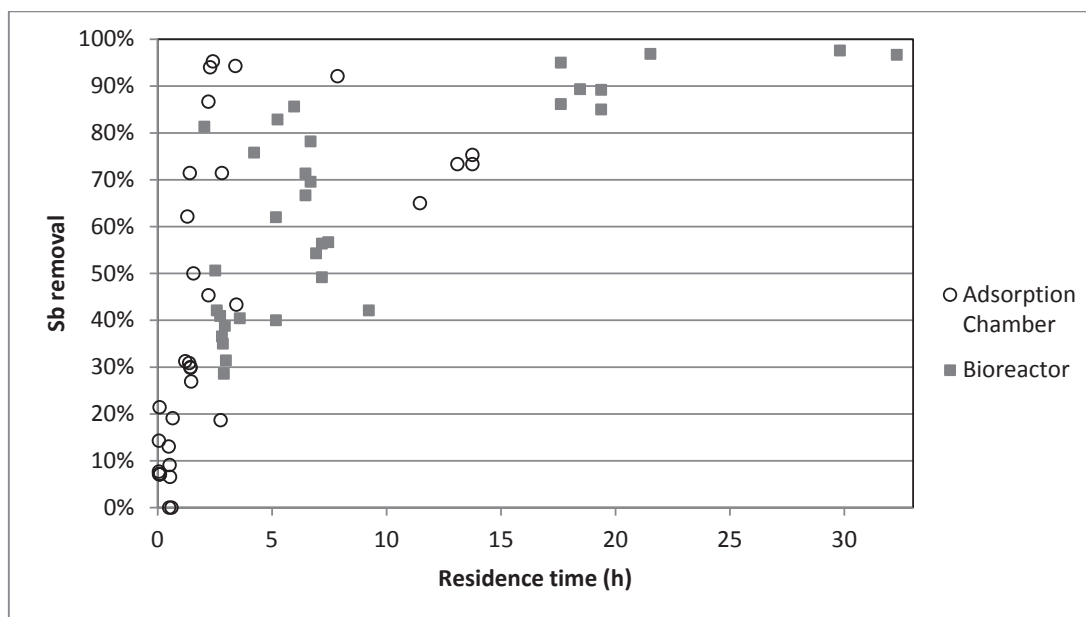


Fig. 5. Antimony removal by adsorption chamber and bioreactor according to hydraulic residence time in the systems

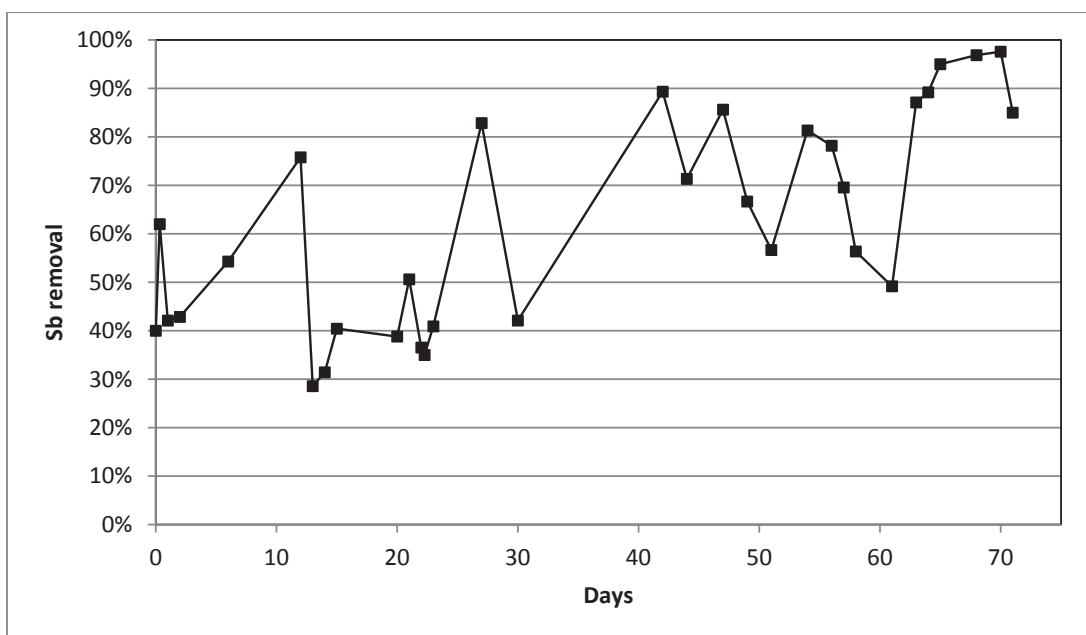


Fig. 6. Antimony removal by bioreactor over the duration of the field trials

At HRTs below about 5 h, Sb removal was about 40%, and at HRTs between 20 and 30 h, Sb removal was up to 98% (Figure 5). Antimony removal rates ranged from $0.002 \text{ moles m}^{-3} \text{ d}^{-1}$ to $0.034 \text{ moles m}^{-3} \text{ d}^{-1}$ and are correlated with HRT. Solid samples from the bioreactor

analysed by SEM with EDS confirmed the presence of stibnite, suggesting successful sulphate reduction.

Removal of Sb with a sulphate-reducing bioreactor may be a better long-term treatment technique than removal through adsorption onto iron oxyhydroxide precipitate. Preliminary work here shows that for similar sized systems, a bioreactor will outperform an adsorption chamber over time. Additional studies are underway to better determine long-term performance for both systems.

4.0 CONCLUSIONS

These results show that Sb dissolved in mine water can be removed through adsorption onto ferric hydroxide precipitates which are sourced from coal mine AMD. This study also shows that Sb can be removed through sulphate reduction in a sulphate-reducing bioreactor. For treatment by adsorption, Sb concentrations can be lowered by 14% at HRTs of only 3 min and by 95% at HRTs above 2.5 h. Performance declines in an adsorption chamber as the adsorption capacity of the treatment media is approached, therefore, HRT must be extended over time to compensate.

In a bioreactor, treatment of Sb occurs through sulphate reduction and formation of Stibnite. Antimony concentrations can be lowered by 81% at HRTs of only 2 h and by 98% at HRTs above 20 h. Treatment performance increases with time as the microbial population expands and sulphate reduction improves. Therefore, HRTs should be longer initially and can be lowered as treatment performance improves.

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INSTRUMENTATION IN WASTE ROCK DUMPS: GOING DEEPER

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ABSTRACT

Instrumentation has been used for many years as part of the assessment and monitoring of cover systems on waste rock dumps, typically involving the use of instruments such as soil moisture probes placed to shallow depths to monitor factors such as net percolation rates. As a result a great deal of valuable information (practical field data) has been gathered from these monitoring programs regarding the performance of cover systems, which has greatly advanced the understanding of unsaturated zone hydrogeology within cover systems in general. However, to date the installation of instrumentation at greater depths within waste storage facilities is far more limited (with the notable exception of tailings dams). It can therefore be argued that our understanding of unsaturated zone hydrogeology and geochemistry within the waste rock mass is therefore (in comparison) biased more towards theory and standardised conceptual models rather than practical field data.

With the development of sonic drilling technology, there is however, considerable opportunities for placing instrumentation at depth within waste landforms. Instrumentation placed at depth can be used to provide the same kind of valuable field data as has been gathered for cover systems previously. In addition this instrumentation can provide field data for long term monitoring of waste dump hydrology, geochemistry, gas composition, temperature etc. which provides significant information for the prediction of AMD.

O'Kane Consultants Pty. Ltd. has completed the installation of over 90 instruments to depths of 40m within waste dump landforms as part of a long term monitoring and assessment program. Instruments were installed using the sonic drilling technique and include galvanic oxygen probes, soil matric potential sensors, temperature sensors, and vibrating wire piezometers. The use of sonic drilling allowed the structure of the dump to be assessed in detail during drilling which allowed the targeting of instrument placement in specific zones.

The gathering of data from the instrumentation allows the conceptual model of the hydrology and geochemical evolution of waste rock dumps to be both tested and refined and there are opportunities for future predictive modelling to be better calibrated. Data from instrumentation installed is presented herein which indicates that standard conceptual theories of dump hydrology such as "wetting fronts" may not be universally applicable.

1.0 INTRODUCTION

Waste rock dumps (WRD) are a consequence of most mining processes and usually consist of uneconomic low grade ore materials and non-mineralized "waste" materials as a result of resource extraction. Historically relatively little attention/effort has been afforded to the characterization of materials placed in WRDs, and to the construction process of forming them. As a result many significant environmental legacy issues have developed relating to WRDs ranging from structural failure (erosion) to acid and metalliferous drainage (AMD) discharges. As a result of these legacy issues having caused significant environmental/social impacts a large number of scientific studies have been completed to gain an understanding of mine waste material and related environmental risk factors resulting from placement in

WRDs. As part of this scientific research and development general conceptual models and theories have been developed to explain for example the causes of AMD from WRDs. Resulting from these generalised conceptual models and theories has been the development of legislative controls by government(s) as part of environmental management, and a better understanding in the industry of what “best practice” means with respect to waste management. This has seen a significant improvement in waste management practices in recent times. In general, engineering design and closure plans for WRDs generally address the concerns of long-term erosion stability as WRDs are considered increasingly as an “engineered landform”. In addition, the use of engineered “cover systems” for example has been widely adopted by the industry as a means to reduce net percolation rates into WRDs (thereby reducing the potential for AMD discharges). However, the success of “best practice” approaches has not been uniformly successful in reducing environmental liabilities caused by WRDs. Three common reasons for this ‘patchy’ success include:

- The development of “best practice” in industry for WRD management and therefore construction is based on the assumption that the underlying conceptual models/theories regarding WRD structure and stability, hydrogeology, geochemistry, gas regime etc are sound. However the understanding of unsaturated zone hydrogeology and geochemistry within the waste rock mass is generally biased towards theory, standardised conceptual models, and predictive models/lab testing rather than from practical field data. This has resulted in engineering solutions being designed based on generalised conceptual WRD models that in some instances are not appropriate.
- In contrast to other waste management industries such as domestic landfill, WRDs are generally not subject to active monitoring by way of internal instrumentation being installed (with the exception of cover systems). This “walk away” approach has resulted in the creation of potential environmental hazards as processes like AMD take many years to develop and often will go undetected for decades until a significant AMD discharge occurs without warning.
- In addition (and again in contrast to domestic landfills), construction and regulator approval of WRDs commonly does not include detailed engineering specifications, as built drawings and QA/QC documentation. This has resulted in many WRDs being constructed poorly and often with limited planning/thought having been made with respect to materials management and placement (for example the prevalence of end tipping).

The third factor listed above can be considered purely an engineering design issue and therefore will not be addressed as part of this study; however, the preceding two factors are addressed herein as these relate to scientific study.

The study presented herein is based on a sonic drilling program of works completed as part of an investigation of decades old WRDs. The sonic drilling allowed the installation of monitoring equipment (instrumentation) throughout the depth profile of a series of WRDs with the following objectives:

- To test conceptual ideas of WRD hydrogeology, structure, and gas flow in the vadose zone
- To provide a means for long term monitoring of the WRDs.

O’Kane Consultants Pty. Ltd. completed the installation of over 90 instruments to depths of up to 40m within waste dump landforms (from iron ore mining). Instruments were installed using the sonic drilling technique and include galvanic oxygen probes, soil matric potential

sensors, temperature sensors, and vibrating wire piezometers. The use of sonic drilling allowed the structure of the dump to be assessed in detail during drilling which allowed the targeting of instrument placement in specific zones. The installations were connected to a telemetry system so that automated and “real time” data collection could be achieved from the instruments.

2.0 INVESTIGATION OF AMD PRODUCING WRDS

A key data set that can be used to guide the development of a conceptual model of a WRD includes the investigation of WRDs that have produced AMD. Although not common, mining companies are now motivated to perform these detailed investigations of WRDs to gain an understanding of the internal hydrologic properties of the waste and evaluate how the *in-situ* waste material controls AMD production and potential release to the surrounding environment. By undertaking these investigations, an understanding of WRD hydrology such as recharge and underdrainage, and gas movement can be developed allowing to better understand the internal controls for AMD production.

A key feature of these investigations is that they allow the selection of material from within WRDs that has been *in-situ* for years, perhaps decades. Testing and characterisation of this material provides valuable data that can be used to retrospectively determine key processes operative in WRDs.

3.0 INSTRUMENTATION

Significant work in the mine closure field has been completed in the performance monitoring of engineered cover systems over mine waste (O’Kane 2011). This includes the installation of detailed instrumentation equipment such as soil moisture sensors, temperature sensors etc to monitor key processes such as:

- Net percolation
- Heat transfer
- Vadose zone gas composition and oxygen migration
- Salt uptake
- Soil water characteristic curves

With the advancement of sonic drilling technology the state of the art methodologies utilized at the near surface can be transferred deep into WRDs to evaluate:

- Internal hydrologic conditions and the response of the WRD to climatic variables such as incident precipitation and pressure
- Internal processes such as heat generated through sulfide oxidation
- Movement and replenishment of oxygen through dump structure
- Effect of localized features related to dump structure such as rubble zones, underlying historical drainage pathways etc

As well as looking at what “has happened” the installation of instrumentation within WRDs allows future monitoring of WRD internal conditions. Instrumentation provides monitoring data that can be both fed into models for predictions of future developments, but also provides “real time” data that may aid with analysis of any significant events as they happen (for example unexpected seepage events)

3.1 *In-Situ Moisture Conditions*

Dielectric water potential sensors (suction) MPS2 sensors and multi-level vibrating wire piezometers (VWP) were installed at various depths within the WRD profile to measure matric suction/temperature, and pore-water pressure/temperature at different zones in the borehole; respectively. The soil suction sensors have a measurement range of -10 to -500kPa and the VWPs have the capacity to measure a positive pressure of 350 kPa (3.5 Bar) and also slightly negative pressure conditions.

Soil suction sensors are used in this study to monitor recharge in the WRDs and the propagation of the wetting front (if present) in response to rainfall events. Given the presence of both fine grained and coarse material bands within WRD construction it is anticipated that the advancement of wetting fronts will be more pronounced within fine grained material. The locations selected for soil suction sensor placement will allow for a detailed evaluation of this hypothesis.

VWPs installed within the WRDs will allow the evaluation of basal flow within the coarse grained material commonly found at the base of WRDs due to segregation during end dumping. VWPs installed throughout the WRD profile will give an indication of the development of perched water tables and seepage.

3.2 *In-Situ Gas Concentrations*

ICT International ICT02 galvanic oxygen probes were installed within the boreholes to give an indication of oxygen levels within the WRDs. Given that oxidation is an oxygen consuming process, it is anticipated that a drop in oxygen levels will be observed following the advancement of wetting fronts. In addition, depressed oxygen levels may be observed within pyritic shale bands. Oxygen replenishment is thought to occur in coarse waste horizons therefore installation of probes in these zones at various depths will allow this theory to be tested. Manual gas sampling ports were installed at the surface to provide additional gas monitoring and to verify readings made by the automated system.

3.3 *In-Situ Temperature Monitoring*

Temperature probes were installed throughout the WRD profile at various depths. The temperature probes, in addition to the temperature readings provided by the VWP and MPS2 sensors will allow for a temperature profile to be developed with depth. The oxidation of sulphide materials is an exothermic process; therefore, it is anticipated that the introduction of meteoric water to the waste material will result in a temperature spike as the wetting front progresses through the WRD. Temperature gradients within the WRD will also be evaluated in terms of introducing thermal gradients resulting in convective gas movement.

4.0 STUDY SITE

The presented study is a mine site in the Pilbara, Western Australia. The site is made up of multiple WRDs which have been constructed mainly by end dumping.

The WRDs do not have cover systems installed, and some surfaces of the WRDs comprise barren coarse waste rock. A significant proportion of the site is covered by relatively dense mature vegetation that appears to have established directly on the coarse waste rock surfaces (natural revegetation of many areas of the site appears to have been very successful).

The Pilbara region consists of a climate classified as arid-tropical. Summers last from October to April and mild winters occur from May to September (Gentili 1972). Sporadic and intense thunderstorms are typical for the region from January to March, and tropical cyclones can result in daily rainfall amounts of up to 200mm over a 24hr period.

4.0 WRD STRUCTURE & AMD PRODUCTION / RELEASE

Generic conceptual models for WRDs have been published by many authors, and many guidance documents contain “industry standard” schematics (MEND, INAP etc). However, what may appear to be a useful concise summary “picture” is potentially dangerous if these are assumed to be based on complete and robust technical models that accurately represent the hydrogeology/geochemistry/gas regimes pertaining to WRDs on a specific site. Some authors have attempted to incorporate field data and scaled laboratory experiments to make these models more technically robust (notably the work of Ward Wilson, e.g. Wilson 2011), however this approach has yet to translate to the more generic “industry standard guidance”.

Figure 1 is a reproduction from Ritchie, 1994, which is included in the Australian guidance (DITR 2007), which clearly shows the limits of a generic model. For example the oxidized zone is shown as a simple halo around the edges of the waste, and infiltration is indicated to be vertically uniform. Whilst potentially applicable to homogeneous waste materials such as tailings this model is not likely to be reflective of waste rock dumps. As will be demonstrated in this paper broad assumptions like these can be misleading.

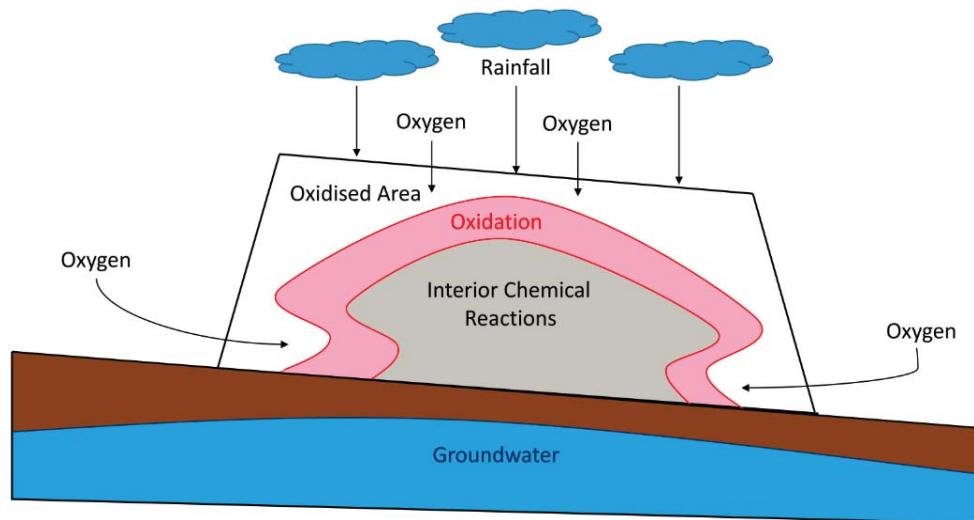


Fig. 1. Generic model of AMD production and contaminant migration from a waste rock dump after Ritchie 1994

An aspect of WRD construction that has been (and arguable still is) commonly overlooked is the internal structure created as a consequence of the prevalence of end tipping material, and the resulting hydrologic characteristics which control oxygen and water flow throughout the waste material (Wilson 2011). Given that oxygen and water flux are major controls in the production and release of AMD to the receiving environment, this is a major oversight in the

conceptual model of WRDs and in their overall closure plan. Figure 2 is a conceptual model showing a cross section of a typical WRD constructed by end dumping material. The segregation of coarse and fine grained material into parallel bands along tip faces is a common feature of end dumped WRDs. The segregation of material as shown in Figure 2 has been confirmed by WRD excavations in the work of Wilson (2011).

The conceptual cross section shows that the infiltration of water enters the WRD at the top of the pile, percolating down through areas of fine grained materials due to their ability to retain water and the low air entry value of coarse grained material. During heavy rain events such as cyclones, water will also enter the WRD in coarse grained sections quickly percolating to depth. Oxygen ingress primarily will enter the WRD at the bottom of the pile moving upwards through the free draining course material layers by the process of thermal advection.

The presented conceptual internal structure of WRDs constructed by the common practice of end dumping is an ideal scenario for the production of AMD given the ample supply of atmospheric oxygen and water.

The conceptual model shown in Figure 2 is an improvement over that shown in Figure 1 as some elements of structure are considered; however, some of the concepts are based on limited field data. Some areas of uncertainty include:

- How efficient the oxygen replenishment is within the dump
- The presence of a “wetting front” within the dump
- The dominance of fine or coarse grained materials with respect to seepage transfer to the base
- The influence of “underdrainage” where dumps are constructed on old drainage pathways or on natural slopes (e.g. escarpments)

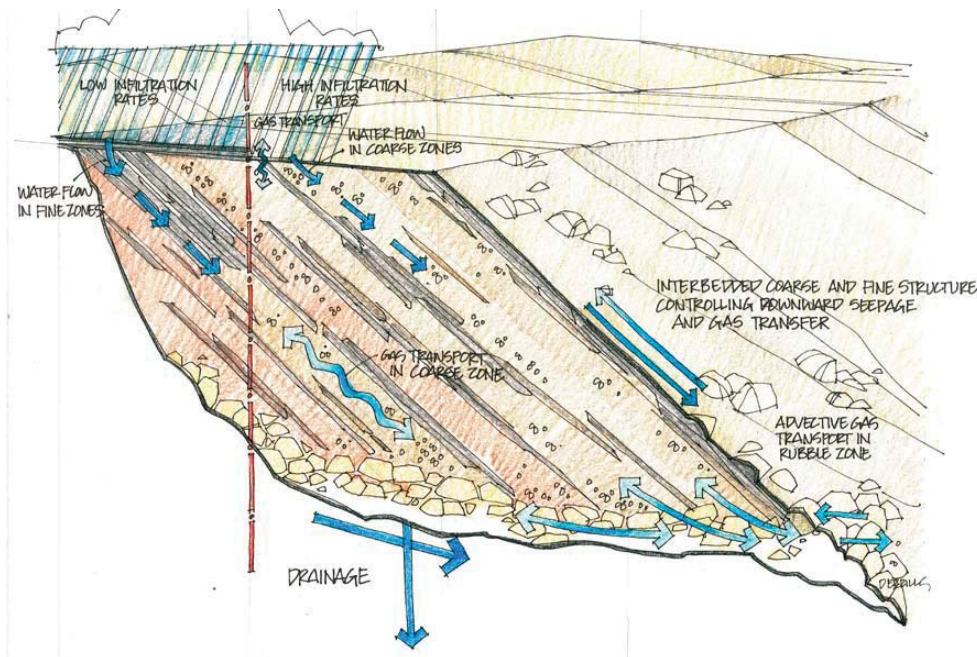


Fig. 2. Conceptual cross section of end dumped WRDs illustrating oxygen and water transport

5.0 SITE MONITORING SYSTEM

OKC completed the installation of over 90 instruments within the WRDs up to a depth of 40m. The site monitoring system includes 19 boreholes equipped with instrumentation to measure *in situ* moisture, pore-gas concentrations, pore-water pressure and *in situ* temperature within the waste rock piles. In addition to automated instrumentation, groundwater sampling wells and manual gas sampling ports were also installed at selected locations. A conceptual layout of the monitoring equipment is shown in Figure 3. Instrumentation was installed in conjunction with a sonic drilling program for material and site characterisation. Fully automated data acquisition systems (DAS) were installed and commissioned in 2013 and are equipped with remote communications to enable real time monitoring of internal WRD conditions. DASs were installed as “low profile” stations to limit the probability of lightning strike during cyclone events, and also to supply additional stability during high wind conditions.

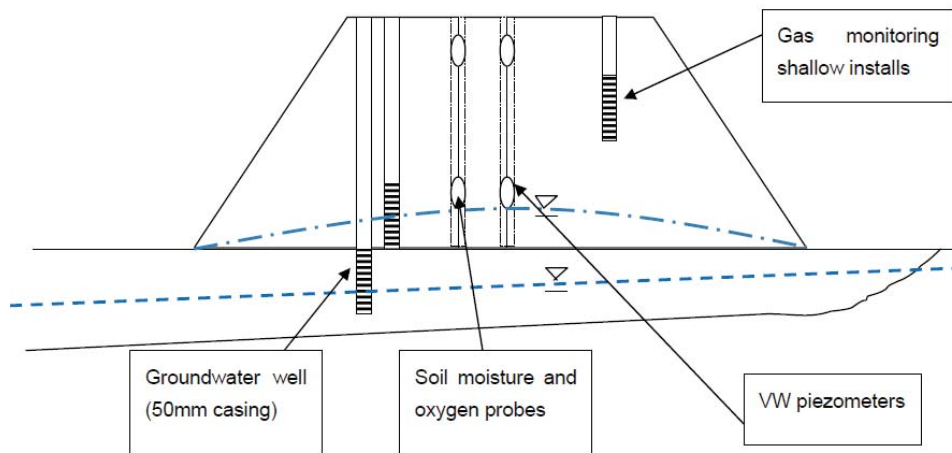


Fig. 3. Conceptual layout of monitoring equipment

5.1 Borehole Drilling and Sensor Installation

Sonic drilling was utilized to construct the boreholes onsite due to the ability to collect accurate, continuous, and relatively undisturbed core samples in unconsolidated material (Barrow 2007). Drill holes were comprised of 150mm diameter holes (outside casing diameter) and 100mm core with depths ranging from 5m to 42m. Because of the ability of sonic drilling to produce (relatively) undisturbed core samples, this allowed the internal structure of the WRDs to be assessed in detail and depths for sensor installation to be chosen based on observations in the field; targeting specific zones such as bands of coarse or fine grained material.

Figure 4 shows a typical profile through the dump based on data gathered from sonic drilling. The following observations are made with respect to this data:

- A coarsening of materials towards the base of the dump is noted
- Natural ground is identified by sharply higher fines content under the rubble zone (clay underlies the dump)

- Moisture contents are low through the profile and do not show an obvious wetting front

The data from the drilling indicates that the dumps are generally dry throughout the profile indicating that the much hypothesized “wetting front” has not developed even though the dumps have no cover and are decades old. However, seepage is noted at the base of the dumps therefore an infiltration mechanism must be active. Given this observation instrumentation was installed to test a theory that seepage relates to fast drainage through coarse layers rather than through “slow drainage” through finer materials as a vertically migrating wetting front.

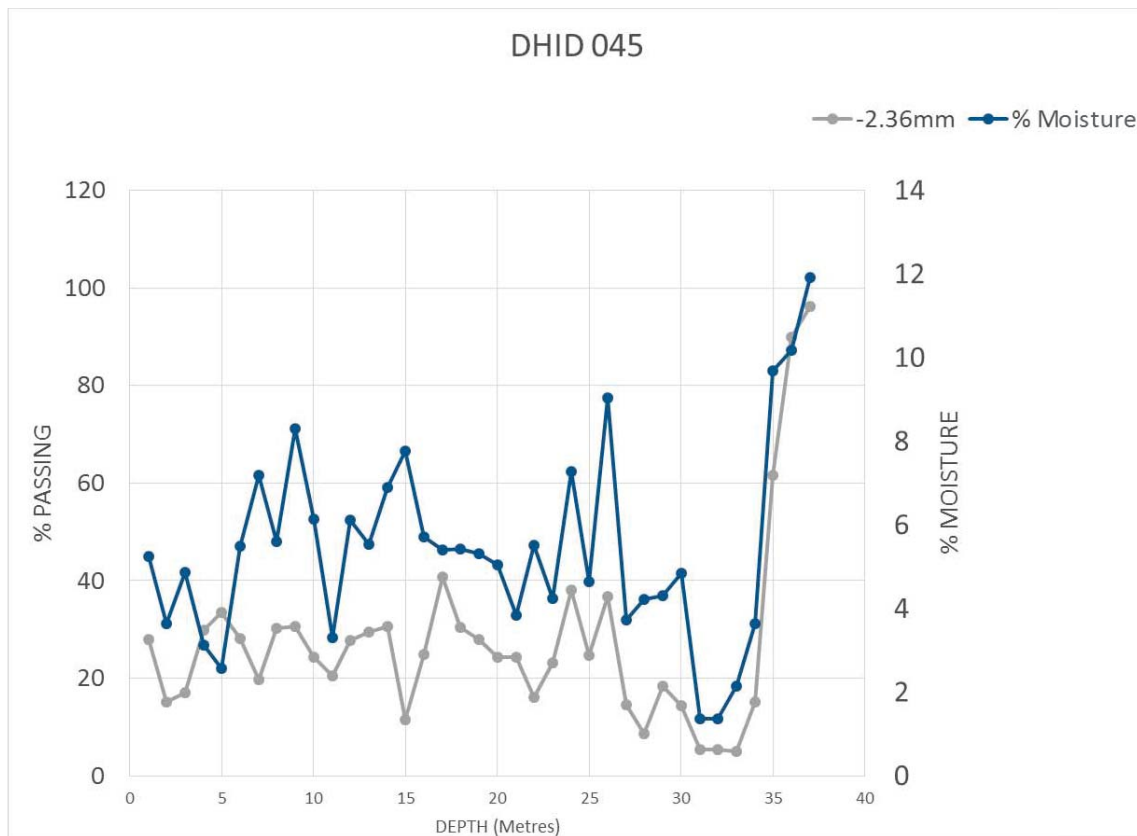








Fig. 4. Typical moisture content, grain size profile through waste dump

Core logging summarized in Table 1 (and PSD data shown in Figure 4) clearly shows the alternating coarse/fine grain horizons present in the dumps resulting from end tipping. In addition material type segregation is noted throughout the profile with bands of shale/BIF/dolerite noted, bands range from as little as 5cm to 6m in thickness. Table 1 shows the rationale for sensor installation at particular depths. In general sensors were installed in the following horizons:

- Shallow depths to monitor oxygen, temperature and moisture content conditions
- Mid depths to monitor oxygen, temperature and moisture content conditions (including pyritic shale bands)

Table 1. Typical profile through dump and rational for placement of sensors

Core photograph	Depth, material info	Placement of sensor
	1-2m Gravely Sand of mainly BIF	MPS2: 2m Sensor placed to determine shallow depth moisture conditions
	4-4.5m Clayey gravel with cobbles; black pyritic clay	O2/Temp: 4m Sensors placed to determine shallow oxygen concentrations and temperature in sulfide zone.
	8.5-9.5m Sandy gravel with cobbles. BIF and hematite	O2/Temp/MPS2: 9m Sensors placed to determine oxygen concentrations, temperature and moisture conditions in coarse material zones
	15.5-16.5m Gravely clay; BIF with grey shale	VWP/MPS2: 16m Sensors placed to determine pore pressure and moisture conditions changes in finer grained clayey zones
	35-36m Silty sandy gravel with cobbles; Hematite, BIF, black shale;	O2/VWP: 35m Sensors placed to determine oxygen flow and pore pressure in coarse waste near base of dump
	36.5-37.5m Silty clay with cobbles; Sandstone;	VWP: 36.5m Sensor placed to determine seepage through natural ground from base dump

Sensors were installed surrounded by a 1-2m thick pack of 1-2mm clean silica sand on the outside of the centralized 42mm PVC casing (used to route cables). Sensor installations were isolated with a bentonite pellet backfill. The top of the boreholes were sealed with a cement grout mixture. Details of a typical borehole installation can be seen in Figure 5

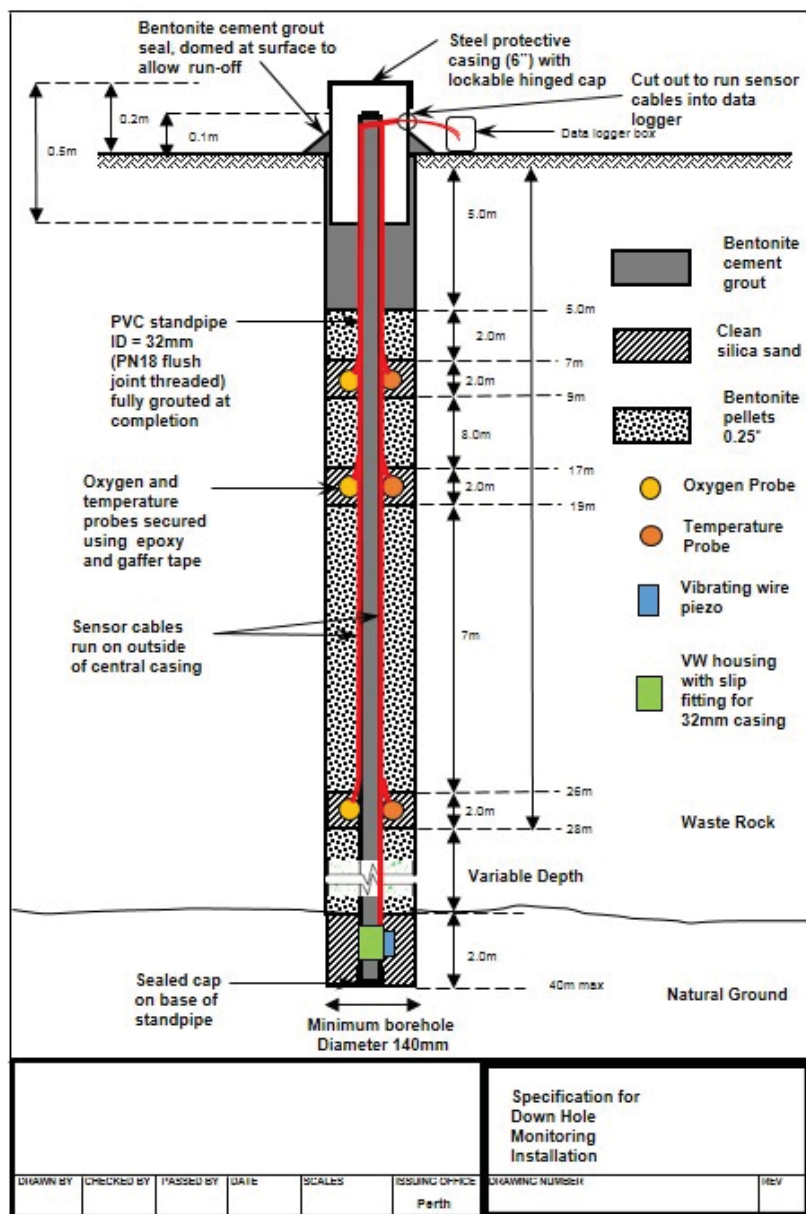


Fig. 5. Typical borehole installation

6.0 DATA ANALYSIS / INTERPRETATION

Preliminary data sets are presented in this section. Significant amounts of data are collected from the study site; however, at this time limited data is available for inclusion in this analysis. Selected data is presented as an initial review with long term monitoring planned for the future.

Figure 6 presents temperature data measured within the waste rock at a selected borehole location. Temperature readings are available from November 1st to November 19th. Temperature in the upper 2 m of the waste has increased fairly constantly over the 19 day monitoring period rising from 33.6°C to 34.8°C, while temperatures from 8 m – 25 m have remained relatively constant between 30 – 32 degrees Celsius. The data indicates the following:

- Temperatures are increasing at shallow depths, this is likely related to increase in ambient air temperatures due to move from spring to summer period
- Temperatures are lowest close to the base of the dump, this is not unexpected as temperature gradients are expected to be inversely proportional to depth given advective forces
- Temperatures are consistent through the middle of the dump and are relatively high. 31 degrees Celsius is a relatively high ambient temperature for the core of the dump and likely explains the low moisture contents given high potential for moisture loss due to evaporation. The consistent temperature profile with depth indicates a high level of convective heat flow through the dump which likely relates to high levels of interconnected advective gas flow

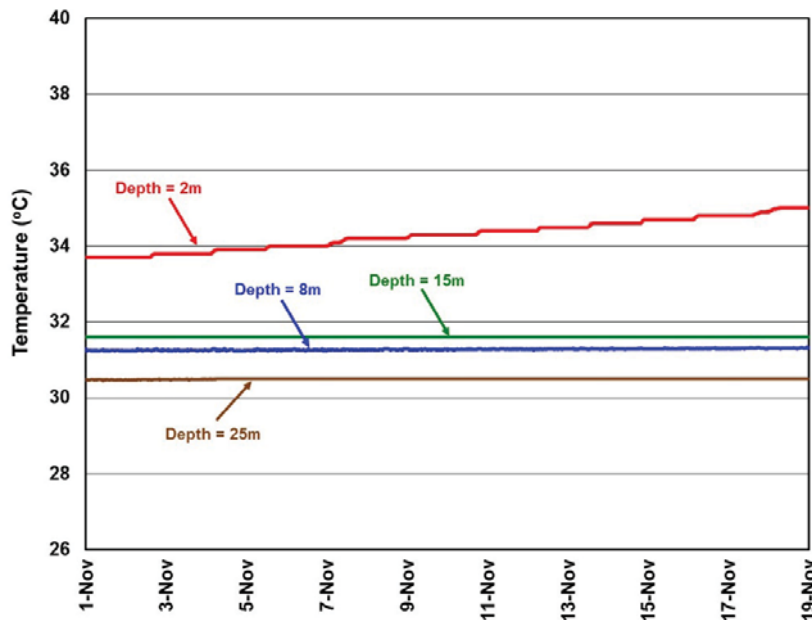


Fig. 6. Temperature profile within the waste rock at selected borehole location

Figure 7 shows the soil suction profile measured by MPS2 dielectric soil suction sensors at a selected borehole location. The reading of 100,000 kPa results from material which is air dry,

indicating that the sand pack which the sensors at 8 m and 15 m are installed in are completely dry. The reading of 12 kPa and 21 kPa for sensors at 2 m and 25 m, respectively, shows that these sensors have moisture present within their sand pack. It should be noted that the sand packs were installed dry so any water is likely a result of migration post installation.

Moisture present at 2 m is not unexpected given the influence of precipitation at the near surface. Moisture readings at 25 m however, gives an indication that water is present and is being replenished at the base of the waste rock pile. The fact that the MPS2 sensors are reading air dry in the middle of the pile would suggest that:

- Water has not percolated uniformly from the top of the pile to the base, but has reached the base through basal seepage or preferential flow paths.
- Supports the evidence from moisture content profiles through the dump that show a “wetting front” has not developed
- Supports the temperature data that indicate that high levels of moisture loss may be occurring due to evaporation as a result of advective gas flows through the core of the dump

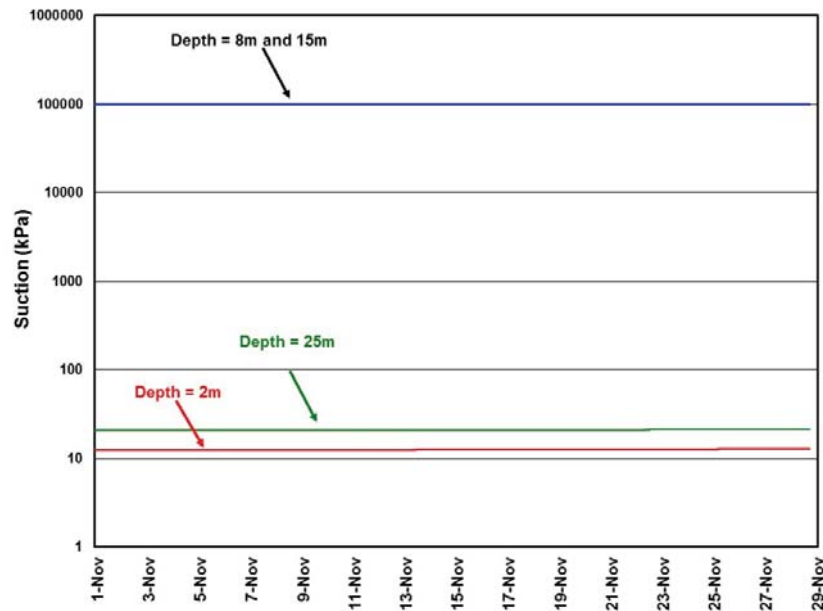


Fig. 7. MPS2 soil suction profile at selected borehole location

Figure 8 shows positive pore-water pressure measured by VWP sensors at a borehole in close proximity to the MPS2 data presented in Figure 7. Data was available at this borehole from October 18th to November 25th. The VWP data shows positive pressure (to a maximum of 10 KPA) conditions developing near the base of the WRD (sensor is installed in the rubble zone just above the contact with natural ground) at a depth of 27m. VWP sensors at 12m and 9m remain relatively unchanged during this time period (readings indicate unsaturated conditions as negative pore pressures are recorded); however, a slight increase in pressure can be seen on November 1st at 9m. The sudden drop in pressure at 27m on November 10th can be attributed to influences on the VWP during initial wetting. A decrease in pore-water pressure can be observed starting November 17th as the waste begins to de-saturate. In

total, the time from initial pressure increase (October 28th) to when decreases in pore-water pressure were observed is 20 days. This is notably a fairly rapid response of the base of the dump to a seepage event.

The MPS2 soil suction data in combination with the VWP pore-water pressure results show that wetting is not occurring uniformly with depth (i.e. wetting front is not developing). Rainfall data from a meteorological station in proximity to the study site reported 26mm of rainfall on October 17th likely leading to the pore-water pressure observed (11 days later on the 28th October). Initial trends in the data suggest that the interior of the WRD is remaining relatively dry while the upper and lower extremities experience changes in moisture. The increase of positive pore-water pressure at the base of the dump gives an indication of potential basal seepage which may be occurring, while the time from initial pressure increase to de-saturation can give an idea of seepage rates. The collection and analysis of a larger data set will allow for a more detailed examination of the WRD response to incident precipitation.

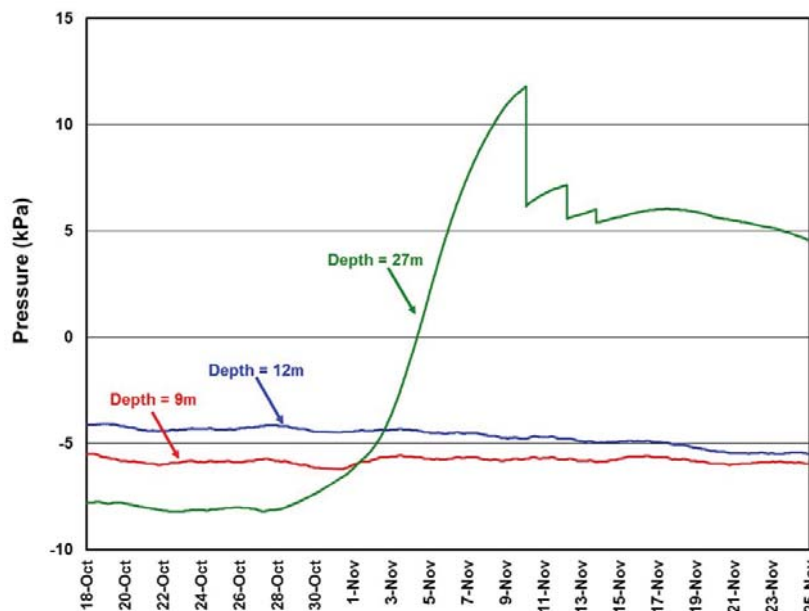


Fig. 8. Pore-water pressure recorded by VWP sensors at selected borehole

Figure 9 shows oxygen concentration with depth at a selected borehole from October 11th to October 29th. The oxygen concentration is slightly depressed in comparison to atmospheric (20.9%) at all depths; however, oxygen levels are above 17% at all depths. The relatively high oxygen levels present throughout the dump profile indicates that significant oxygen replenishment is occurring within the dump. This supports the temperature and moisture data which indicates a high level of air movement (convection/advection) through the dump is occurring

It is noted that the oxygen concentrations oscillate with a daily frequency. These oscillations are more pronounced at 4m depth while at 9m and 16m changes are more subtle. The muted oscillations in the 9m and 16m readings seem to 'mirror' those at 4m in that increases in oxygen concentrations at depth are accompanied by a decrease at 4m. Core log

information shows that the oxygen probe at 4m is located in a zone of black clay which is highly weathered pyritic black shale. It is worth noting that temperature data does not show significant levels of daily fluctuation as is seen on the oxygen data. This indicates that the loss/gain in oxygen at 4m is not likely to be directly related to daily variations in the supply/consumption of oxygen and rate of oxidation of pyrite in the shale as temperature and moisture conditions are constant. Rather the fluctuations are more likely to be a function of air flow.

The regular (day/night) changes in oxygen levels indicate the dump is 'breathing', however the causes of the 'breathing' mechanism have not been determined. Some observations are noted as follows. During the night time oxygen levels appear to be increasing at shallow depths indicating the dump may be breathing out (venting to the surface) as a result of convective gradients (due to internal dump temperature being higher than ambient air temperature) bringing higher oxygen air from depth to the surface. During the daytime oxygen levels decrease at shallow depths indicating the flow upwards of air in the dump is stagnated/reduced (as a result of reduced thermal gradients due to higher ambient air temperatures than dump temperatures driving convective air flow) and consumption may be occurring as a result of pyrite oxidation.

It is important to note that air flows within the dump will be complicated by factors such as the stack effect which depends on local wind conditions therefore this breathing model should be viewed as a high level concept of oxygen flows at this time until more data is gathered.

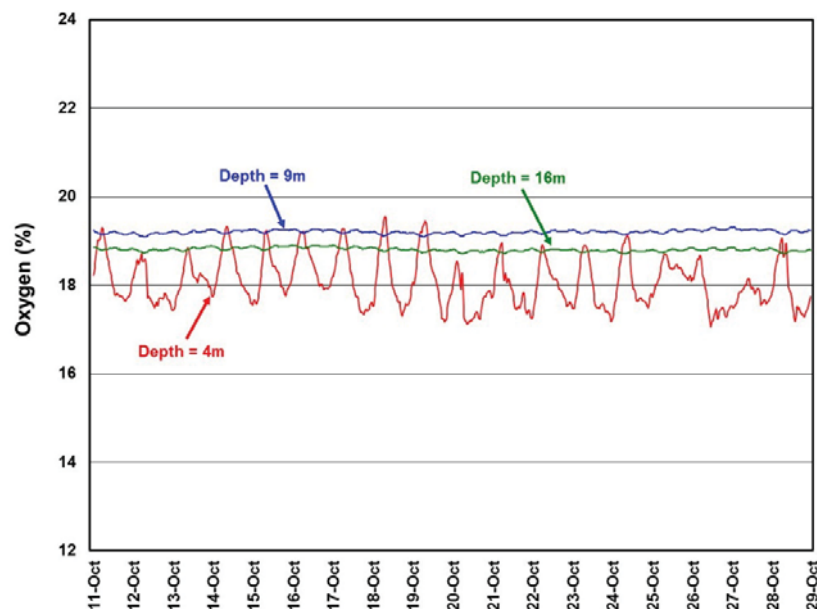


Fig. 9. Oxygen concentration at depth within a selected borehole

7.0 SUMMARY

A monitoring system comprised of 90 sensors was installed and commissioned in a number of WRDs in the Pilbara, Western Australia in 2013. The monitoring system records temperature, oxygen concentration, matric potential and pore water pressure data at depth within the waste rock. Installation of the monitoring system equipment utilised sonic drilling

which allowed for a detailed log to be recovered of the internal structure of the WRDs and specific zones to be selected for sensor installation. The monitoring system in conjunction with detailed logging of WRD structure allows for an understanding of WRD hydrology to be further developed, and for controls of AMD production to be understood as well as releases to the surrounding environment to be predicted. Remote communications allows the real time monitoring and evaluation of the WRDs response to climatic forcing and the effect this has on AMD production.

A preliminary review of monitoring system data indicates that sensors are responding as anticipated, and are producing valuable data. Initial trends suggest that:

- Wetting of the WRD's is not occurring uniformly throughout the waste, i.e. a wetting front is not present as would be suggested by typical WRD conceptual models
- The core of the WRD's have a high internal air flow which is providing ample supply of oxygen to the whole waste rock profile
- Elevated internal dump temperatures and high (and connected) internal air flows are resulting in effective air drying of the waste mass throughout much of the profile preventing a significant size wetting front from developing
- The WRD's appear to be breathing as a result of the response of the dump internally (convection/advection) to daily ambient temperature fluctuations
- Basal seepage is occurring as a result of fast drainage of infiltrating waters through coarse waste horizons (possibly as a result of underdrainage)

On-going monitoring of the study site will provide a unique dataset for the assessment of end dumped coarse waste rock piles and the production of ARD under site specific climatic conditions.

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THE DYNAMIC NEUTRALISING BEHAVIOUR OF ALKALINE WASTE ROCKS (CHLORITE CALCITE SCHIST) IN FLOW-THROUGH TREATMENT OF AMD WATERS

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ABSTRACT

At Savage River in Tasmania, the alkaline (non-acid forming) chlorite calcite schist (A-type rock) is one of the waste rock products of Grange Resources magnetite mining operations. The A-type waste rock has been used successfully as side covers and alkaline flow-through materials to alleviate serious AMD from the legacy (sulfide containing) B-dump. The Old Tailings Dam is also a historic AMD site at Savage River. Its seepage is commonly associated with low pH (≈ 2.6) and responsible for $\approx 50\%$ of the total mine site acidity load.

In this study, the A-type rock was fully characterised and then utilised in a laboratory flow-through system to treat acidic waters from the Old Tailings Dam. Such a flow-through system has the potential for low cost AMD management at the site. It was found that, at the fixed flow rate of 3 L/kg/day, the waste can effectively raise the seep water pH from 2.6 to near neutral (pH 7-8) with a loading of 157 L/kg. The concentrations of dissolved Al, Cu and Fe were decreased by almost 100 %. Changes in the neutralising behaviour of the waste, in particular, shifting neutralisation from dissolution of carbonates to silicates after 60 days of treatment was observed and is discussed in detail in the paper.

1.0 INTRODUCTION

The Savage River iron ore mine, owned and operated by Grange Resources, is located in NW Tasmania, Australia. To date, mining operations have generated in excess of 50 million m³ of ore and 300 million m³ of waste rock during the last 40 years. Current mining plans allow for a further 185 million m³ of waste rock over the next 18 years; however, AMD emanating from legacy (sulfide containing) waste rock dumps and tailings dams at the mine site, has caused environmental harm to Savage River and its tributaries. It has been reported that AMD has negatively impacted the original water quality and sediment characteristics, eliminating 90 % of the major taxa (families) of aquatic macroinvertebrates and decreasing overall invertebrate abundance by 99 %, with severely depleted native fish fauna in the river to about 30 km downstream of the mine site (Hughes et al. 2009).

Waste rock at Savage River is classified into one of four categories: i) A-Type – alkaline materials, non-acid forming (NAF) very durable rock; ii) B-Type – effectively neutral; iii) C-Type – clay or highly weathered rock; and iv) D-Type – potentially acid forming (PAF). In 1999, lithological and structural mapping of North Pit and South Lens Pit at the Savage River

Mine identified a large variety of potentially competent rocks (A-Type) that might be used for AMD mitigation, including the abundant chlorite-calcite schists with some dolomite, magnesite, talc-carbonate schist, dolerites and reactive silicates in mafic rocks, amphibole-chlorite-albite schists, serpentinites and tonalite (Thornt, 1999).

The Savage River Rehabilitation Project (SRRP), which was initiated in 1996 as a co-operative venture between the Tasmanian Government and previous owners Australian Bulk Minerals (now Grange Resources), has successfully implemented and tested the application of available A-type materials for AMD mitigation. In 1998, the Broderick Creek flow-through dump was constructed to divert creek flows through a lower coarse magnesite rock or calcite chlorite schist (A-type) drain; over which potentially acid forming waste rock was placed and encapsulated to maximise the volume of materials being placed in the steep valley adjacent to North Pit (Hutchison et al., 2009). The alkaline flow through drain imparts alkalinity to Savage River water, resulting in pH values in the river exceeding 7. The average alkalinity levels in Broderick Creek have more than doubled since its construction and the resulting near neutral water pH has contributed to a 30 % increase in the macroinvertebrate communities within Savage River downstream of its confluence with Broderick Creek (Hutchison, 2009). In 2006, combined water shedding and alkaline (calcite-chlorite schist) covers were constructed over the historic B-dump, which was identified as a significant source of AMD at the site. Five years after construction, the covers have effectively reduced the sulfide oxidation rate within the dump by about 43 %. Alkalinity from the covers has been migrating down into the acid-forming waste, forming passivating layers of silicate-stabilised iron oxy-hydroxide on pyrite grains (Li et al. 2011, 2012). Sulfate, Al and Cu flux in Main Creek have decreased by 50 %. The discharge from Main Creek to the river has neutral pH, low sulfate (218 mg/L) and metal concentrations, e.g. Al 200 µg/L and Cu 30 µg/L, which are below the SRRP fish target concentration enabling biota 20 km back up the river (Hughes et al. 2009).

The Old Tailings Dam (OTD) is also a historic AMD site at the Savage River iron ore mine. Its seepage is commonly associated with low pH (< 3) and is responsible for ~50 % of the total mine site acidity load (Ray 2007). Table 1 shows typical seep water analysis. The aim of this study was to investigate whether an alkaline flow-through system using fresh A-type waste mined from North Pit could be used to reduce acidity and metal concentrations in seepage from the OTD. This paper describes the results from characterisation of the A-type waste and the flow-through test undertaken as part of the AMIRA International funded project "Alternative Treatment Options for Long-term ARD Control" (P933A).

Table 1. Water quality (mg/L unless otherwise stated) of seepage from the OTD (27th January 2012).

pH	Eh (mV, SHE*)	Acidity (mg CaCO ₃ /L)	Al	C	Cu	Fe
2.6	800	425	4.2	210	0.14	53
K	Mg	Mn	Na	SO ₄ ²⁻	Si	Zn
11	140	27	12	1470	3.8	0.39

*SHE – Standard Hydrogen Electrode.

2.0 MATERIALS AND EXPERIMENTAL

2.1 Seep Water Sampling, Solid Sample Preparation and Experiment Set up

Two tonnes of seep water was sampled at the OTD on 27th January 2012. The water was stored in two large plastic tanks and shipped to University of South Australia.

The A-type waste was sampled at the North Pit on 22nd April 2010. The waste, light grey in colour, was pre-screened and the < 20 mm fraction collected. This fraction was dried at 50 °C and sub-sampled. The sub-sampled material was crushed to < 4 mm for the flow-through neutralisation experiment.

Beaker experiments were initially conducted to determine a flow rate at which the neutralisation would be effective. A-type waste samples of 500 g (< 4 mm) each were put in glass beakers and a known amount of seep water was added into the beakers without agitation at solid to liquid ratios of 1:1, 1:2, 1:3 and 1:4 (wt./wt.). The pH of the supernatant was measured as function of time in each beaker. The rate in L/kg/day at which the seep water pH increased to neutral was determined from these experiments.

Seepage from the OTD was passed through a column containing 1 kg of A-type material (< 4 mm) (Figure 1). The seep water was continuously and evenly distributed onto the solid bed surface at a constant flow rate determined from the batch experiments described above. The flow-through effluent was collected and effluent pH, Eh (SHE), acidity/alkalinity as well as dissolved solute concentrations were monitored as a function of time.

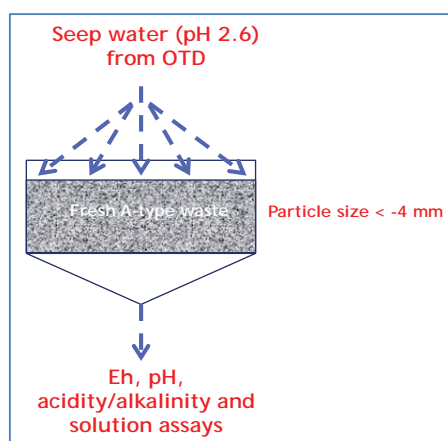


Fig. 1. The flow-through neutralisation experiment set up.

2.2 A-type Waste Material Characterisation

Less than 4 mm, A-type material was pulverised to <75 µm for bulk assay, acid-base accounting (ABA) and mineralogy determination by XRD. The bulk assay was carried out by XRF with lithium borate fusion. Total S was measured by LECO furnace. Pyritic sulfur was measured by the chromium reducible sulfur (CRS) method. Total C and organic C were measured to calculate inorganic C. Net acid generation (NAG) and acid neutralising capacity

(ANC) of the samples were measured based on methods described in the AMIRA ARD Handbook (Smart et al., 2002). The results are shown in Tables 2-4.

The A-type material has relatively high, readily available acid neutralisation capacity contributed to by calcite (4 wt.%) and dolomite (6 wt.%). It also contains substantial amounts of chlorite (66 wt.%) and other silicates which may contribute significant acid buffering capacity (Table 4). Total S (0.44 wt.%) and CRS (0.42 wt.%) indicate very little sulfide oxidation in the waste. Pyrite was not detected by XRD due to its low content (≈ 0.8 wt.% calculated from CRS) but SEM/EDS identified pyrite as the only sulfide phase in the waste material. The Sobek ANC value was relatively high at 190 kg H₂SO₄/t, but surprisingly the NAGpH was 11.6 indicating the possible presence of immediately soluble alkaline material such as brucite (Mg(OH)₂). Because of the high ANC and low MPA (13.5 kg H₂SO₄/t) values, the NAPP value was strongly negative (-177 kg H₂SO₄/t) and this material is classified as NAF.

Table 2. XRF, carbon and sulfur analysis results for the A-type waste (wt.%).

Al	As	Ca	Co	Cr	Cu
6.81	<0.01	6	0.008	0.014	0.009
Fe	K	Mg	Mn	Na	Ni
9.68	0.196	6.18	0.149	1.03	0.008
Pb	Si	Zn	Total	LOI	
<0.009	19.05	0.023	100.95	10.51	
Inorganic C	Organic C	Total C	S	CRS	
1.77	<0.02	1.78	0.44	0.42	

Table 3. ABA and NAG test results from analysis of A-type waste.

MPA (kg H ₂ SO ₄ /t)	MPA* (kg H ₂ SO ₄ /t)	ANC (kg H ₂ SO ₄ /t)	NAPP (kg H ₂ SO ₄ /t)	NAPP* (kg H ₂ SO ₄ /t)	NAG	Classification
13.5	12.9	190	-177	-177	11.6	NAF

Note: NAPP = MPA (total S) – ANC ; NAPP* = MPA* (CRS) - ANC.

Table 4. Mineralogy of A-type waste.

Mineral Name	Chemical Formula	Wt. %
Chlorite	Mg _{2.5} Fe _{2.5} Al ₂ Si ₃ O ₁₀ (OH) ₈	66
Quartz	SiO ₂	10
Calcite	Ca _{0.95} Fe _{0.05} CO ₃	4
Dolomite	Mg _{0.8} Fe _{0.2} Ca(CO ₃) ₂	6
Albite	NaAlSi ₃ O ₈	8
Actinolite	Ca ₃ Mg ₂ Fe ₂ Si ₈ O ₂₂ (OH) ₂	2

2.3 Determination of a Flow Rate for Effective Neutralisation

Figure 2 shows that the OTD seep water pH is raised from pH 2.6 to pH > 6 when contacted with A-type waste at high solid to liquid ratios (1:1 and 1:2) and to pH \geq 5.5 at low solid to liquid ratios (1:3 and 1:4) as soon as the water contacts the solid. The supernatant pH values from different beakers increased with time and reached a similar value of 7.8 at 170 h for all the tests at different solid to liquid ratios. The results in Figure 2 show that after 24 h and at a solid to liquid ratio of 1:3 (3 L/kg/day), the pH of the OTD seep water was increased to almost 6.5 and so this flow rate was chosen for further testing in the flow-through column.

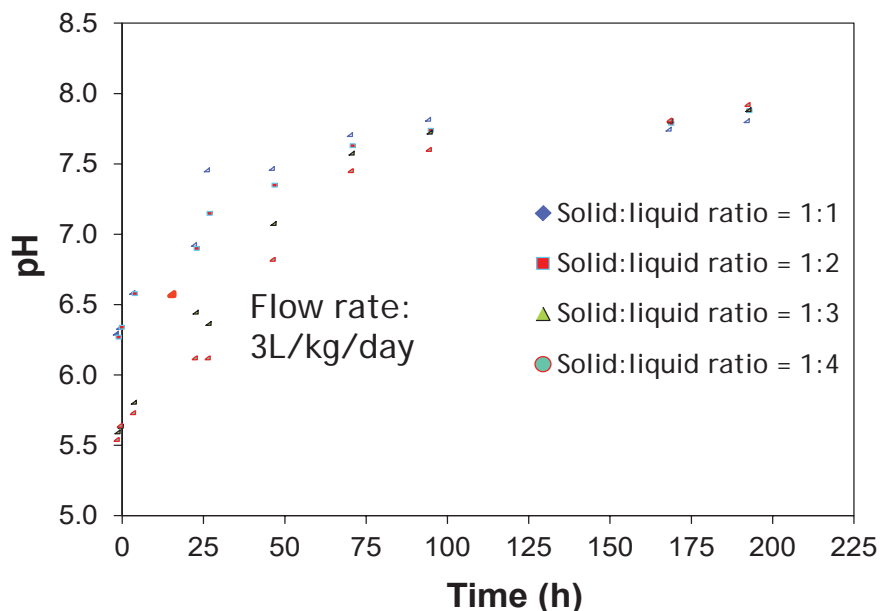


Fig. 2. Variation of supernatant pH as function of solid (A-type waste) to liquid (OTD seep water) ratio and time. The seep water pH at time zero was 2.6.

3.0 RESULTS AND DISCUSSION

3.1 Effluent Water Quality

Figure 3 (left) shows the effluent pH profile during the flow-through experiment. The initial effluent pH was 7.9 when the seep water (pH 2.6) went through the A-type waste. The effluent pH was maintained in a range of 7.7 – 8.0 during the first 39 days and dropped to 7.1 at 54 days. The pH then further quickly decreased to 3.3 from 54 days to 70 days then gradually decreased to 2.7 from 70 days to 104 days. Solids in the column were mixed on the 104th day in order to reduce possible water channelling effects on the neutralisation. The effluent pH rose to 4.6 after mixing and decreased to 2.8 during the last 28 days investigated.

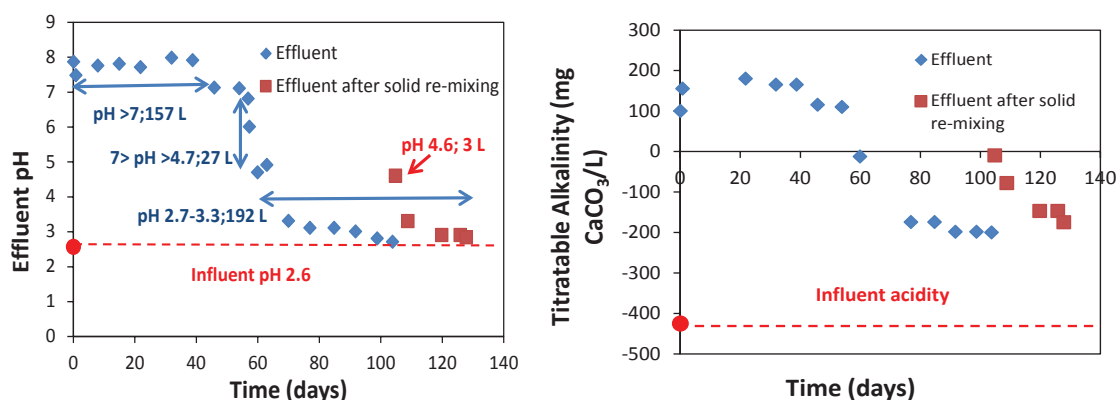
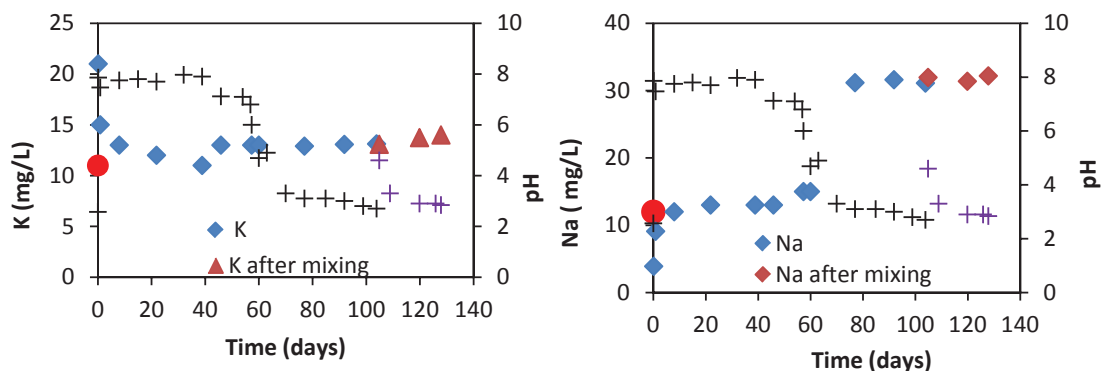


Fig. 3. Effluent pH (left) and titratable alkalinity (right) profiles of the flow-through neutralisation of OTD seep water using the A-type waste at the flow rate of 3 L/kg/day. Note negative alkalinity values are measured acidity.

The corresponding, titratable alkalinities in the effluents were around 110 – 180 mg CaCO_3/L during the first 54 days (Figure 3, right). There was no excess alkalinity in the column effluent after 60 days and acidity varied between 175 mg CaCO_3/L and 200 mg CaCO_3/L between 75 and 128 days. Mixing of the solids on the 104th day resulted in a brief decrease in acidity to 10 mg CaCO_3/L . The results indicate that at the flow rate of 3 L/kg/day the alkaline (A-type waste) flow-through can neutralise up to 150 L of seep water to pH > 7 or 185 L of seep water to pH > 4.7 per kg of A-type waste. After the effluent returned to low pH (2.7 – 4.7), the A-type waste continued to reduce acidity in the OTD seepage by more than 50 % (Figure 3).

The major solution constituents in the effluents were measured in the course of the test as shown in Figures 4-7. The initial K concentration of ≈ 20 mg/L in the effluent was greater by a factor of 2 than K concentration (11 mg/L) in the influent due to rapid dissolution, most likely from a salt in the alkaline material (Figure 4, left top). Thereafter, the effluent K concentration was maintained at a level around 13 mg/L, similar to the concentration in the influent. This observation suggests that neither K-containing secondary minerals precipitate in the system nor K-containing carbonates or silicates exist in the original solid. This is also confirmed by the K assays of the original solid and leached residue.



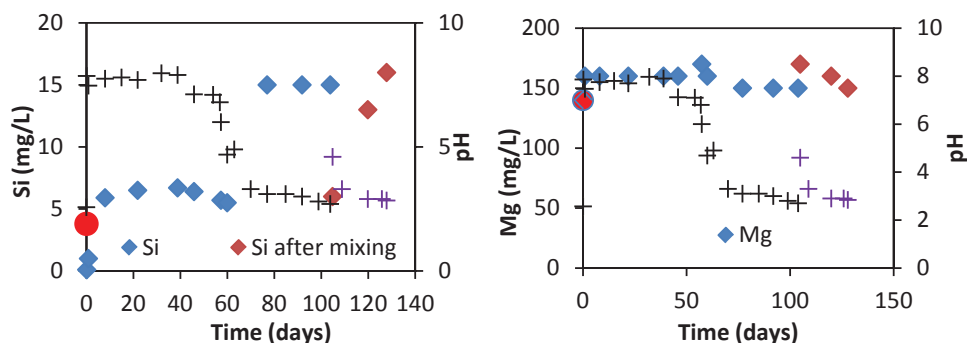


Fig. 4. Variation of K, Na, Si and Mg concentrations in the effluents as a function of time. The red dot in the figure represents the element concentration in the influent. Effluent pH (+) is included to indicate how changes in solute concentration correlate with pH change.

Na concentrations (Figure 4, right top) trended similarly with Si (Figure 4, left bottom) in the effluent indicating that the Na is likely predominately associated with the dissolution of silicates. However, a rapid reduction of Na and Si concentrations (lower than the concentrations in the influent) was observed within the first hour. Solution speciation calculation, using PHREEQC software, predicts that more than ten Na-containing aluminosilicate phases (e.g. montmorillonite-Na) could have been precipitated at pH > 7, consistent with the low Si (0.1 mg/L) and Na (3.9 mg/L) concentrations measured for the effluents collected within the first 20 h (0.83 days). High Na and Si concentrations observed after 70 days appear to be due to relatively fast dissolution of silicates in the alkaline waste at low pH (<4). High Mg concentrations over the pH range investigated were most likely derived from dissolution of dolomite and chlorite, and possibly actinolite (Figure 4, right bottom).

Figure 5 (left) shows considerably greater Ca concentrations in effluents than in the influent up to 60 days, due to rapid dissolution of calcite and dolomite in the neutralisation process; and relatively slow Ca released from the alkaline waste materials at pH <4, suggesting possible passivation of the carbonates. Initial increases in effluent Mn concentrations (Figure 5, right) suggest rapid dissolution of a Mn-containing phase. SEM analysis of the leached residue (see the next Section) indicates that additional Mn is released during dissolution of mangano ferrous dolomite ($\text{Ca}_{0.53}\text{Fe}_{0.16}\text{Mn}_{0.02}\text{Mg}_{0.29}\text{CO}_3$). Very low concentrations of Mn (≈ 5.4 mg/L) at low pH (<4) were observed but the solution speciation calculations do not indicate possible precipitation of Mn containing phases. It appears the reduction of Mn is associated with formation of schwertmannite or jarosite rather than its adsorption on secondary iron precipitates which only occurs at a pH greater than 5 (Millward and Moore, 1982).

The sulfate concentration in the effluent at neutral pH (≈ 7) was around 1500 mg/L, slightly greater than the concentration (1470 mg/L) in the influent, possibly due to dissolution of pyrite in the alkaline material (Figure 6). When the effluent pH decreased to ≈ 3 , the sulfate concentration dropped to ≈ 1410 mg/L, probably due to precipitation of sulfates, most likely schwertmannite at this pH. Solution speciation calculations suggest the formation of (K, Na) – jarosites. Schwertmannite-like phases of iron hydroxyl sulfate particles were identified by SEM/EDS (see next Section). Mixing of the solid resulted in an increased pH (4.6) and increased sulfate concentration due to the dissolution of the precipitates.

As anticipated, metal ions of Al, Fe, Zn and Cu in the influent were precipitated at high pH and the precipitates formed were dissolved at low pH (Figure 7). The Al, Fe and Cu concentrations reduced by almost 100 % at pH >7 up to 50 days.

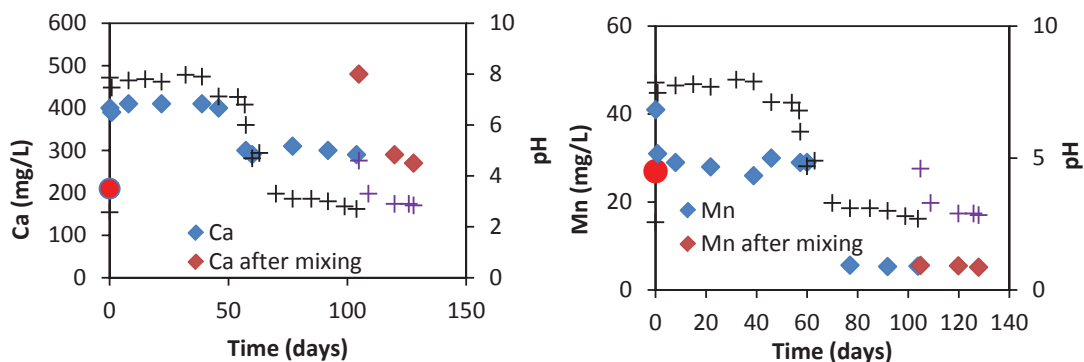


Fig. 5. Variation of Ca and Mn concentrations in effluent as function of time. The red dot in the figure represents the element concentration in the influent. Effluent pH (+) is included to indicate how changes in solute concentration correlate with pH change.

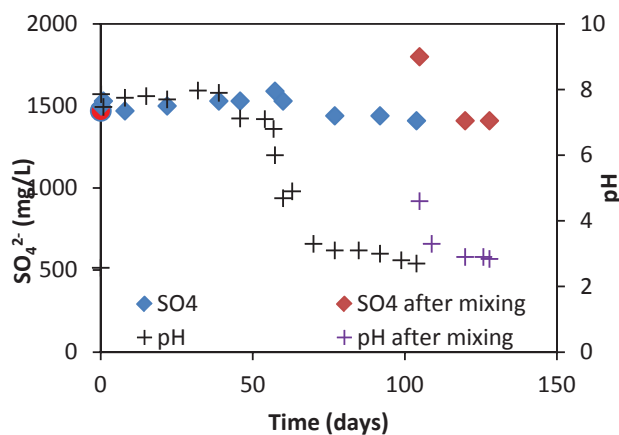


Fig. 6. Variation SO_4^{2-} concentrations in the effluents as function of time. The red dot represents the element concentration in the influent.

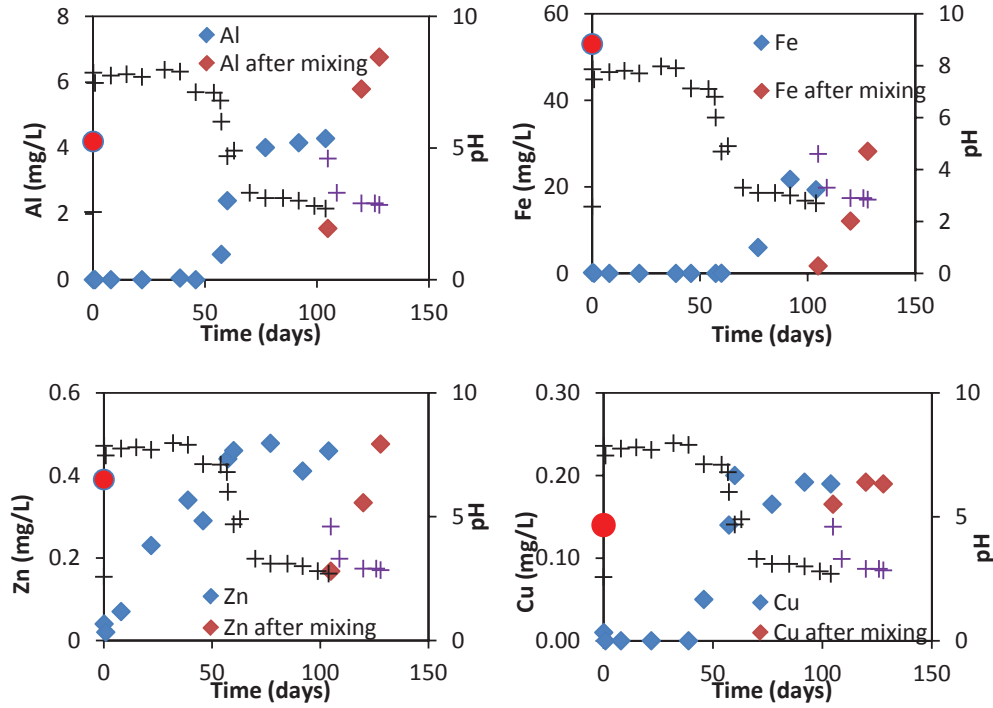


Fig. 7. Variation of Al, Fe, Zn and Cu concentrations in the effluents as function of time. The red dots represent the element concentration in the influent. Effluent pH (+) is included to indicate how changes in solute concentration correlate with pH change.

3.2 Acid Neutralisation Rate versus Acid Generation Rate

The acid neutralisation rate (ANR) and acid generation rate (AGR) in the alkaline flow-through system were calculated as a function of time (Figure 8). The ANR is defined as the acid neutralisation rate at which the alkalinity was released from the A-type waste in the system, as expressed in Eqn. [1]. The AGR is defined as the sum of acidity supply rate from the influent and acid generation rate from oxidative dissolution of pyrite in the A-type waste (Eqn. [2]). However, when effluent pH is less than 4.5, precipitation of schwertmannite/jarosite in the system becomes significant. In this case, the AGR was estimated by subtracting ANR in the influent from the sum of titratable acidity rate measured and ANR in each effluent (Eqn. [3]).

$$\begin{aligned} \text{ANR}_{(\text{Alkaline flow-through})} &= \text{ANR}_{(\text{Effluent})} - \text{ANR}_{(\text{Influent})} \quad [1] \\ \text{ANR}_{(\text{Effluent})} &= M_{w(\text{CaCO}_3)} \times (3 \text{ L/kg/day}) \times 7 \times \left[\frac{K}{M_{w(K)}/2} + \frac{\text{Na}}{M_{w(\text{Na})}/2} + \frac{\text{Ca}}{M_{w(\text{Ca})}} + \frac{\text{Mg}}{M_{w(\text{Mg})}} + \frac{\text{Al}}{M_{w(\text{Al})}/(2/3)} \right], \text{ where K, Na, Ca, Mg and Al (mg/L) are elemental assays of effluent.} \\ \text{ANR}_{(\text{Influent})} &= M_{w(\text{CaCO}_3)} \times (3 \text{ L/kg/day}) \times 7 \times \left[\frac{K}{M_{w(K)}/2} + \frac{\text{Na}}{M_{w(\text{Na})}/2} + \frac{\text{Ca}}{M_{w(\text{Ca})}} + \frac{\text{Mg}}{M_{w(\text{Mg})}} + \frac{\text{Al}}{M_{w(\text{Al})}/(2/3)} \right], \text{ where K, Na, Ca, Mg and Al (mg/L) are elemental assays of influent.} \end{aligned}$$

$$\begin{aligned} \text{AGR}_{(\text{Alkaline flow-through})} &= \text{Acidity Rate}_{(\text{Influent})} + \text{AGR}_{(\text{A-type waste})} \quad [2] \\ \text{Acidity Rate}_{(\text{Influent})} &= (3 \text{ L/kg/day}) \times 7 \times [\text{Titratable acidity of influent (mg CaCO}_3\text{/L)}]. \\ \text{AGR}_{(\text{A-type waste})} &= M_{w(\text{CaCO}_3)} \times (3 \text{ L/kg/day}) \times 7 \times [S_{(\text{Effluent})}/M_{w(\text{S})} - S_{(\text{Influent})}/M_{w(\text{S})}], \\ &\text{where } S_{(\text{Effluent})} \text{ and } S_{(\text{Influent})} \text{ are S concentrations (mg/L) in the effluent and influent.} \end{aligned}$$

When effluent pH ≤ 4.5 :

$$\text{AGR}_{(\text{Alkaline flow-through})} = \text{Titratable Acidity Rate}_{(\text{Effluent})} + [\text{ANR}_{(\text{Effluent})} - \text{ANR}_{(\text{Influent})}]. \quad [3]$$

$$\text{Titrateable Acidity Rate}_{(\text{Effluent})} = (3 \text{ L/kg/day}) \times 7 \times [\text{Titrateable acidity of effluent (mg CaCO}_3\text{/L)}].$$

$$[\text{ANR}_{(\text{Effluent})} - \text{ANR}_{(\text{Influent})}] \text{ is as shown in Eqn. [1].}$$

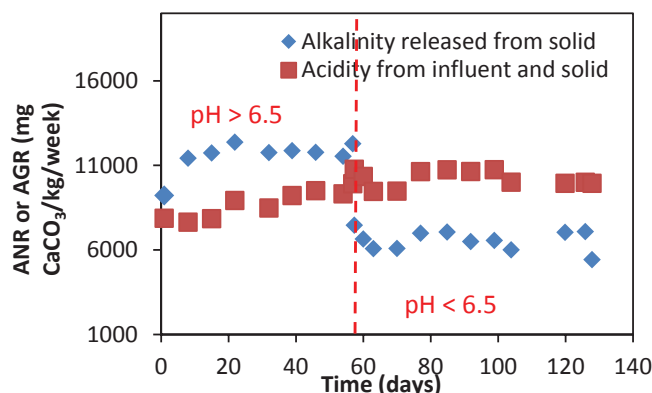


Fig. 8. Calculated ANR and AGR in the alkaline flow-through system as a function of time.

Figure 8 shows that the ANR in the alkaline flow-through system was around 12000 mg CaCO₃/kg/week at effluent pH > 6.5 up to 57 days. The ANR was significantly reduced to 6000 mg CaCO₃/kg/week accompanied by a reduction of the effluent pH to ≈3 during 57–70 days, and then maintained this rate over 128 days. The rapid reduction of ANR at the 57–70 days suggests either exhaustion of carbonates in the A-type waste or surface passivation of the carbonate particles. The AGR was about 9,000 mg CaCO₃/kg/week at effluent pH >6.5 and slightly greater (10,000 mg CaCO₃/kg/week) at the effluent pH <6.5, probably due to the dissolution of pyrite in the solid at low pH (Figure 8). It was observed that achieving neutral pH in the effluent from the system requires a ratio of AGR/ANR <0.9; otherwise acidic effluent will result.

XRD analysis of the solid residue revealed 1 wt.% calcite and 2 wt. % dolomite in the residue (Table 5). The presence of carbonates even after prolonged exposure to low pH solution suggests either the residual carbonates are poorly liberated (limited surface exposure) or that they have been passivated by precipitated iron phases. Significant colour difference between the residue and original solid strongly supports the latter; the A-type material surfaces have been significantly coated by reaction precipitates after 55 days of the test (Figure 9).

Table 5. XRD result (wt.%) of the alkaline flow-through residue.

Mineral Name	Chemical formula	Original*	Residue**
Chlorite	Mg _{2.5} Fe _{2.5} Al ₂ Si ₃ O ₁₀ (OH) ₈	66	70
Quartz	SiO ₂	10	12
Calcite	Ca _{0.95} Fe _{0.05} CO ₃	4	1
Dolomite	Mg _{0.8} Fe _{0.2} Ca(CO ₃) ₂	6	2
Albite	NaAlSi ₃ O ₈	8	11
Actinolite	Ca ₃ Mg ₂ Fe ₂ Si ₈ O ₂₂ (OH) ₂	5	45

*Sobek ANC = 194 g CaCO₃/kg; **Sobek ANC = 36 g CaCO₃/kg.

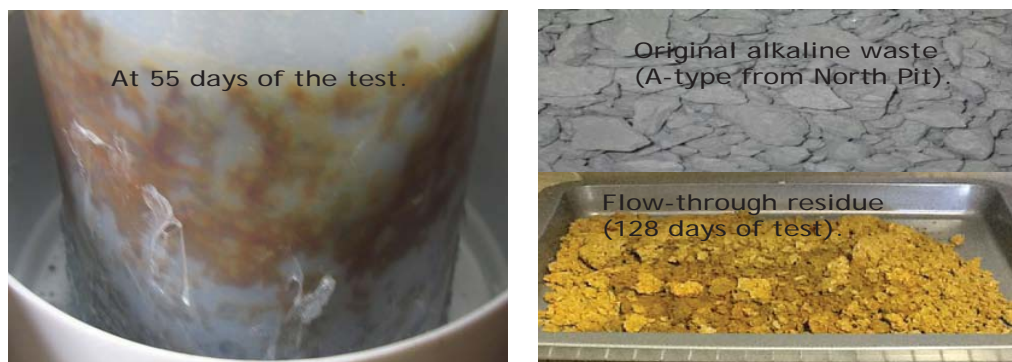


Fig. 9. Colour difference between the residue and original solid used in the alkaline flow-through test.

Total ANC and carbonate acid neutralisation capacity (ANCc) released from the A-type waste over 128 days was calculated (Table 6). The measured Sobek ANC of the original solid (194 g CaCO_3/kg) and residue (36 g CaCO_3/kg) reveals that 80 % of the Sobek ANC in the A-type waste was released over the 128-day test. The released ANC values estimated from the Sobek tests, solid bulk assays and effluent assays give an average value of 156 g CaCO_3/kg with an error of ± 17 g CaCO_3/kg . This ANC value, in comparison with the ANCc of 119 g CaCO_3/kg derived from the inorganic C content in both the original solid and residue, suggests non-carbonate ANC (ANCnc) of 37 ± 17 g CaCO_3/kg , i.e. **$\approx 24\%$ of the total ANC is derived from silicate dissolution**. This ANCnc resulting from silicate dissolution is $\approx 70\%$ of the total ANC (42 g CaCO_3/kg) released from the A-type waste at the effluent pH 2.7–4.7 (Figure 3). This demonstrates a shift of neutralisation from dissolution of carbonates to silicates in the system after 60 days treatment, with negligible silicate dissolution under neutral pH conditions.

Table 6. Total ANC and ANCc (both measured as g CaCO_3/kg) released from the A-type waste over 128 days of the test.

	Solid bulk assay	Sobek	Effluent assay	Average	Max. error	Inorganic C
Original	429	194	-	-	-	147
Residue	289	36	-	-	-	28
Released	140	158	171	156	± 17	119

Note: $\text{ANC}_{\text{nc}} (37 \text{ g } \text{CaCO}_3/\text{kg}) = \text{ANC}_{\text{average}} - \text{ANCc}$. $\text{ANC (released)} = \text{ANC (original)} - \text{ANC (residue)}$.

3.3 Surface Analysis of the Alkaline Flow-through Residue

Surface analyses were performed on the alkaline flow-through residue using SEM/EDS to investigate the surface nature of the carbonates and chlorite remaining in the residue and secondary reaction products precipitated. The results are shown in Figures 10-11. Elemental concentrations determined from EDS analysis are given below each image.

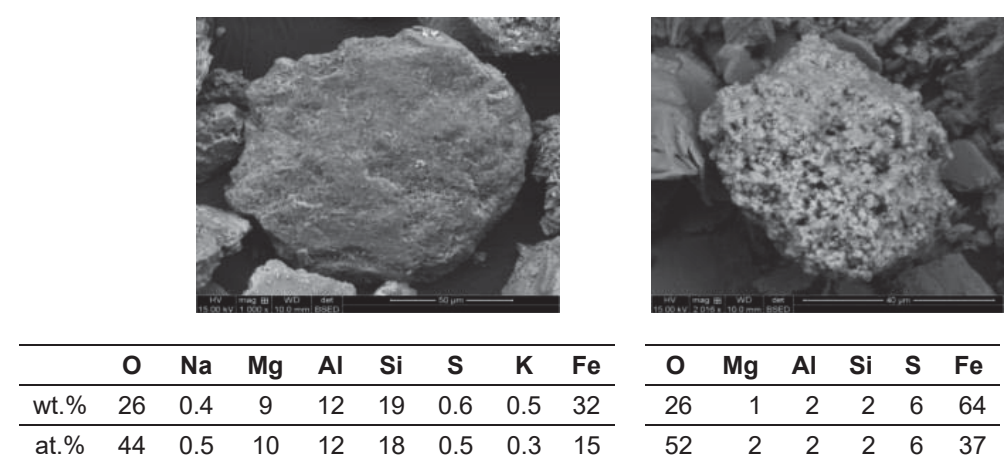


Fig. 10. Backscattered electron (BSE) images of chlorite (left) and schwertmannite (right) particles found in the residue.

Figure 10 (left) shows chlorite particles found in the residue. Na identified in chlorite by EDS together with high Na concentrations observed in the effluent at low pH (≈ 3) (Figure 4), suggests chlorite dissolution increases as the pH decreases.

Dolomite and calcite particles were severely reacted with the acidic influent during the neutralisation process (Figure 11). Mn was found in the dolomite structure ($\text{Ca}_{0.53}\text{Fe}_{0.16}\text{Mn}_{0.02}\text{Mg}_{0.29}\text{CO}_3$) by EDS confirming that the observed high effluent Mn concentration at neutral pH is due to dolomite dissolution. However, we did not detect Mn in any other phases investigated to enable explanation of the observed low concentrations of Mn in the effluents at low pH of ≈ 3 . EDS analyses depths of 1-2 μm will not normally detect adsorbed ions in surface layers (1-2 nm). Hence, it is not clear whether adsorption of Mn ions on secondary precipitates occurred.

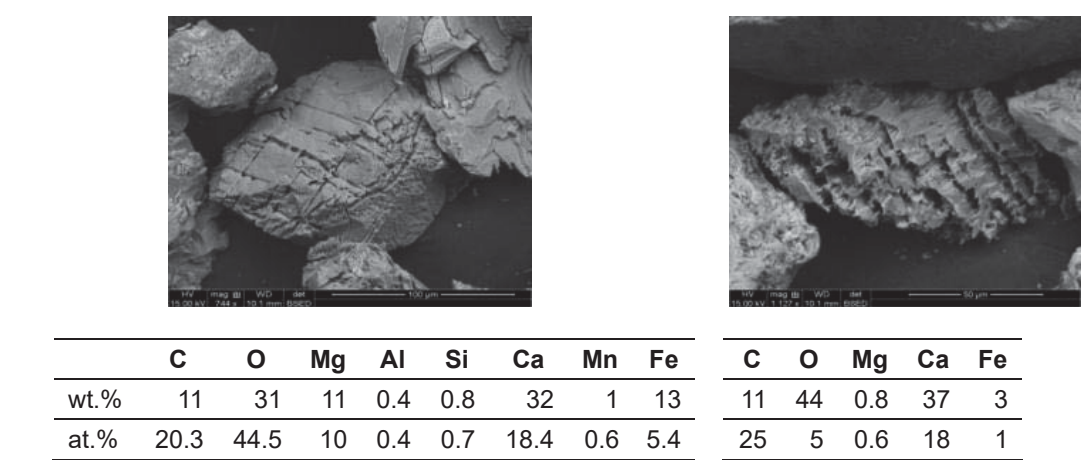


Fig. 11. Mangano ferrous dolomite (left) and magnesium ferrous calcite (right) particles found in the residue. Note, prior to analysis the sample was sonicated in acetone to remove iron hydroxide coatings.

Schwertmannite-like iron sulfate particles were found in the residue (Figure 10, right). The particle has a very porous structure, similar to that of schwertmannite. The element ratio of Fe:S in the particle is 7:1, very close to that of 8:1 for an ideal standard schwertmannite structure. This further confirms that precipitation of the schwertmannite-like phases is responsible for the reduced sulfate concentration in the effluents at the low pH (≈ 3) in the system.

4.0 CONCLUSION

An alkaline flow-through experiment using fresh A-type waste mined from the North Pit at the Savage River iron ore mine was undertaken to investigate the dynamic neutralising behaviour of the alkaline waste in flow-through treatment of the acidic seepage (pH 2.6) from the Old Tailings Dam (OTD). The A-type waste contains 4 wt.% calcite, 6 wt.% dolomite and 66 wt.% chlorite with other silicates, having the Sobek ANC of 194 g CaCO₃/kg. At a flow rate of 3 L/kg/day, the A-type waste can neutralise 157 L/kg of OTD water to pH >7 with Al, Cu and Fe concentrations in the effluent reduced by almost 100 % and Zn concentrations reduced by ≈ 26 %. However, SO₄²⁻ concentration in the effluent was similar to that in the influent (1470-1530 mg/L). The total ANC released from the waste was 156 \pm 17 g CaCO₃/kg with 24 % of the total ANC derived from silicate dissolution over 128 days of the test. The non-carbonate ANC of 37 \pm 17 g CaCO₃/kg released by the silicates at an effluent pH of 2.7–4.7 suggests a shift of neutralisation from dissolution of carbonates to silicates in the system after 60 days treatment. It was found that the carbonates in the waste are not able to maintain neutral effluent pH in the longer-term (more than 60 days), possibly because of passivation of carbonates by iron precipitates. However, silicates in the A-type waste reduced acidity in the water by ≈ 50 % in long-term although the resulting water pH (≈ 3) was still low. Further control measures will be required to raise pH and reduce the dissolved metal concentrations in seepage from the OTD to levels acceptable for environmental discharge.

5.0 ACKNOWLEDGEMENT

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NANOFILTRATION FOR RESOURCE RECOVERY FROM MINE INFLUENCED WATERS

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ABSTRACT

Treatment of mine influenced waters (MIW), including acid mine drainage, is commonly viewed as an end-of-pipe process with the primary objective of meeting discharge criteria. New technologies can concentrate and recover commodity metals present in MIW waters, thus producing positive NPVs. Membrane technology is an established strategy to treat MIW. Nanofiltration (NF) offers economical advantages over reverse osmosis, and allows for concentrating and recovering valuable metals from MIW waters. The rejection performance of NF membranes is influenced by the feed pH, as ion rejection can be significantly impacted when the feed pH approaches the membrane iso-electric point (IEP). Since the IEP of commercially available NF membranes range between pH 3 to 5, thus bracketing the pH range of most MIW streams, understanding the rejection behavior for a particular membrane-MIW problem is critical to ensure compliance with discharge criteria, and to maximize metal recovery for profit. In this study, a NF membrane was tested on a MIW stream containing copper to determine the relationship between feed pH and species rejection. Maximum copper rejection was observed when the NF application was optimized with respect to solution pH, thus maximizing copper recovery and financial return.

1.0 INTRODUCTION

The management of water in mining operations is becoming increasingly scrutinized, with water reuse, water treatment and discharge being major issues faced by the industry (Water for Mining 2010). Johnson and Hallberg (2005) highlight two key points in the choice of suitable technologies to treat mine waters: i) legislation defines discharge criteria that may determine the choice of a system to effectively remove sulfate as well as metals and acidity from mine waters; ii) it is fundamental to consider mine water remediation as a resource, thus encouraging recovery and recycle of the products of mine water treatment. This second aspect is the focus of the present study.

Treatment of mine water is often seen as an end-of-pipe process aiming at producing a clean discharge stream that meets legislation limits of metals, sulfate and acidity. Extensive reviews have been published on treatment options for acid mine drainage and wastewaters containing heavy metal (Johnson and Hallberg 2005; Fu and Wang 2011; Zinck and Griffith 2013). Lime neutralization and biological treatments are recognized as the traditional approaches, while ion exchange and membrane technologies are seen as novel alternatives to treat mine waters (Al-Zoubi et al. 2010b). Different from lime neutralization and biological treatments, these technologies have the ability to not only remove potentially toxic metals and meet discharge criteria, but also to recover valuable commodities such as metals and acid from mine waters. Extraction of copper, nickel and cobalt from acid mine drainage by ion exchange is an example of what is currently practised (Nodwell and Kratochvil 2012).

Membrane treatment by reverse osmosis (RO) and nanofiltration (NF) is an established strategy for heavy metal removal as it is capable of achieving strict discharge criteria while providing high efficiency, easy operation and space saving (Fu and Wang 2011). Recent

studies successfully applied membrane separation to treat both synthetic and real mine water streams (Zhong et al. 2007; Rieger et al. 2009; Al-Zoubi et al. 2010a, 2010b; Mortazavi and Chaulk 2012). The authors explored membrane performance at different experimental conditions, with particular attention to the effects of solution temperature, operating pressure, feed flow and feed concentration on solute rejection and permeate flux. RO and NF provided similar rejection performance for polluting metals, however NF was suggested as the preferable treatment due to lower operating costs, e.g. higher fluxes at lower pressure, and its ability to selectively concentrate and recover metals and sulfuric acid. Relatively unexplored is the effect of mine water pH on membrane performance, although its impact on solute rejections has been reported (Zhong et al. 2007; Al-Zoubi et al. 2010b).

The separation mechanism of NF membranes involves predominantly membrane surface charge, i.e. electrorepulsion, and sieving effects (Childress and Elimelech 2000; Qin et al. 2004). For uncharged molecules, sieving or size exclusion is most responsible for separation and it is controlled by their molecular size in solute form. For ionic species, both sieving and electrostatic interactions are responsible for separation. Electrorepulsion is controlled by the membrane charge density and charge polarity, which are both characterized by the Zeta Potential (ZP) of the membrane surface. This parameter is usually evaluated from streaming potential analyses (Tay et al. 2002; Carvalho et al. 2011). The solution pH has a significant effect on ZP, because it protonates and deprotonates the functional groups of the membrane material and of the molecules in solution, thus modifying the membrane charge polarity and density (Childress and Elimelech 2000; Carvalho et al. 2011). The solution pH at which the membrane charge is zero is the iso-electric point (IEP). The membrane surface is negatively charged when the solution pH is higher than the IEP, and positively charged otherwise. Previous work has been carried out to determine the ZP and IEPs across a range of commercially available NF membranes. These studies were conducted for a range of different solution chemistries and pH values, and resulted in the IEP ranging between pH 3 to 5 (Childress and Elimelech 1996; Tanninen et al. 2004; Artug 2007).

A thorough understanding of the interrelation between membrane performance (e.g. solute rejection) as a function of feed pH is mandatory because as pH changes, so do several of the system characteristics (Childress and Elimelech 2000). Many studies focusing on the relationship between feed pH, membrane charge, and ion rejection, agree on the significant effect of feed pH, with abrupt changes and minimum rejections being expected at the IEP (Artug 2007; Qin et al. 2004; Hagmeyer and Gimbel 1999). Minimum rejections at the IEP are explained as a consequence of the fact that size exclusion is the only active separation mechanism at IEP (Ferreira-Esmi et al. 2013).

Since the IEP of commercially available NF membranes ranges between pH 3 to 5, thus bracketing the pH range of most mine influenced water streams, understanding the rejection behavior for a particular membrane-mine water problem is critical for evaluation of a NF treatment strategy. The objective of this study was to investigate the performance of NF membrane filtration on a mine influenced water stream, in order to i) understand the relationship between solute rejection and feed pH, and to ii) determine whether NF is a suitable technology to treat mine water with the perspective of recovering commodity metals.

2.0 MATERIAL AND METHODS

2.1 Membrane and mine water sample

A Dow NF 270 polyamide nanofiltration membrane was tested in this study because of the availability of published work describing its ZP and IEP, and therefore the ability to compare the current results. NF 270 is considered a 'loose' NF membrane (Al-Rashdi et al. 2012), with

molecular weight cut-off of 270 Da. The published NF 270 IEP range is between pH 2.5 and 4 (Tanninen et al. 2004; Artug 2007; Al-Rashdi et al. 2012).

A sample of mine influenced water was provided by a copper mine in Western Australia, and identified hereafter as MIW. The sample originated as mine runoff during periods of intense precipitation at the mine site. The sample composition is shown in Table 1.

Table 1. Composition of mine water samples as provided by a copper mine in Western Australia

Parameter	Unit	MIW
pH	-	4.5
Aluminium, Al ³⁺	mg L ⁻¹	14
Calcium, Ca ²⁺	mg L ⁻¹	480
Copper, Cu ²⁺	mg L ⁻¹	410
Iron, Fe ³⁺	mg L ⁻¹	0.14
Potassium, K ⁺	mg L ⁻¹	310
Magnesium, Mg ²⁺	mg L ⁻¹	770
Manganese, Mn ³⁺	mg L ⁻¹	440
Sodium, Na ⁺	mg L ⁻¹	2000
Sulfur, S ⁻	mg L ⁻¹	2300
Sulfate, SO ₄ ²⁻	mg L ⁻¹	6900
Chloride, Cl ⁻	mg L ⁻¹	2300

2.2 Experimental set up

A first set of tests was performed to empirically estimate the position of the IEP and the relative membrane charge polarity of NF 270. This test is referred to as IEP Test, and the aim was to determine the position of the IEP during filtration of a NaCl-Na₂SO₄ ion system. Literature suggests that the position of the IEP and membrane charge can be estimated from rejection minima of simple ternary ion systems (Szoke et al. 2002; Artug 2007).

IEP Test was performed on NF 270 membrane, and a synthetic solution was prepared of about 700 mg L⁻¹ NaCl and 15 g L⁻¹ Na₂SO₄. The sulfate concentrations used were within sulfate levels of typical mine water solutions (Al-Zoubi et al. 2010b). The IEP Test was carried out with feeds ranging from pH 5 to pH 2 at 0.2 pH decrements by dosing HNO₃.

A second set of tests is referred to as MW Test. The aim of this test was to determine the impact of feed pH and membrane charge on ion rejection. MIW was filtered by using NF 270 membrane. The test started at the feed original pH (4.5, Table 1), then pH was decreased at 0.2 pH decrements by adding HCl. Feed flow and temperature were kept constant at 225 L h⁻¹.

¹ and 25 °C. Permeate flux rate was equal to $32 \pm 2.5 \text{ L m}^{-2} \text{ h}^{-1}$ (average \pm standard deviation), and feed pressure was set to 7 bar.

The schematic diagram of the cross-flow flat sheet membrane test unit is shown in Figure 1 (membrane surface area of 0.0138 m^2). Both IEP and MW Tests were carried out in batch recirculation mode from a start feed volume of 2.5 L: both permeate and retentate were recirculated to the feed tank unless sample volumes of 30 mL were extracted from the system. The membrane was conditioned for 15 minutes at each pH set point before collection of samples.

All metal and sulfur analyses were conducted using ICP-OES while chloride analyses were conducted using an ion selective electrode. These analyses were performed by a third party commercial laboratory. pH and temperature was monitored during the tests using a TPS Aqua-CPA series combination pH, temperature and conductivity meter. Ion rejection was calculated for each ion as the concentration ratio between permeate and feed sample.

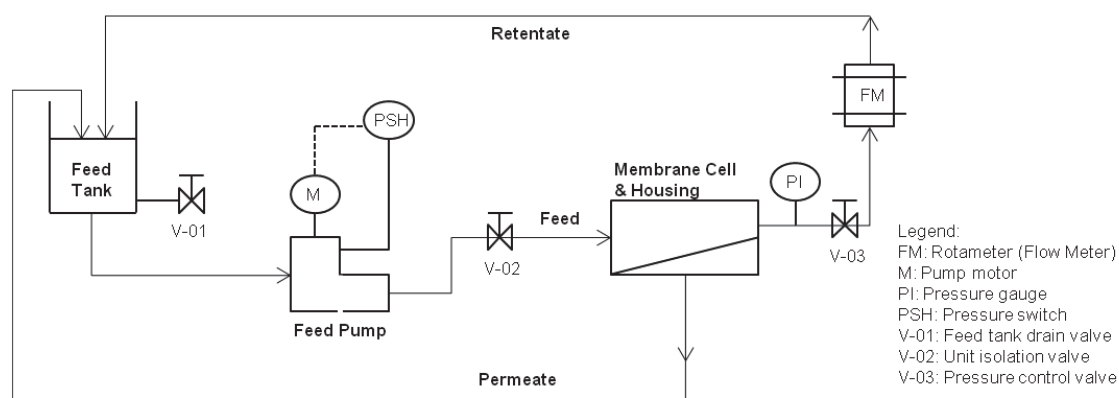


Fig. 1. Schematic diagram of lab-scale NF unit test

3.0 RESULTS AND DISCUSSION

3.1 Iso-electric point and membrane charge

Results of IEP Test are shown in Figure 2 where the relationship between ion rejection and $\text{NaCl-Na}_2\text{SO}_4$ feed solution pH is presented.

Minimum rejections of sodium and sulfur were obtained at pH close to 3.0 (Figure 2), suggesting the IEP was in the vicinity of pH 3 under these conditions. This is consistent with previous studies locating the IEP of NF 270 at about pH 3 (Tanninen et al. 2004; Artug 2007; Al-Rashdi et al. 2012). Minimum rejection at IEP is explained by the fact that sieving effect is the only active separation mechanism, as membrane charge is zero at the IEP (Qin et al. 2004). Rejection minima at the IEP were also found by Szoke et al. (2002) and Artug (2007). It follows that the membrane is positively charged for pH values lower than 3 and negatively charged otherwise. Rejections of sodium and sulfur followed the same trend (Figure 2), as the retention of sodium ions depended on the rejection of sulfur due to the electroneutrality condition (Artug et al. 2007).

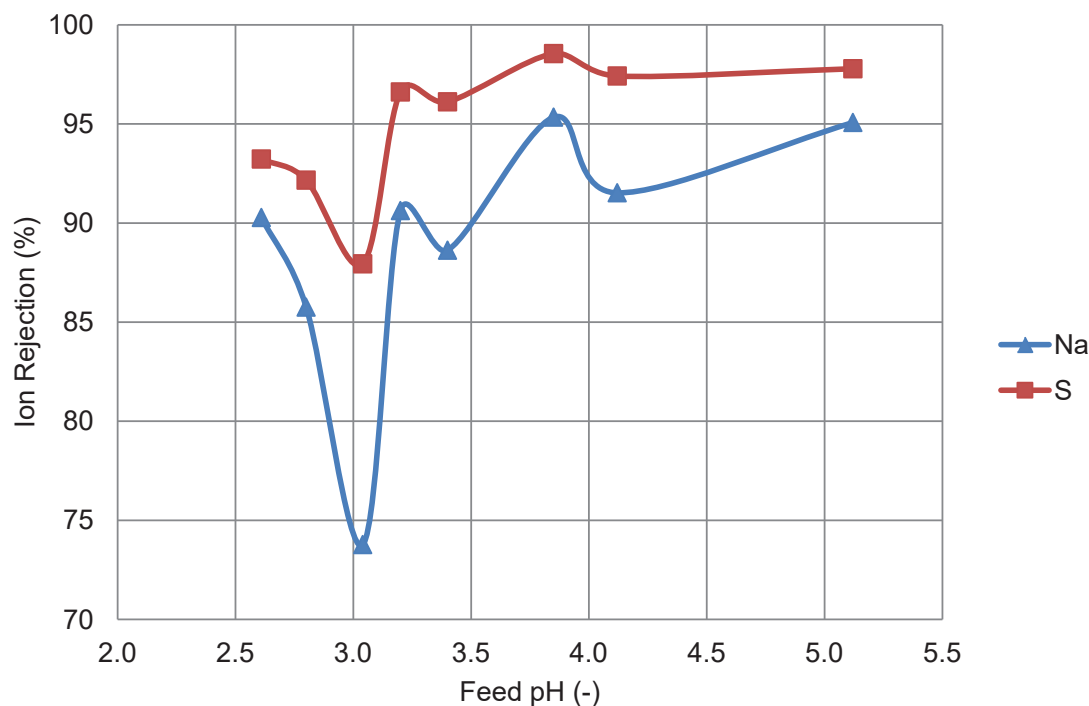


Fig. 2. Ion rejection as resulting of IEP Test on sodium chloride and sodium sulfate solutions on NF 270 membrane

3.2 The impact of solution pH on metal rejections in mine water streams

Results of the MW Test are shown in Figure 3. The rejections of the major cations (calcium, copper, magnesium, manganese, and sodium) and sulfur are shown at different feed pH values.

Filtration of MIW using the NF 270 membrane achieved rejections above 95 % for all multivalent cations at feed pH lower than 3, however rejections decreased as pH increased (Figure 3). Similar trends of lower metal rejections at high feed pH were also found by Zhong et al. (2007) and Al-Rashdi et al. (2012). Cations were highly rejected when membrane was positively charged ($\text{pH} < 3$), but the rejection decreased as the membrane became increasingly negative ($\text{pH} > 3$). An opposite trend was observed for sulfur rejections, with higher rejections at increasing pHs. The observed trend of sulfur rejections can be explained in two ways. First, the membrane is negatively charged at pH higher than 3, thus sulfate rejection increased in accordance with an increasingly negatively charged membrane (Szoke et al. 2002; Al-Zoubi et al. 2010b). Second, at pH higher than 2, sulfur is mostly present as sulfate ion, which is highly rejected by nanofiltration membranes. On the contrary, at low pH, sulfate-bisulfate equilibrium moves towards the monovalent bisulfate ion, and bisulfate ion becomes dominant at pH lower than 2 (Tanninen et al. 2004; Soldenhoff et al. 2005). Rejections of sulfur therefore decreased at low pH due a higher percentage of monovalent bisulfate ions and to a positively charged membrane surface.

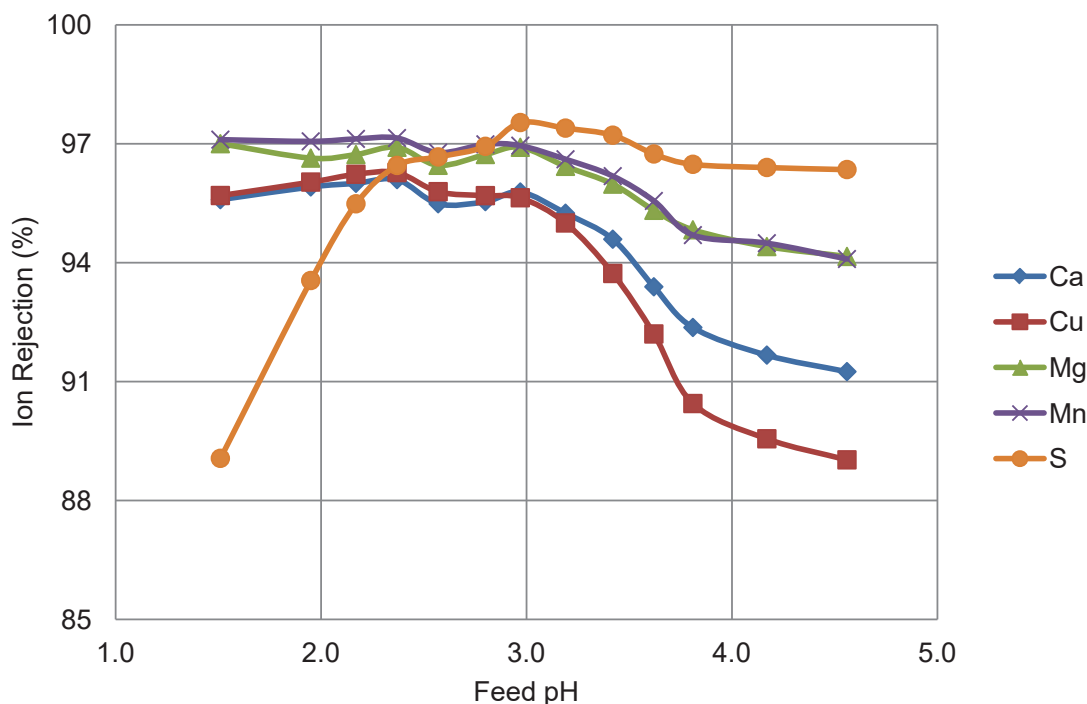


Fig. 3. Ion rejection at varying feed pH. Feed solution: MIW; membrane: NF 270

The results obtained using the NF 270 membrane confirmed the findings of previous studies on the importance of membrane charge to determine ion rejections when treating mine water with NF (Zhong et al. 2007; Al-Zoubi et al. 2010b; Al-Rashdi et al. 2012). NF is a suitable technique to treat mine water as it allows concentrating and recovering valuable metals, while potentially meeting discharge criteria, however the position of the membrane IEP relative to the feed pH has to be carefully considered. It should be noted that NF membranes vary in terms of their rejection characteristics, and metal rejection could be further improved by deploying alternative commercially available NF membranes characterized by higher rejections of multivalent ions. Comparison with alternative NF membranes is the subject of current research conducted by the authors.

3.3 Commodity metals recovery

Given the high rejections of cations achievable by NF as demonstrated in the current and previously published studies (Al-Zoubi et al. 2010b), a conceptual simplified flowsheet is proposed (Figure 4). The objective is to efficiently recover commodity metals, e.g., copper, from mine influenced waters, therefore considering mine water as a resource. The conceptual flow sheet is constituted by:

- Nanofiltration step to pre-concentrate the commodity metals present in the mine influenced water, thus reducing the volume to send to further treatments and improving process efficiencies;
- Sulphide precipitation step to extract commodity metals (Nodwell and Kratochvil 2012).

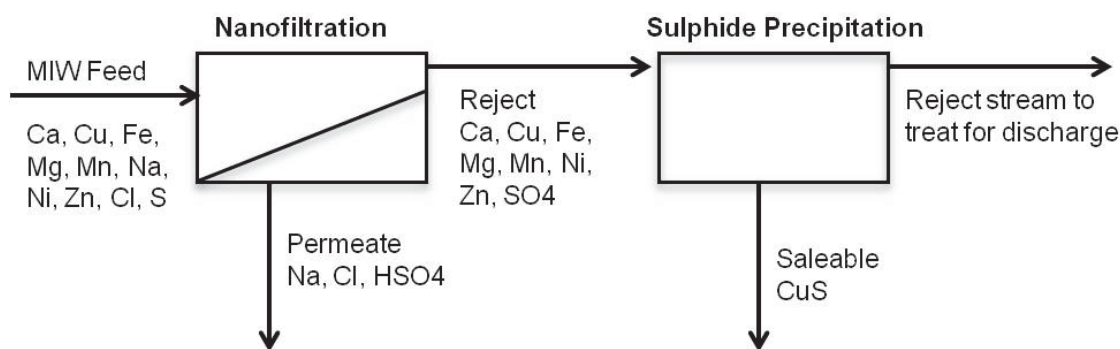


Fig. 4. Conceptual simplified flowsheet for the recovery of commodity metals present in mine influenced waters

Within the context of the proposed flowsheet, the results of MW Test revealed that understanding the interaction between membrane IEP and MIW pH might be considerably important to maximize the rejection and recovery of commodity metals. To achieve the highest recovery of metals, and therefore maximize earnings, the membrane IEP needs to be higher than the feed pH. To further investigate this aspect we have compared the results of a copper mass balance calculated at two different pH values, one higher and one lower than the IEP. pH values equal to 4.5 and 2.1 were selected based on results of MW Test (Figure 3). A mass balance was calculated on copper by considering the following assumptions:

- a typical 100 kL h^{-1} mine water plant was considered and the mass balance was calculated at 75% volumetric recovery (i.e., 75 % of the feed volume was recovered as permeate);
- copper rejections were constant at each recovery and equal to the ones found in the MW Tests, i.e., 96 % and 89 % at pH 2.1 and 4.5 respectively (Figure 3).

The estimated copper concentration in the reject at 75 % recovery is approximately 1560 mg L^{-1} at feed pH 2.10 (Figure 5a), while it decreases to 1400 mg L^{-1} when feed pH is 4.50 (Figure 5b). This is a consequence of the lower rejections of Cu as feed pH become higher than IEP.

A variation of 2.5 pH units caused a possible 4 kg h^{-1} shortfall in Cu not recovered from the reject stream (Figure 5a and b). This mass of Cu is lost in the permeate, thus not sent to sulphide precipitation. At a copper sulphide price of US\$5,600 per tonne (CuS price is considered about 80% of copper price), this difference in the mass of Cu equates to a potential loss of about US\$180k per year. This loss could be avoided if NF separation mechanisms were better understood. Understanding and optimizing the interactions between feed pH, membrane charge and ion rejection translates in economical benefits by maximising the recovery of commodity metals.

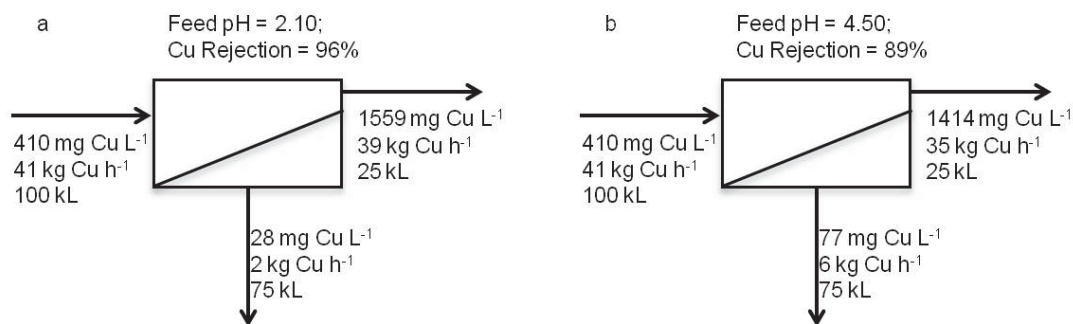


Fig. 5. Mass balance of copper ion at feed pH equal to (a) 2.10 and (b) 4.50. Cu rejections refer to the results of MW Tests as shown in Figure 3

4.0 CONCLUSIONS

Nanofiltration offers advantages over reverse osmosis systems for treating mine influenced water, including selectivity for concentrating commodity metals and lower capital and operating costs. When considering NF, the interaction between solution chemistry, i.e., pH, and membrane charge, i.e., isoelectric point, influences the rejection of all species. Our tests demonstrated that maximum metal rejection occurred when the solution pH was below the isoelectric point, and diminished significantly as the solution pH deviated above the isoelectric point. Anion rejection followed the opposite trend.

Understanding NF rejection mechanisms is important not only to ensure compliance with discharge criteria but also to maximize the potential recovery of commodity metals. Our study demonstrated that to achieve the highest recovery of metals, and therefore maximize earnings, the membrane IEP needs to be higher than the pH of the mine water. Optimization of the membrane performance by understanding the relationship between solution chemistry and membrane isoelectric point can yield significant commercial benefits.

It is recommended that knowledge of the performance characteristics of various membranes under a range of operating environments, in addition to test work to verify and optimize performance targets, is necessary to ensure maximum return on recovered commodity metals. Current research focuses on testing different commercially available NF membranes and on developing a flowsheet that integrates NF and other technologies such sulphide precipitation in order to maximise earnings from the recovery of commodity metals present in mine influenced waters.

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INTEGRATION OF WATER CHEMISTRY AND ACIDITY COMPONENTS: A NEW AMD ASSESSMENT MODEL

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ABSTRACT

The aims of this study were to better understand the reactions giving AMD and quantify the release and transport of sulfate and acidity components from the interaction of water with mine wastes, and in situ mineralisation, at a closed sulfidic mine site in central north Queensland.

Water geochemistry and stable isotope techniques were used to identify origins and associations of different water types. The water chemistry and isotope results showed local mixing of tailings and waste rock seepage with groundwater system along inferred seepage zones. Stable isotope geochemistry of sulfate-sulfur has identified two distinct sources of sulfate contributing to downstream. The background water has clearly distinct $\delta^{34}\text{S}$ -sulfate values compared to those from mine-related sulfides (+4.9 ‰ to +8.8 ‰), alunite group minerals (+8.0 ‰ to +9.8 ‰) and sulfate.

Improved titration methods giving both strong and weak mineral acidity contributions have been applied to determine the relative significance of various acidity components of mine-related water, namely the strong acidity by free proton ion and weak mineral acidity by hydrolysable metals. The acidity of low pH seepage from waste storage facilities is attributed to high concentrations of hydrolysable metals, namely Al^{3+} , Cu^{2+} , Fe(II) , Zn^{2+} and Mn^{2+} . Downstream water, evaporites and sediments have low to negligible acidities, which indicates the effectiveness of site water management measures and neutralising capacity of background surface and groundwater.

The study integrates the results from geochemistry of mine water, stable isotope geochemistry and acidity components to propose a mine water assessment model.

1.0 INTRODUCTION

1.1 Mine water signatures

Mine water assessment is generally carried out by the use of geochemical tools based on geochemical properties namely major cations and anions, hydrolysable metals, and pH. However, the conventional hydro-geochemical approaches may have deficiencies. The main potential deficiencies come from an inappropriate choice of tracing elements (Ghomshei and Allen 2000). Stable isotope compositions of water are increasingly used in mine water investigations (Bird and Mahoney 2000). In the near surface environments, isotopic signatures of surface and groundwater are essentially conserved and will only change as the result of mixing with water of different isotopic signatures (Bird and Mahoney 2000). Therefore, the signature may reflect the geochemical environment through which water has moved and water isotope analysis often allows the separation of water from different sources (Kendall and Caldwell 1998). Heavy isotopes of hydrogen and oxygen are considered as ideal tracers since they are two constituents of water and, therefore, they behave exactly as water is transported through watershed (McGuire and McDonnell 2007). Sulfur isotopes of

sulfate in surface freshwater environment are very stable during isotopic exchange processes (Lewicka-Szczebak et al. 2009). The sources of sulfate are generally determined by comparing isotopic composition of dissolved sulfate and those of sulfide minerals of possible sources found in literature (e.g. Taylor et al. 1984) or those collected and measured *in situ* (Dold and Spangenberg 2005). A combination of conventional hydro-geochemical approaches and stable isotope geochemistry is ideal for the determination of water sources and their connectivity.

1.2 Mine water acidity

There are generally two types of acidities in acidic mine water: (i) strong acidity of free proton (H^+) that is produced from the oxidation of sulfide minerals, and (ii) weak mineral acidity that originates from hydrolysable metals, which are abundant in acidic mine water. While strong acidity can be easily neutralised by neutralising capacity of natural water, residual weak acidity can exist in the environment in the form of evaporites in the dry seasons or precipitates (flocs) in streams. They can readily react in the wet season to generate acidity into the natural streams. Measurement of total and weak mineral acidity of acid mine water can be useful in identification of AMD hotspots. However, the application of types of acidity measurement for AMD fingerprinting at a mine site scale has been rarely studied. Previous studies have mainly focused on the characterisation of the buffering system of acidic mine water. Several methods have been assessed to quantify acidity components of acidic mine water. However, there are some uncertainties associated with those methods that need improvement (Totsche et al. 2003; Totsche et al. 2006).

This study aimed to better understand the reactions giving AMD and quantify the release and transport of sulfate and acidity components from the interaction of water with mine wastes, and *in situ* mineralisation, at a closed sulfidic mine site in central north Queensland. Water geochemistry and stable isotope techniques were used to identify origins and associations of different water types. Improved titration methods have been applied to determine the relative significance of various acidity components of mine-related water, namely the strong acidity by free proton ion and weak acidity by hydrolysable metals.

2.0 METHODOLOGY

2.1 Study Site

The mine site for the study is a closed large open-pit mine producing gold, silver and copper. The mine commenced mining operation in 1986, which was initially producing gold and silver using a heap leach operation of oxide ore. The operation changed to carbon in pulp processing of primary un-oxidised ore in 1989. Mining ceased in early 2001 due to the depletion of the ore body, though mineral processing still operated until March 2002 using low grade stockpiles. During the mining phase, the total annual ore that were processed reached 5.5 million tonnes. Originally, the mine site was developed on granite and metasediments. The ore body is hosted by units of Intrusive and Breccia Complex of highly altered volcanic breccias and felsic dykes. The main sulfides within the ore body and waste rocks are pyrite, chalcopyrite, galena and sphalerite. Sulfide contents are up to 5 % at the mine site proper and generally less than 2 % in the surrounding areas. The main gangue minerals are chlorite, carbonates, mica and quartz. Sulfate-bearing minerals mainly belong to alunite and jarosite groups.

Historic water quality monitoring data show an increasing trend in sulfate concentrations in surface waters of “near facilities” points since late 1980s. Measured sulfate concentrations in all on-site domains of site features usually exceed 1000 mg L^{-1} . The occurrence of acid mine

drainage (AMD) has been reported in some domains namely the dams at the base of the Eastern Waste Rock Dump (EWRD), the Mill Rejects and the Heap Leach stockpiles. pH levels at these sites range from pH 3.07 to 3.92. Sulfate and filtered metals and metalloids including As, Cd, Cu, Fe, Mg, Mn, Ni, Pb, Se, Zn and U are also at high concentrations. In general, the seepage sumps have relatively constant sulfate concentrations (approximately 4000 mg L⁻¹). The tailings sumps have high sulfate (3650 to 4230 mg L⁻¹), and As, Mo and Sr concentrations. Tailing sumps are also characterised by high total alkalinity and salinity (Na and Cl) concentrations.

2.2 Field Sampling and Lab Measurement

Surface and groundwater samples were collected at representative sites among the waste storage facilities at the mine site domains, and along the trajectory of the stream systems associated with the mine. Most of the samples were collected in the wet season from January to March, 2011. Tailings seepages were collected in the dry season (September, 2011) and the wet season (March 2013). Surface water samples were representatively collected at mid-stream as local creeks and mine sumps are relatively narrow and shallow. Groundwater samples were collected in both the dry (June, 2010) and the wet (March, 2011) seasons. Groundwater was pumped using a peristaltic pump from an approximate depth of 20 m and was freely flowed for 5 min to achieve a homogeneous condition. Deposited evaporites, precipitates and various sediments were collected inside the seepage impoundment sumps and along the streambed using simple random composite sampling techniques. Portions of approximately 2.0 to 3.0 g of scattered evaporites or sediments were randomly taken around the located seepage impoundment sumps or streambeds. The portions were mixed to form a composite sample.

Samples for filtered metals, Fe(II) were field-filtered using Millex-MF Millipore 0.45 µm mixed-cellulose-esters membrane filters by syringes. Metals (field-filtered and total) and Fe(II) were preserved by adding approximately 3 mL concentrated HNO₃ and HCl into 1 L water sample to pH < 2 (ALS 2013), respectively, following the standard methods by Clesceri et al. (1998). Samples for ²H and ¹⁸O in water were field-filtered into McCartney 28 mL glass screw cap jars, tightly sealed with zero headspace and were kept in cool at approximately 4°C, dark conditions to ensure no air intrusion. Water samples for sulfur isotope analysis were collected at selected sites including upstream creeks, shallow groundwater and surface water.

Water temperature, pH, electrical conductivity (EC) and redox potential (Eh) were measured in the field using WP-81 TPS pH-EC meter (TPS Pty. Ltd., Brisbane). The pH meter was calibrated in laboratory condition to pH 4.00 ± 0.05 and pH 7.00 ± 0.02 buffer solutions. The EC meter was calibrated in laboratory condition to 2.67 mS.cm⁻¹ ± 1.00 % standard solutions.

Water samples collected in the field were analysed for water quality data by the Australian Laboratory Services, Brisbane (ALS) with NATA ISO/IEC 17025 accreditation. δ²H, δ¹⁸O and δ³⁴S were analysed at the Stable Isotope Geochemistry Laboratory, University of Queensland with appropriate QA/QC procedures. Analytical uncertainties for δ²H and δ¹⁸O and δ³⁴S were better than ±2 ‰, ±0.1 ‰, and ±0.2 ‰ (1 Standard Deviation), respectively.

Acidity titrations for water samples, evaporites and sediment extracts were carried out using a Metrohm 902-Titrando model auto-titrator at a rate of 1 mL min⁻¹ using standard solution low-in-carbonate 0.1 N NaOH (0.1000 ± 0.0005 normality). The titrant (0.1 N NaOH) was standardised by titrating to a set of pH 8.7 with freshly made solutions that were prepared by dissolving approximately 0.1 g of potassium hydrogen phthalate (Chem-Supply, C₈H₅O₄K, 99.95 % purity) with 20 mL Milli-Q water. The error factor was automatically recorded by

averaging the results from three standardisations. All titrations were conducted at laboratory condition that was manually set at 20 °C. Acidity was reported as mg L⁻¹ of CaCO₃ equivalent by titrating to the set pH 8.3. The equivalence points can be identified by plotting average mmol NaOH added against the first derivative of the change in pH as a function of the change in mmol NaOH added ($\Delta \text{pH}/\Delta \text{mmol NaOH added}$). The equivalence points can be used to identify the species contributing to total acidities.

3.0 RESULTS AND DISCUSSION

3.1 Water Chemistry

According to mine water classification based on metal concentrations (Cd+Cu+Pb+Zn) by Plumlee et al. (1999), most creek water and nearby groundwater samples in the study area were classified as near-neutral (pH 5.5 – pH 9.0) and high- (1-100 mg L⁻¹ of metals) or low-metal (<1 mg L⁻¹ of metals). Most sump water samples near the Waste Rock Dump, Heap Leach and Mill Rejects areas were classified as acidic (pH 3.0 – pH 5.5) and having “high hydrolysable metal” or “extreme hydrolysable metal” concentrations (> 100 mg L⁻¹). Groundwater near the Heap Leach stockpiles (MLM21) and tailings seepage (SW23-Sp) was also classified as acidic (pH 3.0 – pH 5.5) and containing high-metal concentrations (1-100 mg L⁻¹). This showed that the Waste Rock Dump, Heap Leach stockpiles and Mill Rejects produced the most acidic and metal-rich seepage. Creek water samples collected in January, 2011, generally had high pH and low concentrations of metals and metalloids. Sulfate concentrations of creek water samples increased above upstream background concentrations as they entered the mine lease boundary that suggested an influence of sulfate-rich sources of water. The above observations indicated that mine water was having limited influence on dissolved constituents in creek water with the exception of sulfate concentrations. The influences of mine seepage to natural creeks could also be limited by the high background neutralizing capacity of creek water with alkalinity values ranging from 313 to 680 mg L⁻¹ CaCO₃ equivalent that might originate from the local basement system surrounding the mine site, which comprises of metamorphosed siltstones, sandstones and greywacke. The limited influence of mine water on creek water might also be attributed to water management practices at the mine site such as pumping back and containment of mine water.

3.2 Stable Isotope Compositions of Water

The stable isotope plot of $\delta^{18}\text{O}$ against $\delta^2\text{H}$ for water samples is shown in Figure 1 with the inclusion of the Global Meteoric Water Line (GMWL (Craig 1961): $\delta^2\text{H} \text{‰} = 8.13\delta^{18}\text{O} \text{‰} + 10.8$) and the Local Meteoric Water Line of Fitzroy region, Queensland (LMWL (Vink et al. 2009): $\delta^2\text{H} \text{‰} = 7.703\delta^{18}\text{O} \text{‰} + 9.018$). The non-evaporated water samples include the waste rock seepage, creek water and the rainwater, which have similar isotopic compositions (-38.0 ‰ to -30.0 ‰ for $\delta^2\text{H}$ and -5.7 ‰ to -4.8 ‰ for $\delta^{18}\text{O}$). Tailings seepage samples are highly enriched in ^{18}O and ^2H (-25.3 ‰ to +1.8 ‰ for $\delta^2\text{H}$ and -3.2 ‰ to +2.8 ‰ for $\delta^{18}\text{O}$). The groundwater samples in seepage zones are distinctly enriched in ^{18}O and ^2H compared with non-evaporated water and are significantly depleted compared to tailings seepage. The stable isotope geochemistry of water defines a local evaporation line (inset Figure 1: $\delta^2\text{H} \text{‰} = 4.97\delta^{18}\text{O} \text{‰} - 9.11$) with a strong correlation ($R^2 = 0.958$). The slope of 4.97 is consistent with the general evaporation line slopes ranging from 3 to 6 (Gibson et al., 1993; Ghomshei & Allen, 2000).

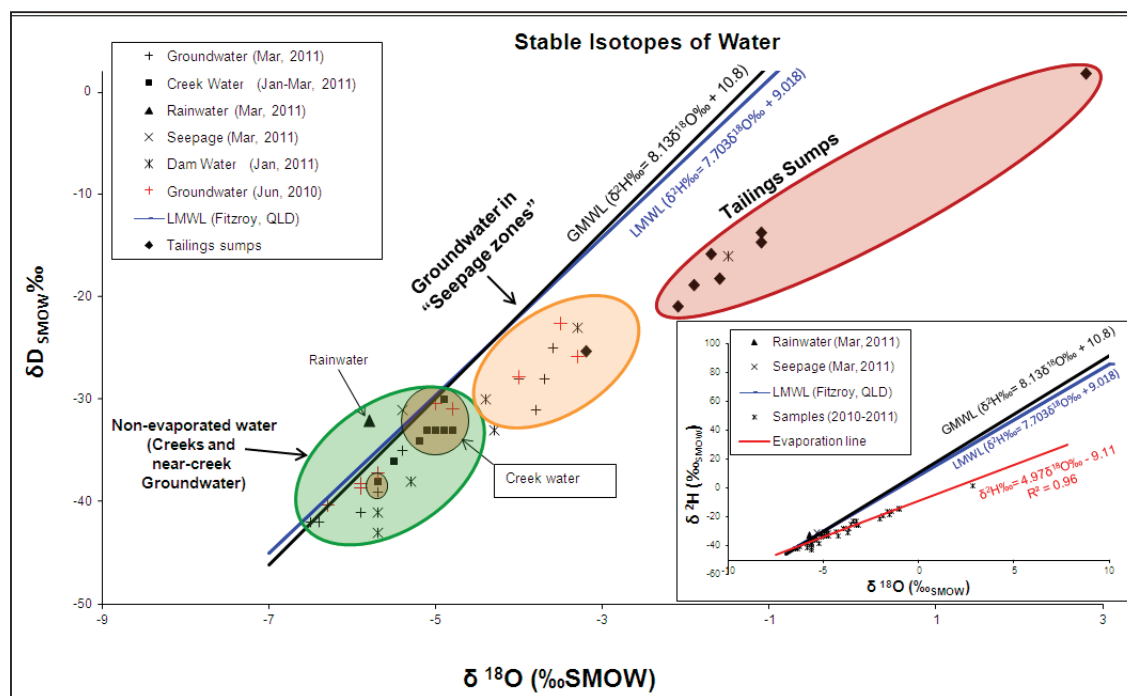


Fig. 1. Stable isotope plot of $\delta^{18}\text{O}$ vs $\delta^2\text{H}$ for water

The similar isotopic compositions of the EWRD seepage and creek water samples compared to the rainwater sample suggested a common origin and drainage history for EWRD seepage and stream water. This also suggested that the seepage from the EWRD was less affected by evaporation, which might be due to seepage management measures where seepage from the EWRD was constantly pumped into the mine pit. The enrichment of tailings sump samples suggested strong evaporation effects. Near-creek groundwater (B1 and B3) had an isotopic composition similar to near-facilities groundwater and was distinctly enriched compared to other near-creek groundwater samples. B1 and B3 also had higher major ion concentrations compared to other near-creek groundwater samples. The combination of the similarities in water stable isotope signatures and geochemical compositions of near-creek groundwater samples B1 and B3 and those near mine waste facilities has confirmed the presence of inferred fracture zones or “seepage zones”.

3.3 Sulfur Isotope Compositions

Most of the surface water and groundwater samples inside the mine lease boundary had sulfate-sulfur stable isotope compositions within the range of sulfides (+4.9 ‰ to +8.8 ‰) and to some extent the alunite groups (+8.0 ‰ to +9.8 ‰). This suggested that sulfate of seepage sumps and groundwater at the mine site originated mainly from sulfide oxidation.

The background water samples had clearly distinct sulfate-sulfur stable isotope signatures (+16.2 ‰ to +22.04 ‰) compared to those of sulfide or alunite groups. This indicated that the upstream creek water contained a different source of sulfate. Water and groundwater of creeks inside the mine lease boundary had similar sulfate-sulfur isotopic signatures lying between those of the background and mine water. This indicated the mixing of background sulfate with mine-related sulfate. There was also an enrichment of sulfate-sulfur isotopes of groundwater in the wet season (March 2011) compared to those in the dry season (June

2010). This would be due to the recharge of surface water to groundwater during the wet season that caused mixing of background sulfate with mine-related sulfate in groundwater.

3.4 Acidity and Acidity Components

The seepage samples from Potentially-Acid-Forming (PAF) materials including the EWRD, Mill Rejects and Heap Leach Stockpile have total acidities, ranging from 3980 to 13,000 mg L⁻¹ (CaCO₃ equivalent). The tailings seepage samples around tailings dams have total acidities ranging from 20.0 to 110 mg L⁻¹. Samples from a major creek downstream from waste rock dump have a decreasing trend in total acidities from 216 to 18 mg L⁻¹ near the mine lease boundary. Small creeks near the mine waste facilities have varying total acidities ranging from 6.00 to 133 mg L⁻¹. Large creeks have negligible water total acidities along their trajectories from upstream background water to inside the mine lease boundary.

The relative contributions of the various components to total acidity estimated based on the evaluation of titration curves, which is improved by the use of first derivative plotting (an example is given in Figure 2), are given in Table 1.

Table 1. Estimated acidity components of representative acidic seepage samples

Sample	Acidity component (mg L ⁻¹ CaCO ₃ Equivalent)			
	H ⁺	Al ³⁺	Cu ²⁺ / Fe ²⁺ / Zn ²⁺ / Mn ²⁺	Total
MMD	48.95	3628	1447	5124
MHD	87.89	5575	4030	9605
MHED	0.000	3347	2417	5764
UPT	64.90	8831	3998	12890
SGD	97.43	3193	686.4	3977

Table 1 shows that strong mineral acidities (H⁺) contributed only small relative proportions to total acidities. The waste rock seepage samples (MMD, MHD and MHED) have a range of relative proportions of strong mineral acidity varying from 0.00 % to approximately 1.00 %. These proportions in seepage samples from mill rejects (UPT) and heap leach stockpiles (SGD) are 0.50 % and 2.45 %, respectively. Hydrolysable metal ions particularly aluminium (Al³⁺), ferrous iron (Fe²⁺), copper (Cu²⁺), zinc (Zn²⁺) and manganese (Mn²⁺) produce a large proportion of the total acidity. In particular, Al³⁺ contributes the largest relative proportion ranging from 58.0 % to 70.8 % in waste rock seepage samples; approximately 64.5 % in mill rejects seepage sample; and 80.3 % in heap leach seepage sample. A combination of other metal ions including Cu²⁺, Zn²⁺, Fe²⁺ and Mn²⁺ contribute to the rest of weak mineral acidities. The doubly charged metal ions have an overlapping range of buffering and therefore, each contribution to the total acidity can't be easily deconvoluted. However, the measurement of the concentration of an individual metal ion could be used to estimate its contribution to total acidity by calculating each contribution. There is a direct relationship between the hydrolysable metal ions in water samples and their respective contributions to acidity. Acidic seepage samples are characterised by very high concentrations of hydrolysable metals (> 100 mg L⁻¹). Thus, acidic seepage samples have distinctly higher total acidities than the tailings seepage and creek water. Tailings seepage and creek water are all characterised by low concentrations of hydrolysable metals (<1 mg L⁻¹ of metals) and thus have low to negligible acidities.

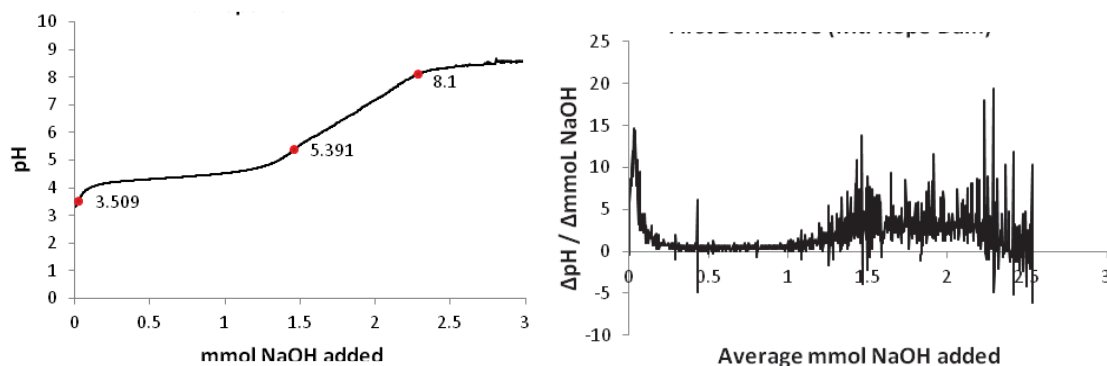


Fig. 2. An example of titration curve (top) and first derivative (bottom) plots of acidic seepage

The evaporites that formed from seepage of waste rocks and mill rejects have extremely high acidity producing capacities of up to 190,300 mg CaCO_3 equivalent kg^{-1} . The precipitates have much lower acidity producing capacities of up to 18,980 mg CaCO_3 equivalent kg^{-1} . The evaporites and precipitates from tailings seepage have negligible acidity producing capacities, while the acidity producing capacities of evaporites and sediments collected along mine-affected creeks are all negligible. However, the creek systems associated with the waste rock and the mill rejects storage facilities have some low acidity producing capacities of up to 72.4 mg CaCO_3 equivalent kg^{-1} .

3.5 Integrated Geochemical Approach for Mine Water Assessment

A new model is proposed to show how the designated contributions to mine water acidity are inter-related. A model can be created by the integration of results for three major components of mine water assessment from source identification, transport elucidation and quantification of mine-site water contaminant release. These components are: (i) geochemistry of mine water; (ii) stable isotope geochemistry; and (iii) acidity measurements. The model will, therefore, serve as an interpretative tool for the elucidation of the source, the transport and the quantification of contaminant release of acidity from a mine site. The conceptual framework of the integrated mine water assessment model (IMWAM) is illustrated by Figure 3.

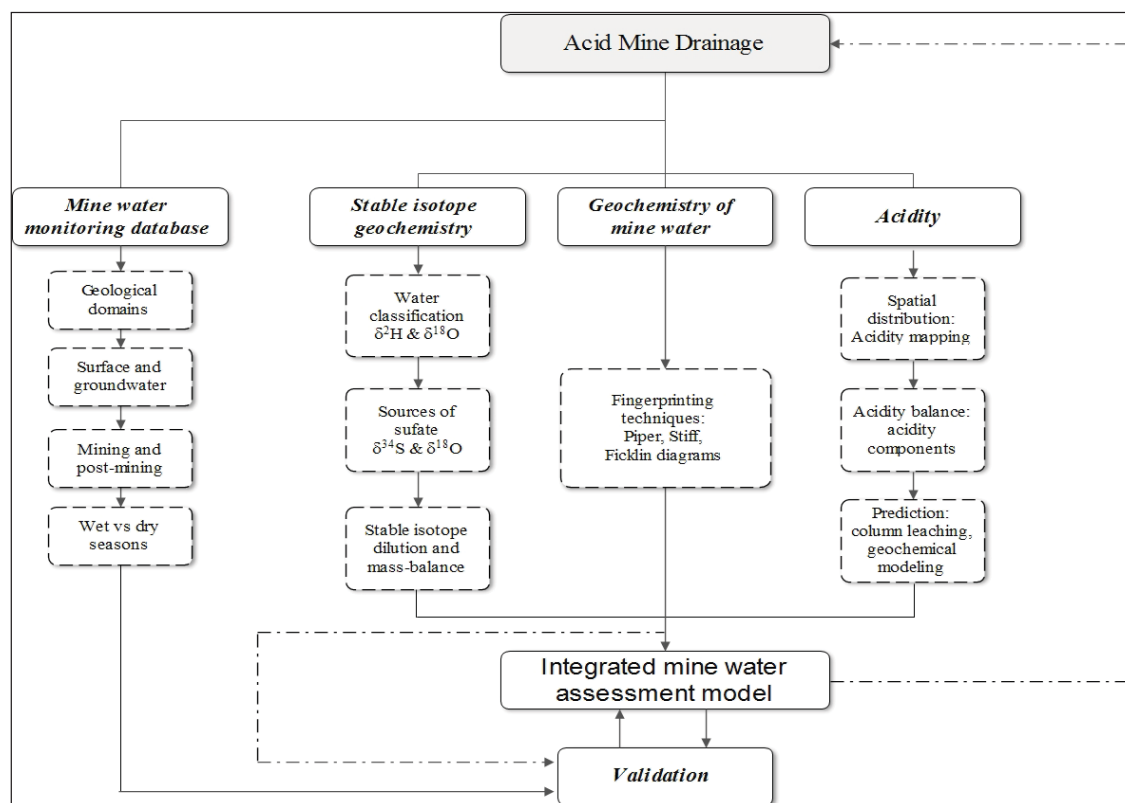


Fig. 3. Conceptual model for the integrated mine water assessment model

The core of the integrated mine water assessment model includes the mine water mixing index (MWMI) and stable isotope geochemistry ($\delta^{34}\text{S}$ -sulfate). The MWMI enables quantification of the total contribution of mine water to a water sample, or in other words, the relative proportion of mine water contribution to a water sample. The total contribution of mine water to a water sample is the summation of mine water contribution values (MWCV) for key parameters such as strong and weak mineral acidity, sulfate concentration and hardness to consider the components of the acid generation and neutralisation reactions, which together define the mine water characteristics. In order to have a quantifiable MWMI for a water sample, the mine water contribution values of key parameters need to have a common unit, which allows the additive effect. Therefore, a simple linear mine water mixing model, which is derived from the sulfur isotope mixing model of Wright and Nordstrom (1999) (Figure 4) is used to calculate the mine water contribution values for the key parameters.

It is identified that most water samples around the mine site will belong to one of the three types: (i) background water; (ii) mine-related water; and (iii) mixed water.

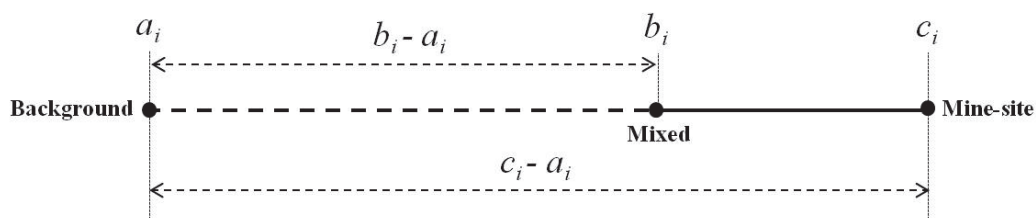


Fig. 4. The conceptual illustration for the linear mine water mixing model

The key parameter i will have two end-member values namely the background value (a_i), and the mine-site value (c_i). The assessed water sample has the value b_i for the parameter i . Thus, mine water contribution value for the parameter i (MWCV _{i}) of the assessed water sample can be calculated by Eqn. [1].

$$\text{MWCV}_i = \frac{b_i - a_i}{c_i - a_i} \quad [1]$$

The summation of mine water contribution values of the designated key parameters is considered to be the mine water mixing index (MWMI) of the assessed water sample. The MWMI can be calculated by Eqn. [2]).

$$\text{MWMI} = \sum_{i=1}^n \text{MWCV}_i \quad [2]$$

The application of the MWMI and stable isotope geochemistry ($\delta^{34}\text{S}$ -sulfate) into the study mine site has clearly classified water types in this study (Figure 5).

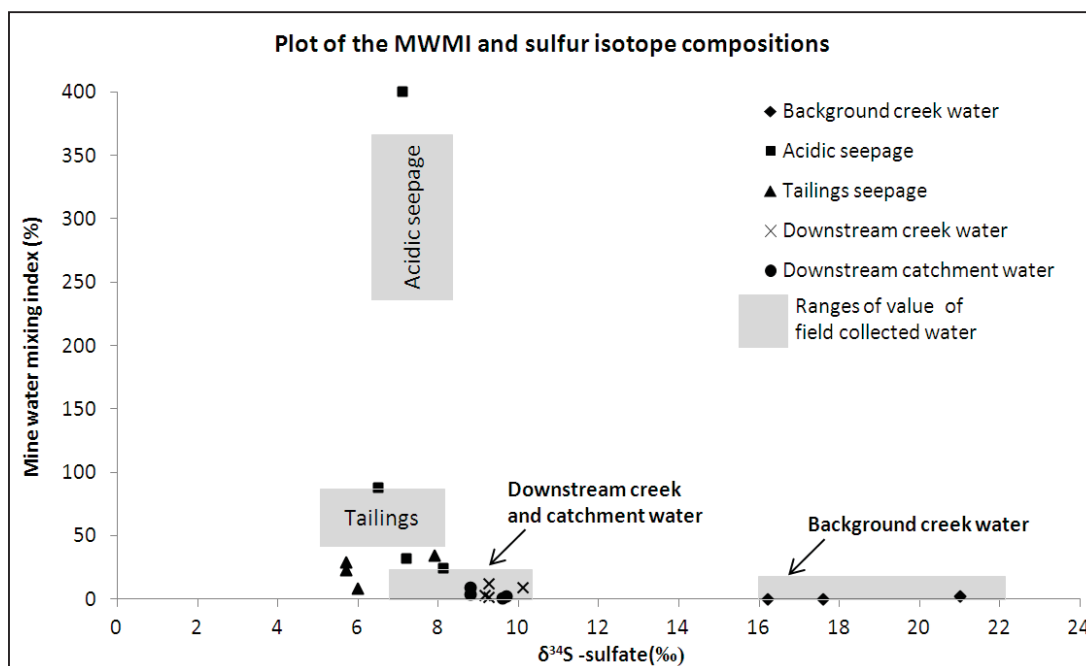


Fig. 5. The IMWAM that plots the MWMI and sulfur isotope compositions

Figure 5 includes the ranges of MWMI values of field-collected water samples and those of selected similar 'monitoring' samples during an extreme rainfall event. The MWMI has distinguished four types of water namely: (i) Background upstream water; (ii) Acidic mine seepage; (iii) Tailings seepage; (iv) Downstream water. The majority of the selected acidic and tailings monitoring samples have much lower MWMI than the field-collected samples. This is probably due to the intensive rainfall during the event that caused dilution of the major chemical components of selected monitoring samples. However, the selected background and downstream creek water monitoring samples have similar MWMI compared to the respective field-collected samples. The MWMI of selected monitoring samples showed a similar pattern of mixing and neutralising processes between background water and mine water that was demonstrated by the MWMI of field-collected samples.

4.0 CONCLUSION

Mine water geochemistry, stable isotopes and acidity titration were effectively used for geochemical fingerprinting of water from different mine facilities at a closed mine. The Eastern Waste Rock Dump (EWRD), Heap Leach and Mill Rejects stockpiles produced seepage with high total acidities, which were attributed by extremely high concentrations of hydrolysable metals, namely Al^{3+} , Cu^{2+} , Fe^{2+} , Zn^{2+} and Mn^{2+} , to the weak mineral acidity. Weak mineral acidity contributes a large proportion (> 98 %) of the total acidity and is retained as secondary minerals and evaporites in the dry season with very high acidity producing capacities that pose a potential long-term risk to natural environment by releasing acidity during the wet season. In contrast, tailings seepage had low hydrolysable metal concentrations, low acidity, which might be due to the use of highly alkaline cyanide solution during gold extraction. Seepage from the mine has locally impacted surface and groundwater especially around the major mine waste facilities along inferred seepage zones. There were also indications of the mixing between mine water and the creeks based on sulfur isotopes. However, downstream water, evaporites and sediments had low to negligible total acidities,

which indicates that the neutralising capacity of background surface and groundwater has neutralised the acidity from AMD. The limited impact or influence that mine water had on the water quality of local streams is also probably related to the effectiveness of site water management measures.

The integrated mine water assessment model integrates three major components of mine water assessment namely (i) water chemistry (hardness), (ii) stable isotope geochemistry ($\delta^{34}\text{S}$ -sulfate), and (iii) acidity. The innovation of the IMWAM is the creation of the MWMI that integrates the water chemistry and acidity components. Therefore the IMWAM that plots the MWMI together with the $\delta^{34}\text{S}$ -sulfate values not only helps to classify different types of water that associate with the mine environment, but also shows the mixing pattern of mine water with background water in the downstream. Such an index will be useful for mapping the extent of potential impact of mine water after extreme rainfall events on downstream surface and groundwater. It can be used as a scientific and accurate basis, to monitor the success of rehabilitation, and for discussions among all stakeholders.

5.0 ACKNOWLEDGEMENT

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GYPSUM SLUDGE AND TAILINGS AT BRUKUNGA MINE USED AS PARTIAL CLAY SUBSTITUTE FOR MAKING CONSTRUCTION MATERIALS: A LAB SCALE STUDY

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ABSTRACT

Gypsum based sludge, the main waste by-product of the Acid and Metalliferous Drainage (AMD) treatment process, is an environmental liability apart from the waste rock and tailings facing the mining industry. The AMD treatment plant at the former Brukunga mine generates thousands of tonnes of sludge each year which are currently stored at the mine site waiting for an environmentally safe disposal to be defined. The growing volume and mass of sludge may lead to severe environmental issues in a long-term perspective as the heavy metals in the sludge are likely to be remobilised to the downstream environment as a result of rain water and the acid runoff in the mine site after its inherent alkalinity has been depleted.

This paper presents a lab-scale study to use the gypsum sludge and tailings sourced from Brukunga mine as partial clay substitute (up to 25% sludge and 20% tailings) for brick making. Test results of weight loss on ignition, water absorption and leachability of fired samples indicate that bricks with 5% sludge and up to 20% tailings have similar properties to waste-free bricks. The high temperature vitrification trial provided an effective way for the stabilisation and solidification of the sludge and tailings, leading to a new option for AMD-derived waste management in mine sites.

1.0 INTRODUCTION

Gypsum based sludge, the main by-product of the Acid and Metalliferous Drainage (AMD) treatment plant at Brukunga mine, is a man-made mine waste material apart from the waste rock and tailings. The sludge has been classified as 'intermediate waste' due to the rich abundance of heavy metals, which makes it impracticable for agricultural use (PIRSA, 2009). Moreover, a potential high levy plus freight cost will make it expensive for a contaminant landfill disposal option. Currently, the dry sludge is transported from the treatment plant to the southern bench next to the high wall where it is stockpiled. The growing volume and mass of sludge may lead to severe environmental issues over the long-term. Until an environmentally safe disposal strategy is found, heavy metals in the sludge are likely to be remobilised to the downstream environment. This is due to repeated seasonal exposure to rain water and acid runoff from the high wall after its inherent alkalinity has been depleted.

To immobilise the toxic metal contents in industrial waste materials, high temperature firing processes have been developed in recent decades and used in numerous studies all over the world by using waste materials as partial clay substitute for making construction materials (e.g., brick, paver and tile) (e.g., Alleman and Berman, 1984; Tay, 1987). These waste materials include, but are not limited to, sewage sludge produced at industrial waste water treatment plants (Weng et al., 2003; Liew et al., 2004), fly ash slag (Lin, 2006), waste steel slag (Shih et al., 2004), kraft pulp production residues (Demir et al., 2005) and sawdust and

tobacco residues (Demir, 2008). The advantage of this method is that wastes can be converted to environmentally-friendly materials and the economic return may compensate the cost.

This paper presents a lab-scale study to use the dry gypsum sludge and tailings sourced from Brukunga mine as partial clay substitute (up to 25% sludge and 20% tailings) for making construction materials. The effects of increasing waste proportion with the clay mix are examined by assessing the qualities and properties of fired brick samples.

2.0 EXPERIMENT

Gypsum sludge, oxidised tailings and clay mix are the raw materials used in this study. The sludge is produced at the AMD treatment plant and the oxidised tailings are sourced from the tailings dam (from surface to ~1 m deep) at Brukunga mine. The clay mix is obtained from a local quarry that is dedicated to the brick/paver making industry. It is a mixture of clay, red sand and slate. The sludge is composed of 65-75% crystallised gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) and 25-35% amorphous solid metal hydroxides and oxy-hydroxides (i.e., Al, Fe, Mn, Cd, Ni and Zn). The tailings consist of sand-like quartz and silicate minerals with less than 1.5 wt% of sulphur as sulphide (pyrite). An organic content of the tailings is summarized of results is found in Table 1.

Table 1. Organic carbon composition of samples of tailings at various depths.

Depth m	Test Pit 1			Test Pit 2		
	DOC ¹ mg/L	TOC ² mg/L	TOC in Soil ³ %	DOC ¹ mg/L	TOC ² mg/L	TOC in Soil ³ %
0.00	5	6	0.82	7	6	0.58
0.10	36	36	0.42	3	3	0.17
0.30	3	4	0.18	4	3	0.15
0.50	3	4	0.12	3	3	0.11
0.75	4	4	0.16	4	4	0.16
1.00	4	4	0.23	4	4	0.12
1.50	3	4	0.19	3	2	0.16
2.00	3	3	0.29	3	3	0.24
2.50	3	4	0.15	3	3	0.12
3.00	4	4	0.15	4	4	0.15
3.50	3	3	0.16	3	3	0.16
4.00	4	4	0.20	3	3	0.17

Notes:

1. EP002: Dissolved Organic Carbon (DOC), LOD⁴ = 1 mg/L
2. EP005: Total Organic Carbon (TOC), LOD = 1 mg/L
3. EP003: Total Organic Carbon (TOC) in Soil, LOD = 0.02 %
4. LOD = Limit of Detection

Standard experimental procedures were followed to make waste-free and waste-bearing bricks as below: (1) clay mix, sludge and tailings were weighed on a dry basis, watered and mixed evenly; (2) the mixed wet clay was placed in a mould and then compressed to be a wet pellet by a hydraulic pump; (3) the wet pellets were dried in an oven for 24 hours at 90 °C; (4) the dried pellets were fired in a lab kiln for ~3 hours at 1090 °C.

Powder X-ray diffraction (XRD) patterns were collected on a Huber Guinier Imaging Plate G670 with $\text{CoK}\alpha 1$ radiation ($\lambda = 1.78892 \text{ \AA}$) generated at 35 kV and 34 mA. Each sample was ground in an agate mortar using acetone, spread on a Mylar film, and mounted onto the oscillation unit. XRD patterns were collected for 30 min, or re-collected for longer periods to

obtain high-quality patterns for qualitative phase analysis. X-ray fluorescence (XRF) data were collected by an Olympus Delta handheld XRF analyser which used a silicon drift detector with X-ray generated by a 40 kV tube. Calibration was carried out prior to data collection to enable the precision of XRF measurement.

3.0 RESULTS AND DISCUSSIONS

3.1 Appearance

Fired brick samples are shown in Figure 1. Compared to the waste-free standard, the colour of sludge-containing bricks is gradually diluted with increasing sludge content, and only samples with 5% sludge are close to the standard. However, there is no visible difference between tailings-containing samples and waste-free control (industry standard mix) and the colour and appearance keep almost unchanged upon tailings addition.

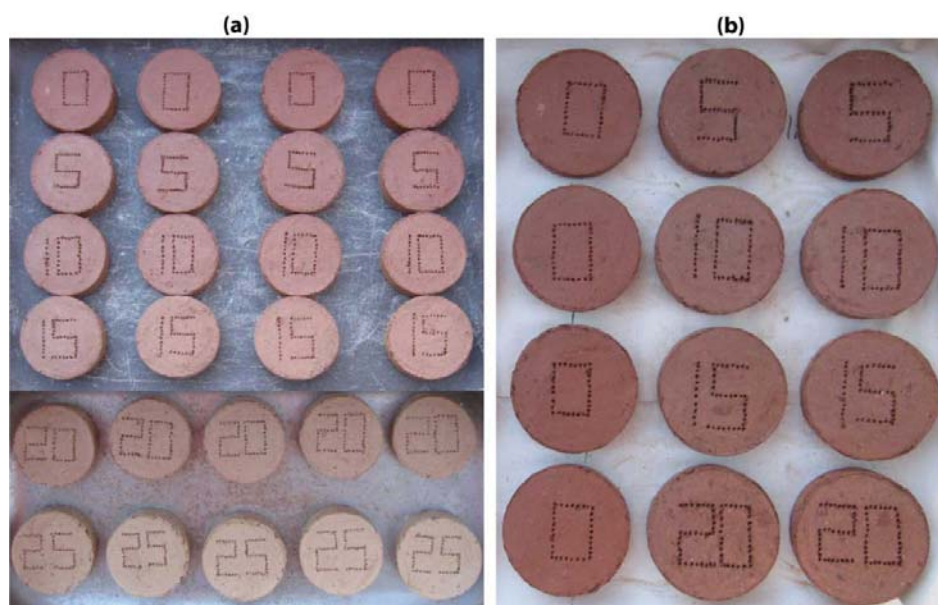


Fig. 1. Appearance of fired brick pellets: (a) waste-free and gypsum/metal hydroxide sludge-containing pellets; (b) waste-free and tailings-containing pellets. Note the numbers on the pellets indicate the waste proportion before firing process.

3.2 XRD Mineralogical Analysis

X-ray diffraction patterns have qualitatively confirmed the mineralogy of sludge, tailings and clay mix raw materials used in this study (Figure 2a). At present, there are no quantitative measurements of composition completed. The main phases of clay are quartz, kaolinite and muscovite and for tailings are quartz, feldspar series minerals and muscovite. A drying process for 24 hours at 90 °C does not change the phase of clay and tailings. Only the crystallised gypsum in the fresh sludge has been identified before the drying process, and there is a main phase transformation from $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$ (before the drying process) to $\text{CaSO}_4 \cdot x\text{H}_2\text{O}$ ($0 \leq x \leq 2$, after the drying process) as a result of dehydration process at 90 °C.

The clay phase transformation from kaolinite ($\text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4$) to mullite ($\text{Al}_6\text{Si}_2\text{O}_{13}$) during the firing process has been well understood and summarised (Brindley and Nakahira, 1954; Chen et al., 2004). This phase transformation is confirmed by the X-ray patterns of waste-free standard before and after the firing process in our study (Figure 2a, b).

X-ray patterns of tailings-containing samples bear strong similarity to that of the waste-free control (Figure 2b). This indicates that quartz in the tailings persists after the firing process while the organic matter and some amorphous materials appear to have been removed during the firing process. Two new phases – anhydrite (CaSO_4) and anorthite ($\text{CaAl}_2\text{Si}_2\text{O}_8$) have been identified in the fired sludge-containing samples and their amounts increase as the sludge fraction increases. The presence of anorthite is a strong evidence that gypsum undergoes a decomposition process (Liu et al., 2013) and can be recrystallised with the aluminosilicate in the clay (i.e., kaolinite) to form new mineral phases (anorthite in our study). It is noted that the fired brick samples with 10% tailings, 20% tailings and 5% sludge have a similar XRD pattern to that of the control.

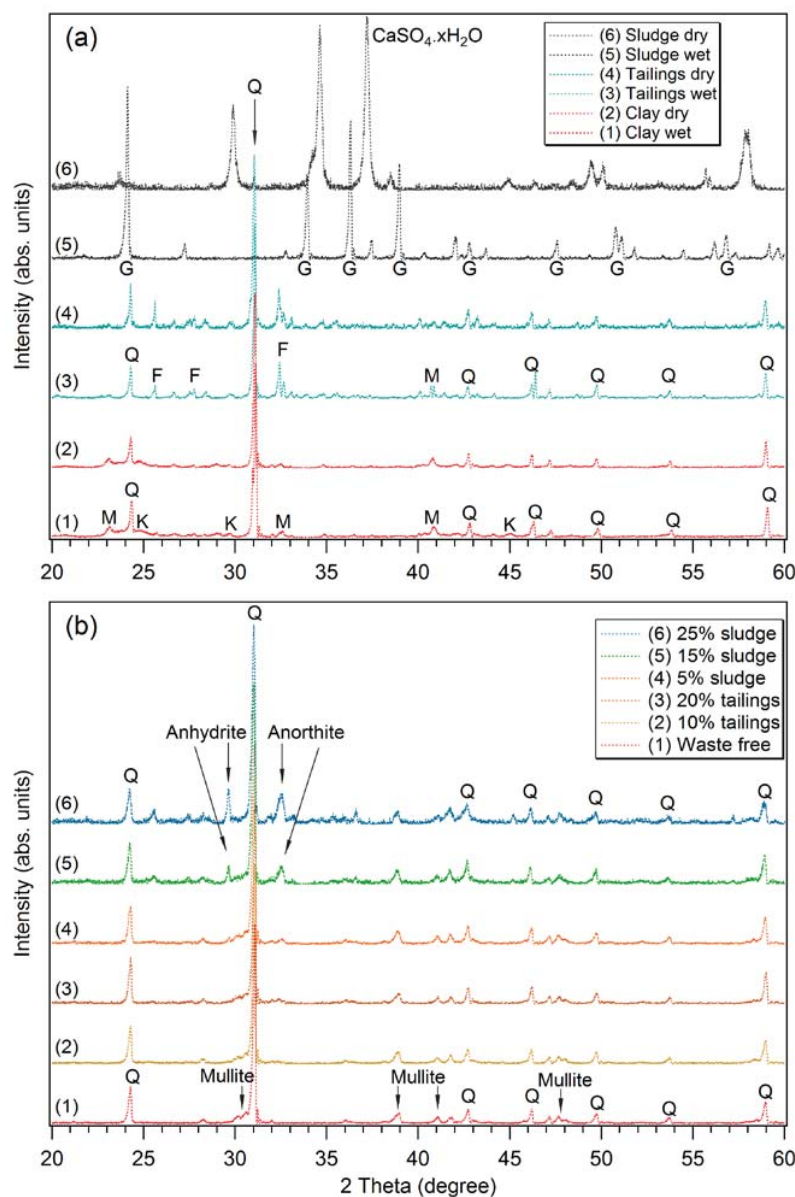


Fig. 2. XRD patterns of (a) raw materials and (b) fired brick samples (Q: quartz; G: gypsum; F: feldspar series minerals; K: kaolinite; M: muscovite). Patterns collected before drying process are labelled as 'wet' and those after drying process labelled as 'dry'. Clay is understood as 'clay mix'.

3.3 XRF Elemental Analysis

The XRF data of some typical elements in sludge-containing samples are shown in Figure 3 as a function of sludge content. With sludge proportion increased from 0 to 25%, there is a decrease of Si, K, Ti and an increase of S, Ca, Mn, Fe and Zn in the fired bricks while Al, V, Cr, Co, Ni, Cu, As, Zr, Cd and Pb keep at near constant level (within errors). This is due to Si,

K and Ti are richer in clay and elements like S, Ca, Mn, Fe and Zn are more abundant in the sludge while the contents of other elements are similar in both materials. There is a dramatic increase of Ca and S with increasing sludge proportion, which is attributed to the contribution of the anhydrite phase in fired bricks that has been identified by the XRD mineralogical analysis (see section 3.2).

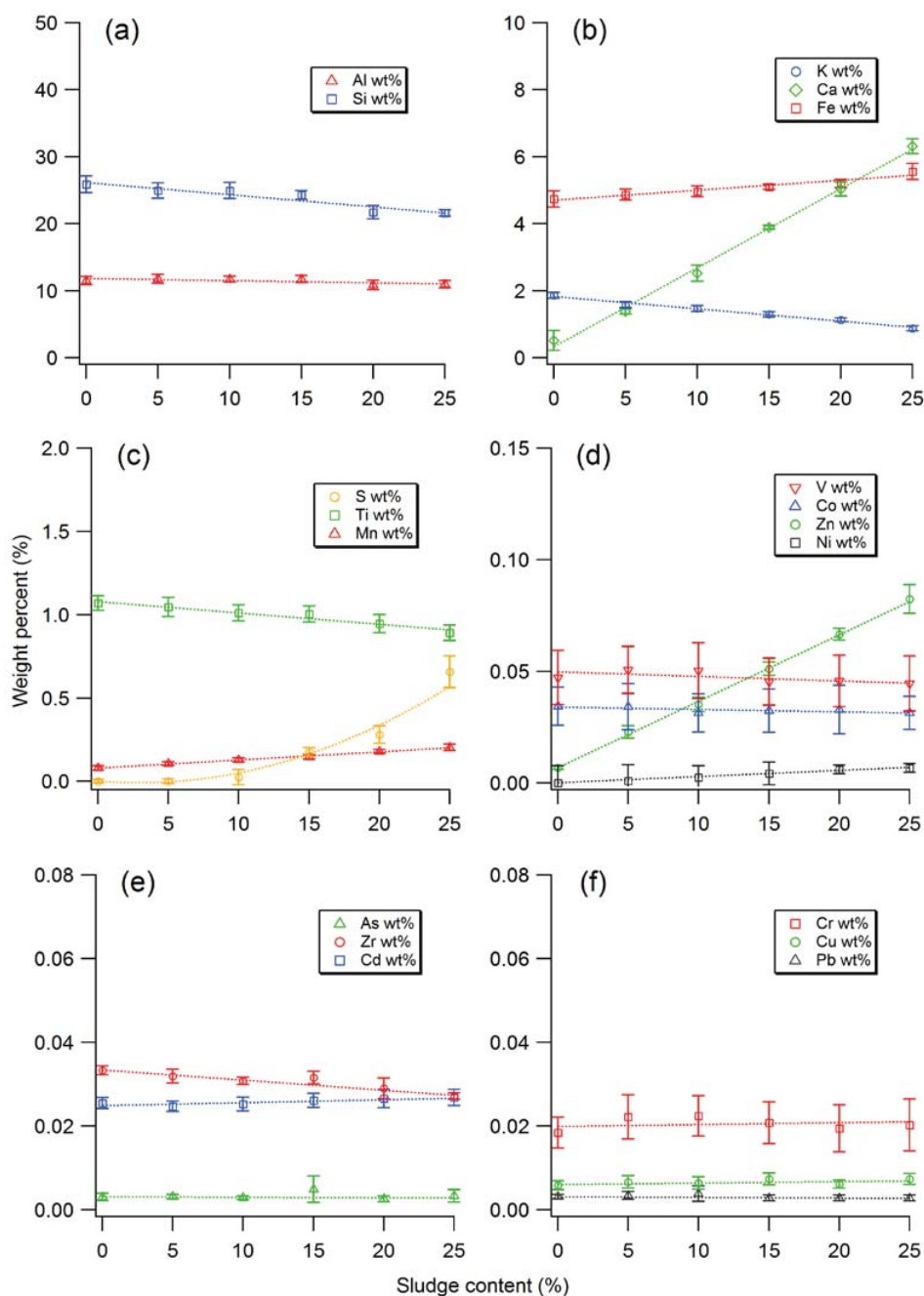


Fig. 3. XRF data of fired samples as a function of sludge content (dotted lines represent the fits of each dataset)

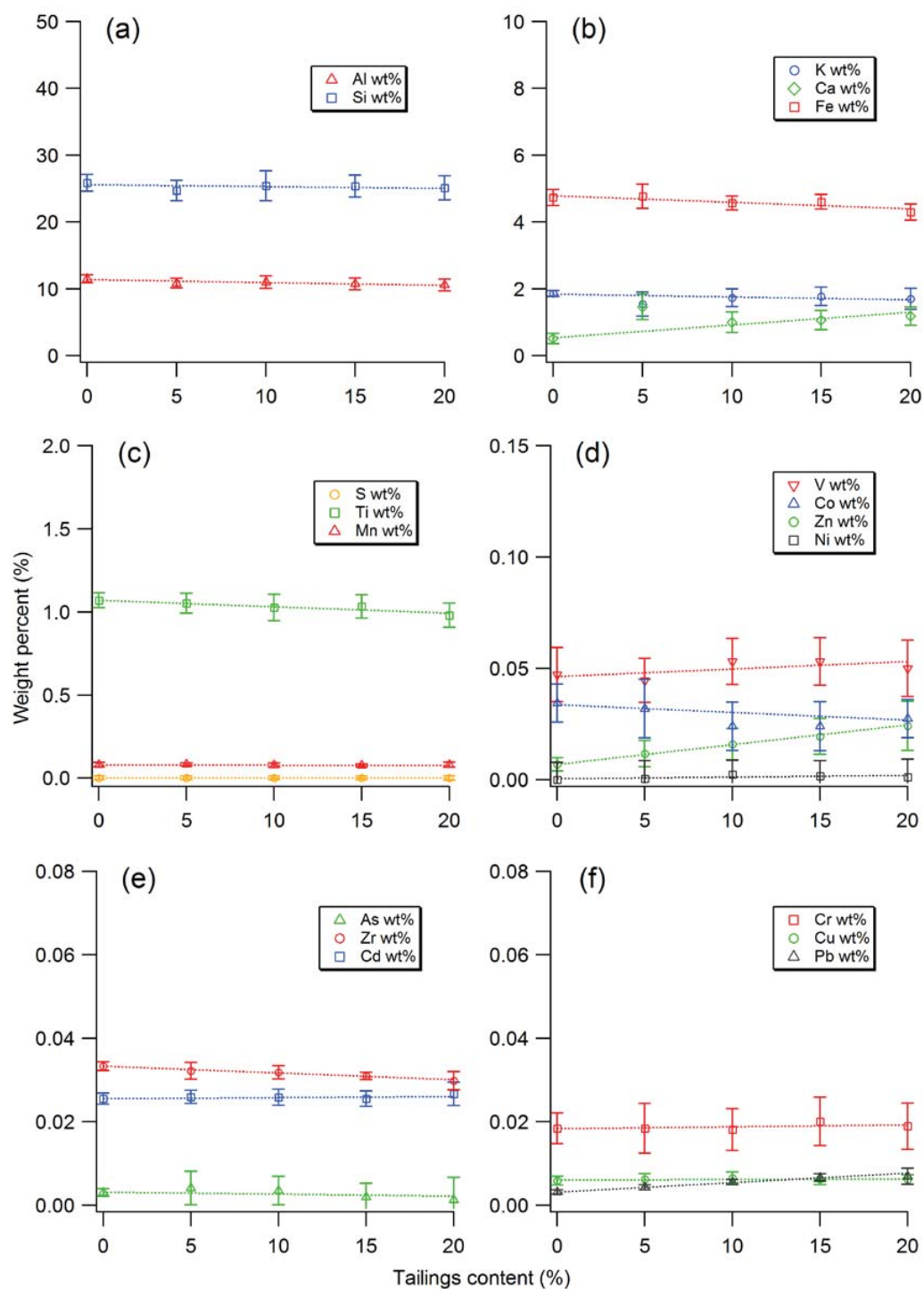


Fig. 4. XRF data of fired samples as a function of tailings content (dotted lines represent the fits of each dataset)

For tailings-containing samples, the XRF data have shown a few different trends with increasing tailings proportion (Figure 4). For example, there is only a slight increase of Ca and Zn upon tailings addition while the weight percentage of other elements keep at constant level or vary in a limited error range. Evidence from the raw material XRD plots for tailings and clay mix (Figure 2a) and some minor variations of elemental composition in the XRF plots for tailings with clay mix (Figure 4), show that clay mix and tailings possess similar mineralogical composition. And more importantly, addition of tailings into clay does not affect the kaolinite to mullite phase transformation upon firing process. Overall, the XRF results are consistent with XRD mineralogical analysis.

3.4 Weight Loss on Ignition and Water Absorption

Weight loss on ignition (LOI) and water absorption are key factors to assess brick quality and durability. LOI is measured during the firing process at a temperature around 1090°C. The lower the LOI, increases likelihood of producing a strong brick, less water infiltration into the brick, better resistance to weathering and corrosion, and longer durability. These two parameters rely on the internal structure of the brick, which is dependent on the chemical and mineralogical properties.

Figure 5a shows that increasing tailings proportion results in a slight decrease in brick LOI for tailings-containing samples. In contrast, brick LOI increases significantly with sludge addition from 0 to 25%. The larger LOI of the sludge containing bricks is most likely due to water losses, decomposition of gypsum before and after the firing process up to 1090°C.

It can be seen from figure 5b that water absorption of tailings-containing bricks remains constant (within error) with increasing tailings proportion. However, water absorption increases dramatically with increasing sludge content. Only brick samples with 5% sludge are comparable to the control in values of water absorption and weight loss on ignition.

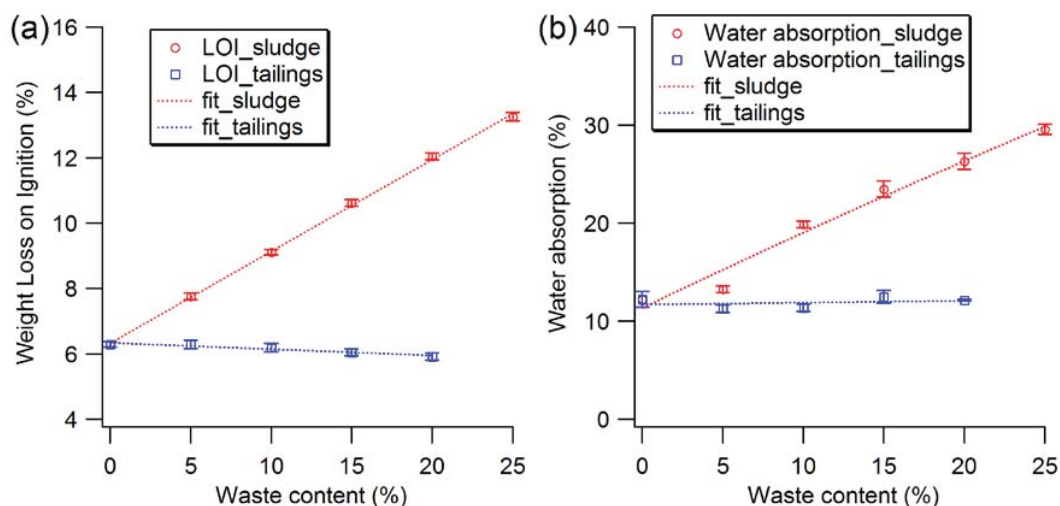


Fig. 5. Weight loss on ignition and water absorption as a function of waste content of the brick (dotted lines represent the fits of each dataset).

3.5 Leachability

The leachability tests of unfired and fired brick pellets were carried out following the Australian Standard of AS4439 bottle leaching procedures (EPA SA, 2010). By comparing the leachability data of unfired and fired bricks (Table 2), it is apparent that the leachability of sludge-containing samples has been largely reduced by the firing process. This is presented as decreased concentrations of SO_4^{2-} , Mg, Al, Ca, Mn, Co, Ni, Zn and Cd. Concentrations of K, Ti, Cr, Fe, Cu and Pb remain unchanged, or vary very little, and only concentrations of V and As increase. Similarly, for tailings-containing samples (Table 3), only concentrations of Al, V, Fe and As are increased by the firing process and most other elements have been effectively immobilised. All leachability data meet the regulation criteria issued by EPA SA (EPA SA, 2010). It suggests that the amorphous metal hydroxides and oxy-hydroxides in the sludge have been largely recrystallised with clay minerals to form stable metal aluminosilicates.

Table 2. Results of the leachability of brick samples as a function of sludge content (mg L^{-1} is used as the unit of concentration)

Sludge content (%)		0	5	10	15	20	25	LOD*
Sulfate as SO_4^{2-}	unfired	15	961	2040	2050	1860	1880	1
	fired	2	142	404	546	883	1740	
Calcium	unfired	14	391	906	905	952	957	1
	fired	8	59	161	218	353	684	
Magnesium	unfired	9	46	99	109	129	137	1
	fired	<1	<1	<1	<1	1	<1	
Potassium	unfired	5	3	4	4	4	4	1
	fired	6	6	8	6	7	6	
Aluminium	unfired	<0.1	4.2	10.5	11.3	1.6	0.4	0.1
	fired	1.1	0.8	0.6	0.9	0.9	0.6	
Arsenic	unfired	<0.005	0.04	<0.005	<0.005	<0.005	<0.005	0.005
	fired	0.023	0.453	0.648	0.51	0.594	0.658	
Cadmium	unfired	<0.001	0.012	0.03	0.035	0.029	0.026	0.001
	fired	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	
Cobalt	unfired	<0.01	0.08	0.18	0.19	0.08	0.05	0.01
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	
Chromium	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	
Copper	unfired	<0.01	<0.01	0.01	<0.01	<0.01	<0.01	0.01
	fired	0.02	0.01	0.01	<0.01	<0.01	<0.01	
Manganese	unfired	0.08	2.76	5.18	5.63	6.21	6.17	0.01
	fired	0.04	0.09	0.06	0.07	0.08	0.11	
Nickel	unfired	<0.01	0.05	0.12	0.17	0.08	0.02	0.01
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	
Lead	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01
	fired	0.12	<0.01	<0.01	<0.01	<0.01	<0.01	
Vanadium	unfired	<0.01	0.12	<0.01	<0.01	<0.01	<0.01	0.01

	fired	0.1	0.36	0.91	1.02	1.5	1.69	
Zinc	unfired	<0.1	0.7	1.4	1.2	0.3	0.2	0.1
	fired	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	
Iron	unfired	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	0.05
	fired	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	
Titanium	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.05
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	

*LOD = limit of detection

Table 3. Results of the leachability of brick samples as a function of tailings content (mg L⁻¹ is used as the unit of concentration)

OT content (%)		0	5	10	15	20	LOD*
Sulfate as SO ₄ ²⁻	unfired	15	46	57	77	104	1
	fired	2	6	11	13	26	
Calcium	unfired	14	27	26	32	43	1
	fired	8	6	16	12	19	
Magnesium	unfired	9	12	10	10	10	1
	fired	<1	<1	<1	<1	<1	
Potassium	unfired	6	7	8	7	8	1
	fired	5	4	4	3	4	
Aluminium	unfired	<0.1	0.2	0.3	0.3	0.3	0.1
	fired	1.1	0.6	0.7	0.9	0.7	
Arsenic	unfired	<0.005	0.036	<0.005	<0.005	<0.005	0.005
	fired	0.023	0.264	0.324	0.504	0.692	
Cadmium	unfired	<0.001	<0.001	<0.001	<0.001	<0.001	0.001
	fired	<0.001	<0.001	<0.001	<0.001	<0.001	
Cobalt	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	0.01
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	
Chromium	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	0.01
	fired	<0.01	0.01	0.01	<0.01	<0.01	
Copper	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	0.01
	fired	0.02	0.01	0.02	0.02	0.01	
Manganese	unfired	0.08	0.66	0.81	0.82	0.87	0.01
	fired	0.04	0.05	0.04	0.04	0.04	
Nickel	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	0.01
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	
Lead	unfired	<0.01	0.02	<0.01	<0.01	<0.01	0.01
	fired	0.12	<0.01	<0.01	<0.01	<0.01	
Vanadium	unfired	<0.01	0.01	<0.01	<0.01	<0.01	0.01
	fired	0.1	0.16	0.15	0.16	0.17	
Zinc	unfired	<0.1	<0.1	<0.1	<0.1	<0.1	0.1
	fired	<0.1	<0.1	<0.1	<0.1	<0.1	

Iron	unfired	<0.05	<0.05	<0.05	<0.05	<0.05	0.05
	fired	<0.05	0.26	0.19	0.27	0.26	
Titanium	unfired	<0.01	<0.01	<0.01	<0.01	<0.01	0.05
	fired	<0.01	<0.01	<0.01	<0.01	<0.01	

*LOD = limit of detection

4.0 CONCLUSIONS

In this lab-scale study, up to 25% sludge and 20% tailings were incorporated into clay materials for brick-making. A combination of mineralogical (XRD) and elemental (XRF) analysis has elucidated the phase transformations of clay-waste mixture during the high temperature firing process, which have been confirmed by results of following weight loss on ignition, water absorption and leachability tests. The tailings can be a good clay substitute due to the lower heavy metal content and limited influence on the kaolinite-mullite phase transformation during the firing process. In contrast, more than 5% sludge addition can have a strong influence on the kaolinite-mullite phase transformation, resulting in gypsum decomposition, recrystallisation with clay minerals and formation of anorthite and anhydrite during the firing process. The anhydrite present in the fired matrix, due to excess of gypsum sludge in the clay mix, can strongly affect the properties and quality of bricks. This is found in a diluted colour and a significant increase in the weight loss on ignition, water absorption and leachability of Ca and SO_4^{2-} with increasing sludge content. In conclusion, fired bricks with 5% sludge and up to 20% tailings have similar properties to the waste-free control.

Full-scale trial for both sludge and tailings incorporating other tests (e.g., compressive strength and volume shrinkage tests) are expected to be conducted to further assess the feasibility of this project. The high temperature vitrification trial has provided an effective way for the stabilisation and fixation of the metals contained in sludge and tailings, leading to a new option for AMD derived waste management in mine sites.

5.0 ACKNOWLEDGEMENTS

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TOOLS FOR ASSISTING WITH THE ASSESSMENT OF ACID AND METALLIFEROUS DRAINAGE (AMD)

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ABSTRACT

Management of Acid and Metalliferous Drainage (AMD) is one of the most environmentally challenging issues facing the mining industry. Early prediction of AMD reduces risk by facilitating intervention and management. An acid-base accounting, spreadsheet-based toolkit has been developed to assist mine planners, geologists and environmental staff with initial characterisation of AMD and management of mine site water quality.

The three main applications in this toolkit permit: a) desktop analysis of the Net Acid Producing Potential (NAPP) of mine materials (Acid Base Accounting tools); b) assessment of the quality of laboratory water chemistry data (Charge Balance tool); and c) identification of water treatment requirements, reagent choice and costs (Acidity tool).

The NAPP tools estimate the potential for acidity generation from geologic materials utilising mineralogical data obtained visually from geological logs (volume % mineralogy) or from X-Ray analysis (weight % mineralogy). This tool can rapidly screen geologic materials for their potential to generate AMD prior to conducting any acid base accounting testwork. In this way it can assist with the selection of the most appropriate samples for acid-base accounting testwork.

The Charge Balance tool enables rapid and rigorous identification of problematic analytical data, thus assisting with quality assurance and quality control on water quality data.

The Acidity tool utilises pH and metal concentration data to estimate acidity values for water samples. The acidity calculation is combined with relevant site water volumes or flow rates to evaluate the Total Acidity Load for a water body or water flow. The Total Acidity Load is indicative of the severity of an AMD issue and can be used to predict treatment reagent requirements (quantities and costs) for a range of potential reagent types. Acidity calculations can also be used to compare laboratory measured values to provide another QA/QC check on laboratory data.

A detailed description of calculations and assumptions inherent in each tool are provided in this paper. The toolkit is provided as shareware at no cost to users, and is regularly reviewed, updated and expanded. Feedback and improvement advice is sought from users. The current version is ABATES v1.4 (Acid-Base Accounting Tool) and is available from the GARD Guide website.

1.0 INTRODUCTION

Management of Acid and Metalliferous Drainage (AMD) is one of the most environmentally challenging issues facing the mining industry. AMD arises when sulfidic materials located below the natural groundwater table become exposed to oxidising conditions as a result of mining and mine dewatering processes. Oxidisation of sulfidic materials can lead to the release of low pH water with elevated concentrations of metals and sulfate into receiving surface waters and groundwater, with impacts that may extend well beyond the operational

life of a mine if not well managed. The complexity and cost of AMD management can be significantly lowered, however, if the potential for AMD is identified and quantified during the early stages of mine planning.

At present, static geochemical characterisation tests (e.g. Acid Base Accounting and Net Acid Generation Tests), combined with AMD block modeling, kinetic geochemical testwork (e.g. column leach and oxygen consumption methods) and water chemistry analysis, are the main tools available to predict and monitor the extent and rate of AMD generation from various mine materials.

Static geochemical characterisation tests rely on specific sample preparation and laboratory-based analytical procedures that are time consuming, with a typical delay of days to weeks between sample collection and analysis. This can limit the value of geochemical data as a tool for day-to-day management decisions (e.g. waste rock segregation and handling). Water chemistry data from mine sites often fails to include Total Acidity analysis. An understanding of Total Acidity concentrations is vital for quantifying the total load of pollutants in a water body or stream. Such data informs the most appropriate treatment method (e.g. passive or active treatment) and can be used to determine the amount of reagent required to fulfil water quality objectives.

An Excel-based program called ABATES has been developed to address some of these issues and assist mine planners, geologists, geochemists and environmental personnel with AMD and water quality assessments. The program includes a series of tools as follows:

- **Acid Base Accounting tools** can be used to estimate Net Acid Producing Potential (NAPP) of geological materials using mineralogical data from geological logs or X-Ray Diffraction (XRD) data.
- **Charge Balance Estimator tool** is designed to provide a quality assurance / quality control (QA/QC) check on analytical chemistry data for any water sample, by estimating the Charge Balance Error (CBE) to verify that anionic and cationic species are correctly balanced.
- **Acidity Estimator tool** enables quantitative determination of the Total Acidity and Total Acidity Load of a water body or stream, based on water chemistry data, in order to assess AMD treatment reagent requirements.

The flexibility of ABATES allows it to be used routinely for estimating AMD potential, assessing water chemistry data and providing a basis for AMD management and treatment at all stages of mine development, as illustrated in Figure 1.

While the program is not a substitute for detailed geochemical testing, it is particularly useful for initial screening of AMD issues and to objectively identify existing issues with the accuracy of geochemical and water chemistry analytical data.

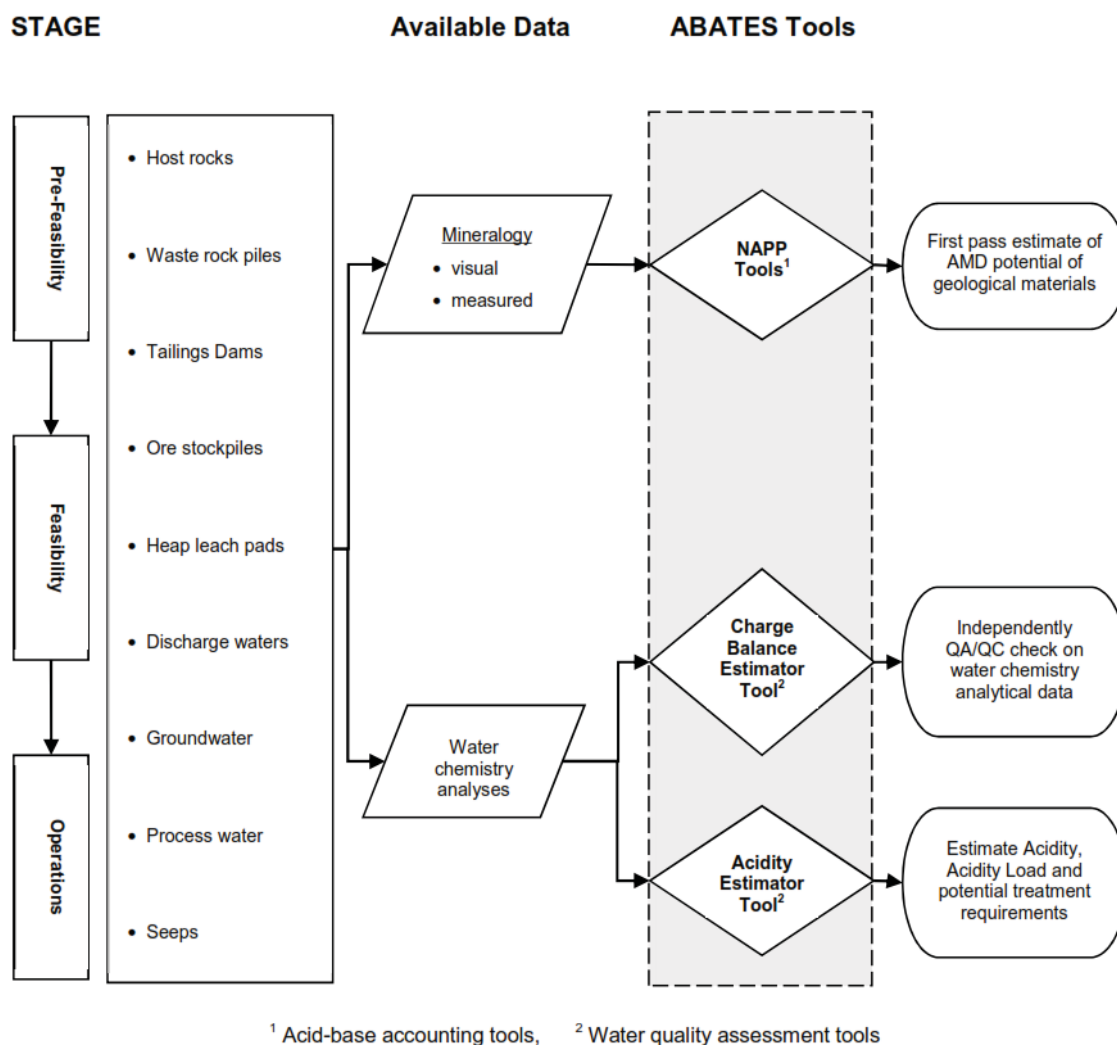


Fig. 1: Simplified flow chart indicating the various stages of mine planning and operation where ABATES can be applied to site geochemical characterisation, AMD monitoring and management programs.

This paper is organised into two main sections: the first section deals with Acid Base Accounting (NAPP tools), while the second deals with water quality assessment (Charge Balance and Acidity tools). Each section contains a brief description of the conventional laboratory-based tests, followed by an explanation of the corresponding theoretical methods used in ABATES. In addition, the section on water chemistry tools describes how the program evaluates the quality of water chemistry data.

2.0 GEOCHEMICAL CHARACTERISATION BASED ON NET ACID PRODUCING POTENTIAL

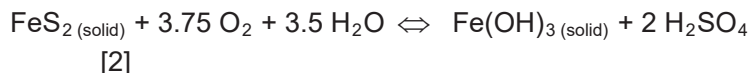
Net Acid Producing Potential (NAPP) is a measure of the difference between the capacity of a sample to generate acid (Acid Producing Potential or APP¹) and its capacity to neutralise acid (Acid Neutralising Potential or ANC). Acid generating reactions are promoted by the oxidation of key sulfide minerals once exposed to atmospheric oxygen, while acid neutralising reactions result from the dissolution of alkaline minerals, mainly carbonates. NAPP is calculated as follows:

$$\text{NAPP} = \text{APP} - \text{ANC} \quad [1]$$

In Australia NAPP, APP and ANC are generally expressed in units of kg H₂SO₄ / tonne of material.

2.1 Analytical Evaluation of NAPP

Laboratory evaluation of NAPP requires two separate analytical procedures to quantify the APP and the ANC of a sample, respectively. These methods are summarised briefly below². APP is determined by analysing the Total Sulfur content (wt.% S) of a sample. All sulfur is assumed to occur as pyrite (FeS₂), which is completely oxidised to produce sulfuric acid (H₂SO₄), as shown in Eqn. [2].



According to the stoichiometry of Eqn. [2], for each mole of pyrite, two moles of H₂SO₄ are generated. Thus, a sample containing 1 wt.% S can generate 30.6 kg H₂SO₄ per tonne. APP is therefore computed as follows:

$$\text{APP (kg H}_2\text{SO}_4\text{/tonne)} = \text{Total Sulfur (wt.\% S)} \times 30.6 \text{ (conversion factor)} \quad [3]$$

APP may be overestimated if a sample contains forms of sulfur other than pyrite, such as commonly occurring sulfate minerals (e.g. anhydrite - CaSO₄, gypsum - CaSO₄.2H₂O, and barite - BaSO₄), native sulfur, non acid forming sulfides (e.g. sphalerite - ZnS, covellite - CuS; Stewart et al., 2003) or weakly acid generating organic sulfur-bearing compounds.

ANC is determined by adding hydrochloric acid (HCl) to a sample and titrating the resulting solution to a pH of 7.0 with sodium hydroxide (NaOH). The amount of acid consumed by the initial reaction between the sample and HCl represents the ANC.

Surface passivation effects on carbonate minerals (ie. precipitate coatings), mineral acidity, incomplete reactions (i.e. oxidation of Fe²⁺; precipitation of Fe, Al and other metals) and the presence of soluble acid salts may influence the “effective” ANC of a sample³.

The main drawback of laboratory-based APP and ANC measurement, and hence NAPP determination, is that sample mineralogy is not taken into account (Jambor, 2003).

¹ APP is commonly also referred to as Maximum Potential Acidity or MPA.

² Detailed descriptions of NAPP analytical methods are provided in Sobek et al (1978), US EPA (1994), MEND (2000), Skousen et al. (2002), AMIRA (2002) and Jambor (2003).

³ These issues are described further in Weber et al. (2004a, 2004b, 2005) and Jambor (2003).

2.2 Mineralogical Evaluation of NAPP Using ABATES

To overcome some of the limitations of laboratory-derived NAPP data as described above, NAPP can be estimated in ABATES when mineralogical data are available. This:

- Avoids the issues associated with using Total Sulfur to compute APP.
- Removes the influences of carbonate passivation and mineral acidity in the ANC test.
- Overcomes the problems of incomplete oxidation and slow reaction kinetics during laboratory testwork.

NAPP values are estimated in ABATES by using mineral abundance data for acid generating and acid neutralising minerals (in wt.% or vol.% units) as a basis for calculating APP and ANC, respectively.

Specifically, the APP is computed based on the theoretical potential of FeS₂ and other sulfide minerals to generate sulfuric acid under oxidising conditions. The calculation draws on a comprehensive reference list (Table 1) which contains the most commonly occurring sulfide minerals and details the stoichiometry of each potential sulfide oxidation reaction. For each reaction, the stoichiometry is used to determine the moles of sulfuric acid produced by complete oxidation of each mole of sulfide mineral. This approach takes into account the fact that not all sulfide minerals react with atmospheric oxygen to generate sulfuric acid.

Similarly, estimation of ANC in ABATES is based on the individual contributions of a range of commonly occurring carbonate minerals and the stoichiometry of their reactions with acid (Table 2). The program accounts for the fact that not all carbonate minerals have equivalent neutralising capacity to CaCO₃, and some even exhibit zero net neutralising capacity. This point is illustrated by the dissolution of siderite (FeCO₃) in the presence of acid (Eqns. [4] and [5]) in which the net effect is neither consumption nor production of acid:



Depending on the mineralogical data available, two options are available to compute NAPP values in ABATES:

- The first option uses visual estimates of mineral abundances (vol.%) from geological logs and/or field observations. The advantage of this option is that geological data are commonly available at any stage of mine development, enabling a rapid first pass NAPP estimate for both in-situ materials and waste rock piles.
- The second option uses laboratory measurements of mineral abundances (wt.%). These can be obtained by X-Ray Diffraction (XRD) and are therefore more accurate, but also more time-consuming and costly than visual estimates.

Key assumptions used in the NAPP calculations are summarised below:

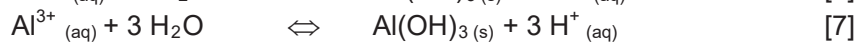
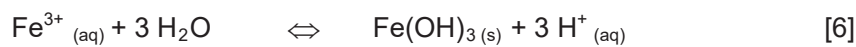
- Mineralogical data are fully representative of the geological materials being assessed.
- Minerals are present as their pure-end member compositions and polymetallic sulfide minerals have fixed molar ratios of metals.
- Only reactive sulfide minerals (i.e. not all sulfides) completely oxidise in the presence of atmospheric oxygen in accordance with the stoichiometric relationships shown in Table 1.
- Each carbonate mineral undergoes complete reaction with acid in the presence of atmospheric oxygen (i.e. no surface passivation of carbonate minerals takes place) in accordance with the stoichiometric relationships shown in Table 2.

- Oxides, sulfates and silicates have negligible effect on acid generation or neutralisation.

3.0 WATER CHEMISTRY ASSESSMENT TOOLS

The assessment of mine site water chemistry begins in the early stages of mine planning (pre-feasibility) and continues through mine development and post-closure. Water chemistry is often routinely monitored to fulfill regulatory requirements, assess options for water reuse or discharge, assess the effectiveness of AMD management, and in some cases assess treatment requirements. The Total Acidity of a water body or stream is probably the best single indicator of the severity of AMD and likely treatment requirements.

Total Acidity, reported as milligrams of CaCO_3 equivalent per litre ($\text{mg CaCO}_3 / \text{L}$), provides an indication of how much CaCO_3 would be required to neutralise an acidic solution, commonly to a pH of 8.3. Total Acidity takes into account both existing H^+ ions (determined from pH) and latent H^+ ions that may be produced by metal hydrolysis and precipitation. Hydrolysis and precipitation reactions for Fe, Al and Mn are shown in Eqns. [6]-[8]:

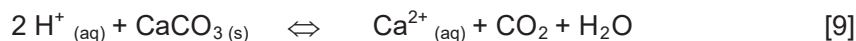


The Acidity tool in ABATES can be used to estimate Total Acidity, Total Acidity Load and likely treatment reagent requirements from routine water chemistry data, providing a quick and low cost method of quantifying an existing AMD issue. A comparison of methods for analytical evaluation of Total Acidity, and the method used in ABATES, is provided below.

This is followed by an explanation of how ABATES can be used to estimate charge balance errors, providing an independent means for evaluating the quality of water chemistry data.

3.1 Analytical Evaluation of Total Acidity

Total Acidity is determined in the laboratory by titrating a fully oxidised water sample with a strong base (e.g. APHA et al, 2005). The sample is oxidised with hydrogen peroxide (H_2O_2) and then titrated to a pH of 8.3 with sodium hydroxide (NaOH) (Eqn. [9]). Results are then equated to an equivalent amount of CaCO_3 , and reported as $\text{mg CaCO}_3 / \text{L}$.



Total Acidity may also be underestimated during laboratory tests as:

- The titration to pH 8.3 may remove the majority of dissolved metal species via precipitation as metal hydroxides, but not all metals (e.g. manganese).
- In some cases, insufficient addition of H_2O_2 may lead to incomplete oxidation and hydrolysis of dissolved metal species.

3.2 Estimating Total Acidity and Total Acidity Load Using ABATES

Within the Acidity tool of ABATES, Total Acidity can be calculated from standard water chemistry analyses using pH and dissolved metal concentration data. This avoids the need to perform additional laboratory test work. Calculation of Total Acidity in ABATES can also be used as a QA/QC check on laboratory-derived acidity measurements.

In ABATES, dissolved metal concentrations are converted to H^+ equivalent concentration, or Latent Acidity, using Eqn. [10], and then Eqn. [11] is used to estimate Total Acidity, in mg/L CaCO_3 equivalent, from Latent Acidity and pH.

$$\text{Latent Acidity} = \sum \left[\text{charge on metal ion} \times \frac{\text{metal concentration (mg/L)}}{\text{metal atomic weight (g/mol)}} \right] \quad [10]$$

$$\text{Total Acidity} = \frac{M(\text{CaCO}_3)}{2} \times (\text{Latent Acidity} + 1000 \times (10^{-\text{pH}})) \quad [11]$$

The calculation of Total Acidity assumes complete oxidation and hydrolysis of all metal species in the sample and is therefore comparable to a theoretical titration with pH endpoint of roughly 9.5, rather than 8.3 as used in laboratory-based acidity measurement. As a result, calculated Total Acidity values are often greater than those obtained using the standard laboratory method, and provide a better (more conservative) estimate of AMD treatment reagent requirements¹.

When entering pH and dissolved metal concentration data, the user should be mindful that:

- Metal concentrations should represent the dissolved (ie. 0.45 µm filtered water analysis) rather than total concentrations to avoid overestimating the Total Acidity.
- Oxidation, hydrolysis and precipitation of dissolved metal species between the time of field sampling and laboratory analysis may alter both pH and dissolved metal concentrations. Use of field pH data provides a better estimate of Total Acidity.
- There are two options for entering Iron (Fe) and Copper (Cu) concentrations, depending on redox state. If this is unknown, all Fe can be assumed to be present as Fe(III), and Cu as Cu(II), to provide a conservative estimate of Total Acidity².

The Total Acidity Load of a water stream can also be calculated in ABATES, using Eqn. [12], and is essentially equivalent to ongoing AMD treatment reagent requirements.

$$\text{Total Acidity Load} = \text{Total Acidity} \times \text{Flow Rate} \quad [12]$$

Total Acidity Load is expressed in tonnes CaCO₃ equivalent per unit time (e.g. 1.0 tonne CaCO₃ per day). Eqn. [12] utilises the Total Acidity value obtained via Eqn. [11] and the estimated stream flow rate in megalitres per unit time (ML / unit time). In the case of a fixed water body (e.g. pit lake), Flow Rate may be substituted for Volume of the water body in Eqn. [12], in which case the Total Acidity Load is expressed in tonnes CaCO₃ equivalent.

Once the Total Acidity Load is known, ABATES can convert its value, on a molar basis, to any neutralisation reagent of choice (e.g. 1.0 tonne CaCO₃ per day = 740 kg Ca(OH)₂ per day) to provide an estimate of AMD treatment reagent requirements and costs.

3.3 Analytical Evaluation of Water Chemistry Data Accuracy

Charge balance calculations are a useful means of evaluating the accuracy of water chemistry data. In natural waters, there is a balance between cations (positively charged species) and anions (negatively charged species). In any water chemistry analysis, calculating the difference between all measured cationic and anionic species, or the Charge Balance Error (CBE), can provide an indication of the quality of the analytical data.

The most accurate way of calculating the charge balance of a solution is by calculating the

¹ Calculation of Total Acidity may also be more accurate than laboratory-derived acidity measurements, as metal concentrations are often measured to a much greater accuracy (e.g. ± 0.001 mg/L) than Total Acidity (e.g. ± 5 mg/L).

² Other redox sensitive metals such as Arsenic (As), Manganese (Mn) and Chromium (Cr) are assumed to be present as As(V), Mn(II) and Cr(VI), respectively.

aqueous speciation of the solution from a water analysis. These aqueous speciation calculations “match” cations with anions to form likely aqueous complexes based on a number of variables including ionic strengths, valence states and activity coefficients. CBE indicates if there are residual cations or anions left unmatched.

Aqueous speciation calculations are performed by complex computer programs, which commonly result in the identification of hundreds of aqueous complexes in a sample. These programs provide a more detailed assessment than is often required for a preliminary evaluation of water chemistry data, as the charge balance of a sample is generally dominated by a relatively small number of major cation and anion species.

3.4 Estimation of Water Chemistry Data Accuracy Using ABATES

ABATES provides a simple alternative to conducting complex aqueous speciation calculations for the estimation of CBE, by focusing on the major cations and anions that tend to be the dominant aqueous species in water samples. While the complexity of true aqueous speciation of any sample is acknowledged, the method used in ABATES (described below) is generally considered to be sufficiently accurate for first pass identification of significant CBE's.

ABATES estimates the CBE by summing cation (including H^+) and anion (including OH^-) concentrations on a milli-equivalent basis. In particular, the analyte concentrations are converted from mg/L to mEq/L (milli-equivalents per litre). For each anion and cation present in solution, its mEq/L value is computed as follows:

$$mEq/L = \frac{mg/L}{molecular\ weight} \times charge \quad [13a]$$

The contribution of H^+ and OH^- to charge balance is computed as follows:

$$\begin{aligned} H^+ \quad mEq/L &= 10^{-pH} \\ OH^- \quad mEq/L &= 10^{-(14-pH)} \end{aligned} \quad [13b]$$

In the above equation, the absolute value of the charge is used so that the milli-equivalent amount is always positive. Milli-equivalents for anionic and cationic species are summed separately and the charge balance error is calculated using Eqn. [14]:

$$CBE = \left[\frac{\sum(mEq) \text{ cations} - \sum(mEq) \text{ anions}}{\sum(mEq) \text{ cations} + \sum(mEq) \text{ anions}} \right] \times 100\% \quad [14]$$

The following assumptions are made in ABATES to estimate CBE:

- Metals are assumed to be present in dissolved form (i.e. 0.45 μm filtered water analysis). Use of total concentrations may lead to overestimation of the CBE.
- Fe, Mn, Cu, Zn and Pb are assumed to carry a 2^+ charge.
- Arsenic¹ is assumed to be present as $H_2AsO_4^-$ at a pH of less than 6.5, or $HAsO_4^{2-}$ at a pH greater than 6.5.
- Chromium¹ is assumed to be present as $H_2CrO_4^-$ at a pH of less than 6.5, or $HCrO_4^{2-}$ at a pH greater than 6.5.
- All ammonia is assumed to be present as NH_4^+ .
- Volatile acids, reported as acetic acid, are assumed to be present as $CH_3CO_2^-$.

¹ The charge of arsenic and chromium, which contribute to the anionic charge balance, is pH dependent.

- Phosphorus is assumed to be present as orthophosphate PO_4^{3-} .

There is no industry, regulatory or academic consensus on what constitutes an acceptable CBE. ABATES considers an error of $\pm 10\%$ acceptable for total anion sums of between 10 and 100 mEq/L. For samples with total anion sums less than 10 mEq/L, an error of $\pm 5\%$ is considered acceptable, however low cation and anion totals decrease the reliability of these calculations for evaluating analytical data. For total anion sums greater than 100 mEq/L, an error of $\pm 15\%$ is considered acceptable in ABATES.

Large errors in charge balance (i.e. excess of positively or negatively charged species) may be attributed to problems with the analytical procedure, exclusion of one or more charged species from the analysis and/or data entry or transcription errors.

4.0 CONCLUSIONS

While ABATES is not intended to replace the need for detailed geochemical testwork, it does provide a quick and low cost method for estimating the AMD potential of geological materials, Total Acidity of mine waters and likely AMD treatment reagent requirements using routinely collected data. It also offers a useful method for QA/QC checks on water chemistry data.

ABATES can be used at any stage of mine development, from pre-feasibility through to post-closure. Estimates provided by ABATES for NAPP and Total Acidity have a number of benefits over more detailed laboratory-based geochemical/AMD test procedures.

NAPP values can be estimated for any sample or geological log for which mineralogical data are available. NAPP estimates based on mineralogical data account for the variable acid producing potential of sulfide minerals (and the fact that not all sulfides are acid producing) as well as the variable acid neutralising capacity of carbonate minerals, unlike laboratory-based NAPP tests. NAPP estimates can also be used to inform sample selection and analytical procedure selection for more detailed static or kinetic geochemical testwork.

Total Acidity values can be readily determined in ABATES for historic and new water chemistry analyses, for which no laboratory measurement of acidity is available. By combining Total Acidity values with field estimates of stream flow rate (or water volume), ABATES can also rapidly assess AMD treatment reagent requirements (quantities and costs). Estimates of Total Acidity are comparable to a laboratory titration to pH 9.5, rather than the standard method for acidity determination (titration to pH 8.3). This leads to slightly higher estimates of Total Acidity than the standard method, but is a more conservative approach to assessing treatment reagent requirements for the purpose of maximising metal removal efficiency.

Identifying the potential for AMD at the early stages of mine planning can dramatically reduce the associated risks and enable more informed design and implementation of specific test work and monitoring programs to assist with AMD assessment and management.

4.1 A note on the availability of the current version

The current version is ABATES v1.4 (Acid-Base Accounting Tool) and is available from the Earth Systems website (<http://www.earthsystems.com.au/resources/acid-drainage>) and can also be accessed via the GARD Guide website (http://www.gardguide.com/index.php/Chapter_7). The toolkit is provided as shareware at no cost to users, and is regularly reviewed, updated and expanded. The current version can be freely distributed, provided the tools are used only as intended and are not modified in any way. Feedback and improvement advice is sought from users, via the contact details provided within the toolkit.

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Table 1: Summary of sulfide mineralogical data used by ABATES for estimating NAPP values. The reactions do not necessarily represent actual or likely reactions in natural systems, but are useful for accurately calculating the potential for acid production and neutralisation for systems in equilibrium with atmospheric oxygen. The final metal oxidation states apply to the first and second metals as they appear in each Equation.

Sulfide Mineral Name	Sulfide Mineral Formula	Molar Mass of sulfide mineral (g/mol)	Sulfide Mineral Density (g/cm ³)	Equation	Mol of sulfuric acid (H ₂ SO ₄) produced per Mol of mineral oxidised	Final Metal Oxidation state
Orpiment	As ₂ S ₃	246.04	3.52	As ₂ S ₃ + 6 H ₂ O + 7 O ₂ = 2 HAsO ₄ ⁻² + 10 H ⁺ + 3 SO ₄ ⁻²	Oxidises	3
Realgar	AsS	106.99	3.56	AsS + 5/2 H ₂ O + 11/4 O ₂ = HAsO ₄ ⁻² + 4 H ⁺ + SO ₄ ⁻²	Oxidises	1
Bismuthinite	Bi ₂ S ₃	514.16	7.00	Bi ₂ S ₃ + 3 H ₂ O + 6 O ₂ = Bi ₂ O ₃ + 6 H ⁺ + 3 SO ₄ ⁻²	Oxidises	3
Greenockite	CdS	144.48	4.49	CdS	Unknown	0
Cobaltite	CoAsS	165.92	6.33	CoAsS + 7/2 H ₂ O + 13/4 O ₂ = HAsO ₄ ⁻² + Co(OH) ₂ + SO ₄ ⁻² + 4 H ⁺	Oxidises	1
Chalcocite	Cu ₂ S	159.16	5.65	Cu ₂ S + 5/2 O ₂ + H ₂ O = 2 CuO + 2 H ⁺ + SO ₄ ⁻²	Oxidises	1
Digenite	Cu ₉ S ₅	146.45	5.71	Cu ₉ S ₅	Unknown	0
Bornite	Cu ₅ FeS ₄	501.84	5.09	Cu ₅ FeS ₄ + 11/2 H ₂ O + 37/4 O ₂ = 5 CuO + Fe(OH) ₃ + 8 H ⁺ + 4 SO ₄ ⁻²	Oxidises	4
Cubanite	CuFe ₂ S ₃	271.43	4.70	CuFe ₂ S ₃ + 6 H ₂ O + 13/2 O ₂ = CuO + 2 Fe(OH) ₃ + 6 H ⁺ + 3 SO ₄ ⁻²	Oxidises	3
Chalcocopyrite	CuFeS ₂	183.52	4.19	CuFeS ₂ + 17/4 O ₂ + 7/2 H ₂ O = CuO + Fe(OH) ₃ + 4 H ⁺ + 2 SO ₄ ⁻²	Oxidises	2
Covellite	CuS	95.61	4.68	CuS	Unknown	0
Tennantite	Cu ₁₂ As ₄ S ₁₃	1479.08	4.65	Cu ₁₂ As ₄ S ₁₃ + 61/2 O ₂ + 19 H ₂ O = 12 CuO + 4 HAsO ₄ ⁻² + 34 H ⁺ + 13 SO ₄ ⁻²	Oxidises	13
Tetrahedrite	Cu ₁₂ Sb ₄ S ₁₃	1666.44	4.95	Cu ₁₂ Sb ₄ S ₁₃ + 57/2 O ₂ + 15 H ₂ O = 12 CuO + 4 HSbO ₂ + 26 H ⁺ + 13 SO ₄ ⁻²	Oxidises	13
Enargite	Cu ₃ AsS ₄	393.82	4.47	Cu ₃ AsS ₄ + 35/4 O ₂ + 11/2 H ₂ O = 3 CuO + HAsO ₄ ⁻² + 10 H ⁺ + 4 SO ₄ ⁻²	Oxidises	4
Greigite	Fe ₃ S ₄	295.80	4.05	Fe ₃ S ₄ + 17/2 H ₂ O + 33/4 O ₂ = 3 Fe(OH) ₃ + 8 H ⁺ + 4 SO ₄ ⁻²	Oxidises	4
Arsenopyrite	FeAsS	162.83	6.07	FeAsS + 4 H ₂ O + 7/2 O ₂ = Fe(OH) ₃ + HAsO ₄ ⁻² + 4 H ⁺ + SO ₄ ⁻²	Oxidises	1
Violarite	FeNi ₂ S ₄	301.49	4.65	FeNi ₂ S ₄ + 11/2 H ₂ O + 31/4 O ₂ = Fe(OH) ₃ + 2 NiO + 8 H ⁺ + 4 SO ₄ ⁻²	Oxidises	4
Pyrrhotite	Fe _(1-x) S _x	85.12	4.61	FeS + 5/2 H ₂ O + 9/4 O ₂ = Fe(OH) ₃ + 2 H ⁺ + SO ₄ ⁻²	Oxidises	1
Troilite	FeS	87.91	4.61	FeS + 5/2 H ₂ O + 9/4 O ₂ = Fe(OH) ₃ + 2 H ⁺ + SO ₄ ⁻²	Oxidises	1
Marcasite	FeS ₂	119.98	4.89	FeS ₂ + 7/2 H ₂ O + 15/4 O ₂ = Fe(OH) ₃ + 4 H ⁺ + 2 SO ₄ ⁻²	Oxidises	2
Pyrite	FeS ₂	119.98	5.00	FeS ₂ + 7/2 H ₂ O + 15/4 O ₂ = Fe(OH) ₃ + 4 H ⁺ + 2 SO ₄ ⁻²	Oxidises	2
Cinnabar	HgS	232.66	8.10	HgS	Stable	0
Molybdenite	MoS ₂	160.09	5.50	MoS ₂	Stable	0

Sulfide Mineral Name	Sulfide Mineral Formula	Molar Mass of sulfide mineral (g/mol)	Sulfide Mineral Density (g/cm ³)	Equation	Mol of sulfuric acid (H ₂ SO ₄) produced per Mol of mineral oxidised	Final Metal Oxidation state
Millerite	NiS	90.76	5.50	NiS	Unknown	0
Pentlandite	(Ni,Fe) ₉ S ₈	771.94	4.96	(Fe,Ni) ₉ S ₈	Unknown	0
Galena	PbS	239.27	7.40	PbS	Stable	0
Stibnite	Sb ₂ S ₃	339.72	4.63	Sb ₂ S ₃ + 4 H ₂ O + 6 O ₂ = 2 HSbO ₂ (aq) + 6 H ⁺ + 3 SO ₄ ⁻²	Oxidises	3
Native Sulfur	S ₈	256.52	2.08	S ₈	Stable	0
Sphalerite	ZnS	97.45	4.05	ZnS	Stable	0

Table 2: Summary of carbonate mineralogical data used by ABATES in estimating NAPP values. The reactions may not necessarily represent actual or likely reactions in natural systems, but are useful for accurately calculating the potential for acid production and neutralisation for systems in equilibrium with atmospheric oxygen. Where two or more reactions are shown, the first is the neutralisation step and subsequent reactions show the hydrolysis of the aqueous metal ion and resultant Latent Acidity produced. In most cases, the number of moles of H⁺ consumed by the neutralisation step is offset by the number of moles of H⁺ produced by hydrolysis of the metal ion. The number of moles of sulfuric acid produced is equal to half the number of moles of H⁺ ions produced. The final metal oxidation states apply to the first and second metals as they appear in each Equation.

Carbonate Mineral Name	Carbonate Mineral Formula	Molar Mass of carbonate mineral (g/mol)	Carbonate Mineral Density (g/cm ³)	Equation	Mol of sulfuric acid (H ₂ SO ₄) consumed per Mol of mineral neutralised	Final Metal Oxidation state
Mangansiderite	(Fe,Mn)CO ₃	115.40	3.80	i. (Fe,Mn)CO ₃ + 3/8 O ₂ + 7/2 H ⁺ = 1/2 Fe ³⁺ + 1/2 Mn ⁴⁺ + 7/4 H ₂ O + CO ₂ ii. 1/2 Fe ³⁺ + 3/2 H ₂ O = 1/2 Fe(OH) ₃ + 3/2 H ⁺ (3 1/2 mole H ⁺ consumed) ii. 1/2 Mn ⁴⁺ + 2 H ₂ O = 1/2 Mn(OH) ₂ + 2 H ⁺ (3/2 mole H ⁺ produced) (2 mole H ⁺ produced)	0	3,4
Witherite	BaCO ₃	197.34	4.30	BaCO ₃ + 2 H ⁺ = H ₂ O + CO ₂ + Ba ²⁺	1	2
Ankerite	Ca(Fe, Mg, Mn)(CO ₃) ₂	206.39	3.05	Ca(Fe,Mg,Mn)(CO ₃) ₂ + 8/3 H ⁺ + 1/4 O ₂ = Ca ²⁺ + 2 CO ₂ + 1/3 Mg ²⁺ + 1/3 Fe(OH) ₃ + 1/3 Mn(OH) ₂ + 1/6 H ₂ O	1.33	3,4
Aragonite	CaCO ₃	100.09	2.93	CaCO ₃ + 2 H ⁺ = H ₂ O + CO ₂ + Ca ²⁺	1	2
Calcite	CaCO ₃	100.09	2.71	CaCO ₃ + 2 H ⁺ = H ₂ O + CO ₂ + Ca ²⁺	1	2
Dolomite	CaMg(CO ₃) ₂	184.40	2.84	CaMg(CO ₃) ₂ + 4 H ⁺ = 2 H ₂ O + 2 CO ₂ + Ca ²⁺ + Mg ²⁺	2	2,2

Carbonate Mineral Name	Carbonate Mineral Formula	Molar Mass of carbonate mineral (g/mol)	Carbonate Mineral Density (g/cm ³)	Equation	Mol of sulfuric acid (H ₂ SO ₄) consumed per Mol of mineral neutralised	Final Metal Oxidation state
Huntite	CaMg ₃ (CO ₃) ₄	353.03	2.70	CaMg ₃ (CO ₃) ₄ + 8 H ⁺ = Ca ²⁺ + 3 Mg ²⁺ + 4 H ₂ O + 4 CO ₂	4	2,2
Olavite	CdCO ₃	172.42	5.03	i. CdCO ₃ + 2 H ⁺ = Cd ²⁺ + CO ₂ + H ₂ O (2 mole H ⁺ consumed) ii. Cd ²⁺ + 2 H ₂ O = Cd(OH) ₂ + 2 H ⁺ (2 mole H ⁺ produced)	0	2
Sphaerocobaltite	CoCO ₃	118.94	4.10	i. CoCO ₃ + 3 H ⁺ + 1/4 O ₂ = Co ³⁺ + CO ₂ + 3/2 H ₂ O (3 mole H ⁺ consumed) ii. Co ³⁺ + 3 H ₂ O = Co(OH) ₃ + 3 H ⁺ (3 mole H ⁺ produced)	0	3
Malachite	Cu ₂ CO ₃ (OH) ₂	221.12	3.80	i. Cu ₂ CO ₃ (OH) ₂ + 4 H ⁺ = 2 Cu ²⁺ + CO ₂ + 3 H ₂ O (4 mole H ⁺ consumed) ii. 2 Cu ²⁺ + 4 H ₂ O = 2 Cu(OH) ₂ + 4 H ⁺ (4 mole H ⁺ produced)	0	2
Azurite	Cu ₃ (CO ₃) ₂ (OH) ₂	344.67	3.83	i. Cu ₃ (CO ₃) ₂ (OH) ₂ + 6 H ⁺ = 3 Cu ²⁺ + 2 CO ₂ + 4 H ₂ O (6 mole H ⁺ consumed) ii. 3 Cu ²⁺ + 6 H ₂ O = 3 Cu(OH) ₂ + 6 H ⁺ (6 mole H ⁺ produced)	0	2
Siderite	FeCO ₃	115.85	3.96	i. FeCO ₃ + 3 H ⁺ + 1/4 O ₂ = Fe ³⁺ + CO ₂ + 3/2 H ₂ O (3 mole H ⁺ consumed) ii. Fe ³⁺ + 3 H ₂ O = Fe(OH) ₃ + 3 H ⁺ (3 mole H ⁺ produced)	0	3
Magnesite	MgCO ₃	84.31	3.00	MgCO ₃ + 2 H ⁺ = H ₂ O + CO ₂ + Mg ²⁺	1	2
Rhodochrosite	MnCO ₃	114.95	3.69	i. MnCO ₃ + 4 H ⁺ + 1/2 O ₂ = Mn ⁴⁺ + CO ₂ + 2 H ₂ O (4 mole H ⁺ consumed) ii. Mn ⁴⁺ + 4 H ₂ O = Mn(OH) ₄ + 4 H ⁺ (4 mole H ⁺ produced)	0	4
Nahcolite	NaHCO ₃	84.01	2.20	NaHCO ₃ + H ⁺ = Na ⁺ + CO ₂ + H ₂ O	0.5	1
Hellyerite	NiCO ₃ ·6(H ₂ O)	226.79	1.97	i. NiCO ₃ ·6H ₂ O + 2 H ⁺ = Ni ²⁺ + CO ₂ + 7 H ₂ O (2 mole H ⁺ consumed) ii. Ni ²⁺ + 2 H ₂ O = Ni(OH) ₂ + 2 H ⁺ (2 mole H ⁺ produced)	0	2
Gaspelle	(Ni _{0.6} , Mg _{0.3} , Fe _{0.1}) CO ₃	108.10	3.70	(Ni _{0.6} , Mg _{0.3} , Fe _{0.1}) CO ₃ + 18/10 H ⁺ + 1/40 O ₂ = 3/10 Mg ²⁺ + 6/10 Ni ²⁺ + 1/10 Fe(OH) ₃ + 15/20 H ₂ O + CO ₂	0.9	2,2,3
Cerussite	PbCO ₃	267.21	6.58	i. PbCO ₃ + 2 H ⁺ = Pb ²⁺ + CO ₂ + H ₂ O (2 mole H ⁺ consumed) ii. Pb ²⁺ + 2 H ₂ O = Pb(OH) ₂ + 2 H ⁺ (2 mole H ⁺ produced)	0	2
Strontianite	SrCO ₃	147.63	3.78	i. SrCO ₃ + 2 H ⁺ = Sr ²⁺ + CO ₂ + H ₂ O (2 mole H ⁺ consumed) ii. Sr ²⁺ + 2 H ₂ O = Sr(OH) ₂ + 2 H ⁺ (2 mole H ⁺ produced)	0	2
Smithsonite	ZnCO ₂	125.39	4.45	i. ZnCO ₃ + 2 H ⁺ = Zn ²⁺ + CO ₂ + H ₂ O (2 mole H ⁺ consumed) ii. Zn ²⁺ + 2 H ₂ O = Zn(OH) ₂ + 2 H ⁺ (2 mole H ⁺ produced)	0	2
Hydrozincite	Zn ₅ (CO ₃) ₂ (OH) ₆	548.96	3.50	i. Zn ₅ (CO ₃) ₂ (OH) ₆ + 10 H ⁺ = 5 Zn ²⁺ + 2 CO ₂ + 8 H ₂ O (10 mole H ⁺ consumed) ii. 5 Zn ²⁺ + 10 H ₂ O = 5 Zn(OH) ₂ + 10 H ⁺ (10 mole H ⁺ produced)	0	2

NEUTRALISATION OF ALUMINIUM-RICH ACID MINE DRAINAGE TO FORM A MAGNETIC SLUDGE

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ABSTRACT

The active remediation of acid mine drainage (AMD) typically employs hydrated lime as the neutralising agent which forms a voluminous low density sludge (typically 1.7 g cc^{-1}). Sludge composition varies depending on initial AMD chemical composition and the neutralising process utilised, but is usually dominated by ferric hydroxide and gypsum. The production of a high density magnetic sludge consisting of magnetite is desirable, as this is relatively low volume, more resistant to leaching and neutral in colour compared to the standard ferrihydrite/schwertmannite sludge. However, high aluminium levels in AMD inhibit magnetite precipitation. To determine the best way to overcome this problem, a two-step neutralisation process was applied to two synthetic AMD solutions with moderate (Fe/Al molar ratio of 12:1) and high (Fe/Al ratio of 4:3) aluminium concentrations. Neutralisation was used to stabilise the pH at ~ 4.6 for 5 minutes to allow aluminium precipitation to occur, followed by seeding with magnetite. Neutralisation recommenced until pH ~ 11.5 was reached. For AMD with a Fe/Al molar ratio of 12:1 this treatment produced a magnetic sludge composed of magnetite with a density of $2.16\text{--}2.7 \text{ g mL}^{-1}$. Results from the synthetic AMD with a Fe/Al molar ratio of 4:3 were less successful, because high levels of calcium (from the hydrated lime neutralising reagent) inhibited magnetite precipitation. This was overcome using sodium hydroxide as the neutralising reagent in place of hydrated lime.

1.0 INTRODUCTION

Acid mine drainage is the outflow of acidic water from (usually abandoned) metal mines or coal mines, and is characterized by low pH and high concentrations of heavy metals. The composition of AMD varies according to the surrounding lithological and hydrogeological environment, with total dissolved solids varying from under 100 mg L^{-1} at Avoca East, Ireland (Bearcock et al. 2011) to tens of thousands mg L^{-1} at Iron Mountain California (Nordstrom 2011).

Considerable research has been conducted on prevention and remediation of AMD, with the most common active treatment, the High Density Sludge (HDS) process, today considered the industry standard. Whilst this effectively neutralizes the AMD, the resultant sludge, usually dominated by ferric hydroxide and gypsum, poses immediate disposal problems of both a financial and environmental nature. Sludge disposal costs are customarily based on volume, therefore the greater the density of the sludge, the lower the disposal cost incurred. In addition, amorphous ferric hydroxide sludge does not exhibit long term chemical stability and when exposed to low pH conditions, leaching of heavy metals can occur (McDonald et al. 2006).

The active remediation of acid mine drainage (AMD) typically employs lime as the neutralising agent, which forms a voluminous low density sludge (typically 1.7 g cc^{-1}). The production of a high density magnetic sludge consisting of magnetite is desirable, as this is

relatively low volume, more resistant to leaching and neutral in colour compared to the orange-red ferric hydroxide sludge.

Previous studies (McKinnon et al. 2000) have shown that aluminium in AMD is detrimental to the formation of magnetite, favouring less stable ferric oxy-hydroxides such as ferrihydrite and schwertmannite. Aluminium solubility is strongly determined by pH, and at normal surface temperatures it precipitates as amorphous aluminium hydroxide at pH between 4.5 and 5.0.

To determine the best way to form a magnetic sludge from aluminium-rich AMD, a two-step neutralising method was applied, and this paper describes the experiments used to determine the optimum conditions for magnetite precipitation.

2.0 EXPERIMENTAL SETUP AND ANALYTICAL METHODS

Two synthetic AMD solutions were used (Table 1), with Fe/Al ratios of 12:1 (typical of conditions at many mine sites; Aubé and Zinck 1999; McDonald et al. 2006) and 4:3 (significantly higher aluminium concentrations found at an existing abandoned mine within Australia).

Table 1. Synthetic AMD concentrations used for small bench top batch reactors

Synthetic AMD	Concentration (mg L ⁻¹)	
	<i>Fe/Al 12:1</i>	<i>Fe/Al 4:3</i>
Aluminium	100	600
Copper	100	30
Calcium	0	300
Iron (II)	1200	800
Magnesium	0	300
Zinc	100	30

2.1 Synthetic AMD Formulation

A small bench top batch reactor consisted of a 5L glass beaker made up to a volume of 4.5L with tap water. Metal sulphates were added and allowed to dissolve under constant stirring. Probes were placed into the solution in the reactor throughout the whole treatment process to monitor pH and redox potential. A TPS WP80D dual channel waterproof pH-mV-Temp meter was used with data logged every 2 minutes through the complete treatment.

2.2 Two-step neutralising process

Neutralising solution was placed in a 200mL glass beaker and constantly stirred with a mixer to stop the slurry settling. A peristaltic pump was used to transport accurate injections (10mL) of neutralising reagent into the reactor. The first step involved constant addition of neutralising reagent until a pH ~ 4.6 was reached; pH was maintained at this level for 5 minutes to allow aluminium precipitation. This was followed by seeding with laboratory grade magnetite. The neutralisation process recommenced with 10mL injections of neutralising reagent every 5 minutes until a pH~11.5 was reached. After completion of neutralisation the reactor was aerated with compressed air piped into the reactor at a pressure of 0.5psi and

continually stirred at a speed of 166rpm for approximately 18 hours. The aeration rate was set at $\sim 0.1 \text{ L m}^{-1}$ using a visi-float air flow meter with a range of $0.05 - 0.5 \text{ L min}^{-1}$. The sludge was then allowed to settle for 24 hours.

2.3 Analytical Methods

2.3.1 Supernatant water analysis

Supernatant water samples were collected before treatment, after neutralisation and after treatment for bench tests using AMD with Fe/Al 12:1, whilst bench tests using AMD with Fe/Al 4:3 included an additional sample immediately after aluminium precipitation step. Samples were filtered through Whatman $0.45\mu\text{m}$ filter paper, placed in a 120mL polyethylene sample vial filled to the top, acidified to a $\text{pH} < 2$ with concentrated nitric acid (HNO_3) then stored in a refrigerator at temperatures of $0-4^\circ\text{C}$. Supernatant water samples were analysed by ICP-OES on a GBC Quantima and a Perkin Elmer Optima 8000.

2.3.2 Sludge analysis

2.3.2.1 Mineralogy identification and quantification

After completion of 24hour settling period the sludge was vacuum filtered using a Welch dry vacuum pump attached to bottle top filter system and Whatman $0.45\mu\text{m}$ filter paper. The sludge was then placed in an oven at 50°C overnight for 16-24 hours to dry. Dried sludge was stored in a 120mL polyethylene sample vial. Dry sludge samples were spiked with $\sim 10\%$ zincite as an internal reference and analysed using a Siemens D5000 X-Ray Powder Diffractometer with mineralogy identification and quantification determined using Bruker $\text{Diffrac}^{\text{plus}}$ Eva V2.1 and Topas V4.1.

2.3.2.2 Magnetism

Magnetism was tested using a neodymium ($\text{Nd}_2\text{Fe}_{14}\text{B}$) rare earth magnet placed on the side of the plastic sample vial and arbitrarily graded according to the portion of sample attracted to the magnet and the resistance to removal of the magnet.

3.0 RESULTS AND DISCUSSION

3.1.1 Bench tests using synthetic AMD with 12:1 Fe/Al ratio

The results from all experiments using synthetic AMD with a 12:1 Fe/Al ratio showed that seeding with magnetite was a pivotal step in the formation of a magnetic sludge. The two-step neutralising process, first stabilising pH at $\sim 4.6-4.8$ to allow aluminium precipitation to occur, resulted in the formation of a magnetic sludge (Fig. 1). Therefore the initial neutralisation removed the inhibiting role that trivalent nonferrous metals play in the formation of a magnetic sludge.

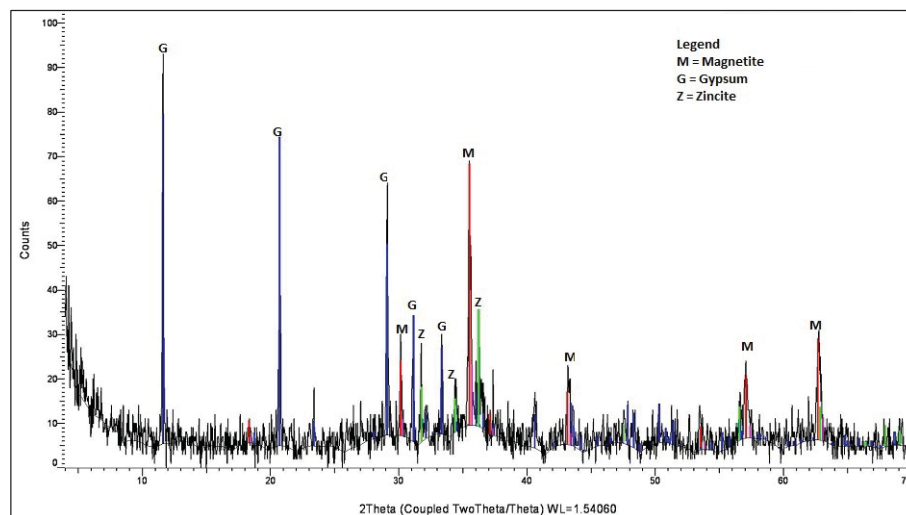


Fig. 1. XRD pattern of moderate aluminium-rich AMD with Fe/Al ratio of 12:1 neutralised with hydrated lime, forming magnetic sludge as shown by prominent magnetite peak

3.1.2 Bench tests using synthetic AMD with 4:3 Fe/Al Ratio

In the initial neutralisation step using AMD with a lower Fe/Al ratio, to remove the aluminium without removing significant amounts of iron the pH needed to be maintained as close to 4.5 as possible. At a pH of 4.5-4.6 the iron concentration decreased by 13% due to co-precipitation with the aluminium, and decreased further as the pH rose (42% at pH ~4.8-5.0). Any decrease in iron concentration during this initial neutralisation was reflected in decreased magnetism of the sludge.

The stage 1 precipitation step produced gelatinous amorphous aluminium hydroxide that was difficult to filter. Future experiments will incorporate flocculants in step one to increase sedimentation rate, aiding settlement and easier removal of the aluminium hydroxide precipitate.

The tests with high-aluminium AMD formed only a weakly magnetic sludge (Fig. 2) even when optimal aluminium removal was achieved, and this was probably due to the high calcium levels in the AMD (both original and from the addition of lime as the neutralising agent). Previous studies (Morgan et al. 2003) suggested that calcium interferes with magnetite precipitation and therefore the formation of a magnetic sludge. A bench test using aluminium-free synthetic AMD neutralised with lime produced less magnetic sludge than using sodium hydroxide as the neutralising reagent.

Neutralising AMD with a Fe/Al ratio of 4:3 using sodium hydroxide as the neutralising reagent overcame the interference of calcium in magnetite precipitation and formed a magnetic sludge composed of magnetite (and lacking gypsum; Fig. 3). However, using sodium hydroxide produces high sodium concentrations in post treatment supernatant water and there is an increased cost, as sodium hydroxide is three times more expensive compared to hydrated lime per tonne of acid neutralised (Taylor et al. 2005).

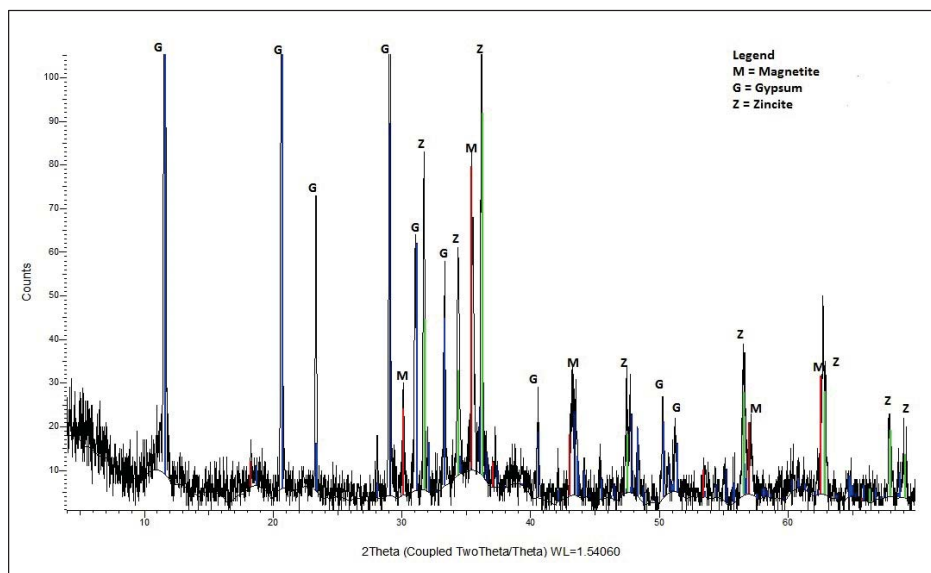


Fig. 2. XRD pattern of high aluminium-rich AMD with Fe/Al ratio of 4:3 neutralised with hydrated lime, forming weakly magnetic sludge. (This scan is only shown from 0-100 counts; gypsum peaks reach counts of 400)

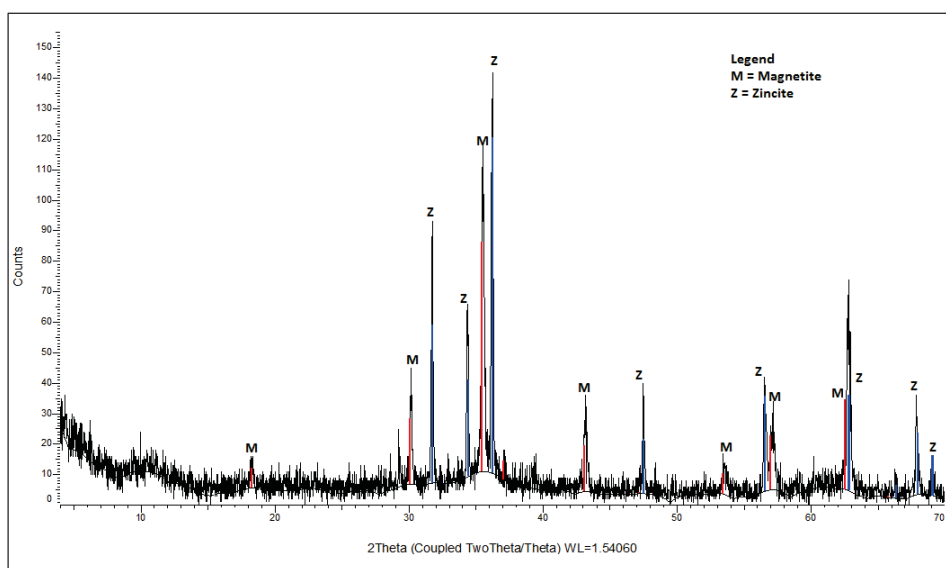


Fig. 3. XRD pattern of high aluminium-rich AMD with Fe/Al ratio of 4:3 neutralised with sodium hydroxide, forming strong magnetic sludge as shown by the prominent magnetite peaks

4.0 CONCLUSIONS

This research has shown that it is possible to precipitate a magnetic sludge by neutralisation of AMD with low levels of aluminium using lime as the neutralising agent, provided that seeding with magnetite is carried out in conjunction with a two-step neutralisation procedure. For AMD solutions with high aluminium levels (4:3 Fe/Al ratio), the most successful neutralising reagent was sodium hydroxide, which although the more expensive option, produced a high density magnetic sludge.

5.0 ACKNOWLEDGEMENTS

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GEOCHEMICAL AND GEOTECHNICAL INVESTIGATIONS AT THE REDDALE COAL MINE, REEFTON, NEW ZEALAND

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ABSTRACT

The Reddale Coal Mine, operated by Solid Energy (NZ) Ltd, commenced operations in early 2012. Acid-base accounting (ABA) techniques indicated the site would produce acid mine drainage (AMD) immediately after mining commenced and a treatment plant was constructed to manage the forecast acid loads. AMD had not occurred even after the removal of 494,000 tonnes of overburden. However, there is evidence to suggest that significant pyrite oxidation is occurring, and that it is possible with time that the site may generate low pH drainage waters. If this occurs it is likely that passive treatment technologies could be employed to treat the low flows derived from the engineered landform (ELF) basal landform.

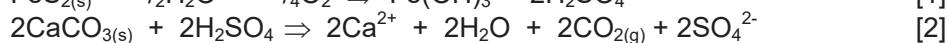
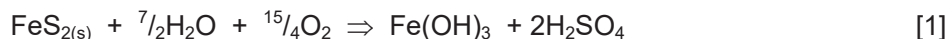
ABA techniques indicated there will be a time lag to the onset of acid mine drainage, in line with other sites having a small, yet significant, amount of CaCO₃. In addition to the inherent acid neutralisation capacity (ANC) present in the overburden, 15 mm (19 kg CaCO₃/m²) of aglime (AP < 2.5 mm CaCO₃) was applied to each 4 m lift of the Ferndale ELF. The benefits of this layering approach were tested using trial pads at site. Results demonstrated a significant reduction in acid load when a limestone layer was present (43.5 g CaCO₃ eq. /m²) compared to the control that was 539 g CaCO₃ eq. /m²), however preferential flow was evident and highest flow rates yielded the highest acid loads.

The permeability of the uncompacted tipheads was 1 x 10^{-6.6} m/s, whilst on the traffic-compacted lifts it was 1 x 10^{-7.1} to 1 x 10^{-6.0} m/s, sufficient to retard significant oxygen ingress into the dump core and thus limit pyrite oxidation and AMD generation rates. Oxygen probes confirmed that only the outer 4-8m of the ELF is oxidising. Paste pH testing of the outer ELF surface confirmed a decreasing pH trend with time. Lysimeter results within the core of the dump, where oxygen is excluded, confirmed circum-neutral conditions, and validated the model.

Although the geochemistry of the ELF will provide a time lag to acid onset, it is likely that the geotechnical properties of the ELF will extend this time lag significantly.

1.0 INTRODUCTION

Acid mine drainage (AMD) is the result of sulfide mineral oxidation, typically pyrite in coal measures, where the pyrite reacts with oxygen and water to produce acidity and metal precipitates such as Fe oxyhydroxides (Eqn. [1]). The acidity generated by pyrite oxidation can be neutralised by materials such as limestone (CaCO₃) as shown in Eqn. [2]. Rocks can contain both pyrite and carbonate, which means that although pyrite may be present, if the carbonate content is sufficient, depressed pH (< 3) may not occur.



Geochemical characterisation of overburden involves a variety of acid-base accounting (ABA) techniques to determine the proportions of pyrite (sulphides) and carbonate within a sample, e.g., the net acid production potential (NAPP) of the sample. This is the difference between the acid neutralisation capacity (ANC), which typically represents carbonates (Eqn [2]), and can be determined by titration methods (expressed in kg H₂SO₄/t equivalent), and the maximum potential acidity (MPA) where $MPA = \text{wt.\% total S} \times 30.6$ (in kg H₂SO₄/t equivalent): note this is a conservative approach as it assumes that the total sulphur determination is present as pyrite. A negative NAPP indicates that the sample has a net neutralising capacity and a positive NAPP indicates the sample has a net acid-generating capacity:

$$NAPP = MPA - ANC \quad [3]$$

One approach to managing incorrect geochemical classification of mine waste rock is to use comparative testing of samples to determine both the NAPP acidity and the net acid generation (NAG) acidity of the sample (e.g., IWRI and EGi, 2002) to provide two methods to assess whether the sample is PAF (potentially acid forming) or NAF (non-acid forming). The NAG analysis method also provides a robust basis for data interrogation and validation, as well as a schematic classification which makes it simple to identify anomalous samples and potential errors.

Part of the management plan for AMD minimisation at Reddale Coal Mine was the construction of the Ferndale ELF by 4m lifts. ELF construction using short lifts reduces chimney effects, which are prevalent in overburden disposal areas that utilise end tipping (where the end tip > 4-6 metres). Significant chimney effects develop in tipheads (> 4-6m) when the coarse and fine materials separate producing coarse grained layers (chimneys) for oxygen ingress and fine grained layers for water ingress with a high permeability rubble layer at the dump base (e.g., Fala et al., 2003; Wilson, 2008). Furthermore, to provide initial acid buffering capacity, and mitigate the initial effects of AMD every 4 m lift had 15 mm (19 kg/m²) of aglime (AP < 2.5mm limestone) applied to prevent the onset of acid mine drainage. An application rate of 19kg/m² equates to 2.5 kg of aglime per tonne (or ~ 2.5 kg CaCO₃/tonne) (based on a cubic metre density of 1.9).

The time lag to acid onset in full scale dumps is dependent on complex scale-up factors that include dump construction (e.g., end tipping, end tip height, paddock dumping), hydrologic processes (e.g., unsaturated dumps and the effects of unsaturated flow), oxygen diffusion into the dump (influenced by waste rock dump permeability, geometry, and thermal regime), temperature (e.g., climate), mineral weathering rates (which are a function of the above and grain size of the waste rock), and microbial activity. Most of these factors are not considered in laboratory based trials and researchers are looking at scale-up of lab results to large scale field trials to evaluate the expected diminishment in acid loads at site.

A number of monitoring systems are used to understand the physical factors affecting the geochemical processes in the field, including lysimeters and leachate monitoring of field trials; monitoring of oxygen concentrations and temperature; and permeability testing using falling head methods. Our paper uses this data to understand the physical factors that are contributing to much slower geochemical reactions in the field.

2.0 MATERIALS AND METHODS

2.1 Site Description

The Reddale Coal Mine is located approximately 1.5 km north east of Reefton within the Brunner Coal Measures (BCM) that is associated with the Reefton Coalfield and is overlain in the area by recent alluvial gravels. The mine commenced operations in early January 2012. Current plans are that 1.3 million bank cubic metres of overburden (920,000 m³ of BCM; 364,000 m³ of alluvial gravels) will be excavated over the current planned mine life (3 years) within a total disturbance area of 29.1 ha.

2.2 Sampling

Two drillholes were used to assess the acid generating potential of rock from the Reddale Coal Mine (Drillholes 662 and 663) and were drilled in April 2005 prior to mining. A total of 36 samples were gathered from these drillholes for ABA testing. These samples were used to characterise the site and develop the AMD management strategy. Another 6 samples were obtained from drillholes DH697, DH699, and DH703, ~325 – 800 m from the Reddale Coal Mine and are considered representative of the area and were obtained in early 2012 after mining at Reddale Coal Mine had started. Two alluvial gravel samples were also tested.

2.3 Acid Base Accounting Techniques

Paste pH testing (e.g., IWRI and EGi, 2002) was conducted to determine the pH of the samples after the addition of water. The method adopted for paste pH involved mixing the sample with water at a solid to water ratio of 1:2 (50g : 100ml) with the pH measured on the slurry after 5 minutes.

ANC (IWRI and EGi, 2002) was not conducted as part of the pre-mining investigations for samples obtained from Drillholes 662 and 663, although this was done for later samples. Total sulfur was determined by LECO analysis. Forms of sulfur were determined by method AS 1038.11-1993. Sulfate sulfur was determined directly as a sulfate in a hydrochloric acid extract of the sample; pyritic sulfur is calculated from the determination of iron soluble in nitric acid following removal of non-pyritic iron by hydrochloric acid; and organic S is determined by difference. MPA was determined by calculation as discussed based on either total S or pyritic S (which is discussed).

The NAG test used and reported here was refined by IWRI and EGi (2002) and is based on earlier similar procedures. This test involves the addition of 250 ml 15 vol.% (unstabilised) hydrogen peroxide (with a measured pH of 4.5) to 2.5 g of pulverised sample (< 75 µm), to encourage the rapid oxidation of reactive sulfides.

2.4 Column Leach Testing

Two column leach tests were undertaken on drill core composites using the methodology developed by IWRI and EGi (2002). Testing commenced 6 months after drilling was completed and operated for 2 years starting in December 2005 and finishing in December 2007. 1.5 kg of BCM material (MPA = 32 kg H₂SO₄/t; ANC not determined) and 1.5 kg of alluvial gravels (ANC = 1 kg H₂SO₄/t; MPA = 12 kg H₂SO₄/t; NAPP = 11 kg H₂SO₄/t) were assessed.

Other columns were set up using the six samples that were obtained from the Burkes Creek Expansion project from drillholes DH697, DH699, and DH703, ~325 - 800 m from the

Reddale Coal Mine as the most suitable unweathered representative samples to rerun column leach tests. These columns started within 1 month of drilling and operated for three months.

2.5 Oxygen Probes and lysimeters

Oxygen probes were installed in the Ferndale ELF upon completion of every second 4 m lift of overburden. The 30 m long, 0.5 m wide trenches were constructed perpendicular from the ELF face. At intervals of 4, 8, 15, 20, and 30 m along the trench, a 0.5 m³ sample collection pit (~0.5 m x 1.6 m x 0.6 m deep) was excavated below the base of the trench. These sample pits were backfilled with large (> 40 mm) rounded gravels and the oxygen probe terminus was within this chamber. The trench was then backfilled and track rolled to recompact. A Teledyne Analytical Instruments portable oxygen analyser with built in vacuum pump was used to extract pore water samples from the five sample collection pits. The pump was run for up to two minutes to purge the conduit before a stable oxygen content was reported by the meter. Between measurements, the oxygen meter was re-calibrated to an atmospheric oxygen content of 20.6%.

Lysimeters were installed in trenches that ran parallel to the oxygen probes but were separated by 5m. Two lysimeters were installed in every second lift at a distance of 10 and 15 m into the ELF. Sample collection from lysimeters is gravity driven. The lysimeters consisted of three components; a leachate collection drum, a conduit and a sample collection bucket that was on the exterior ELF surface.

2.6 Falling Head Permeability

Four falling head permeability tests were carried out on the Ferndale ELF to determine the permeability of different materials under different compaction conditions. Two tests were completed on a machine compacted bench (dozer and dump truck). One test was done on uncompacted overburden which had been pushed up into a windrow. One test was done on compacted gravels. Permeability testing involved driving a 1.7 m long, 300 mm diameter cylinder made of 6 mm steel 300 mm into the material using a digger bucket. The columns were then completely filled with water (~ 1.4 m head) and left to saturate the underlying material overnight. The following day the columns were topped up and the permeability test was started, with daily water level recordings made over several days. The gravel columns drained almost immediately so water level recordings were made over a 90 second period to determine permeability.

2.7 Monitoring of the Ferndale ELF

Thirty random representative samples were collected from the Ferndale ELF on 4 separate occasions 1-4 months apart. Sufficient sample was obtained to undertake a paste pH test at a ratio of 1 part rock to 2 parts water. Paste pH can provide an indication of the amount of sulfide oxidation in a sample and the presence of any subsequent highly soluble stored acidic salts. The method adopted for paste pH involved mixing the sample with water at a solid to water ratio of 1:2 (50g : 100ml) with the pH measured on the slurry after 5 minutes.

2.8 Field trials

Field trials were established in early 2012 on site to compare the benefits of the addition of limestone to the ELF. The trial pad was divided into two 3 m x 3 m sections. Two sets of nine 200mm diameter lysimeters were then arranged in a 3 x 3 grid on the left and right sections with the middle section left empty. The lysimeters drained individually to 20 L

buckets. Clean river rounded gravels were used to form up a 300 mm high flat pad, 3 m wide and 9 m long over the lysimeters; the lysimeters were thus filled with gravels as well. The surface of one of the pads was then dusted with aglime at a rate of 5.7 kg limestone per m^2 , which is proportional to the application rate in the Ferndale ELF over 4 m lifts. Approximately 60 t of run of mine PAF mudstone (~ 0.4 wt.% sulfur) was then spread over the whole trial pad to a depth of 1.2 meters above the lysimeters using an excavator. The surface was flattened off at least 1 m wider than the footprint of the lysimeters to maintain one dimensional flow through the PAF layer above the lysimeters. The leachate collected in the buckets from the preferential flow trial was sampled approximately monthly. The sample bucket was weighed (to determine vol.) and the pH was measured. A sample was taken for acidity determination in the laboratory.

3.0 RESULTS AND DISCUSSION

3.1 Acid-base accounting

The ABA results presented (Figure 1) indicate that the majority of samples, are PAF as determined by NAG testing. For the samples obtained from Drillholes 662 and 663 no ANC was measured so NAPP data is biased towards PAF or Uncertain classification. Even a small amount of ANC can result in a time lag to onset of acidity for PAF samples and this can be predicted from NAG pH versus paste pH testing (e.g., Weber et al., 2006). Figure 2 indicates that the majority of samples obtained from early drillhole samples at Reddale Coal Mine will have a time lag to acid onset.

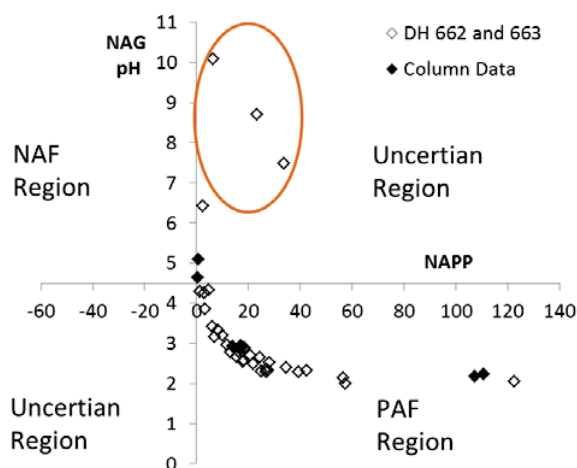


Fig. 1. NAPP versus NAG pH Plot. NAPP in $\text{kg H}_2\text{SO}_4/\text{t}$ equivalent. DH 662 and 663 ABA data obtained prior to mining and NAPP is based on the assumption that $\text{ANC} = 0 \text{ kg H}_2\text{SO}_4/\text{t}$ equivalent.

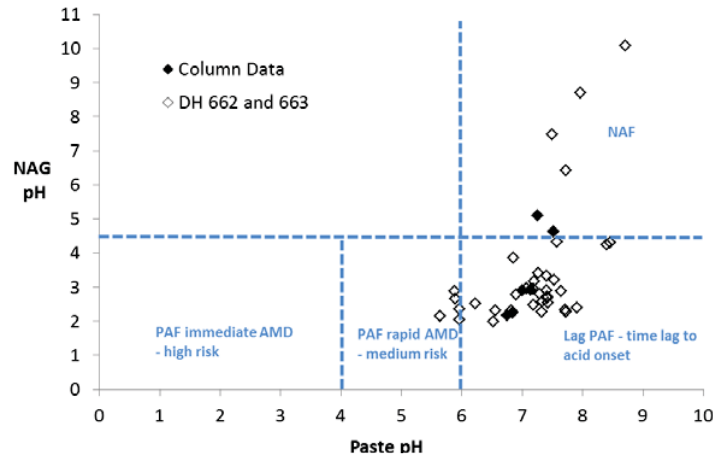


Fig. 2. Paste pH versus NAG pH Plot. DH 662 and 663 ABA data obtained prior to mining. Column data was obtained from nearby drillholes.

Column data samples (Figure 1) were obtained from nearby drill holes and NAPP includes ANC for these samples. There is a good correlation except for the three samples circled, due to there being no ANC data being considered for these samples. Similarly, these uncertain category samples may be due to organic S and thus overestimation of the MPA.

To confirm the new expected geochemical trend (Figure 2) a total of fifty representative grab samples were collected from the Ferndale ELF during the placement of the overburden and were assessed for paste pH, ANC, and total S. Results indicated that a small quantity of ANC was present in both overlying gravels that became the base of the Ferndale ELF (mean = 4.2 kg H₂SO₄/t equivalent) and for the BCM PAF rock (mean = 5.9 kg H₂SO₄/t equivalent), indicating there is more ANC available in the PAF BCM than the alluvial gravels.

Sulfur speciation work indicated that the majority of the S was present as pyrite in both the alluvial gravels and the PAF BCM overburden. However, the results indicated that the non-acid forming organic S fraction is significant, reducing the MPA determination for the BCM PAF overburden from a mean of 33.3 kg H₂SO₄/t equivalent (based on total S) to a mean of 27.8 kg H₂SO₄/t equivalent (based on pyritic S).

Neither ANC nor sulfur speciation was considered during the pre-mining ABA characterisation of the Reddale mine site. The mean ANC for the PAF BCM is 5.9 kg H₂SO₄/t equivalent and MPA calculated from total S is 5.5 kgH₂SO₄/t higher than the MPA calculated from pyritic S. If the NAPP was calculated for this tiphead dataset using only MPA based on total S (and assuming ANC = 0 kg H₂SO₄/t equivalent) then the NAPP would be 33.3 kg H₂SO₄/t equivalent for the BCM. Taking into account ANC and using MPA based on pyritic S indicates the more correct NAPP is 21.9 kg H₂SO₄/t equivalent. The conservatism associated with assuming ANC = 0 kg H₂SO₄/t equivalent for BCM and using total S rather than pyritic S at Reddale alters the acid base accounting balance by about a third and are therefore significant.

3.2 Column Leach Test Results

Two columns were established in December 2005 and were operated for 2 years finishing in December 2007, however the columns were not started until 6 months after drilling and the core was not stored appropriately enabling oxidation processes to occur prior to testing. Acidity and pH results indicated low pH and elevated acidity from the first month and no time lag to acid onset could be determined. In total, 8.8 kg H_2SO_4 /tonne equivalent acidity was leached from the BCM column over the 2 years of operation. The column BCM rock sample had a measured MPA (based on total S) of 32 kg H_2SO_4 /t acidity prior to starting, thus more than a third of the acid load had leached from the remaining sulfides after two years confirming the potential for significant acid mine drainage from the BCM at Reddale.

In total 0.01 kg H_2SO_4 /t equivalent acidity was leached from the alluvial gravels, which is negligible. However, leachate from the alluvial gravels column had near neutral pH and it is likely substantial alkalinity could have been leached from the gravels over the two years of testing, which would help offset any AMD formation. Unfortunately, the alkalinity was not quantified in these lab trials.

To understand whether any time lag to acid onset is possible for rocks obtained from the Reddale site, a number of fresh drill core samples (< 1 month old) from the nearby Burkes Creek site were tested by column leach testing. The samples tested have ANC from 3.7 to 13.7 kg H_2SO_4 /tonne equivalent and have paste pH values greater than 6. This indicates that there would be a time lag to acid onset based on the NAG pH – paste pH classification process. In addition compared to earlier columns there is a time lag to acid onset in these column leach tests (not shown).

3.3 Oxygen Probe Monitoring

Oxygen probe monitoring was carried out monthly (Figure 3) and shows the oxygen content data collected from the two sets of oxygen probes. This data demonstrates that the oxygen content in pore space gas in the dump decreases rapidly from 20.6% (atmospheric) outside the dump to less than 5% at depths greater than 8 m into the dump. The oxygen content of the outer 8 m skin varies between 1 and 9%. The higher oxygen content readings were recorded in the initial monitoring periods, 3 to 44 days after oxygen probe installation. This suggests that initially atmospheric oxygen penetrates the dump to a depth of up to 8 m. After 44 days the oxygen content of the 4 m probes became similar to that of the deeper probes. Over the preceding 44 days, additional material was dumped over the oxygen probes and compacted by heavy vehicle traffic, resulting in a reduction in permeability of overlying material. Following this compaction and stabilisation period, the reactive skin of the dump thins to less than 4 m.

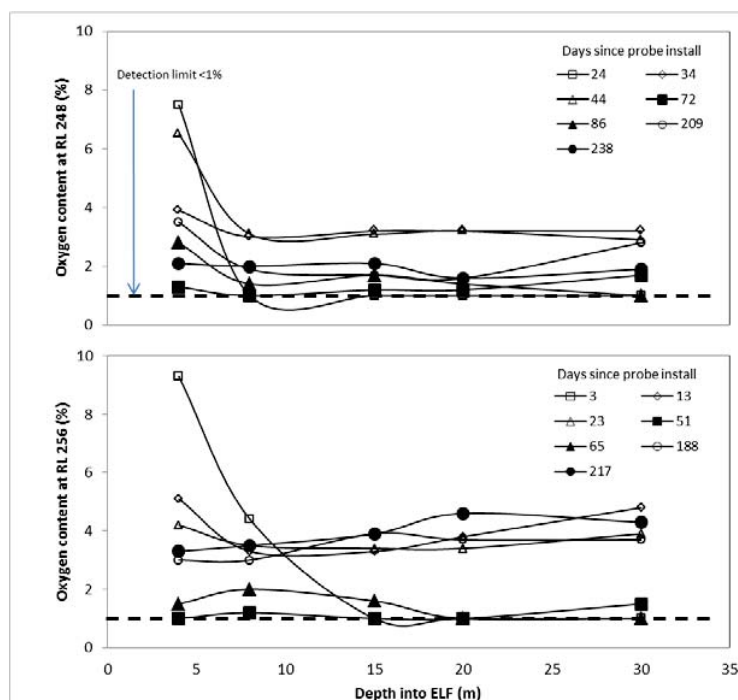


Fig. 3. Ferndale ELF Oxygen content at RL 248 and 256. Data from obtained from 4-5-12 to 24-1-13.

3.4 Lysimeters

Leachate water samples did not report through the lysimeters until the 24th January 2013. The low permeability of overlying material limits percolation through the dump, delaying capture of any drainage. Only a few samples were obtained. Results from 15m horizontally into the ELF indicated that pH remains near neutral (pH 6.3 - 6.6) and acidity (to pH 7.0) was low 5 mg/L CaCO₃. These results indicate that at depth within the ELF the majority of the material is beyond the limit of significant oxygen ingress and indicates the core of the ELF is not producing significant acidity.

3.5 Permeability

The four permeability tests were carried out from the 6th to the 14th November 2011. Results indicated the permeability of the heavy vehicle compacted PAF ($10^{-7.1} - 10^{-8.0} \text{ m.s}^{-1}$) was lower than the permeability of the un-compacted PAF rocks ($10^{-6.6} \text{ m.s}^{-1}$). Thus, vehicle compaction of the PAF materials decreases permeability by an order of magnitude.

3.6 Paste pH Monitoring

The outer surface of the ELF was sampled four times over a period of 9 months and tested for paste pH. A clear trend is identified of decreasing paste pH from 5.2 (6 months after mining started) to 3.0 demonstrating that the outer surface of the dump is becoming more acidic with time and producing acid runoff (Figure 4).

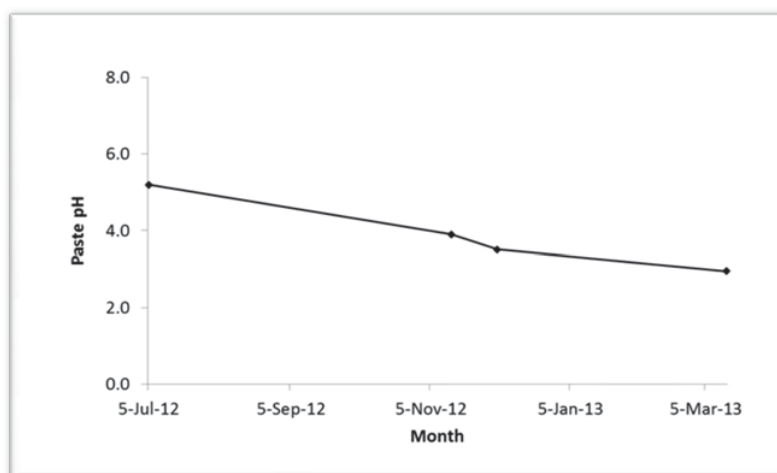


Fig. 4. Paste pH values from the Ferndale ELD surface.

pH has been monitored on a daily basis at a monitoring location downstream of the Reddale Coal Mine site within Burkes Creek. Results indicate the pH has remained fairly consistent, and is comparable to pre-mining conditions of pH ~ 6-7. It may therefore be concluded the onset of depressed pH associated with significant AMD has not eventuated, as predicted by the pre-mining geochemistry analysis.

3.7 Field Trials

The results for the field trial are provided in Table 1. For a well-controlled uniform system it is interesting to note the variation in flow rates, demonstrating preferential flow paths. If such variable flow occurs in a trial, then its occurrence within the Ferndale ELF is almost certain. Results also demonstrate that with increasing flow there is an increase in acid load, which indicates that more secondary acidic oxidation products are being flushed out with higher flow rates. The use of limestone, although not preventing AMD from developing significantly reduced the amount of acid departing the system. The average acid load for the limestone trial was 43.5 g CaCO_3 eq. / m^2 whereas the control was 539 g CaCO_3 eq. / m^2 . Over the course of the 6 month trial the average acid load neutralised by the addition of the limestone was ~500 g CaCO_3 eq. / m^2 . When compared to the starting application rate of 5.7 kg limestone per m^2 this indicates an efficiency of 9%.

After 6 months the measured acidity of the limestone amended pad had increased significantly (not shown) indicating exhaustion of the limestone neutralisation capacity. This suggests the time lag to acid onset created by the addition of neutralent was within 6 months following trial pad construction. By inference there is potential that this time lag to acid onset would be occurring in the ELF where oxygen is freely available (e.g., the outer oxidising zone where oxygen > 10%). Where oxygen ingress is limited in the core of the ELF the time lag to acid onset is expected to be greater.

Table 1. Combined total flow rate, acidity, and acid load for the field trial pads. Data presented on a per m² basis. Monitoring period was from June 2012 to Jan 2103.

Limestone				Control			
<i>Lysimeter</i>	<i>Volume (L/m²)</i>	<i>Average acidity (mg CaCO₃ eq./L)</i>	<i>Acid load (g CaCO₃ eq./m²)</i>	<i>Lysimeter</i>	<i>Volume (L/m²)</i>	<i>Average acidity (mg CaCO₃ eq./L)</i>	<i>Acid load (g CaCO₃ eq./m²)</i>
LA1	104	350	36	A1	81	2901	284
LA2	57	60	5	A2	66	3113	458
LA3	39	38	2	A3	163	2749	531
LB1	147	1813	199	B1	199	4673	880
LB2	80	450	23	B2	104	4223	673
LB3	58	8	0	B3	149	3180	558
LC1	38	536	15	C1	90	3383	426
LC2	61	856	58	C2	195	3908	774
LC3	83	468	54	C3	84	2534	267

4.0 CONCLUSIONS

At this site, the assumption that the Brunner Coal Measures do not have significant ANC is incorrect. As shown here at Reddale there may be sufficient ANC (~5.9 kg H₂SO₄/t equivalent) to provide a time lag to acid onset. Omission of this ANC is conservative and conservatism can lead to over-engineering and greater project costs. Similarly, use of total S for MPA determinations of the Brunner Coal Measures is conservative and MPA can be lower if pyritic S is used. At Reddale the use of pyritic S reduces MPA by ~5.5 kg H₂SO₄/t equivalent compared to using total S content for the MPA calculation. Thus, original ABA data suggested that acid potential could be ~ 33.3 kg H₂SO₄/t equivalent, whereas the inclusion of ANC and MPA based on pyritic S results in a NAPP of 21.9 kg H₂SO₄/t equivalent; close to a 30% decrease in acid potential. Approximately 2.5 kg CaCO₃/tonne has been added to each 4m lift within the Ferndale ELF thereby increasing the total ANC within the dump to ~8.4 kg CaCO₃/tonne. This will further contribute to acid neutralisation within the dump and again extend the time lag to acid onset.

The oxygen gradient to < 5wt% oxygen within the first 4-8m indicates that diffusion is the dominant gas transport mechanism. Paste pH results of the ELF surface indicate that acid generation is occurring on the surface of the ELF where oxygen is freely available; yet at depth, 10m into the ELF, near neutral drainage is still being observed.

The work presented here indicates that the Ferndale ELF can be divided into two geochemical zones, an outer oxidising skin where oxygen is freely available (>5%) and where the dominant sulfide oxidation reactions are occurring, producing acidity; and an inner core that has reduced oxygen content (< 5%) and sulfide oxidation rates are much lower and percolation remains near neutral. It should be noted that the oxidising skin model proposed here is specific to the site. In some waste rock dumps this oxidising skin may encompass the whole dump (Ritchie, 1994).

The field trials demonstrated that preferential flow occurs and that the highest flow rates also delivered the highest acid load. This indicates increased flush rates of acidic salts with increasing flow. Limestone application reduced acid loads and was 9% efficient in this instance. Results indicate that where oxygen is freely available, the addition of the prescribed

quantity of aglime used in the trial will provide about 6 months time lag to acid onset and the trial is probably a reasonable analogue for the outer oxidising zone of the ELF.

The ABA testing indicated that there is likely to be a general delay to AMD onset due to inherent ANC of the PAF overburden, combined with the application of aglime to the ELF. In conjunction with reduced oxygen ingress there is likely to be a greater delay to significant AMD generation. In retrospect, the upfront installation of a $\text{Ca}(\text{OH})_2$ water treatment plant was thus unnecessary and a better option could have been the design of a low cost passive treatment system.

5.0 ACKNOWLEDGMENTS

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THE INERT GAS MIXTURE (IGM) TECHNOLOGY TO PREVENT ACID MINE DRAINAGE

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ABSTRACT

Current methods for Acid Rock Drainage (ARD) mitigation revolve around water covers and dry covers (soils and liners) technologies. Both are of limited effectiveness (e.g. application in limited circumstances, requiring perpetual care and maintenance of water dams infrastructures and related appurtenances) and present significant drawbacks (e.g. high cost, shortages of suitable soil at some sites, uncertainty about long-term effects of root penetration, settling, frost heave, etc.). The basis of both dry and water covers technologies is the insulation of waste from oxygen in the atmosphere with the knowledge that the rate at which oxygen is transported to the sulphide waste controls the rate of acid generation. Both indeed reduce oxygen influx and hence reduce, but do not prevent, ARD generation.

The Inert Gas Mixture (IGM) technology has been designed to overcome the deficiencies of gas transfer into tailings and waste rock materials across the surface and instead block oxidation by controlling the transfer of oxygen from the atmosphere. Inert gas is injected into the mine waste at low flow rates and the positive but low flow purges the oxygen that tends to diffuse into the sulphide waste across the surface. The positive flow of IGM compensates for inward oxygen diffusion and creates a pressure drop across the cover layer that prevents inward diffusion of oxygen and hence acid generation.

The IGM processes are evaluated in this paper: (i) theoretically with a review of the theory of ARD and the current state of knowledge about the performance of notably dry covers as atmospheric oxygen barriers; (ii) via models to predict the performance of the IGM technique; and (iii) practically with a bench- scaled experiment. The results presented in this paper demonstrate the effectiveness of IGM to prevent oxidation of sulphide minerals and ARD in waste rock. The laboratory experiments further identified the key variables to be used to monitor performance and control efficiency of IGM which is now awaiting field-scale application.

The IGM technology is a patented process by Terraquest Technologies Ltd (Canada) in USA, Canada and Australia. The IGM can be cost-effectively manufactured on site with commercially available compressors and gas filtration membranes. Furthermore, it is predicted that low-cost covers can then be utilized with IGM which in turn lowers the overall capital costs of ARD mitigation.

1.0 INTRODUCTION

Acid Rock Drainage (ARD), also known as Acid Mine Drainage (AMD), has adversely impacted the receiving environment at numerous mine sites globally and represents a major financial liability to the mining sector and to some extent to civil rock excavation projects such as roads and tunnels. ARD is generated by a natural process when materials containing iron sulphide and/or metal sulphide minerals are exposed to oxygen and water – the two necessary ingredients that must be present together.

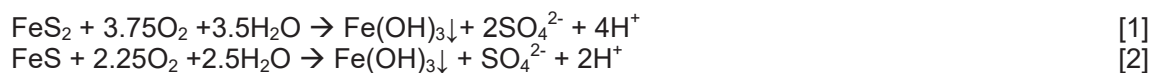
Current methods for ARD mitigation revolve around water covers and dry covers (soils and/or liners) technologies. Both are of limited effectiveness (reducing but not preventing ARD) and also present significant drawbacks.

The Inert Gas Mixture (IGM) technology addresses the deficiencies of notably conventional dry covers. The IGM technology is: (i) a patented process that is intended in part for application to mine waste to prevent oxidation of sulphide or other minerals hence generation of ARD; and (ii) based on the use of an oxygen-deficient gas mixture that can blanket mine wastes in conjunction with low-efficiency or low-cost covers.

This paper presents the theory, modelling and benchscaled experiment of the IGM technique demonstrating the performance of the IGM technology in preventing sulphide oxidation and the generation of ARD in relation to the use of a conventional dry cover and uncovered waste.

2.0 THEORY

ARD results from the natural oxidation of sulphide minerals. When iron or other metal sulphide minerals are exposed to oxygen and water, ARD is generated. Equations [1] and [2] represent for the initial oxidation of pyrite and pyrrhotite, respectively:



ARD is typically characterized by low pH drainage with elevated sulphate and dissolved metals concentrations.

Significant research has been completed in areas related to the prediction, prevention and control, treatment and monitoring of ARD by programs such as e.g. the Mine Environment Neutral Drainage (MEND) program (www.mend2000.nrcan.gc.ca). That body of research and others around the world resulted in the development of the two key technologies presently available to control the generation of ARD from sulphide-bearing rock and wastes: (i) water covers; and (ii) dry covers (comprised of soils and liners). The basis of both technologies is the insulation of waste from oxygen in the atmosphere with the knowledge that the rate at which oxygen is transported to the sulphide waste controls the rate of acid generation.

Underwater storage appears to be one of the most effective methods of managing acid-generating wastes since water can be an effective barrier to the transport of atmospheric oxygen. However, water covers can only be applied in limited circumstances where the water balance is appropriate, among other constraints, and may also require the perpetual care and maintenance of related water- retaining dams, water ponds and appurtenances.

Practical constraints often leave dry covers as the only viable prevention option for many existing and planned waste deposits. Dry covers consisting of soil materials are widely used; however research and monitoring data suggests that they reduce oxygen influx and hence reduce, but do not prevent, ARD generation. Other significant drawbacks with conventional dry covers are high cost; shortages of suitable soil at some sites; and uncertainty about the long-term effects of root penetration, settling, frost heave, etc. on dry cover performance.

Scientific research has assessed oxygen diffusion through soil covers and the relationship between oxygen flux and cover characteristics including but not limited to the moisture content, diffusion coefficient for oxygen, and interstitial tortuous pathway. Nicholson *et al.*

(1989) and MEND 2.22.2a (1996) have shown that a key design objective of dry covers over reactive mine wastes is to limit the ingress of atmospheric oxygen by maintaining a high degree of moisture saturation within the cover void spaces. Yanful (1993) reported that increases in the degree of saturation of a soil cover from 0 to 50% resulted in only minor effects on the diffusion coefficient for oxygen. However, substantial decreases (orders of magnitude) in the oxygen diffusion coefficient occurred at higher degrees of water saturation. Furthermore, Bussi re *et al.* (1995) showed that an 85 to 90% degree of saturation of a soil cover may be required to achieve effective mitigation of oxygen ingress to underlying mine wastes.

Nicholson *et al.* (1989) determined that if the voids of a soil cover are water-filled, then the predominant mechanism for oxygen to travel to the underlying reactive wastes is diffusion through the pore water. Therefore, in the absence of sufficiently water saturated pore spaces within a soil cover, a conventional soil cover is generally not effective enough to mitigate atmospheric oxygen ingress to the point that water quality in the waste is protected. Atmospheric oxygen molecules migrate by diffusion through soil covers by way of complex tortuous pore space pathways. Moisture is present in these preferential pathways and oxygen flux is mitigated since the oxygen flux through water is about 5 orders of magnitude lower than that through air. The IGM technology addresses these mechanisms by creating a small pressure differential across the cover and promotes a tendency for excess inert gas mixtures to migrate from beneath the cover towards the open atmosphere by advection or flow through the same preferential pore space pathways that represent diffusion pathways for oxygen from the atmosphere downward to the wastes. In this way, the IGM counteracts oxygen transport and prevents oxygen ingress through the cover dump.

3.0 MODELLING

Over the past two decades, numerous studies in the field and the laboratory have focused on gas transport in mine waste and in covers that have been constructed to reduce oxygen ingress to the waste material. The sulphide-bearing mine waste acts as a “sink” for oxygen and induces oxygen deficiencies within the waste that create concentration gradients between the atmosphere (where oxygen is about 20% by volume of air) and the pore gas within the waste (where oxygen levels can diminish to negligible values). Oxygen transport into and within coarse uncovered waste rock piles can be supplemented by the natural flow (or advection) of gas, whereas the main mechanism for gas transport in tailings (silty to sandy material) and in soil-type covers is diffusion controlled. The diffusion occurs as a result of the oxygen concentration gradients that develop between the oxygen-rich atmosphere and the oxygen-deficient pore gas in the covered mine waste material.

The flux of oxygen is described by Fick’s first law and has the form: $F = D_{eff} \left(\frac{\partial C_a}{\partial z} \right)$ [3]

where: F = oxygen flux from the atmosphere (kg/m²/s)
 D_{eff} = effective diffusion coefficient (m²/s)
 C_a = concentration of oxygen in the air (kg/m³)
 z = depth from ground surface (m)

Ritchie (1994) indicated that the diffusion coefficient in waste rock piles is typically on the order of 5×10^{-6} m²/s. This value is only slightly lower than the diffusion coefficient for oxygen in air. In contrast, Reardon and Moddle (1985) showed that measured oxygen diffusion coefficients in tailings were very dependent on the degree of moisture in tailings, and ranged from about 4×10^{-6} m/s for dry tailings to about 2×10^{-8} m/s in tailings that were up to 80% saturated with water (80% of the pore volume was filled with water and 20% of the volume remained as gas filled). Elberling *et al.* (1994) confirmed the measurements and presented

Eqn. [4] that relates the effective diffusion coefficient (D_{eff}) to the degree of water saturation (S).

$$D_{eff} = \tau D_a^0 (1 - S)^\alpha + \left(\frac{\tau S D_w^0}{H} \right) \quad [4]$$

where: D_a^0 = diffusion coefficient of oxygen in air (m^2/s)
 D_w^0 = diffusion coefficient of oxygen in water (m^2/s)
 S = water saturation (m^3 liquid / m^3 pore space)
 τ and α = dimensionless fitting parameters
 H = Henry's law constant (dimensionless)

This relationship provides a basis for calculating the diffusion coefficient in tailings or similar silty to sandy "soils", that may be used as covers on mine waste, if the moisture content or saturation value is known or can be predicted. Figure 1 shows this relationship and that the smallest value for the diffusion coefficient for fully saturated condition is about $2 \times 10^{-11} m^2/s$ for oxygen concentration units in air.

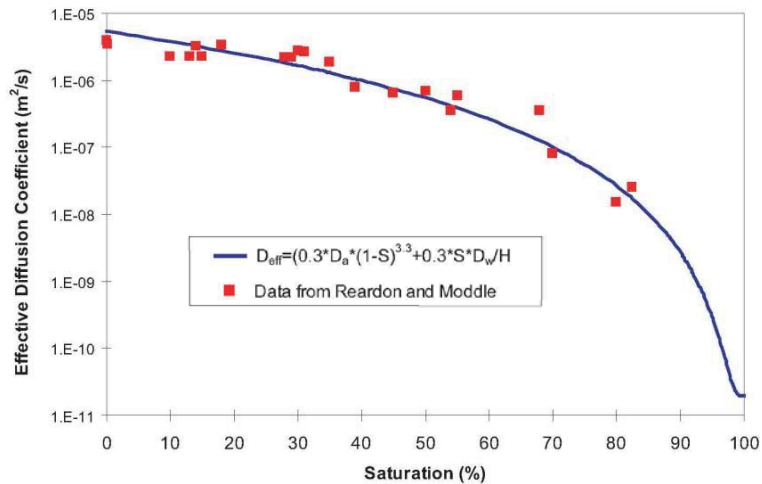


Fig. 1. Relationship between oxygen effective diffusion coefficient and water saturation. Squares are data from Reardon and Moddle (1985).

Nicholson *et al.* (1989) described the use of covers on tailings to reduce oxygen ingress and modelled oxygen diffusion rates through covers for different conditions to demonstrate the theoretical effectiveness of diffusion restriction of oxygen transport and concurrent reduction in oxidation rates within the tailings. The modelling showed that, in theory, oxygen flux values (and oxidation rates) could be reduced by about five orders of magnitude. In practice, however, covers rarely achieve such effectiveness, because the pore spaces are never 100% saturated and there can be zones or areas that are far less saturated.

Elberling and Nicholson (1996) demonstrated a method to measure oxygen flux into tailings or other soils using the Oxygen Consumption Method (OCM). This method used a flux chamber at the soil surface, sealed to the top of a tube that is inserted vertically into the ground. The oxygen concentration in the headspace of the chamber was then monitored with time to observe the concentration decrease as oxygen diffused downward into the tailings or soil. The authors presented the first actual field measurements of oxygen fluxes into mine waste. Tibble and Nicholson (1997) applied the OCM to measure oxygen flux values on a variety of tailings and through covers that were established as experimental test plots. The results showed that engineered cover layers had oxygen flux values that were on the order of about $10 \text{ mol/m}^2/\text{a}$ ($0.32 \text{ kg/m}^2/\text{a}$) compared to uncovered sulphide tailings that exhibited flux values from about 30 to as high as $3000 \text{ mol/m}^2/\text{a}$ ($1 \text{ to } 100 \text{ kg/m}^2/\text{a}$). If we consider the flux through the cover and convert it to a volumetric flux, the value of $10 \text{ mol/m}^2/\text{a}$ would be

equivalent to about $0.224 \text{ m}^3/\text{m}^2/\text{a}$ (1 mole = 22.4 L at STP). Therefore, the volumetric flux of oxygen is about $7.1 \times 10^{-9} \text{ m/s}$.

Gas transport at low velocities and at constant densities (a good approximation for low pressure systems near atmospheric) can be modelled with water transport equations. For the concept of IGM technique, gas transport through the cover was modelled using a one-dimensional advection-diffusion equation for which an analytical solution is available.

Figure 2 shows a schematic of a modelled rock pile with a soil cover where the IGM is being injected into the pile. In such a case, the gas flows very readily through the large voids in the rock pile that has very high gas permeability. The soil cover represents more resistance to flow so that the IGM spreads out through the rock pile and slowly “escapes” through the cover layer to the atmosphere. The gas flow that is occurring through the cover can be very small and still prevent the inward migration of oxygen from the atmosphere.

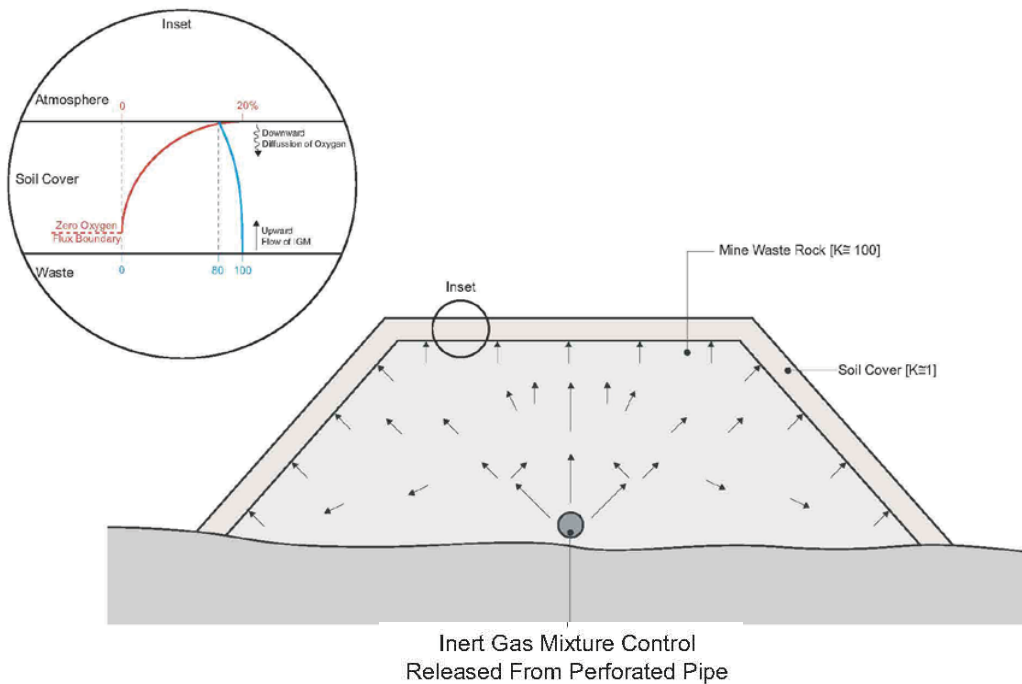


Fig. 2. Schematic showing application of Inert Gas Mixture (IGM) to interior of covered waste rock pile.

In the modelling stage of IGM, a one-dimensional advective-diffusive transport concept, Eqn. [5], was used to investigate the oxygen profile versus cover thickness.

$$\frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial z^2} + v \frac{\partial C}{\partial z} \quad [5]$$

where: C = gas phase oxygen concentration (mol/m^3)
 D = the effective oxygen diffusion coefficient (m^2/s)
 v = gas velocity (m/s)
 z = distance from surface (m)
 t = time (s)

The preliminary modelling assessment included the following specific objectives: (i) determine the range of gas flow required for cover layers with a realistic range of gas diffusion coefficient values (the diffusion coefficient is controlled mainly by moisture content); (ii) estimate the necessary cover thickness to inhibit oxygen ingress by counter-current flow of IGM; and (iii) determine the total gas flow rate that is required for a reasonable full-scale waste rock stockpile.

The model was solved for steady-state conditions so that oxygen profiles in the cover layer can be calculated. Steady-state occurs as a balance between the downward diffusion of oxygen and the upward flow of the IGM (oxygen-free gas). Three steady-state profiles are shown in Figure 3. The vertical axis is the thickness of the soil cover layer on the waste rock, with zero being the surface in contact with the open atmosphere where the relative concentration of oxygen is equal to one (20% oxygen). The relative concentration of oxygen is shown along the horizontal axis as zero to one (or zero to 20% oxygen by volume). A similar oxygen profile is shown schematically in the circled insert in Figure 2 across the cover of the waste rock stockpile to depict the steady-state decrease in oxygen with depth.

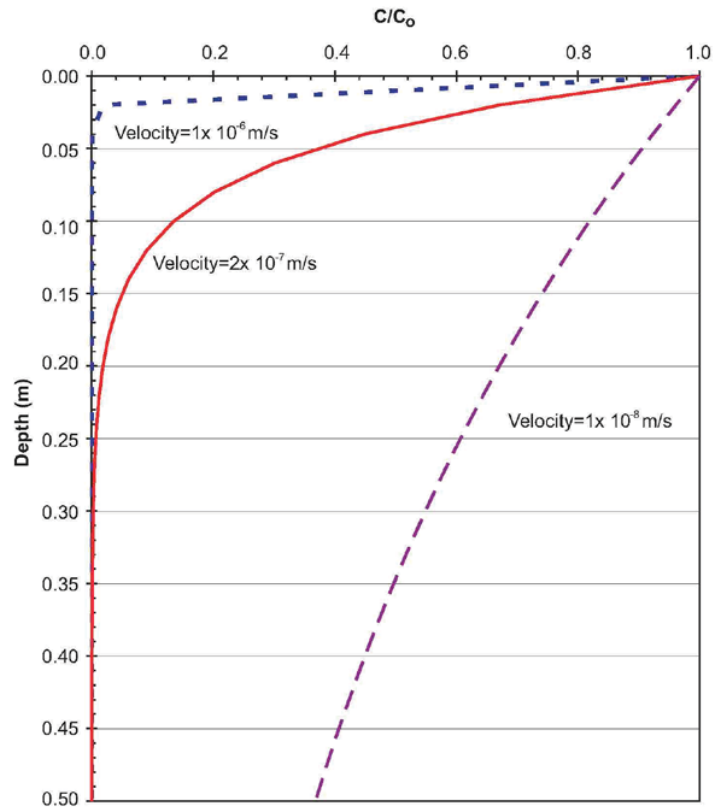


Fig. 3. Steady state relative oxygen concentration vs. depth.

The profiles in Figure 3 correspond to different gas velocities induced by addition of the IGM below a cover. The lowest velocity (1×10^{-8} m/s) shown was insufficient to prevent diffusion of oxygen downward through the bottom of the cover in this example, and the relative oxygen concentration at a depth of 0.5 m is about 0.35 (or 7.35% oxygen). This means that the gas delivery rate for the IGM in this case will not prevent oxygen from entering the waste rock but will reduce the flux of oxygen. At the largest velocity considered in this example (1×10^{-6} m/s), oxygen penetrates the cover to a depth of only about 0.05 m (or 5 cm), and the steady upward flow of IGM prevents the net downward movement of oxygen past that depth. The intermediate IGM velocity also prevents oxygen from diffusing across the bottom boundary of the cover and, in this case, the gas velocity is 2×10^{-7} m/s. For a 10-hectare cover that may be representative of a moderate waste rock stockpile assuming an air filled porosity of 0.3 in the cover, this velocity translates to an IGM flow rate of about $22 \text{ m}^3/\text{hour}$ (or 13 cfm), a rate that can be produced by off-the-shelf compressors that produce the IGM.

The model was then used to calculate the IGM velocities that would be required for different thicknesses of cover layers and different values of diffusion coefficient of the cover material (that is controlled by the amount of moisture in the soil – see Figure 1). It was assumed that

the relative concentration of oxygen (C/C_0) must be less than 0.0001 at the base of the cover to represent an insignificant flux of oxygen through the cover. The calculated velocities are shown as a function of cover thickness in Figure 4 for selected diffusion coefficients in the cover layer. This plot shows that, for a specific diffusion coefficient, the IGM velocity must be greater for a thin cover (0.1 m) and can be smaller for thick cover (0.5 m) to prevent downward diffusion of oxygen. The plot also shows, as expected, that larger gas velocities are required for larger values of the diffusion coefficient in the cover. This is expected because a high diffusion coefficient in the cover layer represents a poor barrier to oxygen and will also be more permeable to gas flow. However, even at moderate diffusion coefficient values in the cover (*i.e.* $1 \times 10^{-8} \text{ m}^2/\text{s}$), a 0.5-m thick cover requires a velocity of only about $2 \times 10^{-7} \text{ m/s}$. These results demonstrate that gas flow requirements, to prevent inward diffusion of oxygen, are in the realm of practical and achievable values.

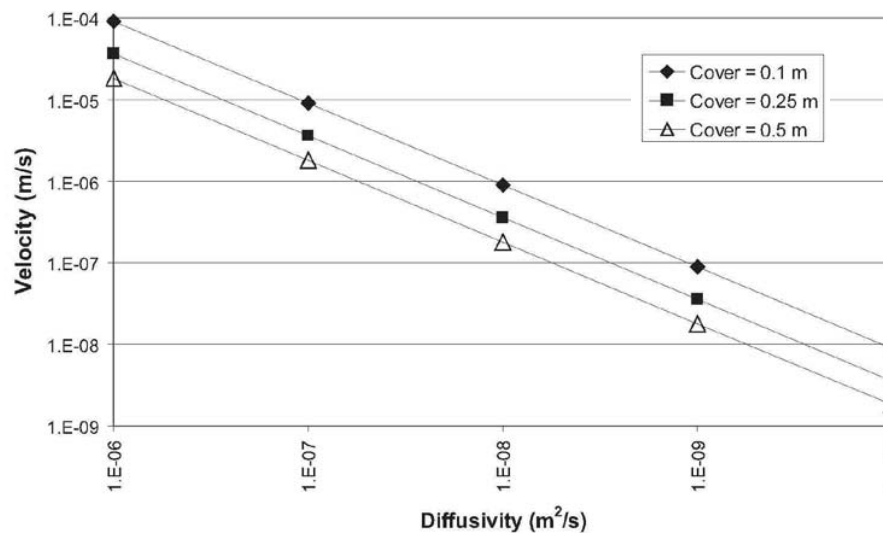


Fig. 4. IGM velocity versus diffusion coefficient for selected cover thicknesses.

A preliminary assessment of defects in the cover layer was also completed. The assessment considered a defect as an area of the cover with an air permeability greater than the design value. Different percentage of the cover area and different ratios between Defect-permeability and Design-permeability were assessed. As an example, assuming a reasonable Defect-permeability/Design-permeability ratio of 100 and a defect fraction over 1% of the total cover area, then the flow needed to compensate for the defect, while preventing oxygen ingress, would be a factor of two greater than the design value with no defects. If the same defect occurred over 5% of the total cover area, then the flow needed to compensate for the defects would be a factor of six times the design flow with no defects. These results suggest that the presence of permeability defects will not result in unmanageable flow rates.

4.0 BENCH SCALE EXPERIMENT

4.1 Experimental approach and material

Three columns were constructed of acrylic to allow visual inspection of the rock and cover and were 0.15 m in diameter and 0.95 m long. All gas delivery and pressure lines consisted of flexible copper tubing with pressure fittings. Each column contained approximately 18.6 kg of reactive rock (the height of which was approximately 80 cm), and those with a cover

contained a 20-cm thick layer of coarse quartz sand. A photo and a schematics of the column test set-up are shown in Figure 5. From left to right in Figure 5a, the columns are:

- Column 1 - IGM. This column was designed to control and assess the IGM system. It contained reactive rock covered by a layer of coarse quartz sand which represented a “poor quality” or low-efficiency soil cover. High purity bottled nitrogen (IGM surrogate) was added to the base of the column in the covered waste rock. This column was continuously monitored as described in the following subsections.
- Column 2 - Control (cover). This column contained reactive rock that was covered by a layer of quartz sand. The column did not involve the use of the IGM technology, and represents a conventional “poor quality” or low-efficiency soil cover.
- Column 3 – Control (no cover). This column contained reactive rock that was left uncovered, and therefore represented an uncovered, exposed waste rock condition.

All columns were loaded with sulphide waste rock and irrigated regularly for about six months to provide a source of moisture to the columns and flush the pore water from each of the columns. The drainage was quantitatively collected and regularly analyzed to provide data on the state of the sulphide oxidation process within each of the columns. The production of metals and acidity in the leachate from each of the columns was used to evaluate the performance of the IGM technology in the lab.

The IGM column (column 1) was instrumented (see schematic in Figure 5b) to also measure the following parameters: gas flow; barometric pressure; differential pressure between the inside and outside of the column; oxygen content in the pore space of the waste rock; ambient temperature and column temperature. The parameter measurements were used to assess relationships among the variables, to demonstrate that values of the key variable were measurable and to assist with scale-up designs for field demonstrations and full scale operations of IGM.

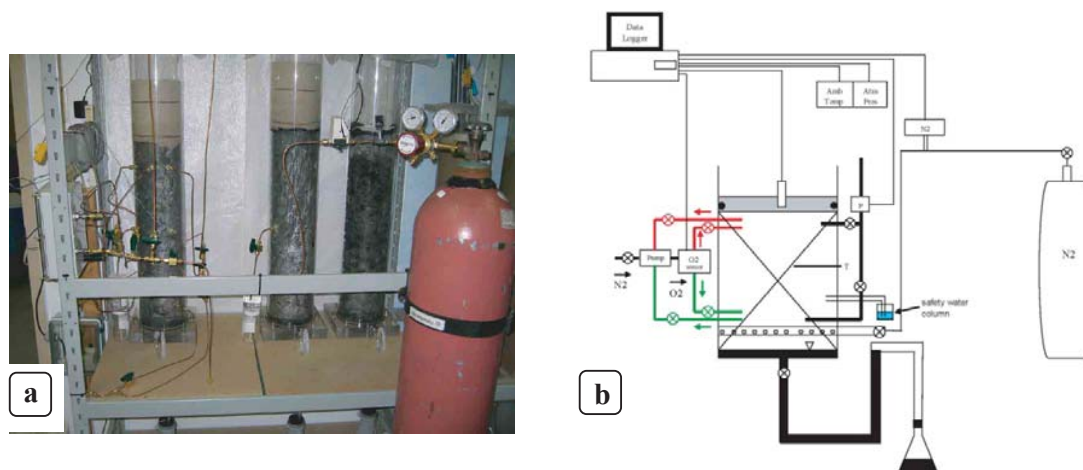


Fig. 5. (a) Columns test setup. (b) Schematic of the IGM column.

The waste rock used for the bench test was a fresh sulphide rock acquired from an active mine located in the Sudbury basin of Ontario (Canada). The rock was a norite with pyrrhotite as the dominant sulphide mineral. Based on test results, the rock sample contained approximately 2.5% sulphide (as S). The Acid Base Accounting (ABA) and elemental

analysis of the waste rock is presented in the Table 1. The ABA characteristics clearly show that the rock will be acid generating if exposed to air and water.

The sample fragments were sized to be larger than one-inch (>25 mm) and less than three inches (<75 mm) across. The crushed rock was initially washed with water, then with a weak nitric acid solution (pH = 4) followed by a final water-wash, and then dried, weighed and loaded into the test columns.

Table 1. ABA and metal analysis of the sample waste rock used as reactive rock for the IGM bench scaled test.

Description	MPA ¹	NNP	NP	Ratio (NP:MPA)	S	Ca	Fe	Ni	Cu	Zn	As
Unit	kg CaCO ₃ /t ore	kg CaCO ₃ /t ore	kg CaCO ₃ /t ore	Unity	%	%	%	ppm	ppm	ppm	ppm
	185.3	-166	19	0.1	2.48	3.41	12.9	7430	1480	105	0.5

1. Maximum potential acidity

Ultra-high pure nitrogen (99.9999% N₂) was used as the IGM surrogate and the flow rate was measured with a digital flow meter. Columns 1 and 2 were covered by a 20 cm layer of coarse non-reactive quartz sand which represented a soil cover. The thickness of the sand was selected to represent a “thin” cover layer. The grain size distribution for the sand is very uniform, with 98% of the material with a grain diameter between 0.43 and 2 mm. The grain size distribution is shown in Table 2. This coarse sand was selected for the cover to represent a material with low water retention capacity.

Table 2. Grain size distribution of the sample non-reactive quartz sand used as cover for the IGM bench scaled test.

Diameter (mm)	< 0.425	0.425 - 0.85	0.85 - 2	2 <
Weight fraction (%)	2	62.4	35.6	0

4.2 Experimental results

Measurements on leachate were taken on all three columns. The time series data (in days from the initiation of column tests) are plotted in Figure 6 and Figure 7 and discussed. Measurements other than those on leachate were only taken on column 1 and included gas flow, oxygen, barometric pressure, differential pressure, temperature and soil moisture. Those later results will not be detailed herein as they would fall beyond the scope of work of this paper.

4.2.1 Trends in pH, Specific Conductivity (SC), Acidity and Calcium

Figure 6a compares the pH of the drainages from the three columns. It is evident that application of the IGM method has prevented acid generation in the waste rock in column 1 that exhibits a neutral pH while the drainages from columns 2 and 3 are acidic, with pH values between 3 and 4.

The pH and SC exhibit an inverse relationship that is observed in Figures 6a and 6b. Lower pH and higher metal laden solutions are associated with higher SC values. Figure 6b shows that column 1 has lower SC than columns 2 and 3 resulting from the lower concentrations of hydronium and metals ions.

Figure 6c presents the acidity released from the columns. The trends are inverse to those of pH. Acidity released under the IGM treatment is almost 20 times less than those under simple cover and uncovered conditions.

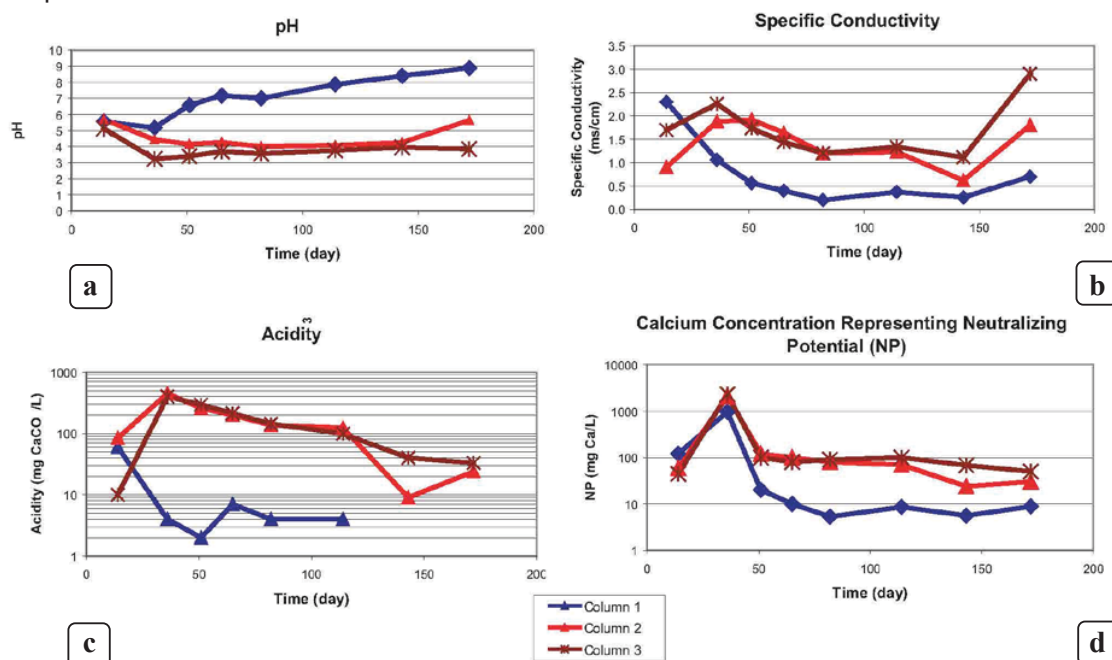


Fig. 6. Chemistry of drainage from the three columns. Column 1 with IGM. Columns 2 and 3 are non-IGM with and without sand cover respectively.

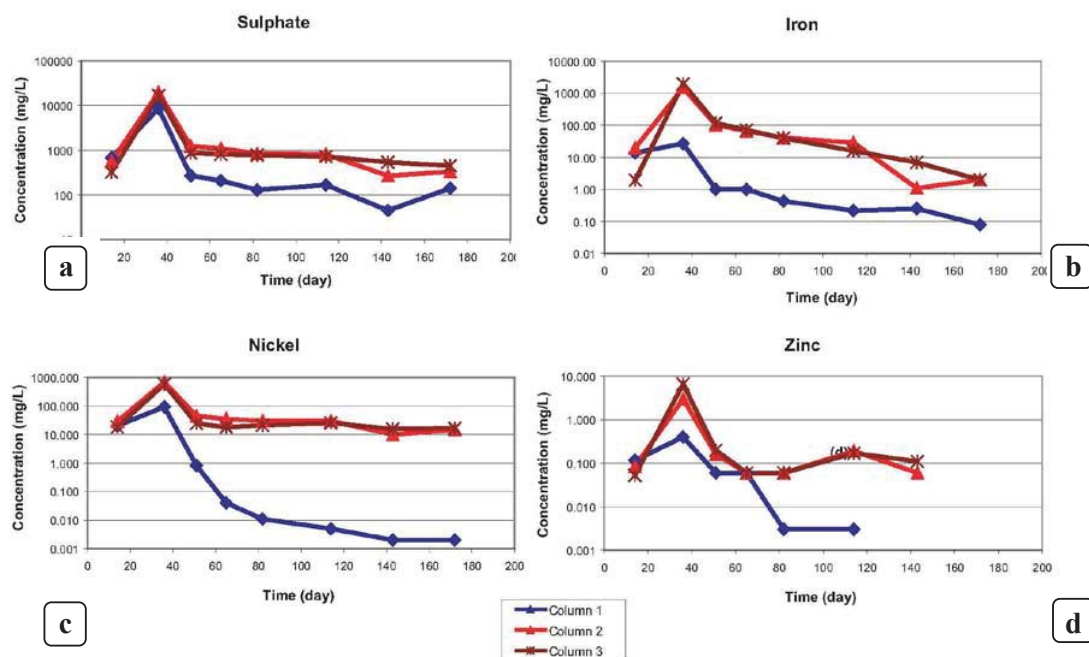


Fig. 7. Elemental concentration in the drainage from the three columns. Column 1 with IGM. Columns 2 and 3 are non-IGM with and without sand cover respectively.

Figure 6d presents the calcium concentrations to indicate the potential consumption of Neutralizing Potential (NP) due to acid generation. More acidic conditions result in greater consumption of NP and a release of higher calcium concentrations. Column 1 drainage has five to ten times lower calcium concentrations than in the other columns that indicate lower consumption of NP because less acid was produced in column 1.

4.2.2 Metal concentrations

The drainage samples from the columns were analyzed for 33 elements. The results of four environmentally important constituents are discussed here and include sulphate, iron, nickel, and zinc.

Figure 7 shows that the concentrations of the constituents in the IGM drainage were two to three orders of magnitude less than the concentrations in the drainage from the two control columns (columns 2 and 3). The sulphate and metal concentrations from the simple covered and uncovered waste rock were similar, showing that the low efficiency, coarse quartz sand cover used in this investigation performed as expected and did not reduce oxidation substantially below levels observed in the uncovered column.

Iron oxide precipitates were observed to form on the inside walls of test columns 2 and 3. Figure 8 shows the occurrence of these precipitates in columns 2 and 3 and that the iron oxide by-products did not form in column 1 in the same time frame. The iron is generally released during sulphide mineral oxidation as ferrous iron (Fe^{2+}) and then can be subsequently oxidized to ferric iron (Fe^{3+}) that will hydrolyze and precipitate as a reddish iron stain that is typically referred to as ferric hydroxide. The visible ferric hydroxide in columns 2 and 3 indicates that substantial oxidation has occurred in those columns.



Fig. 8. The reddish oxidation product concentration in columns 2 and 3 (right) and no precipitation in column 1 (with IGM - left) in the same time frame.

5.0 DISCUSSIONS

5.1. Water Quality - pH and acidity

During sulphide oxidation, pH of the drainage decreases, and sulphate and metal concentrations increase. Therefore, the geochemical characteristics of the drainages provide excellent indications of the extent of sulphide oxidation and a measure of the effectiveness of the IGM treatment.

The acid generation and the neutralization indicators of column drainages are presented in Figure 6. As Figure 6a shows, all columns had initial pore water pH values of about 5.5. There was an initial decrease of the pH in the IGM column drainage. The marginally acidic pH occurred initially as a result of moisture on the crushed waste rock particle surfaces where some oxidation occurred during the column set-up period that extended about two weeks prior to flushing oxygen from the IGM column.

After the initial flushing period of about 30 days, the pH in column 1 increased to about 8.9, while the drainage pH from the two control columns decreased to more acidic values. The pH results show that the IGM technique has prevented oxidation and acidification, while the coarse sand cover had only a slight effect on acidification.

The Specific Conductivity (SC) of the drainage is shown in Figure 6b. Columns 2 and 3 exhibited higher SC values than that in column 1. As the pH decrease, the SC increases for the two columns, indicating that the drainage from the non-IGM columns have higher concentrations of metals and other dissolved solids.

The acidity concentrations for each of the columns are shown in Figure 6c. The acidity from column 1 exhibited values less than 5 mg CaCO_3/L after 40 days, while the acidity values released from columns 2 and 3 were 20 times higher, on the order of 100 mg CaCO_3/L . There was no apparent advantage to covering of waste rock with the low-efficiency coarse sand, based on acidity.

Calcium concentrations are an indirect indicator of acid generation. Calcium is released during neutralizing reactions with acid generated during oxidation. Figure 6d shows that the calcium concentrations in the drainage from column 1 were about one order of magnitude less than those from the two control columns. The slightly higher pH in column 2 than in column 3 resulted in slightly lower calcium concentrations.

The above results demonstrate the efficiency of the IGM treatment for preventing sulphide oxidation and acid generation in column 1. Comparison of the water quality from columns 2 and 3 suggests that the poor water retention in the coarse sandy cover layer results in no substantial differences between the two columns.

5.2. Water Quality - Metal release

Figures 7 a to d show the concentrations of sulphate and heavy metals such as iron, nickel and zinc. Sulphate is a primary product of sulphide oxidation. The three heavy metals are environmentally important and are also important indicators of acid mine drainage.

All profiles corresponding to columns 2 and 3 follow very similar patterns and magnitudes, indicating there are no important differences between the covered and uncovered waste rock. In contrast, the concentrations from column 1 are 10 to 1000 less than those from columns 2 and 3.

The concentrations of sulphate (Figure 7a) in column 1 are near 100 mg/L, suggesting that some oxidation had occurred, likely during the column set-up period that lasted two weeks. As suggested by the nickel and zinc trends (Figure 7c & d), sulfate may not have been washed out by day 165 and further declines in sulphate concentrations in drainage from the IGM column may have been expected. Iron concentrations (Figure 7b) were 100 times less in the drainage from column 1 than in drainages from columns 2 and 3 throughout the test. In addition, iron concentrations appear to be on a declining trend at the end of the test period.

The waste rock in the columns originated from a nickel mine. It was expected that even if only minor oxidation had occurred, there would have been elevated nickel concentrations. However, as Figure 7c shows, the nickel content from column 1 drainage decreased monotonically to values that would be considered acceptable in aquatic environments after wash-out stage (day 35). The zinc concentrations (Figure 7d) shows similar trends and decreased to values below the detection limit after 4 months of operation. These trends are a clear indication that the oxidation products that were initially present on the surfaces of the rock at the start of the IGM test were washed out during the test while the oxidation reactions had ceased producing additional products.

5.3. Elemental Loading

Figure 9 was created to compare the sulphate and metal loading rates from each of the columns. The relative differences in loadings are expected to be similar to the relative differences in concentrations because similar volumes of water were added to the columns weekly. Nonetheless, loadings provide a better comparison for the oxidation rates in each column. The vertical axis in Figure 9 is the loading rate in g/kg-Waste Rock/week.

After six months of operation, the loading rates reached a quasi steady state under all the conditions. The loading rates of sulphate and iron from column 1 are about 1 order of magnitude less than rates from columns 2 and 3. The nickel loading rates from column 1 were on the order of 10^{-5} g/kg/wk after three months while those from columns 2 and 3 stabilized at about 0.2 g/kg/week. The relative differences in loading rates agree well with the differences in concentrations as expected.

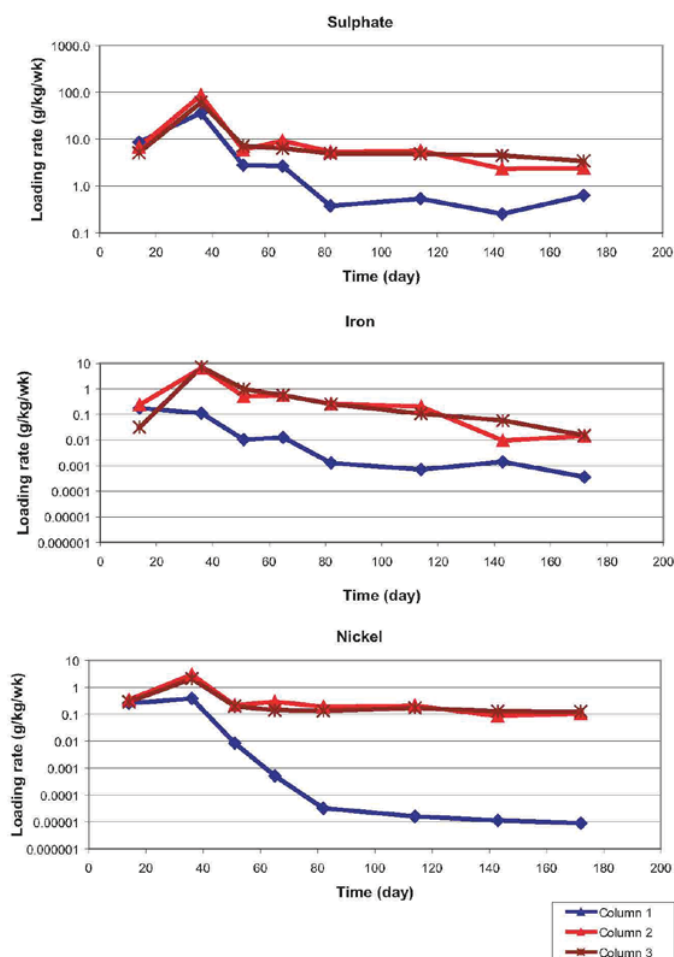


Fig. 9. Sulphate, Iron and Nickel weekly loading.

6.0 CONCLUSIONS

The Inert Gas Mixture (IGM) technology has been designed to overcome the deficiencies of gas transfer into tailings and waste rock materials across the surface and instead block oxidation (*i.e.* creation of Acid Rock Drainage - ARD) by controlling the transfer of oxygen from the atmosphere. Inert gas is injected into the mine waste at low flow rates and the positive but low flow purges the oxygen that tends to diffuse into the sulphide waste across the surface. The positive flow of IGM compensates for inward oxygen diffusion and creates a pressure drop across the cover layer that prevents inward diffusion of oxygen and hence acid generation.

The modelling results to predict the performance of the IGM technique show that, for a simple cover layer that was not efficient at blocking oxygen ingress from the atmosphere, the IGM velocity was on the order of 2×10^{-7} m/s to effectively prevent oxygen ingress. This means that a covered pile with an area of 10 hectares would require a total flow rate of about 22 m³/min (13 cfm), a value that is practically achievable with existing compressors. These results suggest that the IGM technology is theoretically and technically feasible.

Other model calculations show that imperfections or high permeability zones in the cover do not represent an insurmountable challenge. Depending upon site-specific conditions, the IGM could be applied at low levels on a continuous basis, or may be applied intermittently.

The IGM technology is also controllable in that the inert gas mixture injection rate can be monitored and adjusted for changes in cover characteristics or performance (*e.g.* moisture content, snow cover, cover deterioration requiring remedial work). The monitoring of cover performance by way of the IGM technology's monitoring system would also provide mine site caretakers with a practical and effective means of monitoring the performance of the dry covers overlying the wastes.

The bench scaled laboratory experiment clearly showed that the IGM technology prevents oxidation of sulphide minerals in waste rock, and effectively mitigates drainage water quality. The use of the same cover over waste rock in the absence of IGM resulted in similar acidic poor quality drainage that was formed from the interaction with waste rock that had no cover.

The monitoring of other variables (ambient pressure, differential pressure, gas flow rate, temperature, moisture level) of the IGM column evolution during the bench scaled laboratory experiment further demonstrated that the key variables that are involved in gas diffusion and flow can be readily monitored, and that selected variables can be used as feedback to set the IGM flow rate. Those results were not detailed in the present paper. Their assessment however determined that the IGM technique and gas flow requirements are in the realm of practical and achievable values for a field scale application.

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DOMAINING ACID ROCK DRAINAGE RISKS USING GEOMETALLURGICAL DATA

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ABSTRACT

Best practice for acid rock drainage (ARD) risk assessment predominately relies on the geochemical properties of sulphidic rocks. Consequently, there are a plethora of geochemical tests routinely utilised by the mining industry to predict the likelihood of ARD formation. However, due to limitations associated with these tests and their relatively high costs, analysis of recommended best practice sample numbers is rarely achieved, thus reducing the accuracy of waste management plans. This research aimed to address this through identifying potential geometallurgy indicators.

Drill core samples (n=70) obtained from Copper Mines of Tasmania, Australia were subjected to a range of mineralogical analyses, routine ARD geochemical tests (e.g., paste pH; acid base accounting, ABA; net acid generation, NAG), field-based techniques (e.g., portable X-ray fluorescence, pXRF; short-wave infrared spectrometry, SWIR), and geometallurgical analyses (e.g., HyLogger, EQUOtip). Geochemical and mineralogical results were compared against geometallurgical and field-instrument datasets to determine: i) how relative carbonate content measurements (generated by HyLogger) can be used to calculate effective acid neutralising capacity (ANC); ii) if mineral hardness (measured by EQUOtip) can be used to classify lag-time to acid formation; and iii) whether field-portable techniques can be used to domain ARD forming potential.

Several clear relationships between geometallurgical and routine ARD data were identified, thus demonstrating the applications of Hylogger, EQUOtip and pXRF for effective ARD domaining. For example, Hylogger data allows identification of acid-neutralising carbonate minerals. EQUOtip hardness data provide a conservative indication of lag-time to acid formation. pXRF data give insights into the abundance of environmentally significant elements. Consequently, the application of geometallurgical techniques to drill core allows the domaining of ARD risks. Such knowledge permits the development of best practice waste characterisation protocols and waste management plans.

1.0 INTRODUCTION

Undertaking effective environmental ore characterisation at pre-feasibility/feasibility stages is essential for both efficient mine operations and reducing post-closure impacts such as acid rock drainage (ARD). Determining the propensity of a rock unit to produce ARD is possible through the use of established and emerging tools and techniques. This was described by Parbhakar-Fox et al. (2011), where the geochemistry-mineralogy-texture (GMT) approach was proposed. This three-staged protocol allows for low-cost pre-screening techniques to be

undertaken at stage-one before performing established geochemical screening tests (e.g., acid base accounting, NAG tests). The GMT approach is considered an improvement on existing predictive methodologies such as the wheel approach (Morin and Hutt, 1998) and its derivatives, as it optimises applications of existing tests (e.g., paste pH) and field-based technologies (e.g., portable XRF) to aid pre-screening. It also allows for identification of sulphide oxidation controls at stage-three through using stage-of-the-art tools (e.g., laser ablation ICPMS; mineral liberation analysis). Perhaps the most significant advantage of the GMT approach is that through stage-one analyses, it is financially possible to use best practice sample numbers (cf. Price, 2009), for deposit-wide ore and waste characterisation.

Despite the merits of the GMT approach and the use of such simple pre-screening tests for deposit-wide domaining, it can be argued that the undertaking of such specialised ARD focused analyses (e.g., geochemical tests) is financially limiting as these data are fit-for-purpose, and cannot be used to characterise other features of the ore body. Instead, to facilitate deposit-wide characterisation and to add value to already existing datasets, proxies for ARD data must be identified. Geometallurgical tests and data are the most appropriate proxies. Whilst a vast range of data is collected for geometallurgical modelling, no published examples exist of it being utilised for predictive ARD characterisation. Yet, sampling strategies utilised as part of geometallurgical campaigns i.e., 2 m sampling (e.g., Alruiz et al., 2009; Leichter et al., 2011) represent an appropriate sampling interval for deposit-scale ARD domaining.

This study aimed to determine whether geometallurgical data can be utilised effectively to domain acid neutralising capacity and acid forming potential in drill core materials. Geometallurgy data and its environmental implications examined in detail were: i) hyperspectral infrared data to assess the accuracy of acid neutralising capacity (ANC) data, and ii) mineral hardness to determine the weathering rate of rock units. In addition to geometallurgical data, information derived from field-based instrumentation (i.e., SW-IR, pXRF) is also presented and evaluated in terms of application for accurate deposit-wide ARD domaining.

2.0 MATERIALS AND METHODS

2.1 Study Site

Drill core samples from Copper Mines Tasmania, specifically the Copper Chert deposit in the Comstock Valley (located on the northern end of the Mount Lyell mineral field) were used in this study. The Copper Chert deposit is part of the large Cambrian hybrid volcanogenic-magmatic Cu-Au-Ag system at Mount Lyell, which has been exploited for approximately 120 years from 22 separate mine workings. The Mount Lyell mineral field occupies the southern end of a belt of polymetallic base and precious metal deposits within the late Middle Cambrian Mount Read Volcanics (MRV) succession of western Tasmania. The Copper Chert deposit is considered to represent new style of volcanic hosted, copper rich polymetallic mineralisation for the Mount Lyell mineral field (McLoughlin and Morrison, 2013). The mineralisation is largely Cu-Au-Ag but also contains significant intersections of Pb-Zn-Ag. The deposit is hosted in an intense microcrystalline silica alteration package known locally as the Comstock Chert. It also has a stratigraphic control within the lower Tyndall Group and across the underlying contact with the Central Volcanic Complex, within the MRV (McLoughlin and Morrison, 2013).

2.2 Sample Selection

A 70 m interval of drill core (hole ID: CCD007; 304 m to 375 m) from the Copper Chert (CC) discovery in the Comstock Valley was selected for this study. This particular intersect of core was chosen as it represented both carbonate-rich and sulphide-rich material, thus allowing for an in-depth evaluation of the selected geometallurgical and field based techniques. The upper portion of the section (304 m to 345 m) represented the Tyndall Group Lynchford member. This comprises of sedimentary sandstones, limestones and conglomerates, and demonstrated a variety of textures (e.g., fragmental, clastic, veined). The lower portion (345 m to 375 m) sampled the Cambrian Lyell Schist, which demonstrated increasing degrees of mineralisation with depth (i.e., greater proportions of chalcopyrite and pyrite towards the base of the drill hole). Samples were first subjected to geometallurgical analyses, followed by the selection of representative samples every 1 m for geochemical and mineralogical analyses as per stage-one of the GMT approach.

2.3 Geometallurgical Analyses

2.3.1 HyLogger

The Australia-developed HyLogging™ systems are automated platforms to rapidly and systematically collect infrared spectroscopic reflectance data at dense sample spacing from drill core, chips or powders (Huntington et al., 2006; Quigley, 2012). This technique is responsive to the chemical composition and crystal structure of a mineral. Thus, mineral identification is made possible based on the resulting 'spectral fingerprint' or 'signature' observed in the spectral response curve (Huntington et al., 2006; Quigley, 2012). HyLogging has several advantages over other mineral identification techniques (e.g., XRD and automated SEM instruments including MLA and QEMSCAN) through its rapidity (up to 1000 m of core per day), its low cost per sample, and its non-contact, non-destructive approach (Huntington et al., 2006). Based on this, HyLogging was identified as having potential application in the GMT approach.

Forty metres of material (304 m to 345 m: Tyndall Group Lynchford member) were analysed at Mineral Resources Tasmania (MRT). More than 5000 hyperspectral reflectance measurements were collected, with the data acquisition rate c.5 minutes for a three section core tray. Hyperspectral data analysis and mineral interpretation were carried out in version 8.1 of The Spectral Geologist (HotCore). Hylogging- derived mineralogy was intended for comparison with: i) SW-IR data; ii) XRD data; and iii) domaining of acid neutralising capacity using carbonate relative intensity values and total sulphur data.

2.3.2 EQUOtip

Measuring the susceptibility of a lithology to weathering is required to understand how rapidly acid forming minerals will become exposed in a waste rock pile. Therefore, the application of EQUOtip a non-destructive, core-based measurement technique was explored (Keeney, 2008). The Leeb (Ls) hardness value (0 to 1000) is automatically calculated, and is the quotient of the rebound velocity over the impelled velocity multiplied by 1000 (Keeney, 2008). The impact body rebounds faster from harder test samples than it does from softer ones, resulting in a greater value.

Approximately 3000 EQUOtip measurements were collected using an EQUOtip 3 (Proceq) instrument, across the whole 70 m interval. Average values were calculated for each 1 m interval based on measurements taken at 2.5 cm intervals. These values were used alongside total-sulphur values for comparison with NAG pH vs. paste pH data. This has the

potential to allow for a low-cost first-pass understanding of lag-time to ARD on a deposit-scale.

2.4 Geochemical and Mineralogical Analyses

Rapid and accurate measurement of sulphur (wt.%) for comparison against EQUOtip and HyLogger values and calculation of maximum potential acidity (MPA) was performed on all samples using an Eltra C-S 2000 instrument at the University of Tasmania (UTas). The ASTM D4972-01(2007) paste pH method was used following recommendations given in Noble et al. (2013). The pH value of each tested sample (n= 55) was measured in triplicate, with the standard deviation calculated as <0.5.

All samples from the Tyndall Group Lynchford member material (304 m to 345 m) were sent to ALS Brisbane for the calculation of acid neutralising capacity (ANC) by the Sobek method. The multi-addition net acid generation (mNAG) test was performed at UTas laboratories on all samples from the Cambrian Lyell Schist (345 m to 375 m) following the AMIRA P387A method (Smart et al., 2002).

Mineralogical determination of material from each 1 m interval was performed using a benchtop Bruker D2 Phaser X-ray diffractometer at UTas. Samples were analysed for 1 hour (fixed divergence slit: 1 mm; range: 4-900 2theta; 0.020 step size, Fe-filter), with the resulting spectra proceed in Eva 2.1 software, where minerals were identified using the ICDD PDF 2012 database.

2.5 Field-Based Methods

The application of short-wave infrared (SW-IR) spectroscopy in determining pale, fine-grained alteration minerals has been demonstrated in ore-deposit characterisation studies (e.g., Thompson et al., 1999; White et al., 2010). Mineral identification is based on absorption spectra collected from clean, dry, flat, rock surfaces (Gifkins et al., 2005). SW-IR therefore has potential to improve mineral identification of altered drill core samples for ARD characterisation. The TerraSpec spectroradiometer (manufactured by analytical spectral device (ASD) Inc.) with RS3 software (version 4.0.23) was selected for use, with both intact pieces and powders from each 1 m interval analysed.

Accurately measuring chemistry and comparing these data to ARD parameters (e.g., paste pH, total-sulphur) allows for a first-pass indication of potential leachate quality issues which may arise towards the life-of-mine end (i.e., mine closure). This in turn allows the development of appropriate waste management plans. Field-portable XRF (pXRF) instruments have in recent years been used in mine site characterisation studies for determining element concentrations (e.g., Melquiades and Appoloni, 2004; Haffert and Craw, 2010; Higuera et al., 2012). Most recently, Parbhakar-Fox et al. (2013a) discussed its application in mesotextural classification at abandoned mine-sites. In this study, a hand-held Olympus-InnovX instrument was used (UTas laboratories) on powders from each 1 m interval, with reference standards NIST 2781, GXR3-538 and GXR4-2843 used throughout the analyses.

3.0 RESULTS AND DISCUSSION

3.1 Acid-Base Accounting

Acid neutralising capacity (ANC) measurements collected for the Tyndall Group Lynchford member (304 m to 345 m) are shown in Figure 1a. ANC values ranged from 40 kg H₂SO₄/t to

954 kg H₂SO₄/t, correlating with the presence of sandy limestone (e.g., 339 m to 343 m). Paste pH values for select samples of the Tyndall Group Lynchford member (n=21) indicated that, currently, all tested samples are non-acid forming (Figure 1b; range: pH 7.6 to 8.2) when using a cut-off value of pH 5.5 to classify acid forming from non-acid forming materials (Parbhakar-Fox et al., 2011). Downhole total-sulphur values (Figure 1c) identified four distinct zones of relatively high sulphur with potential for acid formation when using a cut-off value of 0.3 wt. % sulphur (Parbhakar-Fox et al., 2011). This indicated that either sulphur is present in sulphide species which are slow to react in water, or lesser acid-forming sulphides dominate in these samples (e.g., galena and sphalerite). Net acid producing potential (NAPP) values classified 15% of samples as potentially acid forming (PAF > 20 kg H₂SO₄/t), with a distinct PAF zone identified from 330 m to 335 m. This was logged as green, chloritic, polymict conglomerate, with fragments of chert, volcanics, limestone and sulphides. These geochemical results confirm that there is sufficient carbonate material in the Tyndall Group Lynchford member to validate applications of geometallurgical data for ANC domaining.

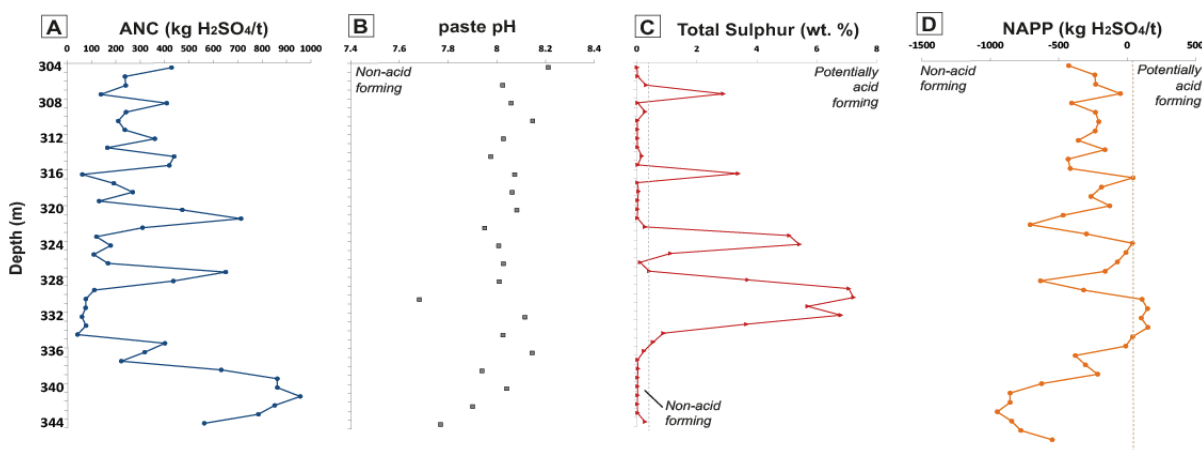


Fig. 1. Downhole static geochemical results from the Tyndall Group Lynchford member samples (n=40): A) Acid neutralising capacity (ANC) values (kg H₂SO₄/t); B) Paste pH values (n=21) with a paste pH cut-off criterion of pH 5.5 used to classify acid forming from non-acid forming materials; C) Total sulphur values (wt.%) with 0.3 wt. % used as the classification cut-off criterion (Parbhakar-Fox et al., 2011); D) Net acid producing potential (NAPP) values (kg H₂SO₄/t) with a cut-off value of 20 kg H₂SO₄/t (Parbhakar-Fox et al., 2011) used to classify non-acid forming from potentially acid forming samples.

3.2 Geometallurgical ARD Domaining

3.2.1 HyLogger mineralogy

Mineralogical data collected by HyLogger for the Tyndall Group Lynchford member (304 m to 345 m) is presented only. The dominant mineralogy was quartz, chlorite, muscovite and carbonate. X-ray diffractometry data collected down hole verified the presence of these minerals, with their relative abundance shown in Figure 2a. Two distinct zones of high carbonate were identified, and correlated directly with two zones of high ANC (Figure 2b).

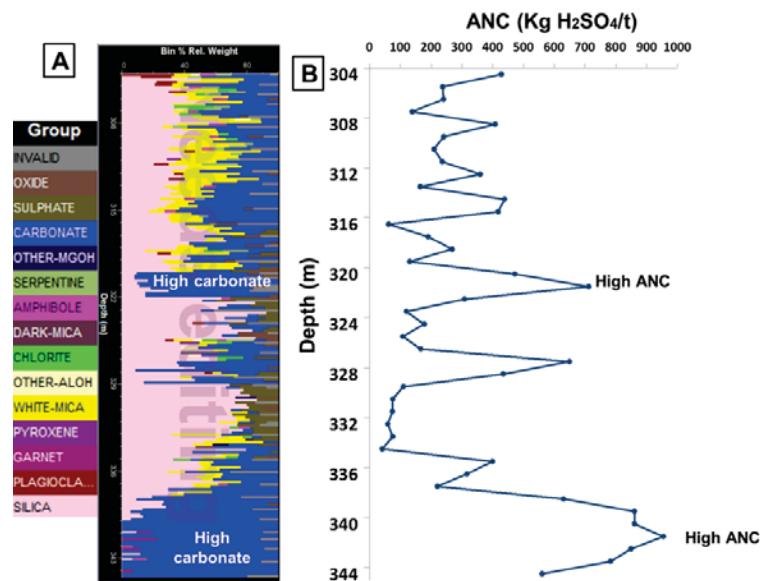


Fig. 2. A) Mineralogical identification performed by HyLogger of drill core material from CCD007 (304 m to 345 m; Tyndall Group Lynchford member); B) Acid neutralising capacity (ANC) values, with zones of high ANC correlating to high carbonate zones as identified by HyLogger.

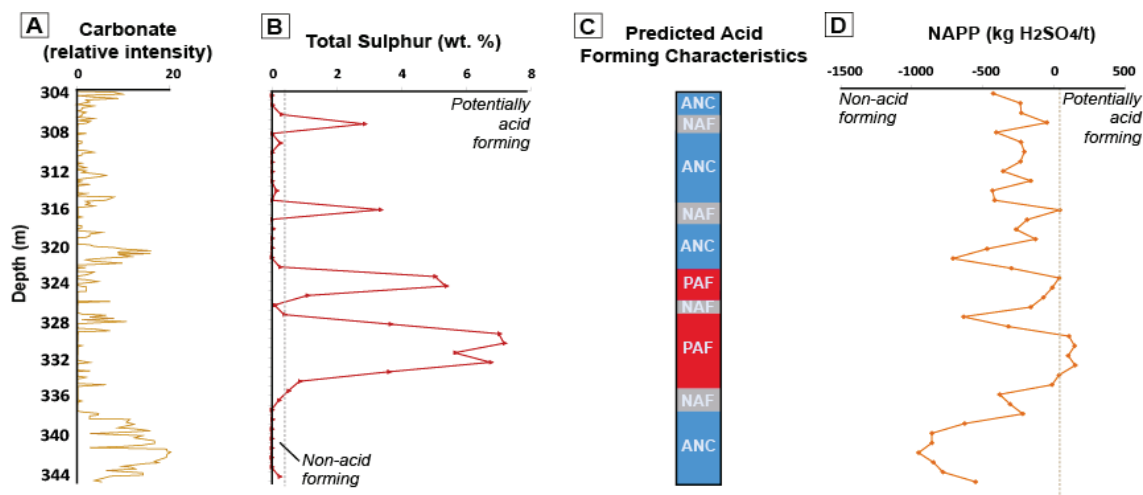


Fig. 3. Downhole static geochemical results from the Tyndall Group Lynchford Member samples: A) Carbonate relative intensity measured by HyLogger; B) Total sulphur values (wt.%; n=40) with 0.3 wt. % used as the classification cut-off criterion (Parbhakar-Fox et al., 2011); C) Predicted acid forming characteristics based on HyLogger relative carbonate intensity data and total-sulphur values; D) Net acid producing potential (NAPP) values (kg H₂SO₄/t; n=40) with a cut-off value of 20 kg H₂SO₄/t (Parbhakar-Fox et al., 2011) used to classify non-acid forming

from potentially acid forming materials. Abbreviations: ANC, acid neutralising capacity; NAF, non-acid forming; PAF, potentially acid forming.

3.3 NAG pH vs. Paste pH

The use of NAG pH versus paste pH values was proposed by Price et al. (1997). However, its application has not been widely demonstrated, despite the fact that it offers an assessment of lag-time to ARD and thus classifies risk. The use of this classification is recommended by the GMT approach (Parbhakar-Fox et al., 2011) and therefore was performed on Cambrian Lyell Schist drill core material (345 m to 375 m; $n = 30$) as shown in Figure 4. All samples but one were identified as either acid-forming or potentially acid forming, and classified as medium-low risk, with the rate of ARD formation determined as rapid, to having a short-lag time (in the case of potentially acid forming samples). This confirmed the suitability of Cambrian Lyell Schist drill core material for evaluating the application of EQUOTip for classifying ARD risk.

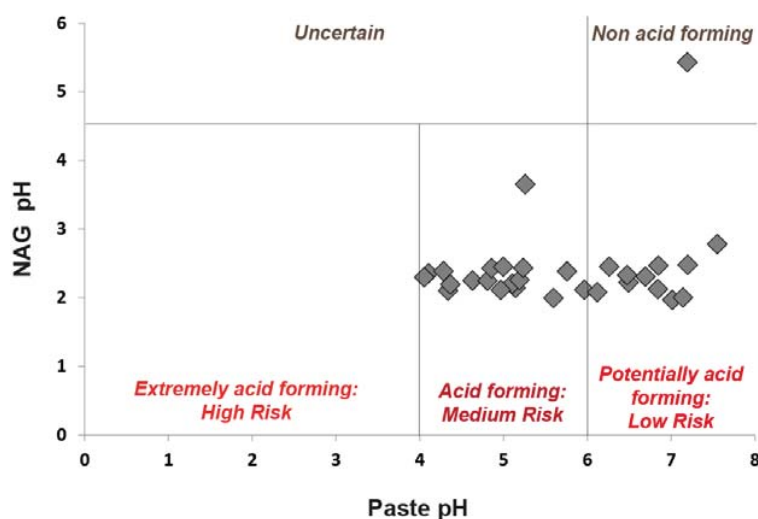


Fig 4. ARD risk assessment screening plot of NAG pH versus paste pH for Cambrian Lyell Schist drill core materials (CCD007, 345 m to 375 m). Modified from Price et al. (1997).

3.4 ARD Risk Assessment: Mineral Hardness

Mineral hardness data measured by EQUOTip for all drill core (304 m to 375 m) was collected and is therefore presented. Detailed interpretations of ARD risk are only given for the Cambrian Lyell Schist. However, consideration of mineral hardness allowed for further assessment of effective ANC for the Tyndall Group Lynchford member samples. Comparison of EQUOTip data against total sulphur and paste pH values are shown in Figure 5. If a mineral hardness value was <648 Ls, it was classified as soft in accordance with Keeney (2008). Thus, samples identified as NAF, by both paste pH and total-sulphur analyses and also classified as soft (e.g., 309 m to 315 m; Figure 5d), are taken to represent the most effective neutralisers. The Cambrian Lyell Schist was classified in a similar manner, with samples identified as PAF by paste pH and total sulphur methods and with soft mineral

hardness (i.e., greater likelihood of acid formation) classified here as acid forming at a relatively rapid rate. These findings are in agreement with data shown in Figure 4, therefore indicating the potential application of EQUOtip when performing ARD domaining. Furthermore, in the absence of paste pH data, mineral hardness and total sulphur data would have been sufficient to conservatively classify the behaviour of these materials.

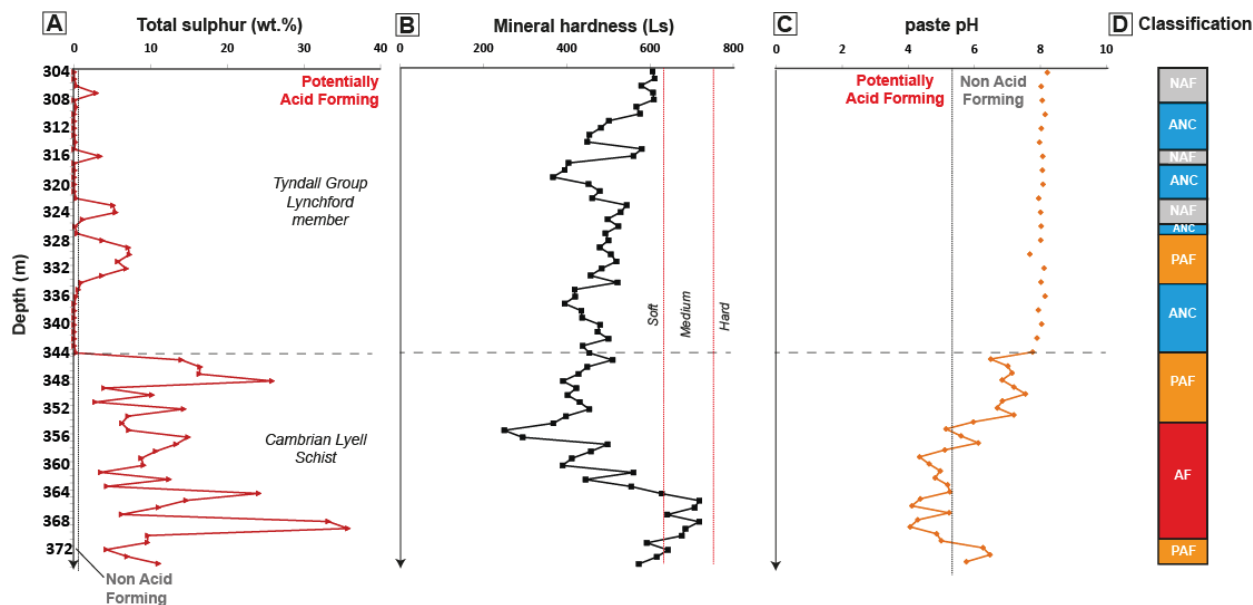


Fig 5. Downhole geochemical and geometallurgical data from drill hole CCD007 (304 m to 375 m sampling both the Tyndall Group Lynchford member and the Cambrian Lyell Schist, with the change of lithology indicated by the horizontal dashed line at 345 m: A) Total sulphur values (wt. %; n=70) with 0.3 wt. % used as the classification cut-off criterion (Parbhakar-Fox et al., 2011); B) Mineral hardness values as measured by EQUOtip and reported in Leeb, with criteria for defining hard, medium and soft samples shown (Keeney, 2008); C) Paste pH values (n=55) with pH 5.5 used as the classification cut-off criterion (Parbhakar-Fox et al., 2011); D) ARD classifications based on total sulphur, mineral hardness and paste pH values. Abbreviations: AF, acid forming, high risk with rapid ARD formation, ANC, acid neutralising capacity, highly effective; NAF, non-acid forming; PAF, potentially acid forming, medium risk with lag-time to ARD formation.

3.5 Mineralogical Determination: SW-IR

The application of a TerraSpec (SW-IR) instrument for mineral identification was tested and its data compared to that of XRD analyses. The TerraSpec was able to confidently identify the presence of calcite, muscovite and chlorite, but occasionally misidentified minerals such as tourmaline and epidote, which were neither logged, nor detected by XRD. The TerraSpec instrument correctly identified the presence of carbonate minerals in 52% of these samples. As these analyses were performed on intact rock samples (three areas analysed, and the mineralogy deduced from the combined results), there is a high probability of conflicts when compared to XRD data. However, if powdered and homogenised samples had been subjected to TerraSpec analyses instead, then much greater agreement with XRD data is anticipated. Based on these results, it is clear there is application for TerraSpec for ANC

domaining, with the best possible results achieved, but only if homogenised powdered samples are used as opposed to analysing intact drill core materials.

3.6 Chemical Domaining: pXRF

Chemical measurements systematically measured downhole for As, Cu, Pb and Zn are presented in Figure 6. In general, concentrations of these elements were greater in the Cambrian Lyell Schist material as anticipated given the increased presence of chalcopyrite and pyrite in this mineralised lithology. Concentrations of As and Cu were low in the Tyndall Group Lynchford member (Figures 6a and 6b). However, localised hotspots of Pb and Zn were identified in the Tyndall Group Lynchford member, e.g., 322 m to 325 m depth (Figures 6c and 6d), which correlated with higher total-sulphur (wt. %) values (Figure 6e). Minor sulphide intercalations were identified by XRD, and are the likely Pb and Zn source. If this lithological unit was uniformly classified in the ARD block model based on a measurement taken at another part of the Tyndall Group Lynchford member, then Pb and Zn variability would likely be masked, with the unit classified as NAF by paste pH testing (Figure 6f). However, elution of Pb and Zn would eventually be likely as processes of ARD evolve, and therefore a management strategy should be developed at this stage of the mining process with this in mind, should materials from this lithology be designated waste.

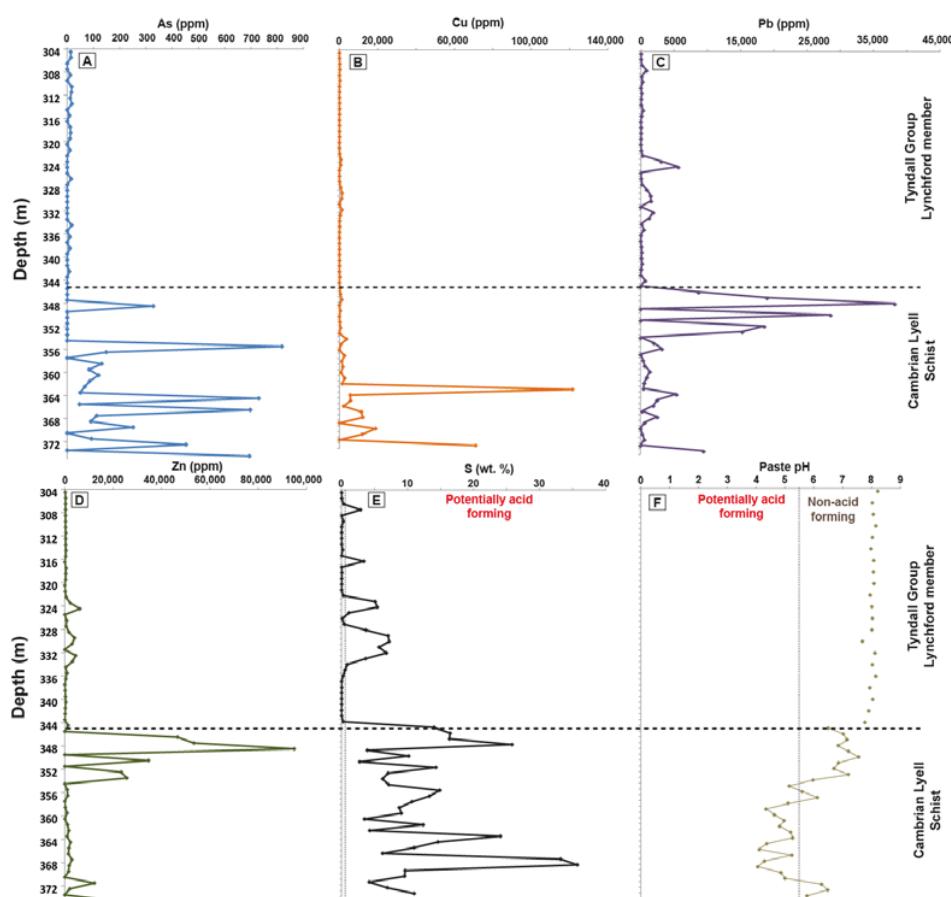


Fig 6. Downhole geochemical data (A to D measured by pXRF; E by CS analyser) from drill hole CCD007 (304 m to 375 m sampling)

both the Tyndall Group Lynchford Member and the Cambrian Lyell Schist, with the change of lithology indicated by the horizontal dashed line at 345 m: A) As (ppm); B) Cu (ppm); C) Pb (ppm); D) Zn (ppm); E) S (wt. %); F) Paste pH values with pH 5.5 used as the classification cut-off criterion (Parbhakar-Fox et al., 2011).

4.0 CONCLUSIONS

The objective of this study was to determine whether geometallurgical data can be utilised effectively to domain acid neutralising capacity and acid forming potential in drill core materials. This study sought to evaluate the application of two geometallurgical techniques (HyLogger and EQUOtip) for domaining ARD characteristics, and to evaluate the benefits of using field-based instruments (TerraSpec and pXRF). These data were compared with routine ARD geochemical data to determine whether geometallurgical data can be used as proxies in ARD prediction, thus allowing for improved deposit-wide ARD characterisation. Site-specific outcomes and implications were as follows:

- Paste pH testing successfully identified materials with significant neutralising capacity, and inferred that calcite was the dominant carbonate mineral in the Tyndall Group Lynchford member drill core materials.
- Thermal infrared data from HyLogger accurately reported carbonate intensity. However, it was less accurate identifying exact carbonate minerals when compared against XRD data. If used in conjunction with total-sulphur values, accurate ANC domaining when compared against laboratory measured ANC values was achieved.
- The EQUOtip hardness/total sulphur geometallurgical classification is best used to provide a conservative indication only of lag-time to acid formation. This approach allowed for the classification of ARD risk for potentially acid forming materials like the Cambrian Lyell Schist.
- Utilisation of EQUOtip hardness/total sulphur method alongside paste pH data also allowed for further domaining of effective acid neutralising capacity.
- Short-wave infrared analysis using a TerraSpec instrument only allowed for correct identification of at least 50% of minerals present in these drill core materials. Therefore, its use should be restricted to use on powdered samples and not intact core.
- Accurate ARD risk domaining was achieved by using pXRF.
- Field tools such as portable XRF and Terraspec instruments need to be calibrated and should only be used as screening instruments.

This case study demonstrates that there are clear benefits of using geometallurgical data for ARD domaining. Firstly, such an approach untangles geological variability. In this study, only 70 m from one drill hole was investigated, yet diverse ARD characteristics were recognised. Secondly, if geometallurgical and sulphur assay data already exist, then ARD domaining can be performed at no extra analytical costs. Thirdly, best practice sampling for ARD prediction can be pursued based on geometallurgical and pXRF data. Fourthly, ARD domaining based on geometallurgical and pXRF data allows the identification of those samples that should be subjected to routine geochemical tests and more detailed ARD risk assessment.

This study has demonstrated that geometallurgical data can be utilised effectively to domain acid neutralising capacity and acid forming potential in drill core materials. This emerging

approach has real potential to enhance and improve the way in which ARD is predicted and modelled, with significant benefits to waste classification and management anticipated.

5.0 ACKNOWLEDGMENTS

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LOW COST ACID MINE DRAINAGE MANAGEMENT USING A MODIFIED BIOLOGICAL REDUCTION PROCESS PRODUCING WASTE HEAP ARMOURING COMPONENTS

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ABSTRACT

The management of acid mine drainage globally is limited by the cost of treatment. This cost also limits mining operations, mineral resource utilization, acid mine drainage treatment and ability of governments to manage legacy sites. A low cost active management system using a modified sequential aeration/oxidation and biological reduction process is proposed that requires a carbon source and will normally not require imported chemicals, not produce stored wastes, will produce a flexible water quality, and usually be cost negative. It reduces mine operating costs and conserves water resources by opening up the possibility of permitting the recycling of the treated acid mine drainage water for mining and agricultural uses. A non-scaling sodium carbonate is included in the treated water to neutralize acidity from dust suppression and wash water due to contact with the acid producing ground and coal. Further treatment may be required to permit its use for irrigation depending on the application and raw water quality. The system converts the acid mine drainage contaminants into the components suitable for lowering of sulphate concentrations and acidity in future groundwater from waste rock and coal heaps by armouring reactive surfaces and converts produced sulphide into recyclable elemental sulphur .

1.0 INTRODUCTION

The global mining industry is under increasing pressure to reduce the volume of acid mine drainage (AMD) on site and reduce the accumulation of waste products from AMD treatment. Simultaneously, they are not permitted to release AMD and may be required to pay into a fund for the treatment of the AMD flow legacy after the mine is closed. When this is combined with the high present capital and operating cost of treatment of AMD and the present low profit margins of mining globally, the result can be mine closures, unresolved pollution events and an avoidance of otherwise valuable mineral deposits where pyrite or sulphide rock occurs.

This has also created unsustainable economic pressure on coal mining as water used in mine operation that contacts some types of coal also becomes acidic. AMD is prevalent throughout Australia (Harries), which relies on mining income to a large extent to support it's economy. The principal mining operations in Australia are gold mining in sulphide deposits in Western Australia, iron ore mining in AMD generating sites in the Pilbara, and coal mining often in pyrite, the handling of coal itself creating AMD water. It is therefore essential to the long term health of the Australian economy that a solution is found to the AMD problem.

Due to SES's 60 year provision of water management solutions to the Australian mining industry, mining companies approached us to develop a total acid mine drainage water management system that would meet the regulators' requirements whilst permitting mining activity to remain economically and environmentally viable. These companies believe there are presently no practical or economic methods available to treat and manage AMD and it's

affects on mining operations. SES has designed the management system described in this paper and has carried out many laboratory based pilot plant trials on actual AMD water from two different mine sites over a period of 6 months. The average results of these trials are at table 1. At the time of writing we are building the first stages of two 6 ML/day plants for separate mine sites.

2.0 CURRENT STATE OF TECHNOLOGY

Prevention and minimization of AMD will always remain the preferred option. However, in many cases in mining, site logistics preclude such management and removal of the resulting AMD water is required. The main treatment methods adopted throughout the world are biological, filtration, chemical dosing and solar and fan evaporation. Unfortunately, all of these systems require the importation of additional chemical products onto a site already overburdened with chemical pollutants. In effect, the very process of treating the water is increasing the long term site pollution. In addition, the cost of the chemicals makes treatment of the AMD unviable in many cases.

3.0 PRESENT SITUATION IN AUSTRALIA

In Australia there are more than 317 mines (Harries) that are producing AMD, both operating and closed. The continuing production of AMD at these sites combined with rainfall often results in the unavoidable overflow entering healthy water catchments. This acid loaded with heavy metals has a serious affect on the quality of the contaminated water.

3.1 Evaporation

In Australia the rate of solar evaporation often cannot keep up with the AMD water production rate, resulting in the present situation where many mines are forced to flood viable mining pits to remain open. The evaporation of the AMD water storages increases the concentration of contaminants and acidity. Eventually the water quality becomes untreatable, the mine runs out of storage space, and closes. The acid mine drainage production continues after mine closure however.

3.2 Water Scarcity

In Australia, this situation is compounded by a scarcity of the high quality water generally required for mining operation. With mines often forced to import processing water at a high cost. In the north of Australia, rainfall occurs in high intensity events such as cyclones or monsoons, making utilization of most of this water impossible. The SES AMD management system permits the mine to empty it's AMD storages. This creates excess water storage space that can be utilized to retain more of this high intensity rainfall thereby reducing operating costs further.

3.3 Legacy of Closed Mines

There are many closed mines around Australia still producing AMD water (Harries). The AMD often results from waste rock and coal heaps. A proposed management technique for waste rock and coal heaps involves chemically armouring the waste rock piles using carbonates and the aluminium silicates of clay in close contact with the rock (Miller). This reduces sulphate concentrations and acidity in the groundwater and permits re-vegetation of these heaps (lee-Daniels). However, this process is very expensive due to the cost of the chemicals and therefore is rarely utilized. The AMD pollution situation therefore continues to persist.

3.4 AMD Management

To be effective, AMD management systems must not only separate the water from the contaminants, it must permanently remove both the water and the contaminants from the site without causing pollution elsewhere, permanently reduce the AMD production and limit its affect on mining operations and costs. Also it must also be affordable, practical and flexible to the constantly changing needs of mining operations. The mining companies believe the patent pending SES management system meets this standard.

4.0 ANAEROBIC BACTERIAL REDUCTION

The SES management system converts the contaminants into products that are useful to the mine and may be able to be recycled by society. Used passively, anaerobic bacteria treatment systems have been successfully used for many decades to treat AMD water. However, the systems eventually fail due to loss of hydraulic conductivity, exhaustion of carbon supply and bacteria death (C Mihaela-Neculita).

5.0 CONTAMINANT CONVERSION AND RECYCLING

To manage all of the above, the SES management system converts the AMD contaminants into metal oxides and carbonates (Figure 1). This is normally achieved without the importation of pH altering chemicals and with the use of imported carbon. The metals from the water are removed using a combination of oxidation and aeration with regulated dosing of recycled carbonates from the treatment process to maintain a constant pH in the anaerobic bioreactor. This process produces iron oxides (plus a lesser concentration of other metals) depending on the chemistry of the AMD water. The iron precipitates as ferrous iron in the aerator and is converted into ferric iron in the settling pond. Following this first stage sulphate reducing bacteria reduce the sulphate in a bioreactor. SES has developed a very low cost carbon source/electron donor which is fully utilized by the bacteria, ensuring no waste products from unutilized imported carbon are developed on site and permitting accurate provision of the nutrient to the bacteria. This process results in the production of carbonates (and high quality elemental sulphur).

This produces system outputs that meet the required quality for recycling by industry or used for chemical armouring of the waste pyrite/ coal heaps which long term testing has confirmed limits the production of sulphates and acidity in the future water runoff (S Miller; W lee Daniels). Hence, the management of acid mine drainage water is achieved without the storage of wastes onsite and importation of chemicals (other than the imported carbon). The coal mining industry has indicated the system actually operates at a negative cost when the cumulative positive impact of the water management system on site operations is assessed.

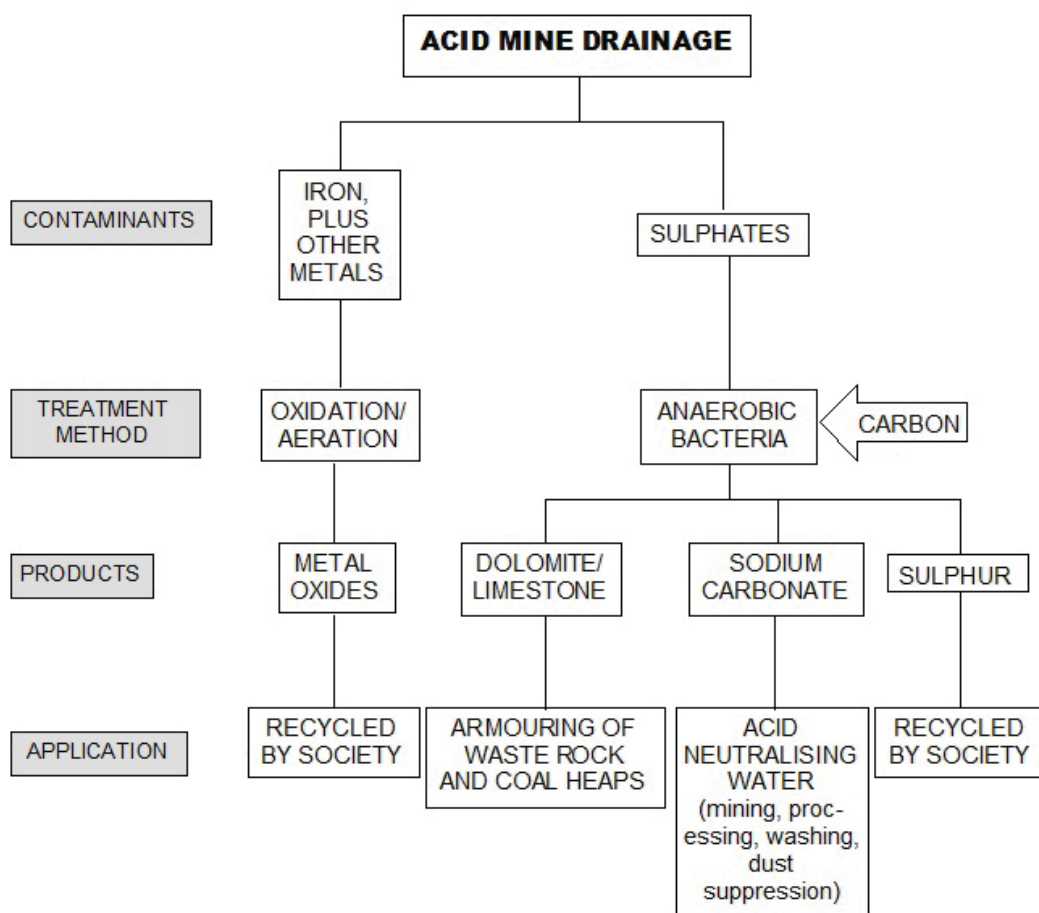


Fig. 1. AMD Management flow chart

Pilot plant trials have confirmed the provision of a specific and stoichiometrically balanced carbon source will result in these reduced sulphates forming carbonates at a low operating cost.

6.0 WATER RECYCLING

Described in table 1 are results designed to the desired water quality from the mine operator. Treated water quality can be tailored to suit each application.

Table 1. 1KI/hr pilot plant trial results on actual AMD water from a coal mine.

CHARACTERISTIC		RAW WATER	AERATION STAGE	BIOREACTOR STAGE	CARBONATE SETTLING STAGE
Electrical conductivity	uS/cm	7245	0	3260	1396
Total dissolved solids	mg/L	8780	0	3950	1692
Chlorides	mg/L	90	9	9	9
pH		2.3	7	7.5	8
alkalinity	mg/L	0	0	2364	200
Sulfate as SO ₄	mg/L	6500	4111	35	35
Carbonates	mg/L	0	0	2450	945
Calcium	mg/L	479	417	417	55
Magnesium	mg/L	474	441	441	50
Sodium	mg/L	623	580	580	580
Potassium	mg/L	20	19	19	19
Aluminium	mg/L	158	0.5	0	0
Cobalt	mg/L	1.64	1	0	0
Copper	mg/L	0.32	0	0	0
Manganese	mg/L	91.8	68	10	8
Nickel	mg/L	3	1.7	0	0
Zinc	mg/L	10.9	0.93	0	0
Lithium	mg/L	0.78	0.63	0	0
Iron	mg/L	794	3.5	0	0

Water is used extensively around mines for all types for dust suppression of mining equipment, road vehicles, conveyors and mine processing plants. In the case of coal mines, dust suppression water increases in acidity when it contacts coal dust.

To reduce operating costs to the mining companies we have designed the management system to produce sodium carbonate in the treated water with greatly reduced concentrations of calcium and magnesium. This non scaling water:

- Neutralises the acidity that results from coal contact, significantly reducing damage to plant, mining and processing equipment.
- Prevents scaling inside the pipes as it is a natural water softener
- Aids in the separation of oils and emulsions from the water
- Increases safety to personnel who work in the wet, acidic conditions



Fig. 2. AMD water before and after treatment by pilot plant.

As an example of the flexibility of the management system, one of the mines where we are installing a plant has a neighboring mine that is purchasing water, and is producing alkaline waste water. SES will adjust the acidity of a volume of the treated water to neutralize this alkalinity, thereby converting unusable waste water into a valuable water resource. As they will no longer need to purchase water, their operating cost will be significantly reduced, and a scarce water resource in Australia is conserved.

TABLE 2. TYPICAL WATER TREATMENT

CHARACTERISTIC		RAW AMD	TREATED WATER
pH		>2	7 to 8
Electrical Conductivity	us	18000	as required
Metals	mg/L each	>1000	0
Sulphates	mg/L	>18000	typically < 50

7.0 TREATMENT PLANT SIZE

As land space on mining sites is often limited, the small treatment plant size of the SES water management system is very useful for the mine or government treating a legacy site. The system is also easily scalable as the volume of acid mine drainage water to be treated changes with mining activity.



Fig. 3. A typical 6 megalitre/day treatment plant

Excluded from the picture is the aeration system, raw water pond, treated water tank and sludge screens. Included are the 17m dia x 3m high tanks of the three stage bioreactor and control room.

8.0 CLOSED MINES

The ability to convert AMD water into a high quality water resource at a very low cost, whilst not requiring imported pH altering chemicals, and producing armouring chemicals, is also valuable for the management of AMD legacy sites around the world. These managed sites will also be able to produce a valuable water resource from the AMD to be utilized by the surrounding stakeholders, and the produced armouring chemicals allow them to significantly reduce the production of future acidic water due to the reduced cost. An Australian state government is currently considering using the SES system to manage a very old AMD legacy site adjacent to a conservation region. Due to the low cost of the SES AMD management system, future AMD water production may be reduced and permanently prevent the pollution of water catchments with heavy metals.

9.0 EFFECT ON FUTURE MINING

There are many sulphide and pyrite mineral deposits that currently are not economically viable due to the resulting acid mine drainage water. The SES management system will permit the re-evaluation of those sites and hopefully result in increased revenue for the mining companies.

10.0 ACID SULPHATE SOIL DRAINAGE

The SES AMD management system can also be applied to the management of acid sulphate soils (ASS) which are prevalent in Australia (CSIRO). These soils create acidity in groundwater when excavated for building foundations. An ASS management plant will convert the groundwater into a condition safe for safe disposal or for use in concrete

production, dust suppression or compaction. The contaminants would be converted into recyclable resources for society.



Fig. 4. SES Self contained 5 KL/hr ASS water treatment bio-reactor for building sites. Separate aeration plant not shown.

11.0 CONCLUSION

SES is receiving a great many requests for assistance from governments and mining companies all around the world. At the time of writing, SES is currently preparing to install a 6 ML/day plant at a coal mine in Queensland and also in New South Wales. To date, we appear to be able to assist the projects we are working on. However, acid mine drainage water from every site has a unique water chemistry and site conditions. Hence it is suggested this is not necessarily the silver bullet needed for management of all AMD. Rather, SES prefers to thoroughly investigate the application of this technology to each site on a case by case basis to validate the management system against water chemistry, site management requirements and location.

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