

Early and Prolonged Limitation of AMD Production via Waste Landform Construction

Andrew Garvie¹, Stuart Miller², Pieter Fransen³, Trevor Matuschka⁴, Jeff Ruddock⁵ and Rory Haymont⁶

¹SRK Consulting (Australasia) Pty Ltd, Sydney, NSW, AUSTRALIA, agarvie@srk.com.au

²EGi Pty Ltd, NSW, Balmain, AUSTRALIA, egi@geochemistry.com.au

³Newmont Waihi Gold, Waihi, NEW ZEALAND, Pieter.Fransen@Newmont.com

⁴Engineering Geology Ltd, Albany, NEW ZEALAND, trevor.matuschka@enggeo.co.nz

⁵Newmont Waihi Gold, Waihi, NEW ZEALAND, Jeff.Ruddock@Newmont.com

⁶Newmont Asia Pacific, Subiaco, WA, AUSTRALIA, Rory.Haymont@Newmont.com

Abstract

Waste rock dumps are usually constructed by end dumping waste rock in lifts. End dumped waste rock often has characteristics that allow oxidation of sulfides and production of acid and metalliferous drainage (AMD) at levels that a) may negatively impact the environment, b) are unacceptable to regulators and c) have bond and relinquishment implications.

Sulfide oxidation commences immediately upon waste rock production and oxidation reaction products are stored in the dump until transported out by water infiltrating the dump. Implementation of strategies to manage AMD during waste rock dump construction often does not arrest oxidation. There is often an expectation that effective strategies can be implemented at the completion of waste deposition. However, at this stage, large masses of reaction products may be stored within the dump and the costs of some management strategies are by this time too large to be implemented.

At the Martha Mine in Waihi, New Zealand construction of an integrated mine waste rock and tailings storage facility recommenced in 1988. The embankment of the facility was designed to limit oxygen supply to waste rock soon after deposition. In this paper we present the design and quantitative evidence of the successful performance of the waste storage embankment.

Key Words: waste rock, sulfide oxidation, Martha Mine.

Introduction

Gold mining at Martha Mine produces potentially acid forming and non-acid forming waste rock. Between 1988 and 1999 waste rock was disposed of in an engineered embankment that is an integrated part of the tailings storage facility known as Storage 2. Objectives of the design and construction of Storage 2 included control of the rate of oxidation of sulfide bearing waste rock, acid neutralisation and control of water leaving the waste rock.

Martha Mine is located in the small rural township of Waihi in New Zealand. Waihi has a temperate climate with an average annual rainfall of 2,135 mm. Average monthly temperatures range between 10 and 19°C and the monthly rainfall varies between 120 and 240 mm throughout the year. Storage 2 is located approximately 3 km south-east of the Waihi township.

Waste Rock Characteristics

Rock types that comprise the waste include andesite, volcanic breccia, clays, tuff and ignimbrite. The oxide rocks (ignimbrite, clays and tuffs) were nearly entirely lacking in sulfide sulfur and were non-acid forming (NAF). The ignimbrite rock was hard but the largest proportion of NAF rock was weak clayey silt and sandy silt. This weaker material required drying before it could be compacted to achieve satisfactory strength in the embankment that forms the tailings storage facility.

The soluble sulfate in the waste rock prior to mining was low. For example, sulfate concentrations in leachate from kinetically tested oxide waste rock were mostly 3 to 5 mg/L.

The partially oxidised and unoxidised andesite and volcanic breccia contain variable amounts of sulfide sulfur (predominantly pyrite) with an expected range of 0.5 to 3 wt% S (Miller and Brodie 2000) and an average value of 1.5 wt% S (Brodie *et al.* 1996). The criteria used to classify NAF and PAF rock types at the Martha Mine have evolved through time. The pre-mining investigations were limited to acid base accounting, batch extraction and column leach test work on a limited number of samples. Based on these results, it was conservatively assumed that only fully oxidised waste rock was NAF and all transition and fresh waste rock was classified PAF. Through time, routine monitoring of waste rock expanded and included determination of the Net Acid Generation (NAG) test which was then used in conjunction with the net acid producing potential (NAPP) to classify waste rock types. The following criteria have been applied:

- NAF: NAPP negative and NAGpH ≥ 4.5
- PAF: NAPP positive and NAGpH < 4.5
- Uncertain (UC): conflicting NAPP and NAG results.

Material represented by UC samples were assumed to be PAF, pending additional test work.

Most of the partially oxidised and unoxidised rocks are potentially acid forming (PAF). In the presence of oxygen the PAF material oxidises. Some of the PAF waste is highly weathered to completely weathered and can be compacted to achieve low permeability fill. The acid neutralising capacity (ANC) of this rock has an expected range of 0 to 40 kg (H_2SO_4)/t, has a positive net acid producing potential (NAPP) and therefore the potential to produce AMD.

Initial geochemical investigations were conducted between 1984 and 1986 prior to the start of mining (GCNZ & SMA, 1986). Study results and site experience identified the need to reduce the rate of formation of AMD in the PAF waste rock and reduce the release of leachate from the mine. Additional geochemical test work commenced in 1993 and continued until Storage 2 was completed.

AMD Control

AMD control is generally undertaken using four approaches:

Hydrological control:	Placement of low permeability layers, evaporation layers and water management structures to control the release rate
Water treatment:	Modification of the drainage chemistry
Geochemical control:	Blending of rock types or addition of neutralising materials to control pH and oxidation rates and
Oxidation control:	Control of oxygen flux to reactive sulfides.

All four of the above approaches were used in managing drainage of Storage 2. During construction of the embankment, the mined waste rock was selected and compacted to reduce oxygen and water flux through the embankment, limestone was added to the PAF waste rock via a lime silo feed over a conveyor belt, and drains were installed to collect leachate which was and continues to be directed to a water treatment plant.

Embankment design and placement

The embankment of Storage 2 was constructed from waste rock excavated during mining. The waste rock was crushed to typically less than 0.3 m, transported by conveyor, loaded into trucks and placed in the embankment.

To achieve the requirements of structural strength, tailings retention and control of acid drainage the embankment structure incorporates various zones. Zone D of the embankment contains the majority of waste rock (Figure 1). Approximately 60% of the waste rock in Zone D was unoxidised and 40% was

oxidised and the mix of oxidised and unoxidised material would result in a spatial variability in the potential for oxidation. Zone E comprised soft and wet oxidised and unoxidised rock that was placed as sub compartments of Zone D.

During construction Zone D material handling specifications were designed to reduce material oxidation and acid production. For example, up to 1997 the surface of PAF material directly exposed to the atmosphere was limited to compartments no greater than 4 ha in area. The compartments were further constrained. They were to be no greater than 5m in height or exposed for longer than 3 months without an intermediate encapsulating compacted layer of low permeability fill. If the compartment was to continue to be raised within two weeks the encapsulating layer was 0.35 m thick, otherwise it was 0.45 m thick.

Since 1997 greater quantities of lime were added to the waste via a hopper discharging onto the conveyor and the need for previous restrictions on waste placement and intermediate sealing layers were eased. In particular 0.25 m thick low permeability sealing layers were only required at the completion of each stage of active construction or in response to geochemical measurements on the waste rock. Upon completion of Zone D three layers of material (Zones F, G and H) were placed over the outside shoulder of Zone D (Figure 1). This was to reduce the transport of oxygen from the atmosphere to the waste rock, to provide a suitable plant growth medium and to prevent erosion. Zone D is up to about 40 m thick.

Zone F is a transition layer between the coarse rock in Zone D and the fine grained soil in Zone G and consists of both oxidised and unoxidised waste rock. It was compacted in lift heights of less than 250 mm to achieve high structural strength to provide a stiff surface on which Zone G can be constructed and to provide improved performance of the cover under the maximum design earthquake loading.

Zone G is a 1.5 m thick cover layer of only oxidised NAF waste rock and it functions to limit transport of atmospheric oxygen in to the waste rock. It comprises selected fine grained soil, placed in lifts no greater than 250 mm thick, capable of being compacted to achieve low permeability (saturated hydraulic conductivity of less than 1×10^{-8} m/s) with high strength and also a high degree of saturation (10 test average greater than 90%).

Zone H is a layer designed to support vegetation and reduce erosion. Zone H was constructed of lightly compacted oxidised NAF waste rock. Zone B is the upstream structural face of the embankment with a saturated hydraulic conductivity of less than 1×10^{-8} m/s. Zone C is a structural Zone of oxidised and unoxidised rock with limits on particle size so as to provide a transition between the finer grained material in Zone B and the rock in Zone D.

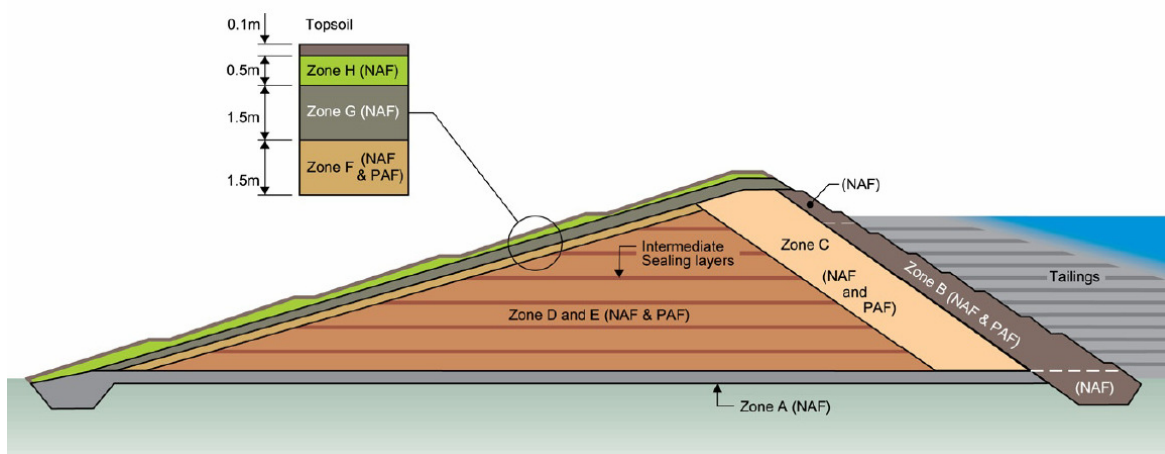


Figure 1. Cross-section Storage 2.

Hydrological control and water treatment

Hydrological control was attained through a combination of material selection, placement and compaction, as described above, and networks of drains and collection ponds. The networks of drains were installed in Storage 2 to collect water from the tailings, Zone D, the subsoil beneath the embankment and the natural groundwater (Figure 2). Leachate drains constructed to intercept and collect the leachate from within Zone D are designated with identifiers that begin with the letter 'L' in Figure 2. Figure 2 also shows surface water collection ponds.

Leachate from the subsoil and Zone D drains flowed by gravity to collection sumps from where it was pumped to the water treatment plant for treatment prior to discharge to the Ohinemuri River. All under drains and leachate drains had goosenecks at their outlets to prevent direct entry of oxygen.

The leachate drains were sampled approximately quarterly at the drain outlets and pH, sulfate and metals concentrations were determined.

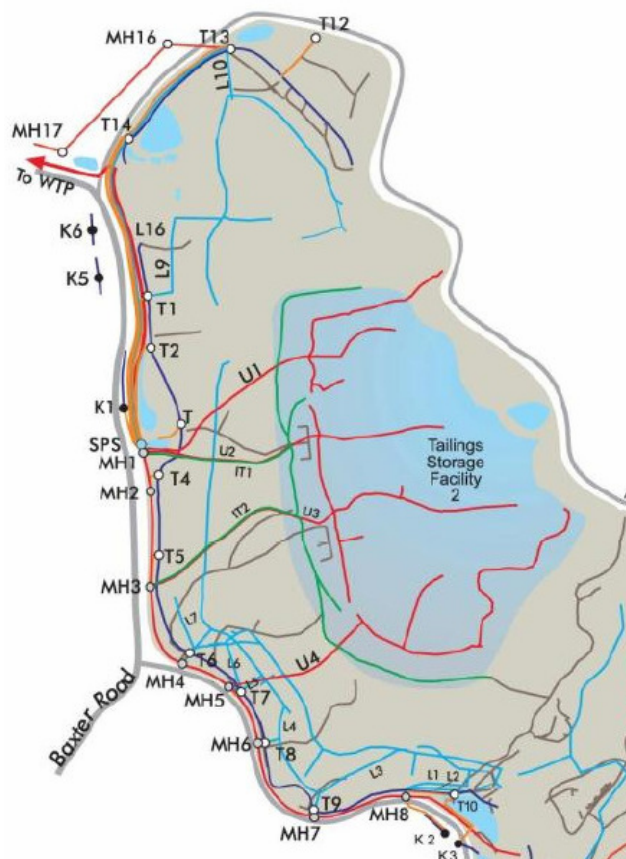


Figure 2. Storage 2 drainage network.

Geochemical control

The ANC available in PAF rock initially buffers the pH, however, as the PAF rock was net acid producing, water in contact with the rock become acidic after a lag time shorter than the time required to place Zone G over the Zone D waste rock. As the addition of limestone would extend the lag time testwork was conducted to determine the appropriate limestone application rates for geochemical control of water quality during construction of Storage 2.

Figure 3 demonstrates the effect of limestone addition on the pH lag period for a fast oxidation PAF andesite sample. Without treatment the pH dropped rapidly to about 4 within 4 weeks whereas with

limestone addition, the lag period was extended to 16 and 25 weeks for 2 and 5 kg CaCO_3/t treatments respectively.

Application of crushed limestone at rates of 5 to 10 tonnes of CaCO_3/ha and sealing by compaction and application of Zones G and H began in May 1994 and was fully operational in 1995. The pH values measured in two surface water ponds, Figure 4, confirm that this application rate was sufficient to achieve circum-neutral to alkaline surface waters most of the time. In early 1995 the pH decrease was due to inadequate liming during the dry season which reinforces the need to apply limestone to all PAF material in all conditions.

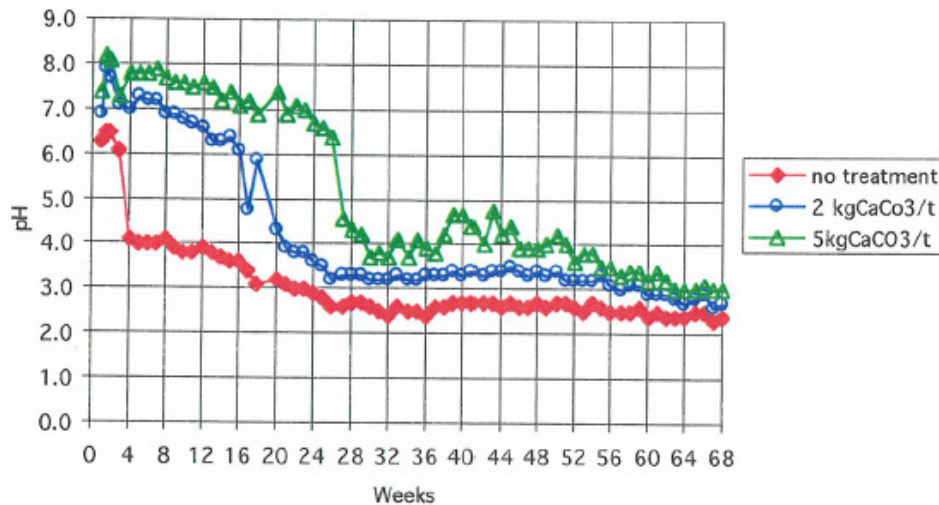


Figure 3. Effect of limestone on the pH lag period for fast reaction PAF andesite waste rock.

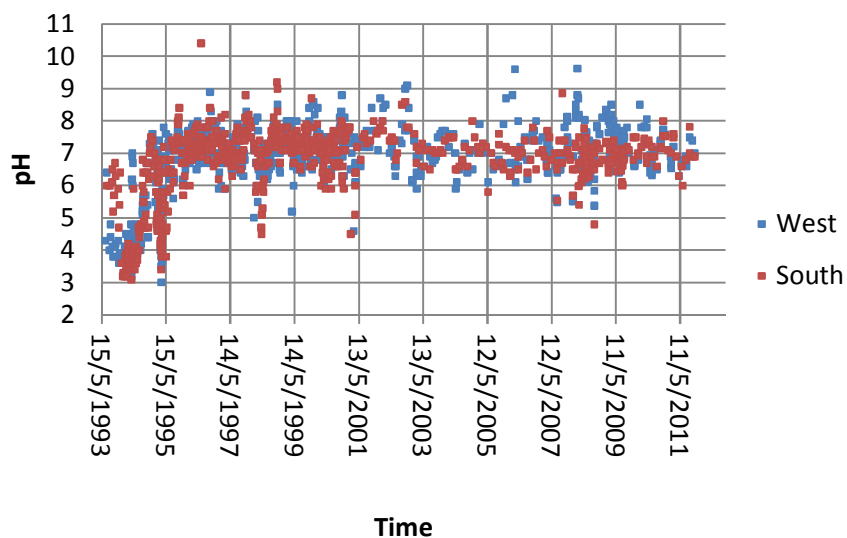


Figure 4. pH values in the South and West collection ponds.

Oxidation control

Diffusion, convection and advection can transport significant quantities of oxygen from the surface of a rock pile to the interior. Diffusion of oxygen is driven by a gradient in the partial pressure (or concentration) of oxygen and is proportional to the oxygen diffusion coefficient. In a pile containing sulfide minerals, oxidation of these minerals generally establishes oxygen concentration gradients.

Convection and advection are driven by spatial gradients in the total gas pressure. The pressure gradients that drive convection are established by internal heating as a result of sulfide oxidation (Ritchie, 2003) or by seasonal changes in surface temperature (Smolensky *et al.* 1999, Phillip *et al.* 2009). Advection may occur due to pressure gradients established by wind (Ann and Pantelis, 1997).

Intrinsic oxidation rate

The intrinsic oxidation rate is the rate of oxidation of material at a point in a sulfidic pile, under the conditions which pertain to that point. It is usually expressed as the rate of oxygen consumption per unit volume. It is a bulk property of the material and it may be a function of a number of variables, including temperature and oxygen concentration. Intrinsic oxidation rates derived from measured sulfate release rates in kinetic columns for Storage 2 rock were in the range 2.4×10^{-9} and 8.5×10^{-7} kg(O₂)/m³/s (Brodie and Miller, 2000). These values are comparable to rates measured in waste rock dumps containing sulfidic rock (Ritchie, 1992).

Oxidation of sulfides produces heat in the pile and consequently temperatures within the pile increase. Temperatures within the pile can therefore provide evidence of the location and rates of sulfide oxidation and the effectiveness of oxidation reduction strategies.

Convection, advection and permeability

The rate of gas transport by advection and convection is dependent on the intrinsic permeability of embankment material. A reduction in intrinsic permeability reduces both the saturated hydraulic conductivity and gas transport rates for given driving forces.

A rigorous quality assurance program was undertaken by Waihi Gold up to November 1998 and subsequently by an independent testing agency to confirm that the specified construction standards were achieved. The Construction Specification required the ten test average saturated hydraulic conductivity of Zone G to be less than 10^{-8} m/s and the degree of saturation to be greater than or equal to 90% or greater than 85% for a single test. The frequency of testing was one hydraulic conductivity test for every 20,000 m³ of Zone G and for twenty degree of saturation tests per hectare for every layer of fill. Zone G was also required to meet air voids criteria (a 10 test average not greater than 8% and a single test not greater than 10%), undrained shear strength (a 10 test average greater than 100 kPa and a single test value greater than 80kPa) and Zone G was required to be placed with a water content that would allow optimum compaction.

The results of hydraulic conductivity tests indicate saturated hydraulic conductivity values generally in the range of 1×10^{-9} m/s to 1×10^{-8} m/s. Testing of the degree of saturation confirmed that the specified degree of saturation was achieved. Fill that did not meet the standard was removed or reconditioned then re-compacted and re-tested. Zone H is a plant growth layer and did not have the rigorous testing that applied to Zone G.

Degree of saturation and oxygen diffusion coefficient

The diffusion coefficient of a soil or similar materials is dependent on the fraction of the pore space filled with water, i.e. the degree of saturation (Figure 5, Millington and Shearer, 1971). An embankment Zone with a high degree of saturation therefore has a relatively low oxygen diffusion coefficient and low rate of oxygen supply by diffusion.

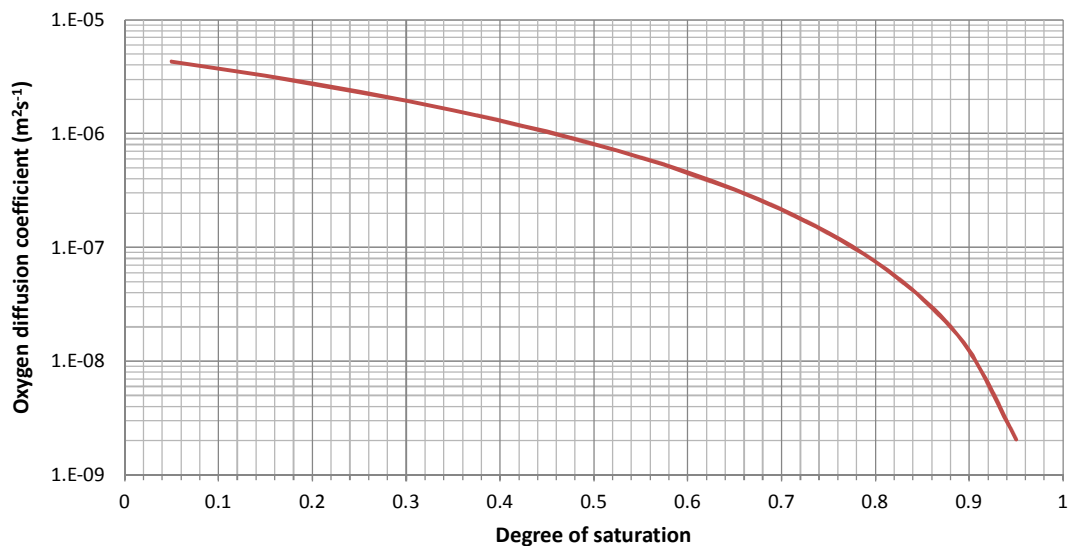


Figure 5. Dependence of the oxygen diffusion coefficient on the degree of saturation.

Field measurements of the oxygen diffusion coefficient were made in Zones G and H between 1994 and 1999 (Garvie, 1999). These included a set of measurements in February 1998 when the rainfall since December 1997 had been unusually low. The measurements were made at 21 locations spread widely across the embankment.

There was variation in the values of oxygen diffusion coefficient, however, it was concluded that Zone G generally had lower diffusion coefficients than Zone H and a diffusion coefficient value of $1.5 \times 10^{-8} \text{ m}^2/\text{s}$ was considered representative for Zone G.

Balloon densitometer and calibrated nuclear densitometers were used to determine the water content and bulk density of Zone G and H materials. The measured values were used with measured specific gravities to calculate the degree of saturation. Measurements were conducted adjacent to six diffusion coefficient measurement locations and 30 other locations in Zone G. The average degree of saturation adjacent to the Zone G diffusion coefficient measurement locations were 0.877 and 0.904 for the balloon and nuclear densitometers respectively. These values correspond well with the representative oxygen diffusion coefficient of $1.5 \times 10^{-8} \text{ m}^2/\text{s}$ determined by measurement (Figure 5). The average calculated values of degree of saturation for other Zone G locations were 0.939 and 0.949 respectively, which also correspond to low oxygen diffusion coefficients.

Oxygen concentration and temperature measurement

Ten vertical probe holes installed in Storage 2 in 1994 were used to measure vertical oxygen concentration and temperature profiles (Clark and Garvie, 1994). The probehole design is shown in Figure 6. Gas sampling tubes terminated at different depths from the surface. Between 0 and 2 m the interval was 0.25 m and below 2 m the interval was typically 1 m. The length of probe holes varied between 7 and 20 m. The backfill was a combination of sand and bentonite layers. A set of thermistors was lowered into the PVC pipe to measure temperatures at a number of depths.

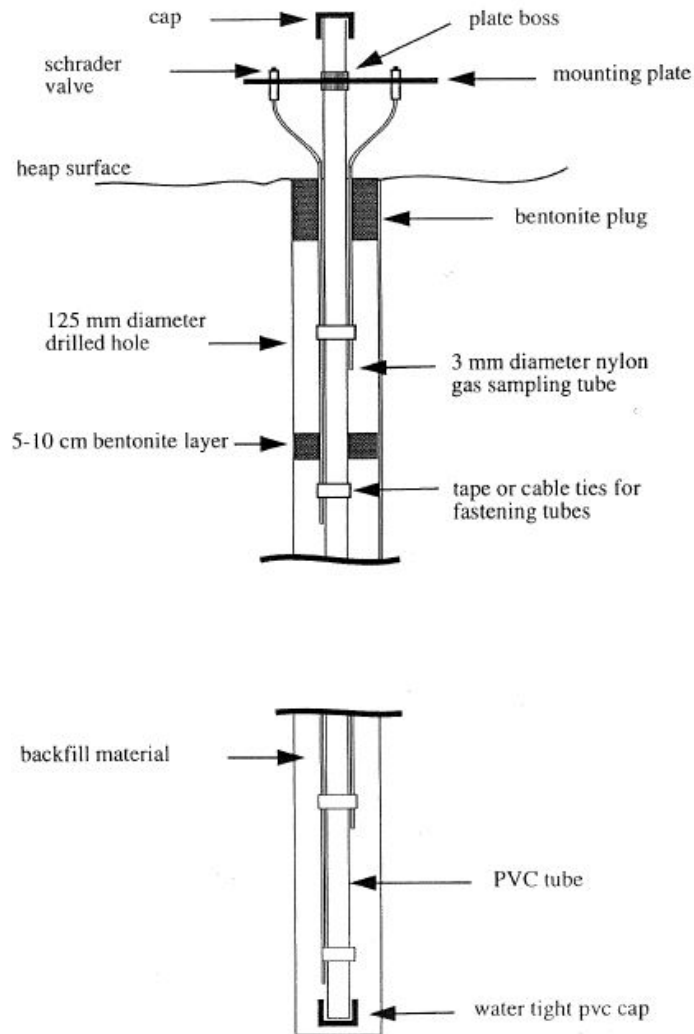


Figure 6. Probehole for measurement of oxygen concentration and temperature.

Oxygen concentration and temperature profiles were measured in Storage 2 in August 1994 and repeated in October 2011. Selected results are presented in Figure 7.

Oxygen concentrations measured in October 2011 were similar to those measured in 1994. Typically oxygen concentrations decreased rapidly with distance from the surface concentration (21 v%). At some locations the oxygen concentration was less than or equal to 1 v% within 5 m of the surface and within 3 m of the base of Zone G(Figure 7c).

There were profiles where oxygen concentrations decreased and subsequently increased with depth (e.g. WI08 at 15 m), indicating that oxygen may have been supplied to and consumed at deeper regions in the embankment. Temperatures at WI08 at 15 m were within 1°C of the long term average temperature. The temperature gradient was zero, within the accuracy of the measurements, and indicates that the oxidation rate approached zero. Assuming that the water infiltration rate into Zone D was 1.6% of rainfall the estimated pore water sulfate (SO_4) concentration in the vicinity of the WI08 at 15 m was less than 2400 g/m³.

In general temperature profiles (e.g. Figure 7 b and d) did not provide evidence of oxidation at locations more than 5 m below the embankment surface in either 1994 or 2011. Therefore oxygen diffusion is considered the dominant gas transport mechanism.

Where diffusion is the dominant gas transport mechanism, the Constant Intrinsic Oxidation Rate Model (Gibson 1994; Ritchie 1995) provides a method of calculating global oxidation and acid sulfate generation rates (ASGR). The ASGR is the rate of generation of acid and sulfate due to the oxidation of sulfides and is reported in the units $\text{kg}(\text{H}_2\text{SO}_4)/\text{ha}/\text{d}$. Calculated ASGR values in Table 1 indicate that Zone G reduced the ASGR by more than 95%.

Miller and Brodie (2000) reported ASGR a value of $1.4 \text{ kg}(\text{H}_2\text{SO}_4)/\text{ha}/\text{d}$ for the covered embankment. This value was calculated assuming a lower oxygen diffusion coefficient value of $0.5 \times 10^{-9} \text{ m}^2/\text{s}$.

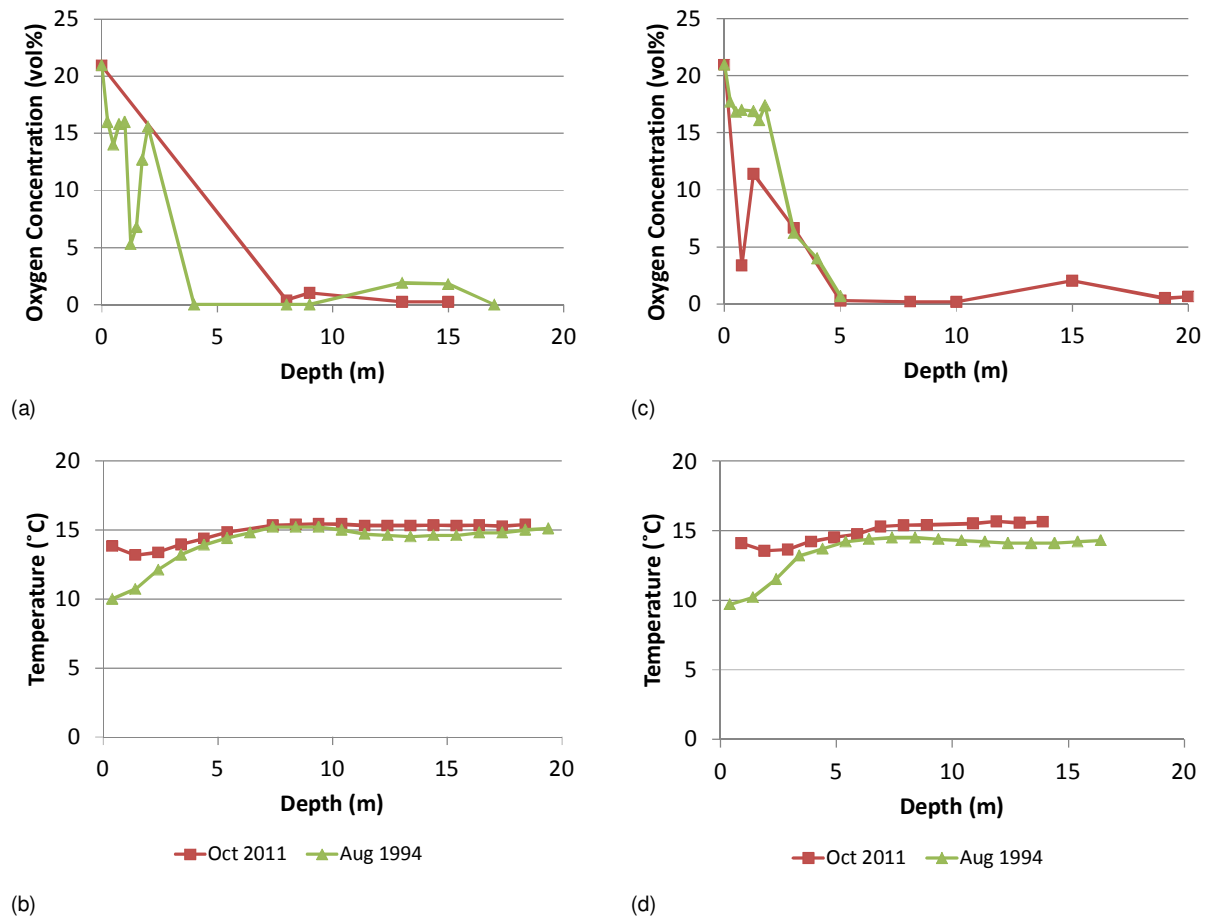


Figure 7. Oxygen concentration and temperature measurements at WI07 (a) and (b), and WI08 (c) and (d).

Table 1. Sulfate generation rates in the embankment.

IOR	ASGR		
	With cover	Without cover	Reduction due to Zone G
kg(O ₂)/m ³ /s	kg(H ₂ SO ₄)/ha/d		%
2.4×10^{-9} ^(a)	3	70	96
8.5×10^{-7} ^(b)	4	1990	99.7

Notes: Diffusion coefficient of the cover and waste rock were 1.5×10^{-8} and 2×10^{-6} m²/s, respectively.
Cover thickness was 1.5 m.

(a) and (b) are Brodie and Miller (2000) minimum and maximum values respectively.

Sulfate Concentrations

Table 2 presents the potential increase in sulfate concentrations in pore water due to oxidation before placement of Zone G and compares them with the increase in sulfate concentration in the pore water after placement of Zone G. During construction of the embankment rain infiltrating Zone D would have dissolved sulfates produced by oxidation. Some of these dissolved sulfates would remain in the embankment in pore water that would subsequently be collected by the leachate drains.

Sulfates produced due to oxidation after placement of Zone G would also have been available to increase the sulfate concentration in the pore water. In estimating the sulfate concentration in pore water before and after application of Zone G it was assumed that there was no solubility control of sulfate (e.g. by gypsum precipitation).

Table 2. Estimates of sulfate concentration in pore water.

IOR	Increase in SO ₄ concentration in pore water	
	Before Zone G placement	After Zone G placement (2011)
kg(O ₂)/m ³ /s	g/m ³	g/m ³
2.4×10^{-9} ^(a)	24	960
8.5×10^{-7} ^(b)	13,500	1229

Notes: (a) and (b) are Brodie and Miller (2000) minimum and maximum values respectively no account was taken of the potential for sulfate solubility controls (e.g. by gypsum precipitation)

Measured concentrations in leachate collected by 14 drains in Zone D are presented in Figure 8. All values are less than the maximum (E1) concentration estimated for the pore water due to oxidation before application of the cover assuming no sulfate solubility controls (Table 2).

Equilibrium speciation modelling using the software package PHREEQC (Parkhurst and Appelo, 1999) indicated that sulfate concentrations in the majority of the drains could be controlled by gypsum solubility. That is, gypsum precipitation places an upper bound on dissolved sulfate concentrations in the leachate. As conditions change within the embankment (for example, pore water dilution due to ingress of rain) the precipitated gypsum may dissolve releasing sulfate into the leachate.

In Figure 8, the blue highlighted region indicates the range of measured sulfate concentrations that may be subject to control by gypsum solubility. The wide range of solubility controlled sulfate concentrations is a reflection of the variability in overall solution chemistry. In particular, magnesium is present at concentrations of several hundred g/m^3 and sustains sulfate concentrations much higher than typically observed in pure calcium – sulfate solutions.

The yellow region in Figure 8 indicates the range of sulfate concentrations that would be expected following cover emplacement (i.e. potentially attained by the water infiltrating Zone D and oxidation in Zone D after placement of the cover layers (Zones G and H)). The predicted concentrations are less than those currently measured in 9 leachate drains. Therefore it is likely that current sulfate concentrations reflect flushing of sulfate generated in the period prior to cover emplacement. As the pre-existing sulfate load is depleted, it is expected that sulfate concentrations would decrease. That process will be limited by the rate at which water flushes through the rock. Ultimately, sulfate concentrations would be controlled by oxidation under Zone G.

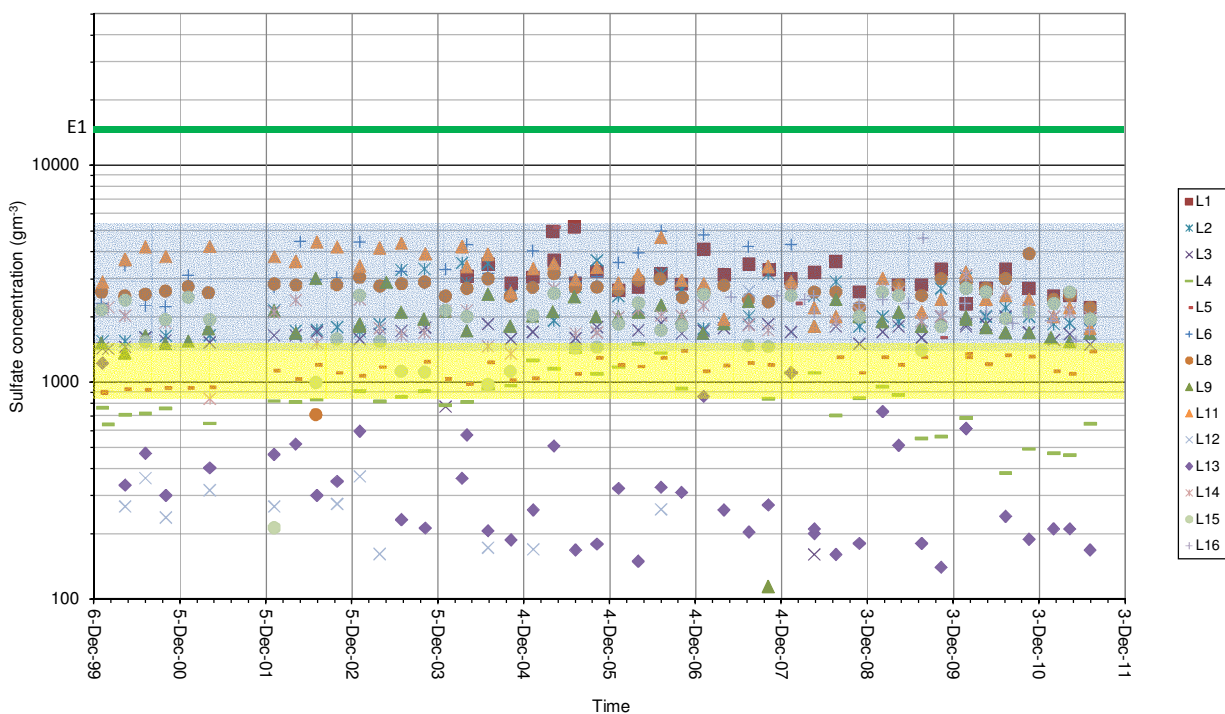


Figure 8. Sulfate concentrations measured in the leachate drains of Zone D.

Line E1 is the estimated upper concentration in the pore water before application of the cover. The blue shaded area indicates range of measured sulfate concentrations that may be subject to control by gypsum solubility. Yellow shaded area indicates the concentration in water that infiltrates the cover due to ongoing oxidation. (One sulfate concentration of 94.7 g/m^3 for drain L15 does not appear on the figure).

Conclusions

A compacted cover layer of oxidised waste rock (Zone G) on the embankment of Storage 2 in Waihi, New Zealand, has reduced the rate of oxidation of underlying sulfide bearing waste rock (Zone D). Zone G was effective because it had a low oxygen diffusion coefficient of about $1.5 \times 10^{-8} \text{ m}^2/\text{s}^{-1}$.

As a result of the effectiveness of Zone G the global sulfate generation rates were 3 to 4 kg(SO₄)/ha/d in 2011. These values are within the range reported by Miller and Brodie (2000) (1.4 kg(SO₄)/ha/d) and Garvie *et al.* 1997 (5.5 kg(SO₄)/ha/d) . Zone G has reduced the rate of acid sulfate generation to less than 6% of that expected for uncovered waste rock.

It is likely that current sulfate levels in the leachate drains reflect flushing of sulfate generated in the period prior to cover emplacement. Over the long term, as the pre-existing sulfate load is depleted, it is expected that sulfate loads would decrease. Ultimately, sulfate concentrations would be controlled by oxidation under Zone G and the rate of infiltration of water into Zone D.

References

- Anne, RD and Pantelis, G, (1997). Coupled Natural Convection and Atmospheric Wind Forced Advection in above Ground Reacting Heaps, Proceedings of International Conference on CFD in the Mineral and Metal Processing and Power Generation, CSIRO.
- Brodie, K, Miller, S and Matushka, T, (1996). Prediction and Control of Acid Rock Drainage and the Martha Mine, Waihi, New Zealand. Personal communication.
- Garvie AM, Plotnikoff, WW and Ritchie, AIM, (1997). *Provisional Assessment of the Global Sulfate Generation Rate During a Period of Average Rainfall and an Extended Period of Low Rainfall at the Martha Mine*, ANSTO/C466 Australian Nuclear Science and Technology Organisation.
- GCNZ & SMA, (1986). *Martha Hill Project. Geochemical Evaluation of Leachates from Waste Rock and Tailings. Final report. July 1986* , Prepared by: Ground Water Consultants New Zealand Limited and Sturat Miller and Associates Pty Ltd.
- Miller, S and Brodie, K, (2000) Cover Performance for the Control of Sulfide Oxidation and Acid Drainage from Waste Rock at the Martha Mine, New Zealand. Fourth Australian Workshop on Acid Mine Drainage, Townsville, 28 Feb. to 2 March 2000.
- Millington, RJ and Shearer, RC, (1971). Aggregated Porous Media, Soil Science, 111, 6, pp 372-378.
- Phillip, M, Hockley, D, Dawson, B, Kuit, W and O’Kane, M. (2009). 8th ICARD, June 22-26, Skelleftea, Sweden.
- Ritchie, AIM, Oxidation and Gas Transport in Piles of Sulfidic Material (2003), In: Environmental Aspects of Mine Wastes. Eds Jambor JL, Blowes, DW and Ritchie, AIM, Short Course Volume 31, Mineral Association of Canada, Vancouver, British Columbia.
- Ritchie, A.I.M. (1992), Environmental geochemistry of sulphide oxidation Ed. Alpers, CN and Blowes, DW ACS Symposium series 550 pp 108-122.
- Smolensky, J, Hockley, D, Lefebvre, R, Paul, M (1999) Oxygen Transport Processes in the Nordhalde of the Ronneburg Mining District, Germany, In: ‘Sudbury 1999- Mining and the Environment II, Sudbury Ontario, 13-15 September 1999.