Australian Government

PREVENTING ACID AND METALLIFEROUS DRAINAGE

Leading Practice Sustainable Development Program for the Mining Industry

September 2016

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Leading Practice Sustainable Development Program for the Mining Industry.

This publication has been developed by a working group of experts, industry, and government and non-government representatives. The effort of the members of the Working Group is gratefully acknowledged.

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Support for the LPSDP was provided by the Australian aid program administered by the Department of Foreign Affairs and Trade due to the reports' value in providing practical guidance and case studies for use and application in developing countries.

Cover Page Left: AMD seepage evaporite deposits downstream of a covered waste rock dump

Cover Page Centre: Partly flooded acidic pit showing oxidising sulfidic wall rocks

Cover Page Right: Lime dosing water treatment plant

All cover photos: D Jones

September 2016.

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ACKNOWLEDGEMENTS

The Leading Practice Sustainable Development Program is managed by a Steering Committee chaired by the Australian Government Department of Industry, Tourism and Resources. The 17 themes in the program were developed by working groups of government, industry, research, academic and community representatives. The leading practice handbooks could not have been completed without the cooperation and active participation of all members of the working groups, and their employers who agreed to make their time and expertise available to the program. The mining companies, government departments and other organisations that contributed case studies to demonstrate the application of leading practice are especially acknowledged. Particular thanks go to the following people and organisations who contributed to the production of this handbook:

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FOREWORD

The *Leading Practice Sustainable Development Program for the Mining Industry* series of handbooks has been produced to share Australia's world-leading experience and expertise in mine management and planning. The handbooks provide practical guidance on environmental, economic and social aspects through all phases of mineral extraction, from exploration to mine construction, operation and closure.

Australia is a world leader in mining, and our national expertise has been used to ensure that these handbooks provide contemporary and useful guidance on leading practice.

Australia's Department of Industry, Innovation and Science has provided technical management and coordination for the handbooks in cooperation with private industry and state government partners. Australia's overseas aid program, managed by the Department of Foreign Affairs and Trade, has co-funded the updating of the handbooks in recognition of the central role of the mining sector in driving economic growth and reducing poverty.

Mining is a global industry, and Australian companies are active investors and explorers in nearly all mining provinces around the world. The Australian Government recognises that a better mining industry means more growth, jobs, investment and trade, and that these benefits should flow through to higher living standards for all.

A strong commitment to leading practice in sustainable development is critical for mining excellence. Applying leading practice enables companies to deliver enduring value, maintain their reputation for quality in a competitive investment climate, and ensure the strong support of host communities and governments. Understanding leading practice is also essential to manage risks and ensure that the mining industry delivers its full potential.

These handbooks are designed to provide mine operators, communities and regulators with essential information. They contain case studies to assist all sectors of the mining industry, within and beyond the requirements set by legislation.

We recommend these *leading practice* handbooks to you and hope that you will find them of practical use.

Senator the Hon Matt Canavan

Minister for Resources and Northern Australia

The Hon Julie Bishop MP

Minister for Foreign Affairs

1.0 INTRODUCTION

Key messages

- Any mining activity that exposes mined materials to air and water has the potential to generate ongoing water pollution.
- The potential risk of contaminated mine drainage should be fully evaluated before mining begins. The primary focus should be on prevention or minimisation, rather than collection and treatment.
- While the cost of active management of sulfidic waste during operations to prevent the inception of acid and metalliferous drainage (AMD) can be significant, it is often small in comparison with the long-term costs that could be incurred to retroactively implement control and treatment strategies.
- Leading practice in this area continues to evolve—there are no quick fixes or one-size-fits-all solutions to AMD problems, and specialist expertise is often needed to develop the most appropriate management strategy.
- Despite examples of leading practice throughout the industry (see the case studies in this handbook), leading practice principles for the management of AMD risks are not universally understood or applied.

1.1 The scope of this handbook and who should use it

This handbook addresses the topic of acid and metalliferous drainage (AMD), which is an important component of the range of issues covered by the Leading Practice Sustainable Development in Mining Program. The program aims to identify the key issues affecting sustainable development in the mining industry and to provide information and case studies that illustrate a more sustainable approach for the industry.

The emphasis in this handbook is on preventing the production of AMD from exposed sulfidic materials, followed by managing existing AMD, followed by treating AMD. This approach is consistent with the contemporary hierarchy for risk reduction.

However, the occurrence of only low concentrations of sulfide minerals, or indeed the absence of sulfides, does not mean that mined materials (or material exposed by mining, such as pit wall rocks) will not produce drainage, such as neutral metalliferous drainage (NMD) and saline drainage (SD) as defined in Section 2, that could be of risk to the receiving environment. For this reason, each type of mined material must be assessed on its own merits as part of a mine material characterisation program to ensure that the waste is managed appropriately.

This handbook covers all phases of a mining project, from exploration and feasibility studies through to operations and closure. It is applicable to exploration properties, operating and decommissioned mines and brownfield and legacy sites. It is primarily intended to be a resource for mine managers, mine planners and site operational staff responsible for developing, implementing and reporting the outcomes of an AMD management plan. However, the content is also a resource for consultants, government authorities and regulators, non-government organisations, interested community groups and students.

AMD is a complex topic, so a substantial amount of technical content is provided to meet the information needs of those on mine sites responsible for the development and implementation of an AMD management plan. A good high-level introduction to the topic, and the key management issues that need to be addressed, can be obtained by reading sections 1–3, and the key messages and introductory elements in each of the remaining sections.

- Section 2 describes the different types of AMD and how they are produced.
- Section 3 is an overview of leading practice for the prevention of AMD.
- Section 4 provides tools to address the management issues raised in Section 3 and describes ways to predict the magnitude of an AMD issue and model it.
- Section 5 discusses the way those factors can be incorporated into an assessment of AMD risk.
- Section 6 outlines strategies to limit the generation and release of AMD.
- Section 7 describes methods to treat AMD that is produced.
- Section 8 addresses the regulatory aspects of AMD.
- Section 9 describes where, when and how to monitor for AMD.
- Section 10 examines 'social licence to operate' and stakeholder communication aspects.

1.2 Overview of the issues

Predicting and managing the occurrence of AMD to minimise risks to human and environmental health is one of the key challenges facing the mining industry. Any type of mining, quarrying or excavation operation that exposes minerals to water and air has the potential to generate water pollution. As some types of mining operations have evolved from high-grade, low-tonnage underground operations to large-tonnage (with typically high stripping ratio), low-grade open-cut operations over the past 30 to 40 years, the volume of surface material with the potential to create AMD has increased exponentially.

AMD can also result from the disturbance of acid sulfate soils, which occur naturally in geologically recent estuarine and mangrove swamp environments. AMD from this source is often associated with the mining of mineral sand deposits in near-coastal or low topographical locations. The characterisation and management of acid sulfate soil material can be similar to that used for sulfide-containing material at mine sites.[1]

All sulfide-containing material has the potential, when exposed to water and air, to produce run-off and/or leachate with increased concentrations of solutes. The key questions to be addressed are the extent to which this may occur and whether the risk to the environment is of a magnitude that needs to be mitigated to produce an acceptable outcome. This handbook presents:

- the tools needed to estimate the likely magnitude of the risk
- the strategies needed to manage mined materials in order to reduce the risk to an acceptable residual level
- treatment processes that can be used to treat poor-quality seepage and run-off, in the event that it occurs.

1 See Dobos (2005) and Ahern et al. (2014) for further guidance.

AMD may be very acidic (low pH) and contain elevated concentrations of metals, metalloids and major ions and low concentrations of dissolved oxygen. Hence, it can present a major risk to aquatic life, riparian vegetation and human uses of the water resource for many kilometres downstream from where it enters a waterway. In many parts of the world, local communities depend on waterways for their livelihood. Clean water is essential for drinking, crop irrigation and stock watering, and is vital to sustain aquatic ecosystems, including aquatic life used for food.

Mining activities in the past have sometimes damaged ecosystems and had very heavy impacts on communities. Today, such poor practice should not occur if mining is to be accepted by society as part of an economically sustainable development framework. Successful management of AMD is vital to ensure that mining activities meet increasingly stringent environmental regulations and community expectations, and that the industry's social licence to operate is maintained.

Once a mining operation has ceased, poor-quality water from the production of AMD may continue to damage the environment, human health and livelihoods for decades or even centuries. A mine site in the Iberian Pyrite Belt in Spain, for example, has been generating AMD for more than 2,000 years.

The crucial step in leading practice management of AMD is to assess the risk as early as possible. 'Risk' includes environmental, human health, commercial, reputation, legal and regulatory risks. The progressive evaluation of AMD risk, begun during exploration and continuing through the feasibility evaluation stage, provides the data necessary to quantify potential impacts and management costs before significant disturbance of sulfidic material. When projects proceed at sites where AMD is a potential risk, efforts should focus on prevention or minimisation, rather than on control or treatment.

At decommissioned and older operating mine sites where the characterisation of mined materials for AMD potential and/or management of resultant drainage has been inadequate, high remediation and treatment costs can continue to reduce the profitability of the operating companies. The term 'treatment in perpetuity' has entered modern mining vocabulary as a result of intractable AMD issues that prevent the relinquishment of mining leases, despite the cessation of mining operations. Such situations are inconsistent with sustainable mining practice and must be avoided at new mines by the application of appropriate materials characterisation and AMD minimisation strategies.

Leading practice management of AMD continues to evolve, and that evolution has been captured in the content of successive editions of this series of handbooks and elsewhere (Jones & Taylor 2008; Jones 2011; Miller 2014). This handbook outlines current leading practice approaches for predicting and minimising the occurrence of AMD from a risk management perspective and contains case studies highlighting the application of leading practice strategies for the management of sulfidic wastes that are currently being, or have been, implemented by the industry. In particular, a number of the case studies document strategies that were implemented during the operating life of a mine, and for which longer term monitoring data is now available to demonstrate good performance over at least a decade. There are also updates of previous case studies that indicate the benefits of the early implementation of leading practice concepts. This contrasts with the previous two editions of this handbook, which by necessity used case studies of management strategies that had just been implemented and for which there had been insufficient time to fully assess their performance.

Specialist expertise may be needed to help plan and implement many of the strategies described in this handbook. It is particularly important that expert advice be sought during the characterisation and prediction process (Section 4), in assessing the risk posed by AMD (Section 5), and before the selection and implementation of long-term minimisation and control strategies (Section 6).

1.3 Sustainable development

Based on the widely accepted Brundtland Commission definition (UNWCED 1987), sustainable development is 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs'. In recent years, sustainable development principles have been applied to the mining sector by governments and community and non-government groups as well as by the mining industry itself.

To provide a framework for articulating and implementing the mining industry's commitment to sustainable development, the Minerals Council of Australia (MCA) developed *Enduring Value: the Australian minerals industry framework for sustainable development.* The framework was initially released in 2005 and was updated in 2015 (MCA 2015a). *Enduring value* is specifically aimed at supporting companies to go beyond regulatory compliance and to maintain and enhance their social licence to operate. Enduring value's risk-based continual improvement approach is reflected in this handbook.

Comprehensive guidance on sustainable mining practices and on strategies for more effective community engagement has also been produced by the World Bank (2007) and the International Council on Mining and Metals (ICMM 2008).

1.3.1 The environment and communities

Directly involved stakeholders and communities and members of communities that may be indirectly affected by mining often place different emphases on the social, environmental and economic aspects of sustainability. For example, cultural and social issues may be more important to indigenous groups. Some stakeholders might be satisfied if engineered rehabilitation measures for AMD meet performance targets for several decades, while others will demand 'in perpetuity' solutions. The size of bonds or financial sureties for mining projects can be substantial, given the scale of the works required and the time frame over which performance targets must be met. The issue of perfomance bonds is addressed in Section 3 of the *Mine closure* handbook in this series (DIIS 2016a).

When considering the environmental implications of AMD and other mine-related impacts, a common principle used by communities, non-government organisations and regulators is the 'precautionary principle'. The principle is defined in section 3 of the *Environment Protection and Biodiversity Conservation Act 1999* (Cwlth) (EPBC Act): 'if there are threats of serious or irreversible environmental damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation'. In practice, this means that lack of scientific evidence for a real or potential detrimental impact should not be used to approve an action or to permit the continuance of an action when evaluating proposals that have the potential to seriously or irreversibly damage the environment. The use of the precautionary principle requires decision-makers to take a risk-based approach to the assessment.

Given the challenges and scientific uncertainties associated with the prediction and management of AMD, the application of the precautionary principle is vital when the need for control strategies is being considered.

Potentially affected communities expect that decisions about the management of AMD will be based on more than just economic costs. While AMD management decisions need to be informed by technical investigations, stakeholder aspirations and values should also be considered when applying a 'whole-of-mine-life' planning perspective to both day-to-day operations and closure planning. In this context, while the operational life of most mines is usually measured in years or decades, the effective life of control strategies and remediation works may have to be measured in centuries. An effective AMD management strategy must integrate social, economic and environmental aspects to achieve an acceptable sustainable outcome for all concerned (see the Kelian mine case study 'Community relationships and the closure of a mine in Indonesia' in the *Mine closure* handbook).

Section 10 of this handbook provides more information about communicating and discussing AMD-specific issues with affected stakeholders and communities. Refer to the following handbooks for more information and guidance about addressing stakeholder and community issues:

- *Community engagement and development* (DIIS 2016b)
- *Working with Indigenous communities* (DIIS 2016d)
- *Social responsibility in the mining and metals sector in developing countries* (DRET 2012)
- *Mine closure* (DIIS 2016a)
- *Evaluating performance: monitoring and auditing* (DIIS 2016c).

1.3.2 Business

At present, despite examples of leading practice throughout the industry (see case studies in this handbook), leading practice principles for management of AMD risks are not universally understood or applied.

The MCA *Enduring value* framework (MCA 2015a) provides mining companies with a vision for sustainable development as well as guidance on its practical implementation. Leading practice companies have also targeted policies and procedures relevant to managing AMD that are binding on management, employees and contractors. Additional commitment to environmental certification, participation in initiatives such as the International Network for Acid Prevention (INAP), the Canadian Mine Environmental Neutral Drainage (MEND) program and the US Acid Drainage Technology Initiative (ADTI), and the regular involvement of AMD experts in operational decision-making all lead to improved performance that can more cost effectively meet or exceed stakeholder expectations.

While the cost of appropriate management of AMD-producing materials during operations can be significant, it is often small in comparison with the long-term costs that could otherwise be incurred to retroactively implement an AMD control and/or treatment strategy. There is ample historical evidence of the consequences of failing to predict and manage AMD for individual operations and for the mining industry as a whole. Those consequences can include significant unplanned spending on remedial measures, damage to reputation, the introduction of more stringent regulatory requirements, and premature closure as a result of inability to finance the increased environmental management costs.

Unplanned cost escalations have frequently been in the order of at least US$80 million where operations have had to retroactively implement an AMD management and/or treatment strategy during the closure phase. In the case of the Ok Tedi mine in Papua New Guinea, the implementation of a process during the operating life of the mine to reduce the sulfide content of tailings and improve closure prospects has cost more than US$500 million (see the Ok Tedi case study in Section 7).

The annual costs of AMD management at operating sites in Australia have been estimated as US$100 million (Ciccarelli et al. 2009). Further examples of the cost of managing the short- and long-term effects of AMD are provided in MEND (1995), USEPA (1997) and Wilson et al. (2003). It has been suggested that the total global cost for the environmental liabilities associated with AMD is of the order of US$100 billion (Wilson 2008).

The risks presented by inadequate AMD management can be significant. Aside from the large scale and cost of remediation and clean-up when things go wrong, inadequate management creates the perception that the industry is unable to avoid harmful impacts. These outcomes do not align with the industry's aim of making a strong contribution to sustainable economic development and earning and maintaining its social licence to operate.[2]

1.3.3 International guidance

Since the last edition of this handbook was published in 2007, there has been substantial growth in international guidance material with a specific focus on issues associated with AMD.

A comprehensive web-based international global acid rock drainage (GARD) guide, produced with funding support from INAP, was launched in 2009.[3] The GARD Guide (INAP 2009) is a state-of-the-art summary (in web-based Wiki format) of practices and technologies to assist mine operators and regulators to address issues related to sulfide mineral oxidation. Its target audience is technical personnel with a reasonable background in chemistry and/or the basics of engineering with little specific knowledge of AMD.

The MEND program, which has been operating since 1989, continues to provide leading edge resources addressing the range of issues associated with AMD prediction and management.[4] Of particular note is the comprehensive *Prediction manual for drainage chemistry from sulfidic geologic materials* (Price 2009).

ADTI in the US promotes advances in acid drainage prediction, prevention, control, sampling, monitoring and treatment, with an emphasis on evaluating and developing cost-effective and practical acid drainage prevention methods.[5]

An innovative approach has been developed by the Interstate Technology and Regulatory Council (ITRC) in the US to assist with the more rapid promulgation and take-up of new mine waste management approaches by industry and their acceptance by regulators. This initiative recognises that a significant constraint on industry's implementation of new remediation methods can be regulatory inertia resulting from different views within and between jurisdictions about the effectiveness of the methods. The ultimate ITRC product is a web-based guidance framework that guides the selection of appropriate technologies for particular applications.[6] The ITRC website contains decision trees, overviews of applicable technologies and more than 65 case studies in which these technologies have been implemented.

Guidance on AMD management in the European Union is produced by the Partnership for Acid Drainage Remediation in Europe (PADRE).[7] The Water Research Commission of South Africa has been very active over the past decade in developing guidance related to the management of AMD-affected waters as a result of developing AMD issues in that country.[8]

Updates on mining environmental management in general and on strategies for more effective community engagement have been produced by the International Finance Corporation (IFC 2013) and the International Council on Mining and Metals (ICMM 2015).

2 Further information about assessing the magnitude of risk posed by AMD is provided in Section 5.

3 GARD Guide, http://www.gardguide.com.

4 http://mend-nedem.org.

5 http://aciddrainage.com/index.cfm.

6 http://www.itrcweb.org/miningwaste-guidance.

7 http://www.padre.imwa.info.

8 http://www.wrc.org.za/Pages/KH_DocumentsList.aspx?su=c7&ms=2;68.

2.0 UNDERSTANDING ACID AND METALLIFEROUS DRAINAGE

Key messages

- The acronym 'AMD' is defined here as acidic and metalliferous drainage and includes acidic drainage, pH neutral metalliferous drainage (NMD), and saline drainage (SD) generally caused by the oxidation of sulfide minerals.
- AMD sources can include waste rock dumps (WRDs); ore stockpiles; tailings storage facilities (TSFs) and tailings dams; roadways and embankments constructed with sulfidic material; open cuts and mine pits; underground mines; heap and dump leach piles; and acid sulfate soils.
- It is the total load (the product of concentrations and flow), not concentrations alone, of acid, metals/metalloids and salts in the source mine drainage that influences the magnitude of downstream impacts and the cost of treatment.
- AMD (as manifested by NMD or SD) can still be an issue even if a site assessment concludes that drainage with a low pH is unlikely to develop.
- The potential for self-heating and auto-ignition of sulfidic waste materials must be considered and assessed where needed.

2.1 Types of AMD

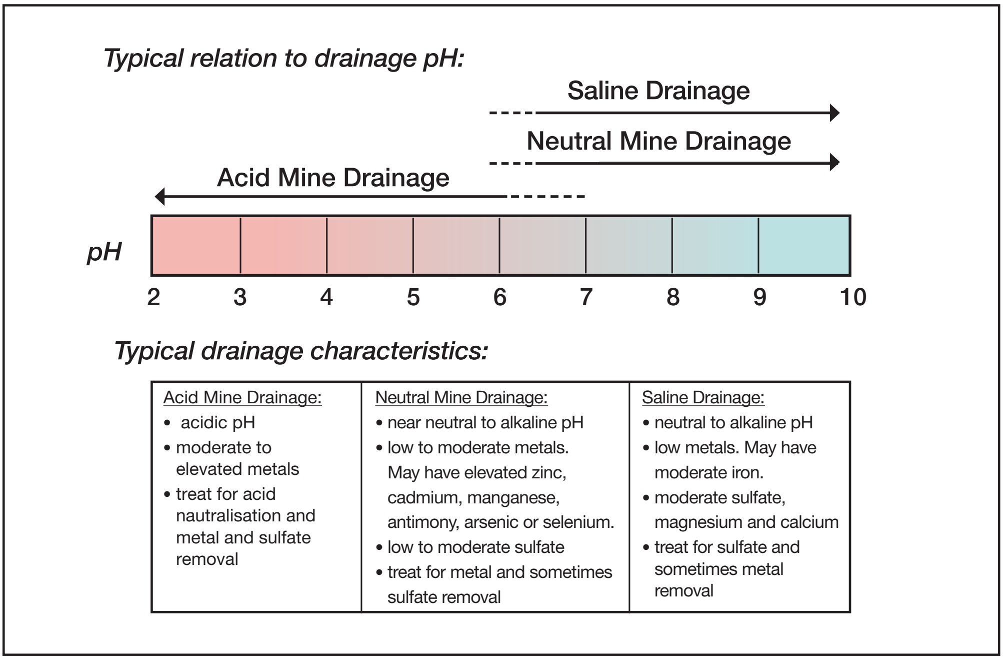
AMD has traditionally been referred to as 'acid mine drainage' or 'acid rock drainage' (ARD). In this handbook, the extended descriptor 'acid and metalliferous drainage' is used to implicitly recognise that not all problematic drainage from the oxidation of sulfide minerals is acidic (see below).

At some sites, near-neutral drainage containing elevated concentrations of major ions such as calcium, magnesium and sulfate and/or dissolved metals/metalloids (NMD and/or SD) can be just as difficult to manage and treat as acidic water. At these locations, either:

- the acid produced by the primary oxidation of sulfides has been neutralised by natural mineral assemblages or in some cases process chemicals, removing the major proportion of the dissolved metals/metalloids, but leaving a saline leachate that may contain elevated concentrations of those metals/metalloids that are soluble at higher pH values, or
- the saline and metal/metalloid-rich drainage has been produced by the oxidation of metal sulfides that do not generate net acidity.

The spectrum of AMD types is summarised in Figure 1.

Figure 1: Range of drainage types produced by oxidation of sulfide minerals



Source: Adapted from the GARD Guide; with permission, International Network for Acid Prevention.

Table 1 complements Figure 1 by summarising the range of pH values and concentrations of major ions and metals/metalloids found in AMD at a number of metalliferous mine sites in the Northern Territory. While the concentrations of metals are often higher in more acidic mine waters, that is not always the case.

2.2 Indicators of AMD

Visual indicators of AMD can include:

- red coloured or unnaturally clear water
- orange–brown iron oxide precipitates in drainage lines (Figure 2)
- dense coatings of green algae filaments on the bed of a stream with unnaturally clear water (Figure 3)
- the death of fish or other aquatic organisms on mixing AMD with receiving water
- precipitate formation on mixing AMD with groundwater inputs into stream channels or on mixing AMD with receiving surface waters, such as at stream junctions (Figure 4)
- poor productivity of revegetated areas (such as WRD covers)
- vegetation dieback or soil scalds (such as bare areas)
- deposits of white or coloured salts forming along the banks of stream channels and along the toes of WRDs during the dry season (Figure 5).

Table 1: Compositions of AMD from six metalliferous mine sites in the Northern Territory

ANALYTE	UNITS	MINE A	MINE B	MINE C	MINE D	MINE E	MINE F
pH		6.6	6.0	4.2	3.7	3.2	2.1
EC	µS/cm	4118	980	1680	4080	11290	–
Iron	mg/L	<0.002	29	0.1	–	550	359
Aluminium	mg/L	0.014	0.32	7.5	217	620	116
Sulfate	mg/L	2650	590	890	4840	9080	2230
Arsenic	µg/L	4	350	83	–	250	11000
Cadmium	µg/L	140	48	–	5570	40	24
Cobalt	µg/L	15	135	–	4240	8000	258
Copper	µg/L	7	0.9	1450	6280	11000	6400
Manganese	µg/L	2560	2190	–	142000	74100	1910
Nickel	µg/L	572	627	–	4310	12800	810
Zinc	µg/L	42600	1310	10390	131000	5330	6940

Source: Team NT (2004).

Figure 2: Orange coatings on rocks and precipitates forming in a drainage line downstream of an AMD source

Source: R Jung, CSIRO.

Figure 3: Algal filaments on stream bed

Source: D Jones.

Figure 4: Precipitate of copper carbonate produced by neutralisation by alkaline groundwater inflows of AMD containing copper

Source: D Jones.

Figure 5: White magnesium sulfate-rich salts forming downgradient from a covered and revegetated sulfidic WRD

Source: Northern Territory Department of Mines and Energy.

Caution needs to be exercised in semi-arid and arid climate zones, where many of Australia's mines are located. In those areas, surface discharges of acidic water are ephemeral and the soil environment under, and closely adjacent to, waste storage facilities (WRDs and tailings dams) is often the primary receptor for run-off and shallow groundwater discharge containing AMD. This may also be the case at mineral sand deposits that are underlain by highly permeable sandy sediments that do not allow water to pool at the land surface.

Under these conditions, the occurrence of sulfide mineral oxidation products on dump surfaces and changes in groundwater quality provide the clearest indicators that AMD is being produced. Although changes in pH may not be initially observed in groundwater beneath mine sites due to the buffering effects of minerals in the aquifer, other readily measured water-quality changes in monitoring bores that indicate that groundwater is being affected by AMD may include:

- progressive increases in sulfate concentrations or the sulfate to chloride mass-ratio (SO4/Cl) in groundwater over time
- progressive reduction in the alkalinity of groundwater over time (which can be readily measured in the field using an alkalinity test kit)
- progressive increases in the total titratable acidity of groundwater samples over time (which can be readily measured in the field using an acidity test kit).

Under conditions in which aquifers have a limited buffering capacity, seepage of AMD can cause groundwater to become acidic and to contain elevated concentrations of metals. This can pose health risks to groundwater users and affect groundwater-dependent ecosystems where the watertable is shallow or where groundwater discharge takes place. Measurements of the alkalinity and pH of groundwater can indicate whether there is a significant risk that groundwater at a mine site will acidify if affected by AMD.[9]

9 Additional guidance on this topic is provided at the website, http://www.der.wa.gov.au/your-environment/acid-sulfate-soils.

A key indicator of future AMD risk is the occurrence of sulfide minerals that will be exposed to air and water. The most common acid-generating sulfide minerals are iron (Fe)-based and include pyrite and marcasite (FeS_2), pyrrhotite (FeS), chalcopyrite ($CuFeS_2$) and arsenopyrite (FeAsS). Not all sulfide minerals (for example, galena (PbS) and sphalerite (ZnS)) are acid-generating during oxidation, but most have the capacity to release metals on oxidation and/or on exposure to acidic water. The acidity contributions of sulfide minerals can be calculated using a spreadsheet-based tool called ABATES.[10] ABATES uses the stoichiometry of oxidation reactions (provided for reference in a worksheet table) to calculate the overall production of acidity for a given sulfide phase.

Some sulfide minerals, such as pyrrhotite, behave in a significantly different manner from pyrite. Although they are fast-reacting following exposure to oxygen, they may generate only a fraction of the acidity generated by pyrite oxidation (Robertson et al. 2015; Schumann et al. 2015).

Situations in which reactive sulfides can be routinely exposed to air and water include the walls of open pits, WRDs, ore stockpiles, TSFs, pits, underground mines, heap leach piles, and roadways and embankments constructed from waste rock (Section 2.4). Leading practice AMD management involves implementing strategies to minimise the interaction between reactive sulfides and air, water, or both.

2.3 Production of AMD

The generation of acid (H^+) occurs typically when iron sulfide minerals are exposed to both oxygen (from air) and water. This process can be strongly catalysed by bacterial activity under the right conditions (acid pH, availability of nutrients and oxygen). The complete oxidation of pyrite to produce sulfuric acid and an orange precipitate, ferric hydroxide ($Fe(OH)_3$), is provided as an example in Reaction 1:

$$FeS_2 + 3.75\,O_2 + 3.5\,H_2O \Leftrightarrow Fe(OH)_{3\,(s)} + 2\,SO_4^{2-} + 4\,H^+ \qquad \textbf{\textit{(Reaction 1)}}$$

Iron sulfide + Oxygen + Water ⇔ Ferric hydroxide (orange precipitate) + Sulfate + Acid

Two key processes are involved in the generation of acid (H^+) from pyrite:

- Oxidation of sulfide (S_2^{2-}) to sulfate (SO_4^{2-}).
- Oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) and subsequent precipitation of ferric hydroxide.

10 Earth Systems, Acid-Base Accounting Tool, v. 1.4, http://earthsystems.com.au/technologies/acid-base-accounting-tool/.

These processes are represented by the following three reactions:[11]

$FeS_{2\,(s)} + 3½\,O_{2\,(g)} + H_2O_{(aq)}$	⇔ $Fe^{2+}_{(aq)} + 2SO_4^{2-}{}_{(aq)} + 2H+_{(aq)}$	***(Reaction 1a)***
Iron sulfide + Oxygen + Water	⇔ *Ferrous iron + Sulfate + Acid*	
$Fe^{2+}_{(aq)} + ¼\,O_{2\,(g)} + H^+_{(aq)}$	⇔ $Fe^{3+}_{(aq)} + ½\,H_2O_{(aq)}$	***(Reaction 1b)***
Ferrous iron + Oxygen + Acid	⇔ *Ferric iron + Water*	
$Fe^{3+}_{(aq)} + 3H_2O_{(aq)}$	⇔ $Fe(OH)_{3\,(s)} + 3H^+_{(aq)}$	***(Reaction 1c)***
Ferric iron + Water	⇔ *Ferric hydroxide + Acid*	
	(orange precipitate)	

Once Reaction 1a above has occurred, it is difficult to avoid oxidation of aqueous ferrous iron to ferric iron and subsequent precipitation of ferric hydroxide (Reactions 1b and 1c) if sufficient oxygen is present and the pH is high enough. The precipitation stage (Reaction 1c) is acid-generating.

The presence of dissolved ferric iron (Fe^{3+}) – typically when the pH is less than 3.5 – can lead to significant acceleration of the acid generation process, as the ferric iron reacts directly with fresh iron sulfide minerals (Reaction 2). This process explains why sulfide oxidation can still occur to a limited extent at depth in a waste stockpile in the absence of oxygen, as dissolved ferric iron percolates downwards in solution through preferred flow pathways.

$FeS_{2\,(s)} + 14\,Fe^{3+}_{(aq)} + 8\,H_2O_{(aq)}$	⇔ $15\,Fe^{2+} + 2\,SO_4^{2-} + 16\,H^+$	***(Reaction 2)***
Pyrite + Ferric iron + Water	⇔ *Ferrous iron + Sulfate + Acid*	

As acid water migrates through a site (for example, through waste rock, ore stockpiles, pit wall rock or groundwater) it will react with other minerals in the surrounding soil or rock material and may dissolve a range of metals and salts. The acid will be progressively neutralised by the minerals it dissolves. This has the effect of increasing the pH.

However, the neutralisation of acid occurs at the expense of increasing concentration of metals (metal acidity) and salts in the resulting drainage. As the acidic water contacts common aluminosilicate (such as chlorite, muscovite) or other sulfide minerals, partial dissolution of the minerals and neutralisation of acidity results. While increases in pH are desirable, the associated rise in dissolved metal and dissolved salt concentrations is not. Ultimately if the pH rises high enough, pH neutral mine drainage results. Such drainage can contain elevated concentrations of major ions (salinity) and metals (for example, Cd, Co, Mn, Mo, Ni, Zn) and metalloids (B, As, and Se) that remain soluble under these pH conditions (that is, NMD).

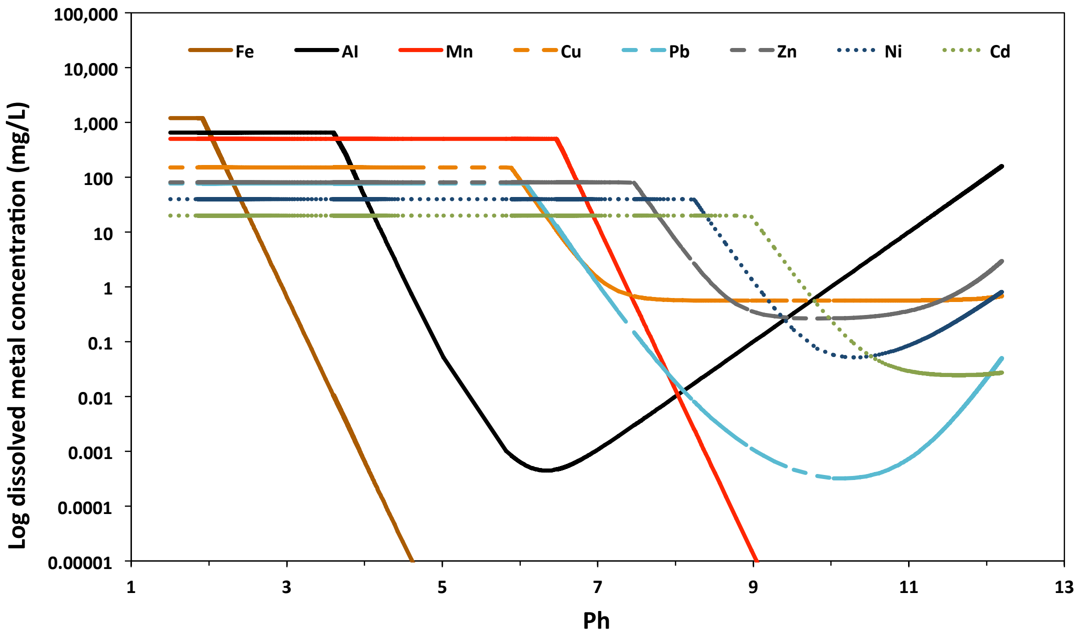
At many sites, there is insufficient natural neutralising capacity within geological materials to raise the pH of drainage to near-neutral levels. Consequently, drainage characterised by low pH and elevated concentrations of metals/metalloids is the most common form of AMD at mine sites.

11 These reactions, when combined, are equivalent to Reaction 1.

2.3.1 Neutral metalliferous drainage

In some cases, the acid generated is completely neutralised by the dissolution of common carbonate minerals such as calcite, dolomite, ankerite and magnesite. Since the solubility of many metals is pH-dependent, the neutralisation process can lead to the precipitation of metals such as aluminium, copper and lead, and thus their removal from the drainage. However, at near-neutral pH, concentrations of elements such as, As, Cd, Mn, Ni and Zn can remain elevated, resulting in NMD. As is the case with AMD, NMD can also contain high (sulfate) salinity. Figure 6 provides an example of the effect of pH on the solubility of metals in a real sample of AMD. Note that the precise effect of pH depends on the starting composition of the AMD, as there are secondary processes such as co-precipitation and adsorption that also control the solubilities of metals and metalloids (McLean & Bledsoe 1992).

Figure 6: Effect of pH (titration with calcium hydroxide) on dissolved metal concentrations in AMD-affected water from a base metal mine



Note: A logarithmic concentration scale is used to capture the full range of behaviours.

Source: Earth Systems.

NMD is less common than acid drainage, due to the requirement for specific sulfide minerals (for example, sphalerite and galena) and/or a local excess of carbonate minerals that neutralise the acid but only partially precipitate environmentally important metals/metalloids.

2.3.2 Saline drainage

In situations in which acidic drainage is completely neutralised by local carbonate resources and the resulting drainage contains no significant residual concentrations of metals, the potential remains for drainage to be a salinity issue by virtue of elevated concentrations of magnesium, calcium and sulfate. The sulfate salinity of the neutralised drainage primarily depends on the relative proportions of calcium and magnesium in the neutralising carbonate materials. If magnesium minerals are the dominant components of the neutralising material, high salinity is more likely to be an issue due to the high solubility of magnesium sulfate. Conversely, if calcium is the dominant component, the precipitation of the much less soluble gypsum ($CaSO_4.2H_2O$) may result in lower salinity levels.

SD generated specifically as a result of sulfide oxidation is relatively rare compared with AMD and NMD. Nevertheless, sulfate salinity can be an important early indicator of developing AMD at mine sites and may require similar management strategies (that is, control of sulfide oxidation).

SD can also be the result of the estuarine provenance of coal measures, in particular. Marine-derived sodium chloride salinity is a feature of many mines in the Bowen Basin in Queensland, and is not the product of sulfide oxidation. However, in common with SD produced from sulfide sources, this salinity still needs to be managed in the context of cumulative impacts on potentially affected river catchments (EHP 2013).

2.4 Self-heating and auto-ignition

If self-heating occurs, the rate of production of soluble oxidation products can be greatly accelerated compared with that expected at ambient temperatures, and additional reaction products may be produced that will contribute further to the solute load being transported by water infiltrating through the material.

The oxidation of iron sulfides is an exothermic (heat-generating) process. The faster the rate of reaction, the greater the rate of production of heat. In certain circumstances, the rate of heat production by an oxidising mass of sulfide-containing material (waste, ore or concentrate) can exceed the rate of heat loss. If this happens, the temperature can rise such that the mass ignites and sulfur dioxide gas is liberated to the atmosphere.

This process can be exacerbated by the presence of a carbon source (for example, graphitic shales or coal) that provides additional fuel. This condition occurs in fine-grained masses of pyrite present in the McRae shale at a number of iron ore mines in the Pilbara region of Western Australia[12] and some coal seams in the Hunter and Lithgow regions of New South Wales. In coalmines, this process is widely known as 'spontaneous combustion'. Tailings containing appreciable amounts of pyrrhotite can also be susceptible to self-heating if that material is allowed to dry out in contact with the atmosphere.

The second (less common) type of heat-generation is caused by a galvanic process (Payant et al. 2011). In this case, a combination of different (usually fine-grained) sulfides present in the ore body act like a shorted battery, producing heat. If the rate of galvanic heating is greater than the rate of heat loss, then self-heating can occur. The combinations of sulfides most likely to undergo self-heating by this galvanic mechanism are pyrite-galena, and pyrite-sphalerite occurring in massive sulfide ore bodies.

2.5 Acid and acidity

The concentration of free hydrogen ions (acid) in solution is measured by pH. However, the total acidity of mine drainage consists of both hydrogen ion and mineral (or latent) acidity. Mineral or latent acidity refers to the potential concentration of hydrogen ions that could be generated by the precipitation of various

12 See case study in Section 5 and Waters and O'Kane (2003).

metal hydroxides by oxidation or neutralisation. For example, Reaction 1c above illustrates the latent acidity that is released as ferric iron is converted to ferric hydroxide.

In general, acidity increases as pH decreases, but there is not always a direct relationship between total acidity and pH. In this context, it is possible to have AMD with an elevated acidity due to the presence of dissolved metals, but with neutral pH values. It is therefore important to quantify the contributions of both hydrogen ion concentrations (acid) and dissolved metal contributions (latent acidity) in order to determine the total acidity (acid + latent acidity) that is present. Acidity is generally expressed as the mass of calcium carbonate ($CaCO_3$) equivalent per unit volume (for example, mg $CaCO_3$/litre) needed to neutralise the acid.

The concentration of free acid can be easily measured in the field using a calibrated pH probe. Total acidity can also be measured in the field or laboratory (for example, by titration with sodium hydroxide solution) or using inexpensive commercially available test kits. Acidity can be estimated from water-quality data using a formula such as Equation 1 below, which is broadly suitable for coalmine drainage.[13] If more comprehensive input water-quality data is available, shareware such as AMDTreat or ABATES can be used to obtain more accurate estimates of total acidity (see Glossary).

$$\text{Acidity } (mg/L\ CaCO_3) = 50 \times \{3 \times [\text{Total Soluble Fe}] / 56 + 3 \times [Al^{3+}] / 27 + 2 \times [Mn^{2+}] / 55 + 1000 \times 10^{-(pH)}\} \qquad \textbf{(Equation 1)}$$

Note: [ ] denotes concentration, mg/L

2.6 Acidity loads

Acidity load is the product of the total acidity (acid + latent acidity) and flow rate (or volume) and is expressed as 'mass of $CaCO_3$ equivalent per unit time' (or mass $CaCO_3$ equivalent for a given volume of water). If flow rate or volume data is available, then the measured or estimated acidity values can be converted into acidity load as shown in Equation 2, or using the ABATES or AMDTreat shareware.

$$\text{Acidity load } (tonnes\ CaCO_3/day) = 10^{-9} \times 86{,}400\ (conversion\ factor) \times \text{Flow rate } (L/s) \times \text{Acidity } (mg/L\ CaCO_3) \qquad \textbf{(Equation 2a)}$$

or...

$$\text{Acidity load } (tonnes\ CaCO_3) = 10^{-9}\ (conversion\ factor) \times \text{Volume } (L) \times \text{Acidity } (mg/L\ CaCO_3) \qquad \textbf{(Equation 2b)}$$

Acidity load is the principal measure of potential AMD impact at a mine site. Consequently, AMD management planning needs to focus on those areas of the site that can potentially release the greatest acidity load.

Typically for an operating mine site, waste that is exposed to air and water (for example, uncovered WRDs, ore stockpiles and pit walls) provides the largest potential source of acid load. Tailings are unlikely to be a

13 Equation 1 is applicable to sites such as coalmines where Fe, Al and Mn represent the dominant components of acidity.

source of AMD during the operating life of the mine because tailings circuits are typically operated at elevated pH, deposited tailings can be inundated with fresh tailings, and the bulk of contained sulfidic material is generally maintained under water-saturated conditions. However, tailings can be an operational source of NMD and/or SD, either from sulfate or from other major ions and metals/metalloids in the ore or in the process fluids, including forms of cyanide.[14]

Following the decommissioning of a TSF, the watertable may drop and the surface layers of the tailings can desaturate, with resultant exposure to oxygen and the migration of an oxidation front into the tailings over time. Thus, the potential for tailings to be a future source of acidity load must be addressed as part of mine closure planning.

2.7 Sources of AMD

To assess the acidity load likely to be produced at the site, it is essential to have a good understanding of the local geology, mineralogy and geochemical characteristics (Section 4) of all materials that are exposed, handled or processed during mining operations. The exposure of unconsolidated materials (such as waste rock and tailings) or bedrock (such as the walls of a pit or underground workings) to air and water has the potential to generate AMD.

Carbonates are generally the only alkaline minerals that can occur naturally in sufficient quantities to effectively neutralise acidity and decrease metal concentrations over the short term. Silicate and aluminosilicate minerals (for example, biotite and chlorite) have significant intrinsic neutralisation capacity, but their slow rates of reaction may render them largely ineffective in the short term. Reliance on the presence of aluminosilicate minerals to neutralise acidity, for example, is not generally recommended because of the risk that elevated concentrations of soluble Al (produced by decomposition of the aluminosilicate at acid pH) will be released into surface water run-off and groundwater.

However, not all sulfide minerals produce acidic drainage; nor are all carbonate minerals effective neutralisers of acidity. As noted above, low-quality drainage may also persist at near-neutral pH due to elevated concentrations of metals/metalloids (see Section 2.3.1).

Assessments of the geologic lithologies of the ore body and its surrounds and of the process streams in the mineral extraction circuit are essential to underpin the development of effective management strategies for handling sulfidic mining wastes. It may be possible, for example, to implement strategies such as segregation, selective placement, co-disposal or blending and/or encapsulation as the waste material is being placed during the operational life of the mine (Section 6). Such proactive management can substantially reduce the future AMD legacy during the operating period, when it can be done most cost-effectively.

The overall goal of AMD management strategies should be to minimise or, wherever possible, eliminate the exposure to air and/or water of reactive sulfidic material, now and into the future. This can only be achieved if site planners and managers have a thorough understanding of the AMD risks of the materials disturbed (or exposed to air) as a result of mining and integrate appropriate management and mitigation strategies into the mine plan.

14 See the *Tailings management* (DIIS 2016e) and *Cyanide management* (DRET 2008) handbooks in this series.

2.7.1 Waste rock dumps

WRDs are generally placed above ground, where the bulk of the material remains unsaturated, containing about 5–10% water irrespective of the climatic setting. In the semi-arid climate in which many Australian mines are located, the waste rock may only wet-up to about 5–10% water. In the wetter climates of western Tasmania and Northern Australia, the waste rock may wet-up to a greater degree, particularly in Northern Australia during an extreme wet season, resulting in the formation of a perched watertable or watertable mounding within the WRD and subsequent production of seepage from the WRD.

Alternatively, waste rock may be co-disposed with tailings or be returned to a pit or mined-out underground workings, where it may be partially or wholly inundated. An intermediate situation is provided by a valley-fill waste rock dump, where the base and lateral margin of the waste may be exposed to more water. In each of these cases, any unsaturated zones of sulfidic waste rock are susceptible to oxidation. AMD may seep from the toe of the WRD or migrate beneath it into groundwater. This can have adverse impacts on water quality during operations and after closure.

The processes controlling AMD generation and transport for a WRD are represented schematically in Figure 7. The behaviour of a given system depends not only on the materials' mineralogical and geochemical properties, but also on physical properties such as porosity, grain size (surface area), diffusion coefficient, gas permeability, hydraulic conductivity and thermal conductivity. Many of these parameters change through time as a result of in situ physical and chemical processes. The time course of accumulation of sulfide oxidation products and their subsequent rate of leaching is illustrated in Figure 8.

Geographical location determines the importance of driving factors such as rainfall, temperature, prevailing winds, vegetation and the extent of seasonal variation in each of those factors.

Figure 7: Schematic representation of AMD generation and pollutant migration from a WRD

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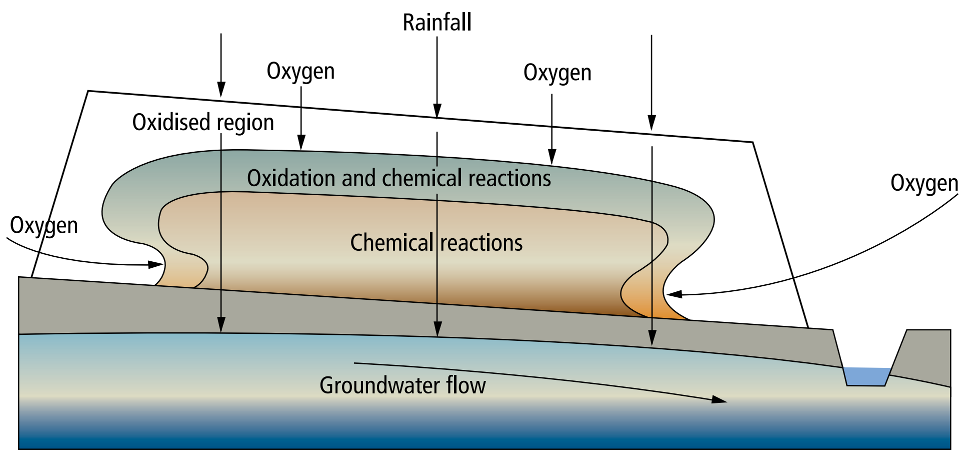
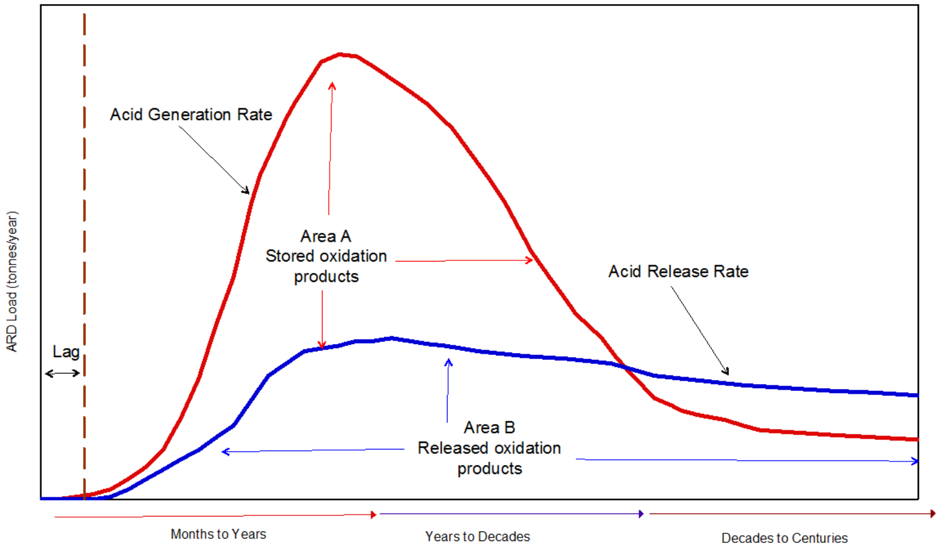


Source: Ritchie (1994).

Figure 8: Schematic of the time evolution of AMD oxidation product generation and release from a waste storage facility



Source: Miller (2014).

2.7.2 Ore stockpiles

Run-of-mine (ROM) ore stockpiles are physically similar to, albeit typically much smaller than, WRDs. However, sulfide concentrations can be higher than for the waste. Since the average exposure time of the ore is generally relatively short as material is reclaimed for processing, there is a much shorter period of time available for initiation and evolution of the AMD process. However, in specific circumstances where the ore is highly reactive, or prone to self-heating, special attention must be paid to minimising the residence time of ore in the stockpile (from both resource recovery and AMD management perspectives).

2.7.3 Low-grade ore stockpiles

In contrast to run-of-mine ore stockpiles, low-grade ore stockpiles may be present for decades, depending on metal prices. Thus they are potentially long-term AMD sources. Attention needs to paid to the effective interim management (for example, covering with a geomembrane to prevent infiltration of rainfall and limit ingress of oxygen) and method of construction of such stockpiles (see Section 6), as they can contribute substantial amounts of poor-quality water to the site's water management circuit. In addition to water-quality issues, the oxidation of contained sulfides may result in substantial reduction in the grade of the stockpiled ore, change the characteristics of the material so that it can no longer be economically processed, or both. If a low-grade ore stockpile is not processed at the end of the mine's life, that material will need to be managed appropriately (for example, by incorporation in a covered WRD) to minimise future AMD risk.

2.7.4 Tailings storage facilities and tailings dams

Tailings produced during ore processing are typically placed in a TSF in slurry form. Sulfidic tailings can be a significant future source of AMD due to their fine particle size.

Seepage from TSFs is generally to groundwater, while surface water is often re-used on site (during operations) or may be discharged via a spillway structure (after closure). AMD generated in TSFs can adversely affect both surface water and groundwater quality, on and off site. The magnitude of this issue must be minimised by implementing appropriate rehabilitation strategies.[15]

2.7.5 Pits or open cuts

Wall rock in pits or open cuts[16] may contain sulfidic minerals that have the potential to generate AMD. The extent to which groundwater is lowered around a pit during mining can affect the amount of sulfidic material exposed to air and the resulting acidity load that is generated. AMD from the wall rock may seep into the pit or the local groundwater system. This can affect the quality of water pumped from pit sumps or groundwater bores during operations. AMD produced from the oxidation of wall rocks and in catchment drainage lines reporting to the pit can also have significant long-term impacts on pit water quality after mine closure (Section 6).

In some cases, pits may be partially or completely backfilled with waste rock. The volume of waste rock remaining in the variably unsaturated zone above the recovered watertable in the pit can be a substantial source of AMD to the groundwater. This potential source needs to be considered to ensure that it will not compromise closure objectives.

2.7.6 Underground mines

Issues associated with wall rock in underground workings are similar to those for pits. Sulfides exposed to air as a result of dewatering are a potential source of AMD. This can affect the quality of water that is collected underground and re-used, treated or discharged during operations. At the completion of mining, flooding of the underground workings can prevent further AMD generation. However, the mined-out voids may already contain poor-quality water as a result of sulfide oxidation before and during the flooding process and, as noted in Section 2.3, sulfide oxidation can still occur in the absence of oxygen if dissolved ferric iron is present in solution. A particular issue that needs to be addressed in this context is the potential for downgradient discharge of such water, particularly if the mine is in mountainous terrain.[17]

15 Further discussion about management options for sulfidic tailings is in Section 6.2 of this handbook. The *Tailings management* handbook in this series (DIIS 2016e) contains extensive information about the various strategies that can be used to place and contain tailings to minimise future risk to the environment.

16 Surface mines are commonly referred to as pits or open cuts. The term 'pits' is used throughout this handbook for consistency.

17 See Section 6.5 for further discussion about this issue.

2.7.7 Heap and dump leach piles

Heap leaching of low-grade base metal sulfide ores is gaining favour as this technology matures and the size of operations increases. At the time of decommissioning, the remaining sulfides present in the piles of spent material are a potential long-term source of AMD. The presence of an engineered low-permeability liner underneath a leach pile allows all drainage to be collected during decommissioning and after closure, facilitating the management of the AMD. However, in the case of dump bioleach operations, where no effective liner exists, the generation and transport of AMD from the spent dump material to the environment may be similar to that from sulfidic WRDs. There is a paucity of published information on the successful decommissioning of sulfidic heap leach operations in Australia.

2.7.8 Construction materials

An often overlooked source of AMD onsite is the materials that are used for the construction of pads, roadways and embankments. Indeed, the typically high-strength characteristics of primary (unoxidised) rock are often seen as ideal for construction purposes. There are many examples in which the use of sulfide-containing rock for construction has resulted in disseminated sources of AMD across mine sites. This can cause a significant operational water management issue and is especially likely to increase the costs of cleaning up and rehabilitating the site at the end of the mine's life. The geochemical characteristics of all material proposed to be used for construction (including for use in rehabilitation) should be assessed in advance to determine whether the material is geochemically fit for its intended purpose.

2.8 Factors influencing the extent of AMD generation

Many factors can influence the generation of AMD and its transport and thus the ultimate concentrations and load of pollutants at a point downstream of a source. The primary factor driving AMD generation is the oxidation of sulfide minerals. The chemistry of the AMD changes as the initially produced solution of oxidation products moves through the system and interacts with other geologic materials.

Factors affecting sulfide oxidation and subsequent transformations of oxidation products include:

- the concentration, distribution, mineralogy and physical form of the sulfide minerals
- the rate of supply of oxygen from the atmosphere to the reaction sites by advection, convection and/or diffusion
- the chemical composition of pore water in contact with the reaction sites, including its oxygen content, pH and ferrous/ferric ion ratio
- the temperature at the reaction sites
- the water content and degree of saturation at the reaction sites
- the microbial ecology of mineral surfaces.

Factors affecting the subsequent transformations and load of the initially produced AMD include:

- the concentration, distribution, mineralogy and physical form of neutralising and other reactive minerals
- the flow rate and flow paths of water
- the chemical composition of pore water.

3.0 OVERVIEW OF LEADING PRACTICE FOR PREVENTING AMD

Key messages

- The characterisation of AMD risk should begin during exploration and continue through pre-feasibility, feasibility and operations phases.
- Mining should only proceed if investigations conducted during the feasibility phase demonstrate that AMD can be managed from technical, economic and reputational perspectives, including during the closure phase and beyond.
- An AMD management plan for site operations and closure should be developed during the feasibility phase and be implemented and updated during the operations phase in response to increased knowledge and/or change in the project scope.
- Monitoring data should be used to periodically assess the ongoing performance of the initially implemented AMD management strategy, and changes should be made if the required performance is not being achieved.
- The effectiveness of strategies being used for operational management and proposed to be used for the post-closure minimisation of AMD risk should be tested by numerical modelling and validated by in situ monitoring and results from field trials well in advance of closure to confirm their effective performance.

The focus of this section is on leading practice mine materials management to prevent the occurrence of AMD. However, mining invariably takes place in a social and regulatory landscape comprising potentially affected communities, other stakeholder groups and government agencies. The way these interactions are addressed, managed and communicated is also part of modern leading practice.[18]

The development of a mine proceeds through a number of phases, commonly described as exploration; pre-feasibility; feasibility; development and construction; operations; and closure. The understanding needed to assess and manage the potential for AMD should be gained as early as possible in the life of a project and further refined as it proceeds.[19]

Such are the costs involved that a high risk of AMD can potentially prevent a project from progressing beyond the feasibility phase and can have significant implications for project performance if the operation proceeds. The sequence of leading practice actions needed to minimise AMD risk throughout the mine life cycle is summarised in Table 2 and discussed further in Miller (2014).

18 These aspects are specifically addressed in Section 10.

19 Methods for predicting the likely magnitude of an AMD issue are presented in Section 4.

3.1 Pre-mining

Leading practice decision-making for AMD begins with an understanding of the site geology, geomorphology and geochemistry. An indication of the potential for AMD for a given mineral deposit type can be obtained by reference to geo-environmental models that have been developed based on extensive past experience with different types of ore bodies (USGS 2002). Geo-environmental models provide a variety of information about the geological and geochemical setting of mineral deposits, mining and mineral processing technology as they relate to the generation of mine waste, and the environmental behaviour of mineral deposits in the broadest sense.

Few mineral resources are homogeneous, and it is necessary to develop an increasing understanding of them and their host rocks in the pre-mining phases to identify the likely materials that will be exposed and the constraints this will place on the mining operation (Scott et al. 1997). To develop this understanding, it is important during the pre-mining phases that the project team, including geologists, mine planners, environmental scientists and AMD experts, ensures that an adequate geological and geochemical database is compiled to clarify baseline conditions and to estimate the risk of AMD. The requirements for a progressive test work program are summarised in Table 2. Methods for identifying and predicting AMD are provided in Section 4.

As shown in Table 2, operational AMD and closure plans that properly address the identified risks of AMD must be developed and costs estimated for the site during the feasibility phase. Great care should be taken in using net present value (NPV) calculations to decide between options, as the conclusion from this type of analysis is often that it is better to defer the cost of AMD management to the end of the mine's life. However, the consequence of not implementing an initially more costly, but effective, control strategy could be an AMD problem that has escalated during operations to the extent that there may no longer be an economically viable solution. Decoupled NPV (DNPV) may be a more appropriate method to use for this purpose (Espinoza & Morris 2013).

If there is a high risk of a substantial AMD issue, detailed design and higher confidence cost estimates will be needed, as the costs involved could seriously affect the economics of the project. The closure plan developed in the feasibility phase will remain a 'living document' and be subject to regular reviews and updates to address changing mine conditions, community expectations, stakeholder inputs and the emergence of new, more cost-effective AMD management methods as the mine proceeds.

3.2 Operations

During operations, the management of acid-generating materials can be a complex process involving a number of different strategies depending on the characteristics of the ore and waste, local climate and landscape (Section 6). It is essential that the preliminary AMD management plan developed during the feasibility phase be revised and updated in response to increased knowledge (such as more detailed waste and ore block models) and changing conditions (for example, changes in the distribution of AMD waste types or alterations to the project scope) during mining operations.

Day-to-day management of AMD risk can involve the identification, characterisation, scheduling, transport, segregation, selective placement, co-disposal and sometimes blending of sulfidic and carbonate-bearing materials, as well as an appropriate level of monitoring. This process must be integrated with mine planning and mine operations and can be time consuming and labour intensive. However, the consequences of not doing it can drive up project costs into the future if unforeseen mitigation works and ongoing treatment are needed.

Material for construction and rehabilitation most often comes from the pit and the footprints of WRDs, TSFs and process plant. All these materials should be characterised chemically and physically and incorporated into the material balance and mined material schedule. Excluding them from the material balance and planning process is one of the root causes of poor rehabilitation outcomes (for example, soil is lost or there is no material stockpiled for the construction of covers or for rock armouring of rehabilitated slopes and drains).

Successful implementation of the AMD management plan requires serious commitment by mine management and staff and positive collaboration across all relevant sections of the business unit. For sites with an identified AMD risk, leading practice recommends the establishment of a dedicated AMD management group chaired by the general manager of operations and comprising senior members of the mining, mine planning, waste management (rock and tailings), water management, and environmental management teams, with external expert input as required.

Regular in-house performance evaluation (Section 9) is an important function of the AMD management group, which is charged with ensuring the currency and effectiveness of the AMD management strategy. Meetings of the AMD management group should be scheduled at least quarterly and include special meetings convened as necessary to address:

- changed circumstances that could reduce the effectiveness of the current AMD management plan, and/or
- monitoring results that indicate that the AMD management strategy is not performing as expected.

Performance evaluation should provide feedback for regular updates of the closure plan.[20] If initially proposed management strategies are found to be incapable of meeting long-term AMD performance objectives, alternatives should be developed and implemented while suitable equipment and experienced personnel are still available onsite, so that the alternative approach can be implemented at lowest cost. New developments in AMD abatement technology should be considered for evaluation and testing as part of the evolving AMD management and closure planning process. Regulators and community groups also need to be consulted during closure planning so that their evolving needs can be considered.

20 See the *Evaluating performance: monitoring and auditing* (DIIS 2016c) and *Mine closure* (DIIS 2016a) handbooks in this series.

Table 2: Leading practice actions to minimise AMD risk through the mine life cycle

See Glossary for definitions of acronyms used in this table and Section 4 for details of the mine waste characterisation methods that are mentioned.

Mine phase	**Category**	**Actions** (refer also to sections 5 and 8)	**Comments**
Exploration: reconnaissance	Visual	Evidence of AMD (e.g. sulfidic minerals, seepage stains, precipitates of iron and aluminium hydroxides).	Even if there is no visual indication of potential for AMD at this stage, further assessment is still necessary and should be extended during prospect testing (see below).
	Water quality	Analysis of baseline surface water and groundwater levels and quality.	
Exploration: prospect testing	Visual	Evidence of the potential for AMD (as above).	Even If there is no indication of potential AMD risk at this time, more detailed characterisation is still required during resource definition (see below) as problematic material might not have been intersected by initial drilling. Baseline water-quality data is collected to inform the subsequent development of water-discharge criteria during operations and water-quality objectives and criteria for closure, as they will need to accommodate previous disturbances on the site and/or naturally occurring concentrations of metals and salts. The limited geochemical characterisation dataset available at this time may provide some indication of AMD potential but is unlikely to provide a robust statistical basis to evaluate AMD potential or risk.
	Water quality and hydrology	Determine baseline surface water quality and groundwater levels and quality.	
	Geochemistry (preliminary static test work)	Static geochemical characterisation test work, including analysis of sulfur and carbon in drill chips (if available) and outcropping lithologies (Section 4). If techniques such as handheld X-ray fluorescence (XRF) are used by exploration geologists for drill core assessment, sulfur and carbon should be included in the analysis suite. At least 3–5 representative samples should be tested for each identified key lithology/alteration type.	
Exploration: resource definition	Water quality and hydrology	As above.	If there is an indication of AMD potential, more focused investigations will be required in the next phases of project development. Even if there is no indication of potential AMD risk at this time, further characterisation will be needed at the feasibility stage as better coverage of ore body and waste zones is obtained. By the end of the resource definition phase, there should be enough information to provide an indication of the AMD potential of the ore body (high- and low-grade ore) with reasonable accuracy. The development of a preliminary ore block model may be possible if sufficient data is available (Section 4.4). More information will be needed in the next stage of project development for the adequate characterisation of waste rock and process tailings, since neither the boundaries of the mining operation nor the details of processing have been defined at this time.
	Geology/ mineralogy	Identification of geological/lithological types and mineral phases in the mineralised and waste rock categories using traditional methods. Drill core logged and carefully stored so that unaltered material is available for future more detailed geochemical analysis if necessary.	
	Geochemistry (more detailed static test work)	NAPP/NAG test work, including analysis of sulfur (as sulfide) and carbon (as carbonate) minerals in drill chips for different geological types and mineral phases. If techniques such as handheld XRF are used by exploration geologists for drill core assessment, sulfur and carbon should be included in the analysis suite. At least 5–10 representative samples should be tested for each key identified lithology/alteration type. Note that the number of identified lithotypes will probably increase in this phase owing to greater knowledge about the deposit. The initial scope of tests may be constrained but increase in complexity if AMD is identified as an issue.	

Mine phase	Category	Actions (refer also to sections 5 and 8)	Comments
	Geophysical	Methods such as induced polarisation/self-polarisation (IP/SP, to detect disseminated sulfides), magnetics and electromagnetics (EM, to detect massive sulfides) may be used to better define the extent of AMD potential.	
Pre-feasibility	Water quality and hydrology	Establish baseline water quality (acidity, alkalinity, metals/metalloids, major ions, including sulfate and salinity) and environmental values of surface water and groundwater resources potentially affected by the project. This information is a vital part of the environmental approvals process. Develop preliminary site water balance model.	Potential AMD impacts and associated management costs should be evaluated at least at a qualitative level for a range of mining, processing and closure options using defined risk/opportunity assessment processes and a technically robust multidisciplinary team. The selection of potential mining and mineral processing methods to minimise AMD generation from waste materials and tailings should be considered as part of the project options evaluation. Indicative capital costs and ongoing operating and closure costs to manage AMD must be factored into project financial analysis to help distinguish between options and ensure a proactive approach to AMD management. This enables a preferred project option to be selected and taken to the feasibility phase. The preferred option will be the basis for applications for approvals to conduct the project. It is possible that the outcome of the pre-feasibility assessment will be that the costs to manage the AMD risk are judged to be too high for an economically viable project. Consultation with regulators and the affected community is recommended during pre-feasibility to provide feedback to help select project options.
	Geology/ mineralogy	Start to develop and/or refine preliminary block model (Section 4.4), depending on the extent of understanding of deposit geology.	
	Geochemistry (detailed static test work; preliminary kinetic test work)	Up to several hundred (depending on the size of the resource and geological complexity) representative samples of high- and low-grade ore and waste rock may need to be collected for static geochemical test work. The exact number of samples depends on the geology of the deposit and the indicative AMD risk. If metallurgical test work is carried out for this step, samples of potential tailings material should also be tested. Sufficient data points to develop a preliminary block model with a reliable distribution of static geochemical data for ore, waste and wall rock. See Downing and Giroux (2014) in Section 4. Ideally, kinetic tests should be established for at least one or two representative samples for each key lithology/alteration type identified to have an actual or potential (e.g. uncertain category from static test work—Section 4) AMD risk.	
	Pit lake assessment	AMD characterisation of the lithologies that may be exposed in the pit walls. This will be at a preliminary level because the definition of the pit shell is not likely to be completed until towards the end of the feasibility phase. Water balance modelling to indicate maximum lake depth and risk of decant, and whether the lake will be a terminal sink or a through-flow environment.	At this stage, it will be known whether a pit lake is being considered as a closure option. Sufficient information must be gained to identify likely risks and develop management options.
	Rehabilitation concept	Conceptual designs for WRDs, TSFs and open pits that address the current status of knowledge.	Apply leading practice design concepts for waste management (Section 6).

Mine phase	Category	Actions (refer also to sections 5 and 8)	Comments
Feasibility	Water quality and hydrology	Update water-quality baseline as more time series data becomes available and develop and/or refine site water balance model.	All AMD data should be reviewed by a multidisciplinary team to develop a preliminary AMD management plan that is well integrated with the mining plan. AMD minimisation strategies should be developed to a sufficient level of detail to allow realistic costings to be made. The manager/custodian of the characterisation program and AMD management plan needs to be specified at this time to ensure internal accountability for the process. The estimated cost of implementing this plan should be input to the project costs. Great care should be taken in using NPV calculations to decide between options, as the conclusion from this type of analysis is often that it is better to defer the expenditure to the end of mine life. However, the consequence of not implementing an initially more costly, but effective, control strategy could be an AMD problem that escalates during operations to an extent that there may no longer be an economically viable solution. Decoupled NPV (DNPV) may be a more appropriate method to use (Espinoza & Morris (2013). Planning and operational approaches to managing AMD need to be well detailed and backed by sound technical arguments. Procedures for ongoing monitoring of AMD management performance need to be developed. A closure plan that is seen to be workable and convincing must be developed. Closure costs need to be factored into project financials, and need to be known to +/–30%, based on a refined conceptual model with detailed designs (see caution above on the use of NPV calculations). A robust technical approach and transparency in communications are key factors in expediting approvals and facilitating progress to the operations phase. A project concept that minimises impacts and provides a safe and stable landscape after closure is consistent with the objective of sustainable development. The preparation of an environmental impact assessment (EIA) is completed and the approvals process is normally begun just after the end of the feasibility phase. At this point, a preferred project has been selected but may still be subject to revision during final design work and as a result of feedback from stakeholders (community and regulators) during the approvals process. The key at this stage is having a good understanding of what stakeholders expect for the management of AMD risk.
	Geology/ mineralogy	Develop and continue to refine block models (Section 4.4). Use models to develop life-of-mine waste mining schedule and to optimise mine plan for selective handling, placement and management of waste. Inventories and scheduling of production of benign materials needed for interim containment of AMD material and for rehabilitation should also be included in the development of the life-of-mine schedule.	
	Geochemistry (detailed static and kinetic test work)	Review previous geochemical data for high- and low-grade ore, waste rock and tailings and any other materials that will be disturbed or formed as a result of mining operations. If required, improve the density of static geochemical test data to develop or refine block model, and conduct sufficient complementary analyses (e.g. NAPP and NAG) to cross-check geochemical data for key lithologies (Section 4). Establish and/or continue kinetic tests on waste rock and tailings samples. Kinetic tests using blends of different materials (e.g. acid-generating and acid-consuming materials) may be established to explore AMD management options. AMD issues and management techniques from other mines with similar location, climate and geology should be reviewed in detail if they are available. If there is still insufficient data to assess AMD potential and provide a convincing management plan for approval, additional sampling, test work and refinement of block models will be required.	

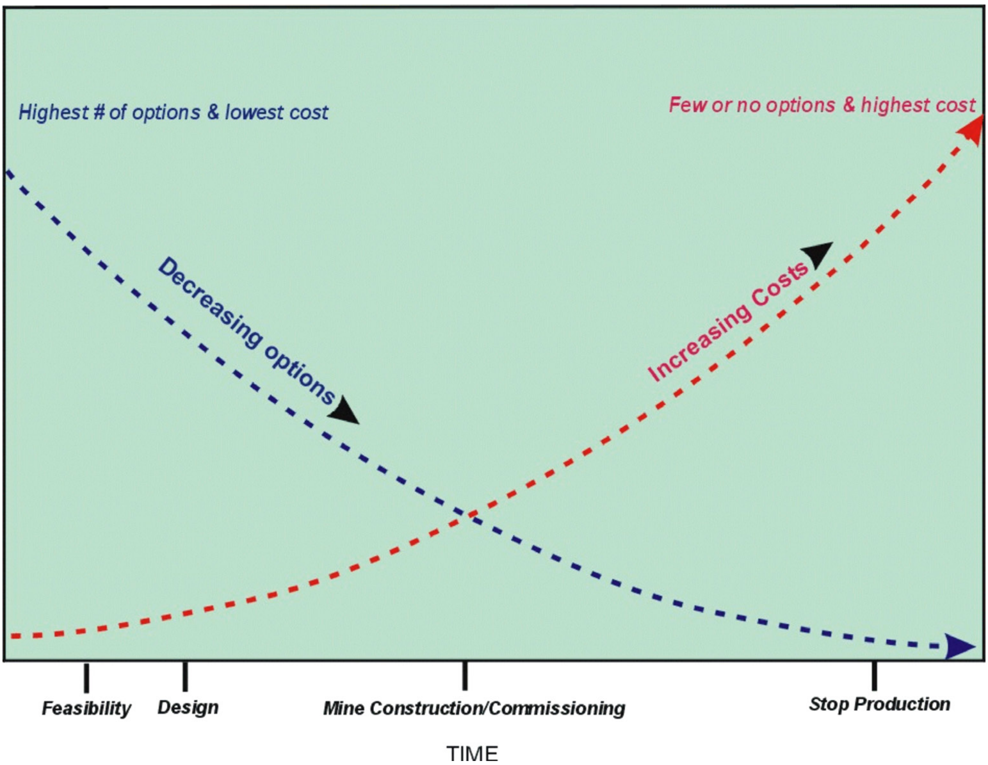
Mine phase	**Category**	**Actions** (refer also to sections 5 and 8)	**Comments**
	Operational AMD management, conceptual landform design and rehabilitation strategy	Waste encapsulation strategy and landform design (dump and dam design, covers or backfilling of voids, walk-away water management design or pump and treat? (sections 6 and 7)). Field-scale trials should be initiated to test the effectiveness of proposed AMD management strategies.	A sufficient level of technical rigour is needed to provide assurance that the design will effectively control the source of AMD and to derive a realistic cost estimate for implementing the strategy during the operational life of the mine and beyond.
	Pit lake	Ensure a good geochemical understanding of host rock and all wastes that will be stored in the pit lake catchment. Evaluate the composition of water that will be likely to collect in the base of the pit and in surface water flow paths in and around the pit void (Section 6).	The likely future water quality of the pit lake should be able to be predicted at this time and compared with the proposed closure objectives for the water body. Closure objectives may need to be changed if the water quality is not clearly likely to meet the defined future value for the lake. The locations of waste management facilities in the catchment of the lake may need to be reconsidered (Section 6.6).
Operations	Visual	As above, but extending to seepage zones at the toes of WRDs and ore stockpiles, open-pit walls, and embankments and roadways constructed from rock containing sulfidic material.	AMD data for both waste rock and tailings acquired as part of the ongoing mining operations should be imported into the waste block model, which should be regularly reviewed to ensure that the waste management plan is able to accommodate any significant changes (positive or negative) to the site's AMD risk profile. The closure plan must be regularly reviewed to ensure that it is consistent with the site's current risk profile, which may have materially changed as a result of additional AMD characterisation data or a substantive change in the scope of the project. Although the closure plan must be seen to be workable for the purposes of approvals, field trials and other research will need to be conducted during mining operations to demonstrate the long-term viability of the strategies that have been proposed. Any significant variations to projected closure costs need to be factored into project financials. Closure costs based on detailed designs need to be known to +/–15–20% within 5 years of closure. A project that minimises impacts from AMD during operations and after closure is consistent with the objective of leading practice sustainable development. See the *Mine closure* and *Mine rehabilitation* handbooks in this series for more information about the development of closure plans and designs for rehabilitation.
	Water quality and hydrology	Periodically revise site water balance model to incorporate changes to originally proposed water management plan and in response to changes in the scope of the mine plan. Install groundwater monitoring bores in footprints and downgradient of WRDs and TSFs to provide advance warning of the production of AMD seepage and impacts on groundwater. Implement surface water quality monitoring program to complement existing baseline program.	
	Geology/ mineralogy	Continue to develop and refine block models in response to the more comprehensive data produced as mining progresses. Use block models to refine the life-of-mine waste mining schedule and optimise the mine plan for the selective handling, placement and management of waste.	
	AMD management	To minimise the production of AMD, waste must be placed according to the strategy in the AMD management plan. This may involve encapsulation or placement below water of the most reactive material, for example (Section 6).	There *must* be regular reviews and audits of operating practice against the requirements and objectives of the AMD management plan. Early warning monitoring data (Section 9) should also be reviewed regularly to ensure that the implemented management strategy is meeting its performance objectives. If the monitoring data indicates a developing issue, the waste management strategy may need to be changed.

Mine phase	Category	Actions (refer also to sections 5 and 8)	Comments
	Progressive rehabilitation and closure planning	Undertake progressive rehabilitation where possible using human and capital resources available during the operations phase. Progressive rehabilitation needs to be distinguished from the management of waste during operations to minimise the inception of AMD. It involves the installation of covers and revegetation of those parts of the waste storage areas that have reached their final design height and are no longer active areas of placement. Field trials of proposed cover designs will probably be needed to confirm expected (from modelling) long-term behaviour. Use predictive modelling to assess the expected long-term performance of the rehabilitated landform (run-off, seepage, erosion) and compare the modelled results with the results of field trials.	Progressive rehabilitation will further reduce the medium-term risk of leaching of AMD products from the waste. An important benefit of progressive rehabilitation is that the effort needed to stabilise the site will be reduced in the event of unexpected closure or the placing of the site into care and maintenance. Material for construction and rehabilitation will most often come from the pit and footprints of WRDs, TSFs and process plant. All these materials should be characterised chemically and physically and incorporated into the material balance and mined material schedule. Ensure that understanding of likely performance of final waste landforms is sufficient to assess the magnitude of future risk. If field trials of the design concept and/or the outputs from modelling indicate potential issues with performance, an alternative rehabilitation strategy will need to be developed.
	Pit lake	Refine understanding of long-term pit lake water balance and quality by monitoring groundwater flows and quality and climatic data. Update water-quality predictions as required. Model water balance, incorporating projected climate change effects out to several hundred years. Undertake forward water quality assessment using tools ranging from simple mixing models of water inflow and contact geochemistry to more detailed water-quality predictive modelling if high water-quality is needed to meet closure objectives.	Ensure that all information is available to enable a comprehensive assessment of pit lake closure risk. The original closure objectives for the pit lake may need to be revisited or changed based on the findings from this more detailed assessment.
Closure and completion		Refer to the *Mine closure*, *Mine rehabilitation* and *Evaluating performance: monitoring and auditing* handbooks in this series for the necessary key actions during and at the end of the mine life to ensure minimal ongoing impact on the environment.	

3.3 Closure

At the time of mine closure, most of the preparatory work, including the development of closure objectives and criteria needed to protect the environment into the future, should have been done as part of a well-conceived AMD management strategy tested in field trials and implemented throughout the operations phase. If that is not the case, there could be a significant risk of adverse impacts, including higher costs to retrofit solutions at such a late stage. Invariably, as time goes on, the number of options available for effective management decreases and the costs rise (Figure 9).

Figure 9: Relationship between rehabilitation options and costs over time



Source: Team NT Toolkit (2004), courtesy of M Fawcett.

A review of 73 mine closure plans for the 2007–13 period noted how much more time mining companies took to meet their closure objectives when water management design for the long term had not been an integral part of planning and design (Byrne 2013). Furthermore, post-closure impacts from AMD have been identified as the dominant primary environmental impact from rehabilitated mine sites (Laurence 2006). The additional cost and other issues associated with closure planning deficiencies at the Woodcutters Mine in the Northern Territory have been described by Dowd (2005). General closure issues are dealt with comprehensively in the *Mine closure* handbook in this leading practice series (DIIS 2016a).

Because many AMD management technologies are still relatively new (less than 30 years old), there are few long-term cases that can demonstrate success in achieving stable and environmentally safe landforms. In the planning stages leading up to the design and implementation of the AMD closure strategy, the likelihood of success is indicated by results obtained from predictive modelling and from field trials.

However, the ultimate indicator of success will always be the long-term monitoring record. A company operating a site at risk from post-rehabilitation AMD should provide adequate financial and technical resources to run a robust long-term post-rehabilitation monitoring program (Section 9) and to undertake remedial works if necessary.

Because there can be a long lag time before AMD problems become evident, it will often be necessary to monitor the effectiveness of waste containment systems and impacts on surface water and groundwater for many years until good evidence of performance is available and sign-off can be obtained from the regulator. Such a responsible approach will enhance the reputation of the industry and help to maintain its social licence to operate.

However, largely because of past mining legacies, Australian regulators are currently reluctant to accept the return of mining leases at the end of mining. This means that potential financial liabilities associated with AMD and other risks at closed sites can remain on company balance sheets for a long time.

Case study 1 is a leading practice example of the application of the life-of-mine closure planning paradigm at a site at risk from AMD. Control measures for potential AMD-producing waste were implemented from the start and continued through the operating life of the mine. Ten years of post-decommissioning water-quality monitoring has shown that water leaving the site meets all agreed closure criteria quality objectives.[21]

Case study 1: Life-of-mine planning for closure and post-closure monitoring, Kelian goldmine, Indonesia

Context

The Kelian Equatorial Mining (KEM) gold mine, operated between 1992 and 2005 by Rio Tinto, was located in the East Kalimantan Province of Indonesia. Ore was mined from a single open pit. A diversion channel was constructed to redirect the original course of the Kelian River around the open pit whilst mining was in progress to allow expansion of the pit to the north. Gold was recovered from crushed and milled ore using a conventional cyanide leaching process. Approximately 90 million tonnes of tailings and almost 230 million tonnes of waste rock were generated during mine life.

Early characterisation work indicated that much of this mineral waste would be potentially acid-forming (PAF). KEM also receives almost 4 m of rainfall per year, so acid and metalliferous drainage (AMD) and water management were key focus areas during both the operational and closure periods. This study describes the proactive AMD management strategies implemented at KEM and provides monitoring data from the 10-year post-closure period, which indicates that the management strategies are performing as designed. Additional detail is provided in a paper by Palon (2014).

Mineral waste geochemical characteristics and classification

The Kelian ore body was an epithermal gold deposit hosted in volcanic and sedimentary rocks including tuff, andesite, rhyolite, mudstones, siltstones and sandstones. Pyrite was the most common sulfide mineral associated with the mineralisation, and total sulfur in the mineral waste typically averaged 2–4%. Calcium, magnesium, iron and manganese carbonates were also common in the mineralised rock. Neutralisation of the produced acid by rhodochrosite ($MnCO_3$).

21 See also Case study 7 in Section 6.

can release significant concentrations of manganese into solution. Manganese is relatively soluble at neutral pH and was recognised early as a primary contaminant of concern at KEM.

Waste rock was segregated based upon its sulfur, carbonate and manganese contents measured in an onsite laboratory to allow rapid classification. Non-acid-forming (NAF) waste rock was defined as having a negative net acid producing potential (NAPP) that contained less than 0.3% manganese. The dominant carbonate minerals in the NAF rock were calcite and dolomite, which provide more effective acid neutralisation capacity and do not release manganese into solution. PAF waste rock was defined as having a positive NAPP and/or manganese content greater than 0.3%. All tailings were designated and managed as PAF. Approximately two-thirds of the pit high walls above the post-mining watertable were also recognised as being PAF.

AMD management strategies

Given the extremely wet tropical climate at KEM, the overarching management strategy for AMD control focused on limiting oxygen ingress through the use of flooding and maintenance of saturated conditions. Subaqueous disposal is a proven management strategy which typically reduces sulfide oxidation rates by several orders of magnitude compared to conventional dry methods of mineral waste disposal (see Chapter 6). Waste rock segregation began in 1994 with PAF waste rock placed into the specially constructed upper Nakan waste rock dam where it could be rapidly flooded. Saturation was also maintained in the PAF tailings throughout operation by subaqueous deposition of tailings into the dam impoundment. At closure, the following mineral waste and ARD management areas were present on the site (Figure 1):

1. Namuk Tailings Dam—The dam was constructed as a permanent water-retaining structure with a clay core, rock fill shell and grout curtain installed in the foundation. The structure and spillway were constructed to International Commission on Large Dams (ICOLD) standards for a high-hazard dam. The structure impounds approximately 80 million tonnes of PAF tailings which are stored beneath a permanent shallow lake (Figure 2).

2. Upper Nakan waste rock dam—The interior of this repository contains approximately 160 million tonnes of saturated PAF waste rock covered by approximately 1–1.5 m of water. The embankment for the Upper Nakan dam was constructed as a water-retaining structure with compacted clay and muddy breccia cores, filter zones and rock-filled shells.

3. Lower Nakan waste rock repository—The repository contains over 40 million tonnes of predominantly NAF waste rock excavated during the diversion of the Kelian River and from the open pit.

4. 1280 and SP24 waste rock repositories—Less than 10 million tonnes of predominantly PAF waste rock was placed into these two small repositories at the start of mining before the Upper Nakan waste rock dam was constructed. These small repositories were capped with multilayered covers designed to limit both oxygen and water ingress.

5. Open pit—Approximately 10 million tonnes of tailings were placed into the bottom of the open pit after mining was completed. The pit was rapidly flooded, creating a permanent pit lake, and controlling oxygen ingress to tailings at depth and to the pit walls below the spill point. Only about 25 hectares of the pit walls are still exposed above the lake surface (Figure 3).

During operation the site had a positive water balance and excess water needed to be discharged to the Kelian River. Where needed, contact waters from the above facilities were treated with hydrated lime and allowed to settle in a series of settling ponds. The neutralised supernatant waters were then mixed in two main polishing ponds before being tested for compliance with water-quality criteria and discharged into the Kelian River downstream of the mine site. Lime additions continued as a short-term management measure during the immediate pre- and post-closure periods as the mineral waste repositories were flooded or capped.

Post-closure performance

In the post-closure period water is discharged directly from the Namuk tailings dam and mixed with flows from the waste rock repositories and open pit that are first discharged through a large constructed wetland (Figure 4). This wetland was constructed shortly after closure to passively polish (primarily for Mn) the mixed flows before discharge. Average annual flows are approximately 1,350 L/s from the Namuk tailings dam and 280 L/s from the wetlands. The combined flows then enter a final polishing pond before being discharged into the Kelian River from a single discharge point.

Discharge water quality meets Indonesian class I (drinking water or equivalent uses) river water-quality standards without the need for lime addition or other forms of active water treatment. Tables 1 and 2 present mean water-quality data for the Namuk tailings dam, and for the mixed pit and waste rock outflow from the wetlands, respectively. Table 3 shows water quality at the point of discharge to the Kelian River after all of the flows have been combined. This time-series WQ data is especially noteworthy since it shows that the compliance standards were met immediately after closure, with sustained performance thereafter. This illustrates what can be achieved by early implementation of control strategies and rigorous attention to integrated management of potential ARD sources throughout mine life.

Conclusions

KEM's early implementation of proactive AMD management strategies allowed for the successful closure of the mine without the need for perpetual active water treatment. The use of water covers and permanent saturation has effectively controlled sulfide oxidation rates and strongly limited AMD loadings within the mine footprint. This strategy has successfully protected the aquatic ecosystems and water resources in the Kelian River during the post-closure period while also minimising post-closure financial liabilities. The success of the KEM closure highlights the importance of early characterisation and control strategy selection, followed by effective implementation and performance monitoring to demonstrate success.

Table 1: Namuk tailings dam: mean 2008–14 annual monitoring results

PARAMETER	UNIT	QUALITY STANDARD	AVERAGE/YEAR OF MONITORING MINE A						
		LAKE WATER CLASS II	2008	2009	2010	2011	2012	2013	2014
Total dissolved solids	mg/L	1000	56	55	66	53	51	53	52
Total suspended solids	mg/L	50	7	6	8	8	4	5	7
Sulfate (SO4)	mg/L	400[1]	16	14	11	11	12	9	NM
pH		6-9	7.12	7.11	7.31	7.11	7.12	7.00	7.24
Iron (Fe)	mg/L	1.03	0.041	0.046	0.039	0.038	0.044	0.037	0.036
Manganese (Mn)	mg/L	2	0.036	0.018	0.008	0.012	0.010	0.018	0.065
Zinc (Zn)	mg/L	0.05	0.009	0.012	0.011	0.011	0.007	0.007	0.007
Mercury (Hg)	mg/L	0.002	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Lead (Pb)	mg/L	0.03	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Arsenic (As)	mg/L	1	0.001	0.001	0.001	0.002	0.002	0.001	0.002
Total cyanide (CNt)	mg/L	0.05	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	NM
Free cyanide (CNwad)	mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	NM

NM = not measured.

Table 2: Passive wetland treatment system: mean 2009–14 annual monitoring results

PARAMETER	UNIT	QUALITY STANDARD	AVERAGE/YEAR OF MONITORING MINE A					
		DISCHARGE WATER CLASS I	2009	2010	2011	2012	2013	2014
Total dissolved solids	mg/L	2000	1020	898	754	726	691	772
Total suspended solids	mg/L	200	10	8	7	5	4	7
Sulfate (SO4)	mg/L	-[1]	769	586	456	389	367	NM
pH		6-9	6.64	6.81	6.82	6.86	6.76	7.00
Iron (Fe)	mg/L	5	0.028	0.043	0.041	0.030	0.048	0.030
Manganese (Mn)	mg/L	2	0.0066	0.170	0.164	0.166	0.055	0.120
Zinc (Zn)	mg/L	5	0.031	0.024	0.012	0.007	0.008	0.005
Mercury (Hg)	mg/L	0.002	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Lead (Pb)	mg/L	0.1	0.049	0.044	<0.001	<0.001	0.002	0.002
Arsenic (As)	mg/L	0.1	<0.001	0.001	<0.001	<0.001	0.001	<0.001
Total cyanide (CNt)	mg/L	0.05	<0.01	<0.01	<0.01	<0.01	<0.01	NM
Free cyanide (CNwad)	mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	NM

NM = not measured.

[1] No discharge standards for sulfate at this location.

Table 3 – Mean 2009-2014 Annual Monitoring Results for Final Site Discharge to Kelian River (WS05)

PARAMETER	UNIT	QUALITY STANDARD		AVERAGE/YEAR OF MONITORING MINE A					
		DISCHARGE WATER CLASS I	RIVER CLASS I	2009	2010	2011	2012	2013	2014
Total dissolved solids	mg/L	2000	1000	198	314	254	174	208	312
Total suspended solids	mg/L	200	50	13	12	9	8	32	10
Sulfate (SO4)	mg/L	-[1]	400	113	193	153	99	99	NM
pH		6-9	6 - 9	7.03	7.04	7.00	6.91	6.75	6.94
Iron (Fe)	mg/L	5	0.3	0.056	0.037	0.047	0.046	0.042	0.035
Manganese (Mn)	mg/L	2	0.5[2]	0.017	0.167	0.045	0.054	0.047	0.078
Zinc (Zn)	mg/L	5	0.05	0.010	0.013	0.011	<0.005	<0.005	<0.005
Mercury (Hg)	mg/L	0.002	0.001	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Lead (Pb)	mg/L	0.1	0.03	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
Arsenic (As)	mg/L	0.1	0.05	0.002	0.002	0.001	0.001	0.001	0.001
Total cyanide (CNt)	mg/L	0.05	0.02	<0.01	<0.01	<0.01	<0.01	<0.01	NM
Free cyanide (CNwad)	mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	NM

NM = not measured

[1] No discharge standards for sulfate at this location.

[2]sulfate Determined from site background levels in ANDAL (1990).

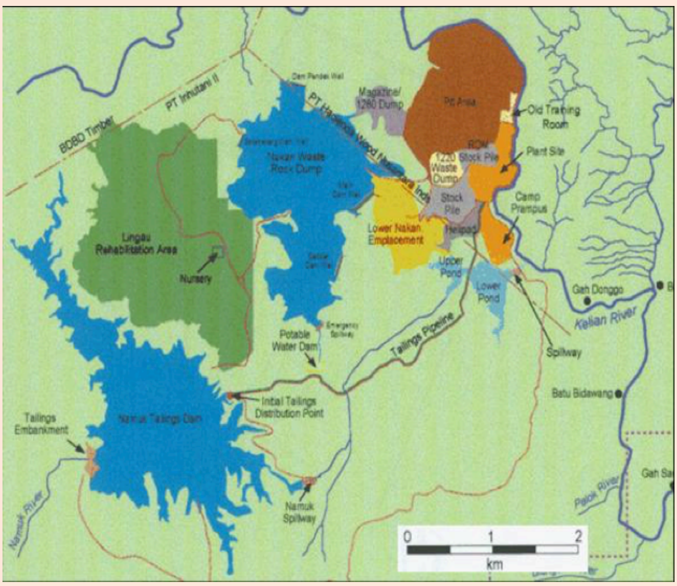


Figure 1: Site layout during operation showing open pit (upper right), waste rock repositories and tailings impoundment (lower left)

Figure 2: Namuk tailings storage area showing constructed embankment

Figure 3: Flooded open pit

Figure 4: Wetlands passive treatment system

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4.0 CHARACTERISING MINE MATERIAL AND PREDICTING AMD

Key messages

- The science for predicting the likely extent of AMD risk before mining is well advanced.
- No single AMD characterisation test will be sufficient to fully assess the AMD risk across the range of material types typically present at mine sites, with multiple test methods being required.
- An increasingly detailed sampling and materials characterisation program is required as a project with some AMD risk evolves from an exploration to a mining phase.
- It is essential to develop a three-dimensional 'AMD block model' (see Glossary and Figure 10) and associated waste materials management schedule where AMD risk warrants this. This model should be integrated with the ore block model and the mine plan to optimise materials handling on site and to define final pit wall rock types.
- The failure of mines to identify and subsequently manage AMD risk can be due to not having similar levels of resolution for the waste and ore block models and not integrating them at the earliest possible stage in the mine planning process.
- There needs to be close coordination between the personnel who are developing the AMD block model and the mine planners. Shortcuts taken in waste deposition to achieve immediate cost savings may be substantially offset by subsequent long-term AMD and water-quality management costs.

4.1 Overview

The key aims of mine material characterisation are to determine:

- the potential *extent or magnitude* of AMD generation
- the potential *rate and timing* of AMD generation
- the likely contaminants of concern in leachate produced from the oxidation of sulfidic materials.

It is critical that a staged mine material sampling and geochemical assessment program is carried out to ensure that sufficient data is available at all phases of a project. This applies to both greenfield and brownfield mine sites.

Leading practice management of mine materials can only be achieved through:

- careful characterisation
- the classification of mine material types
- appropriate planning and scheduling for containment of material at risk of AMD throughout the mine's life.

In this context, 'mine material' is all sulfidic materials that can be exposed to water and air, either directly as a result of being excavated and exposed or indirectly as a result of groundwater drawdown (desaturation of sulfidic strata) as a consequence of dewatering.

Expert advice should be sought at an early stage of project development to help site personnel (especially mine planners) interpret results, define geochemical material types and understand the implications for operations, materials management and closure.

Table 3 summarises the key components of a mine materials characterisation program for a new project. It shows the stages in the mine's life at which the various types of characterisation should be initiated and further developed, and the general areas of responsibility for running and managing the characterisation program. While this table was produced specifically to summarise the geochemical characterisation component of the G4 resource modelling and mine planning process undertaken by Kinross Gold Corporation in the United States, its content is generically applicable. Descriptions of the various test methods are provided in below in this section.

Table 3: Geochemical characterisation and stage in mine life

DATA OR ANALYSIS	LEAD	INITIAL	INFILL	SCOPING	PRE-FEASIBILITY	FEASIBILITY	OPERATIONS
Design/review of data collection strategy (selection of analytical method and sampling density and distribution)	Expl						
Total sulfur and carbon	Expl						
Multi-element data	Expl						
ABA (acid base account), NAG (net acid generation) test, water extraction	Expl / Env						
Kinetic investigations	Env						
Identification of materials management requirements	Env / M- Planner						
Monitoring & data collection strategy (sampling frequency, distribution, analysis)	Env / M- Planner						

Expl = Exploration; Env = Environment section; M-Planner = mine planner

Progressively more detailed/advanced

Full implementation

Use/maintain/update as necessary

Source: Reproduced with permission, Kinross Gold Corporation, USA, from Williams et al. (2015).

A basic screening level investigation should begin as early as possible. In the first instance, reference to other mining operations in the region, particularly those extracting from comparable stratigraphic or geologic units, may provide empirical information on the likely geochemical behaviour of ore types, waste rocks and wall rocks, as well as the potential for AMD production. Geological logging of drill core (or drill chips) should be carried out to identify the presence of reactive minerals such as sulfides and carbonates, their abundance and their mode of occurrence.

For many sites, early indications of AMD potential can be provided by measurements of total sulfur and total carbon content in collected materials. Screening for total sulfur in the field can be done using a handheld X-ray fluorescence (XRF) analyser. The need for and scope of more detailed investigations depends on the findings from this initial screening.

Where screening indicates that further assessment is warranted, the leading practice approach is to collect and test enough samples to progressively develop a statistically valid spatial model of the AMD potential of waste, ore zones and other relevant mine materials (as defined above) to quantify the potential extent or magnitude of the AMD. Spatial AMD models include block models (usually in hard rock mines) or grid/layer models (usually in coalmines) that can be developed to represent both in situ mine materials (for example, pre-mining ore and waste rock and in situ wall rock) and ex situ materials (such as ore stockpiles and WRDs).[22] In the remainder of this handbook, block models and grid/layer models are collectively called 'block' models.

An AMD block model (Downing & Giroux 2014) is a three-dimensional cellular representation of the level of AMD risk presented by different types of mine materials. The size of each cell relative to the lateral heterogeneity of the waste (and ore) determines how accurate the model will be. Each cell is populated with relevant material properties. Visual representations of cross-sections through an AMD risk block model are shown in Figure 10. This shows the locations of materials of different types (different colours) in the spatial context of the final pit shell.

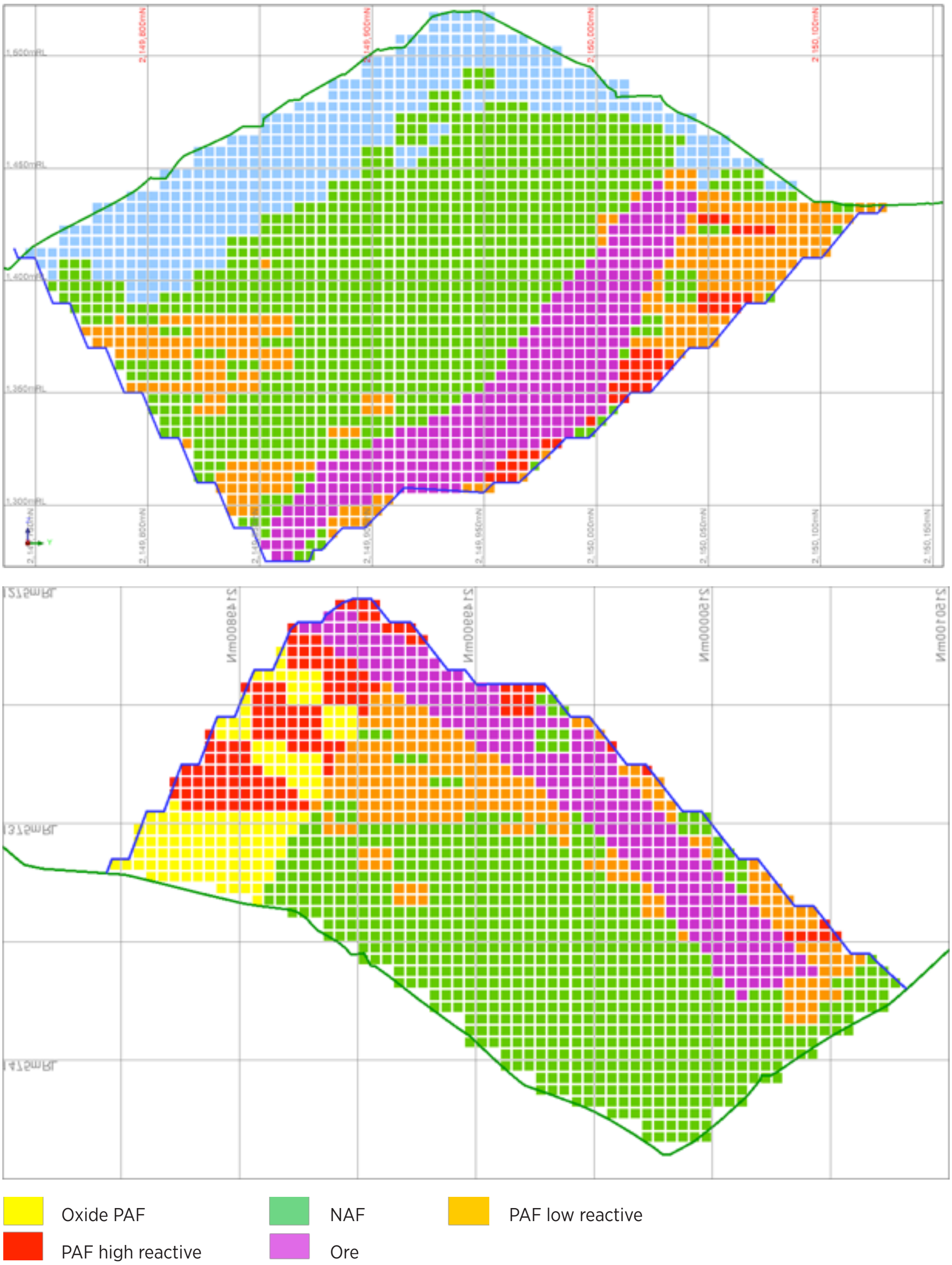
Where possible, an AMD block model (or environmental geochemical layer within the mine model) should be based on the existing mine geology block model. The application of a range of geochemical tests (for example, AMIRA 2002; Coastech Research 1989; INAP 2009; MEND 2005; Miller et al. 1997; Price 2009; Australian Standards AS4969.0–AS4969.14 for acid sulfate soil material) will be needed to develop and further refine the classification system and AMD block model, as greater definition of the resource and its associated waste and surrounding materials is obtained.

Note that no single characterisation approach is valid for all mineral deposit types. However, once parameters are calibrated using a range of geochemical tests, it is likely that only one or two parameters (such as total sulfur content or NAPP data) can ultimately be used to simplify inputs into the block model.

The results from the characterisation test work should be used to produce an AMD risk classification layer in the block model. This layer identifies those zones of the waste and in situ materials that pose the highest risk and hence require the most management when mined (Figure 10). Knowing in advance the volumes of problematic material, and when it will be produced by the mining schedule, will facilitate its placement in the most appropriate locations and ensure that enough benign material is available to encapsulate or cover it within a reasonable time frame (for example, within the lag period preceding acid generation) after it is placed.

22 For simplicity, the term 'AMD block model' is used hereafter in this handbook to cover all forms of spatial AMD models.

Figure 10: Visual representation of an AMD block model—two slice cross-sections showing relationship of waste types to proposed open-pit shell (ore body shown in purple)



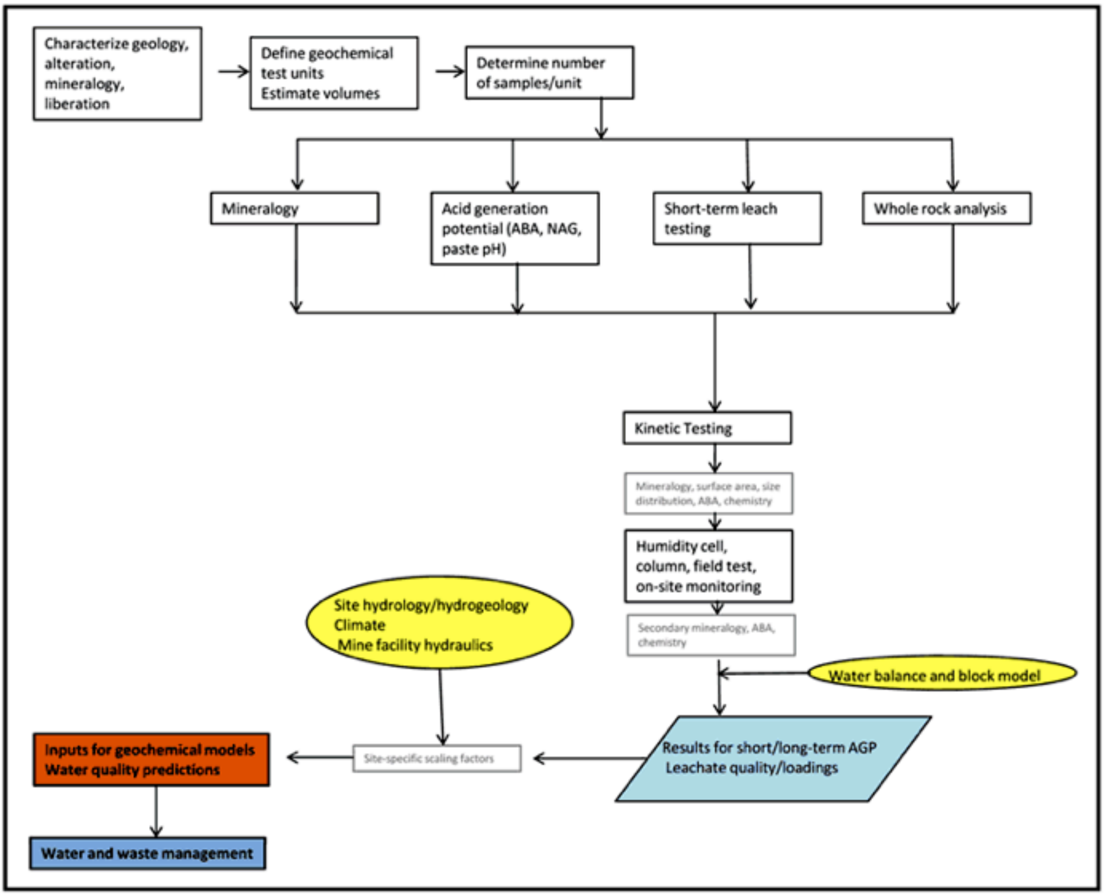
PAF = potentially acid-forming; NAF = non-acid-forming

Once an AMD risk classification system has been developed, the rate and timing of AMD generation can be assessed by combining the AMD block model and the mine materials production schedule with the results of kinetic geochemical test work for each of the key mine materials and relevant lithologies (see below).

Kinetic test work involves measuring the rates of reaction of sulfidic materials, identifying potential soluble contaminants of concern and their release rates, and estimating the potential lag times before the onset of acid generation and the predicted longevity of acidity generation. Kinetic tests include column leach, humidity cell and humidity cell and xygen consumption methods and field test piles.[23]

The sequence of geochemical tests and the flow of information to inform the development of a waste and water management strategy for a mine site are summarised in Figure 11. The use of a well-developed AMD block model and associated mine materials schedule, supported by static and kinetic geochemistry data, provides a robust approach for predicting the potential magnitude of water-quality impacts associated with AMD and for identifying and prioritising management measures for sulfidic materials.

Figure 11: Flowsheet for the conduct of a geochemical characterisation program



Source: Reproduced with permission, Kinross Gold Corporation, USA, from Williams et al. (2015).

23 The application of kinetic test data to a specific project design or proposed waste management strategy is discussed further in Section 4.5.

The data produced by this approach can be fed into more complex site- or domain-level water-quality models to develop more detailed water chemistry predictions to assess potential impacts. However, the value of such modelling must be assessed in the light of the sensitivity and uncertainty of results that can be achieved.

4.2 Sampling for characterisation

4.2.1 Overview

Selecting samples and the methods used to collect them are critical tasks that need careful consideration at all stages of the project. As the project progresses from exploration through to feasibility, the range of samples tested should ultimately represent each type of material that will be excavated or exposed throughout the mine's life and after its closure. Enough samples should be obtained for each of the types of exposed material to determine the extent of significant variability in their material properties. Key mine material types to be sampled include waste rock, overburden, ore (run-of-mine and low-grade) and wall rock (open-cut/underground). While overburden might not contain much sulfide, it could still be geochemically enriched in metals and require specific management.

Each mine material type may consist of several different lithologies. In addition, each specific lithology can be weathered or oxidised to differing extents and, as this will influence the reactivity and AMD potential, the altered zones also need to be sampled and characterised. Even if sulfide concentrations are low or absent due to weathering processes, metalliferous drainage may still be an issue at some deposits. This highlights the importance of conducting geochemical sampling (and characterisation) of all material types.

Conducting representative sampling involves identifying the number and types of lithologies and the alteration and weathering sub-variants that constitute the bulk (>95%) of the total tonnage of materials to be excavated or exposed to oxidising conditions. Where possible, this sampling should span the full lateral and vertical extent of the ore deposit and associated waste and surrounding materials. It is important for expert advice to be sought at an early stage of project development to provide input into the design of the exploration drilling program and associated sampling for geochemical characterisation.

Representative samples of process/product/concentrate streams and tailings or other process waste streams should be obtained from metallurgical test work conducted during the feasibility and development stages of the project. Those samples are needed in addition to samples of ore, waste and surrounding materials to provide geochemical characterisation information for the tailings and product components of the AMD waste management system. Sulfide mineral concentrates can be very reactive and may require specific management where they are stockpiled (on site, during transport, or in intermediate storage at ports).

The numbers and type of samples selected will be site-specific and depend on the phase of the project (see Table 2 for guidance on sample numbers by project phase). Sufficient samples must be obtained to represent the variability/heterogeneity in each mine material as described above. Factors such as habit, grain size, structural defects, extent of alteration and dissemination or veining of reactive minerals (for example, sulfide minerals, carbonate minerals) must be addressed as part of the sample selection process to ensure that the full range of relevant properties is captured for each type of mine material.

Sample collection and handling requirements also need to consider the degree of weathering (for example, fresh, partially oxidised or fully oxidised) at the time of sampling, and implications for subsequent static or kinetic geochemical test work results. This is particularly important for aged drill core materials, weathered samples from brownfield or legacy sites, and sulfidic materials that are highly reactive and/or prone to spontaneous combustion. Ideally, samples from drill cores should be taken and tested as soon as possible after drilling. If that is not possible, the drill core should be stored under cover (for example, in a core shed) to minimise exposure to weathering processes until sampling and testing can occur. Drill chips should be stored in sealed heavy-duty plastic sample bags to minimise the potential for oxidation before testing.

4.2.2 In-place mine materials

Although drilling and sampling tends to focus on ore zones in the exploration, pre-feasibility and feasibility phases, sufficient samples of potential wastes and exposed wall-rock material should also be collected to confirm that the future risk of AMD from those sources is not substantially underestimated. As the project develops, samples of waste rock and wall rock should be increasingly represented. This progressive development ensures that adequate data is available to produce a robust AMD block model and associated mine materials management schedule. Geostatistical analysis will ultimately be able to be used to inform and optimise the sampling strategy and to refine the block model.[24] However, a sufficient number of samples representing the different lithologies, and the lateral distribution of properties within those lithologies, must initially be available before such analysis can be applied with a reasonable degree of confidence.

Sampling guidelines for static and kinetic geochemistry test work for initially in-place (that is, yet-to-be-mined) materials are summarised in Table 4. Indicative sampling frequencies are provided in Table 2.

Samples collected for AMD assessment during operations are normally obtained from holes drilled for blasting or underground development. The inferred waste and ore boundaries are marked up using the geological block model on a map of the current bench plans for an open-cut mine (or on a map of the drive and stope development plans for an underground mine) and checked before and after blasting. The results from the characterisation of the blast hole samples are used to reconcile the geological model and AMD block model.

24 See Case study 3.

Table 4: Sampling guidelines for in-place mine materials

- Multiple bulk samples, for both static and kinetic geochemical test work, are required to represent each lithology and weathering type from different parts of the deposit (that is, to ensure representative lateral and vertical variation).
- Continuous drillhole sampling should be completed where discrete samples collected for characterisation comprise a single lithology and weathering type, rather than spanning more than one type.
- Drillholes for sampling should be selected based on an even grid spacing and, in the case of metalliferous mines, should be sampled from surface (hanging wall) to the footwall of the ore body. Waste rock, ore and surrounding rock should be sampled. Diamond core provides the best source for sampling (since the intact material can be visually logged), followed by reverse circulation (RC) drill chips.[a] A similar sampling regime should be employed for coalmines, although sulfidic materials are often concentrated near and within coal seams, so the sampling intensity may be skewed towards those areas.
- Each bulk sample should be taken from a drillhole interval length sufficient to sample a single lithology (typically 0.5 m to 10 m), unless differential patterns of alteration indicate otherwise. Multiple lithologies should not be mixed. At coalmines, it is important to include specific coal seam roof, floor and major parting samples in the range of samples taken down a drillhole.
- Each bulk sample should comprise at least quarter core (minimum) extracted from the entire length of the selected drillhole interval to ensure that the sample is fully representative of that interval.
- Samples for geochemical test work should generally not be composited within each drillhole unless the subsamples used to produce the composite are obtained from the same lithology and drillhole interval. Sample compositing (for example, if a larger sample size is required for kinetic tests) can sometimes be undertaken after the results from initial geochemical screening tests have been acquired and interpreted,
- Each bulk sample should be crushed to <20 mm aggregate (or finer) to facilitate representative subsampling by 'splitting'. Representative subsamples cannot be achieved by grab sampling small masses of material from the bulk. Splitting using standard equipment (such as a rotary splitter or riffle splitter) and procedures produces representative subsamples of the required mass for static and/or kinetic geochemical analysis.
- For static geochemical test work, a minimum representative bulk subsample mass of 1 kg of aggregate is generally sufficient to submit to a laboratory. Additional sample preparation by laboratories includes further crushing to <2 mm or <4 mm, riffle splitting and pulverising to <100 µm, and the resultant pulp is subsampled for analysis.
- Bulk sample quantities are required for kinetic test work (Section 4.5). The samples can be around 2–5 kg for oxygen consumption test work, up to 35 kg for column leach test work, and up to 100 kg for oxygen diffusion test work.

[a] In most metalliferous deposits, RC drilling or open-hole rotary air blast (RAB) drilling is commonly used to drill through the waste rock material, and coring using a diamond drill bit is used to sample the ore material. Thus, the bulk of the mine material available for sampling and testing for waste properties may be RC or RAB drill chips, with limited diamond core available for testing waste.

- For kinetic geochemical test work, samples should ideally be prepared and tested in a manner that most closely simulates field conditions. However, for laboratory-based tests a smaller sample size means that waste rock needs to be crushed to ensure an adequate particle surface area and contact time with the leaching solution (typically deionised water or site rainwater). Bulk samples are typically crushed to pass a top size ranging from 5 mm to 40 mm, depending on the dimensions of the leaching apparatus. Tailings should be tested at the milled size that will be used for the process.
- Representative subsamples of the material used for kinetic geochemical test work (rather than similar/comparable mine material types) should be collected for static geochemical test work to assist in the interpretation of the findings.

4.2.3 Existing exposed mine materials

Existing exposed mine material is typically found at brownfield and legacy sites where mined materials and process tailings have been exposed for some time and a proportion has already undergone significant oxidation. However, this situation may also occur at sites that have had a long operating life and have waste dumps, possibly of uncertain composition, in which the oxidation of sulfides is well advanced.

For legacy sites, in particular, there may be little or no prior characterisation data that will provide information about the primary compositions and disposition of different waste types within the waste management facilities. Even if such prior information is available, the extent of reaction that has occurred may mean that the original geochemical characteristics of the materials have changed to such an extent that the original characterisation data is no longer applicable. The term 'forensic geochemistry' has been applied to the process that needs to be followed to define the current characteristics of such facilities.

WRDs, in particular, can be highly heterogeneous (both in geochemistry and in particle size distribution), depending on how the mining and waste deposition were carried out. Lithologies may no longer be present in defined spatial arrangements and in intact layers, as is the case for yet-to-be-mined materials.

The sampling of such masses requires defining the distribution of material types and their current geochemical characteristics by obtaining sufficient numbers of profile samples, typically by obtaining composites over 1 m vertical intervals. There are close similarities between what is required to define the spatial variance of geochemical properties in these facilities and a contaminated site assessment. Hence, reference should be made to the National Environment Protection Measure (NEPM) for the assessment of contaminated sites to provide guidance on the principles of sampling design.[25]

In the case of waste rock, drilling or test pitting provides the samples. Conventional, auger or sonic drilling is used for tailings. The additional complexity of particle size needs to be addressed as part of the sampling regime for mined waste rock. The <2 cm fraction is typically considered to provide the best initial indication of bulk material properties for initial characterisation. However, whole samples may subsequently be needed to provide greater definition of materials' physical and AMD properties as a function of particle size.

The geochemical characterisation of already exposed material (for example, an existing WRD) typically proceeds in stages, the first stage being a pilot-scale investigation to obtain a measure of likely variance in

25 http://www.scew.gov.au/nepms/assessment-site-contamination.

properties. The second stage involves additional sampling to more clearly define the distribution of material properties.

The masses of samples needed and the type of sample preparation required for static and geochemical test work on these already exposed materials are identical to those provided for in-place materials in Table 4.

4.3 Geochemical static tests

A core suite of standard procedures has been developed to assess the potential for AMD generation from sulfidic materials. In general, the procedures are designed to take account of both:

- acid-generating reactions, which are promoted by the oxidation of reactive sulfide minerals once exposed to atmospheric oxygen
- acid-neutralising reactions, which result from the dissolution of reactive alkaline minerals, mainly carbonates (Section 2).

The types of geochemical static test methods and their strengths and weaknesses are described below. Finally, the use of static test data to develop an AMD classification scheme for the range of mine materials assessed is described.

4.3.1 Field measurements

Simple and very useful measurements to make in the field are pH and EC: pH provides an indication of the amount of free acid, and EC is the electrical conductivity of the sample. Both parameters can be easily measured in the field using calibrated pH and EC probes coupled to a handheld meter. A 1:1, 2:1 or 5:1 liquid-to-solid slurry of the sample (typically the <2 mm size fraction sieved from a bulk sample) is made with deionised water and the pH and EC values are measured after a defined period of time. The same liquid:solid extraction ratio should be used for all samples to provide a common basis for comparisons between samples.

The pH value indicates whether the oxidation of sulfides has exhausted the neutralising capacity of the material (acid pH), and the EC value provides a measure of the amounts of soluble salts (salinity) available to be leached from the material. These measurements are especially useful for characterising the status of already partially oxidised material (for example, sulfidic material exposed at the surface, present in existing waste storage facilities, or present in acid sulfate soil horizons).

Field-based mineralogy assessments (Section 4.3.2), NAG tests (Section 4.3.5) and hand-held XRF analysis can also be applied to screen and guide the selection of samples for subsequent more rigorous characterisation using the suite of methods described below.

4.3.2 Mineralogical analysis

Mineralogical analyses most commonly use XRD (X-ray diffraction) techniques that are conducted on 1–2 mg of representatively subsampled finely crushed sample material. The limit of detection for a given mineral type is routinely close to 1–2 weight% (that is, 10,000–20,000 mg/kg) for common mineral phases. Hence, while accurate mineralogical assessments are the ultimate goal of geochemical test work, XRD techniques almost always need to be supplemented with more sensitive methods that infer rather than directly measure the concentration of a specific mineral.

For greenfield sites, the key reactive minerals are normally dominated by sulfides and carbonate minerals. Since not all sulfide minerals generate acidity and not all carbonate minerals can effectively neutralise acidity, it is important to understand the characteristics and distribution of these key minerals as revealed by XRD.

As reactive geological materials oxidise, the number of secondary mineral phases increases, and they may coat the surfaces of the primary sulfide minerals, thus decreasing the rate of reaction of the bulk mineral assemblage with oxygen. Some of the secondary mineral phases are of low solubility (for example, alunite and jarosite) and can store acidity. This natural evolution has significant implications for both measuring and managing AMD risk at brownfield or legacy sites, and highlights the need to clearly understand the nature and distribution of secondary acidity generating mineral phases, particularly at those sites where significant oxidation of sulfide minerals has already occurred.

The AMD-generating and AMD-neutralising capacity of common sulfide and carbonate minerals from mineralogical data collected via geological logs or XRD analysis can be calculated using the ABATES shareware (Waters et al. 2014; see Glossary).

Mineralogical analysis can be conducted via field-based or laboratory methods. Field methods use visual estimates of mineral abundances (volume%) obtained from geological logging of drill core, documented in the drill logs, and/or gained from field observations. The advantage of this is that geological data is commonly available at any stage of mine development, enabling a rapid first-pass estimate of AMD potential for both in situ materials and waste rock.

New technology is available for field-based analytical measurements of mineral abundances (weight%) using benchtop XRD equipment. Laboratory XRD analyses can achieve greater accuracy and lower detection limits than field-based measurements, but remain time consuming and limited in resolution.

Mineralogical data is most effectively used in conjunction with the results from other geochemical test methods (see below).

4.3.3 Elemental composition

The elemental composition of representative samples of each mine material should be determined and assessed in relation to the degree of enrichment or depletion relative to background soils and rocks. Two methods can be used to do this: the elemental enrichment factor (EEF) and the geochemical abundance index (GAI). Both measures can be used as part of a leading practice characterisation program because they provide complementary information.

The EEF compares the concentration in the sample to the background for the local area, while the GAI compares the concentration with median global soil/rock element abundance data, using a geostatistical approach based on a log scale. These comparisons are used to identify elements (especially metals and metalloids) that occur at concentrations above normal background values and that may require further investigation, such as by analysis of water extracts and/or NAG test supernatant (Section 4.3.5) or by kinetic test work (Section 4.5), to further assess their potential environmental significance.

The use of the GAI alone may provide an over-conservative indicator of the significance of an occurrence of an elevated GAI value if the local background value for a given element is high compared to the global average. However, even if the local background for a particular element is elevated, that does not mean that the significance of its occurrence can be discounted. For example, exposure of that element contained in mined material could introduce an additional load of the element to the environment via accelerated leaching induced by the oxidation of sulfides.

Elemental composition data can also assist with interpretation of mineralogy data, due to the greater precision and lower detection limits of analytical methods used to measure elemental concentrations, such as handheld and laboratory-scale XRF, ICPAES (inductively coupled plasma atomic emission spectroscopy) and inductively coupled plasma mass spectrometry.

Some elements, such as arsenic, mercury, selenium and cadmium, may be a concern at concentrations that are not substantially elevated relative to background concentrations, since those elements can biomagnify through food chains. It is therefore important that the leachability and/or bioavailability of these elements be considered further, if required, as part of the assessment. Their concentrations in a water or controlled pH extract can be compared with the applicable protection of aquatic ecosystem guidelines and livestock drinking water guidelines in ANZECC–ARMCANZ (2000a). In some instances, the metal/metalloid concentration values in solids can be compared with the concentration values for metals in soils in the Australian guidelines for contaminated site assessment (SCEW 2013). The latter guidelines were recently updated and now contain expanded guidance on how to undertake risk assessments that include accounting for the bioavailability of metals in soils contaminated by mining wastes or AMD.

4.3.4 Acid base account

The acid base account (ABA) estimates the balance between the potential for a material to generate acid and to neutralise acid. The output from an ABA is a value known as the net acid producing potential (NAPP), expressed in units of kilograms of sulfuric acid per tonne (kg H_2SO_4/t).

The NAPP test involves determining the maximum potential acidity (MPA) and the maximum inherent acid-neutralising capacity (ANC) of a sample. The total sulfur content is commonly used as a conservative estimate of pyritic sulfur (that is, all S is assumed to be present in the form of pyrite) to calculate the MPA (MPA = weight% S x 30.6). The use of total sulfur is a conservative approach because some sulfur may be present in forms other than pyrite.

Some sulfate–sulfur bearing secondary minerals such as gypsum, anhydrite and barite are non-acid-generating, while others such as melanterite, jarosite and alunite are acid-generating, although to a lesser degree than pyrite. The latter group of acid sulfate (secondary) minerals are products of the oxidation of pyrite. There are other metal sulfides, such as chalcocite (Cu_2S) and covellite (CuS), that yield less acidity upon oxidation than pyrite, and others that are non-acid-generating, such as sphalerite (ZnS) and galena (PbS). The ABATES freeware (see Glossary) accounts for the different acid-generating potentials of the range of sulfide mineral types that may be present.

The mineral types (both primary and secondary) that are present and their concentrations can be identified or inferred by a combination of XRD mineralogy and sulfur speciation test work (sections 4.3.2 and 4.3.6, respectively). If the amounts of these other forms of sulfides present in a sample are determined, allowance can then be made for their contributions to provide a more refined estimate of the MPA.

The ANC is typically determined by the addition of a known quantity of concentrated hydrochloric acid to a sample, followed by back-titration with sodium hydroxide to quantify the maximum amount of acid consumed by the inherent neutralising capacity of the material. See Price (2009) for a discussion about the different methods that can be used to measure the ANC of a sample.

The determination of ANC is not precise and may overestimate the ANC that is available to neutralise AMD. While the measurement of total carbonate carbon is useful in this context, care does need to be taken with this parameter. In particular, unless the presence of iron- and manganese-bearing carbonates such as siderite, ankerite, ferroan dolomite and rhodochrosite is accounted for, the carbonate-based neutralising capacity will be overestimated. This issue can generally be resolved through a combination of mineralogical (Section 4.3.2) and elemental analysis (Section 4.3.3) test work.

A more refined estimate of the effective ANC of a sample can be provided by the acid buffering characteristic curve method. The sample is slowly titrated with dilute acid to measure the extent of buffering provided by carbonate minerals. However, care needs to be taken for those materials that contain significant amounts of siderite (FeCO3) or rhodocrosite (MnCO3), which are carbonate minerals that do not provide net acid neutralising capacity.

Once sufficient characterisation data becomes available for a given deposit, it may be found, for example, that there is a good correlation between effective ANC and the sum of acid extractable Mg and Ca. In such a case, the generally much more frequently determined Ca+Mg value could be used to infill the distribution of ANC in the block model.

Two measures of the ABA are calculated from the MPA and ANC: the NAPP and the ANC/MPA ratio. The NAPP is a qualitative measure of the difference between the capacity of a sample to generate acid (MPA) and its capacity to neutralise acid (ANC).[26] The NAPP, MPA and ANC are expressed in units of kg H_2SO_4/t and the NAPP is calculated as follows:[27]

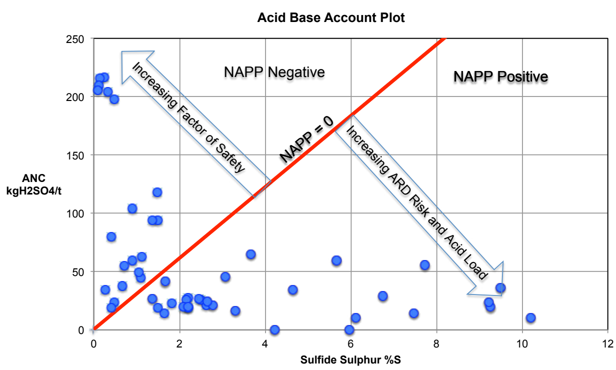
NAPP = MPA - ANC

If the MPA is less than the ANC, then the NAPP is negative, indicating that the sample may have sufficient ANC to prevent acid generation. Conversely, if the MPA exceeds the ANC, then the NAPP is positive, indicating that the material may be acid-generating.

The ANC/MPA ratio provides an indication of the relative margin or factor of safety (or lack thereof) for a given material. Various ANC/MPA values are referenced in the literature for indicating safe values for the prevention of acid generation. Those values typically range from 1.5 to 3. As a general rule, an ANC/MPA ratio of 2 or more signifies that there is a high probability that the material will remain near-neutral in pH and should not be problematic in terms of acidity generation and resultant acidic drainage. However, NMD and SD may still be issues that need to be addressed.

The relationship between ANC and sulfide content for the range of samples obtained by an AMD characterisation program is displayed on an acid base account plot (Figure 12). The plot shows the distribution of samples between the higher and lower risk (of generating a net acidic pH) domains.

Figure 12: Example of an acid base account plot



Note: ARD is synonymous with AMD for these materials.

26 There are several nomenclature variations for static test parameters in the literature. For example, net neutralisation potential (NNP) refers to the difference between the neutralisation potential (NP) and acid potential (AP). The NNP is generally expressed as kg $CaCO_3$/t.

27 The NAPP can also be estimated using the ABATES shareware (see Glossary).

In Figure 12, the samples are distributed over the NAPP positive and negative domains. Whether there is sufficient NAPP negative material to counterbalance the NAPP positive material depends on the overall masses, reactivity, mineralogy and scheduling of the respective classes of materials that are present. This information can be provided by the AMD block model, geochemical and mineralogical characterisation data and mine planning data.

If most of the samples represented in Figure 12 had been located in the higher risk domain, that would indicate:

- a high risk of a net acid outcome
- that there is likely to be limited net neutralising material available to encapsulate the likely acid-generating material.

Because some sulfur minerals do not generate acid (but may contribute to NMD and SD, as indicated above) and there are different forms and reactivities of AMD-generating and AMD-neutralising minerals, there is a level of inherent uncertainty in prediction based solely on reliance on the reported ABA.

While the NAPP value (and ANC/MPA ratio) provide an indication of the potential for acid generation from a sample, additional test work is needed to predict the potential for NMD or SD and the lag time before substantial AMD is produced (see below).

4.3.5 Net acid generation test

The net acid generation (NAG) test, once calibrated, is one of the simplest and generally most reliable geochemistry test methods for first-pass estimation of AMD potential. It involves the reaction of a sample with hydrogen peroxide (H_2O_2) to rapidly oxidise any sulfide minerals (AMIRA 2002). Both acid-generation and acid-neutralisation reactions occur simultaneously, and the net result represents a direct measure of the amount of acid released from the sample. A pH after reaction (NAG pH) of less than 4.5 generally indicates that the sample is net acid generating. The amount of acid released is determined by titration to pH values of 4.5 and 7.0 and expressed in units of kg H_2SO_4/tonne.

NAG tests are generally done in the laboratory, but can also be used in the field to facilitate day-to-day materials characterisation and handling. A similar method has been developed for characterising and handling acid sulfate soils (Ahern et al. 2014).

Several variations of the NAG test have been developed to accommodate the wide geochemical variability of mine materials and to address potential interferences. The two main static NAG test procedures currently used are the single addition NAG test and the sequential NAG test (AMIRA 2002). The sequential NAG test may be required for high sulfide/sulfur samples to provide a measure of the total acid-generating capacity, and for samples with high total sulfur and high ANC.

Analysis of the supernatant produced by the NAG procedure can provide valuable information about the relative concentrations of metals that are likely to be contained in leachate produced by the oxidation of the sulfides present in waste. In particular, the sequential NAG test can inform the sequence of release of metals as the sulfides are progressively oxidised.

The pH and composition of the NAG supernatant can also be used to infer whether the material being tested could be at risk of producing NMD or SD.

The NAG test is suitable for a wide range of mine materials, but can be unreliable for samples with high organic carbon content such as coal-bearing materials or bituminous shales associated with some base metal deposits. Modified NAG tests are available for those types of materials, but the results need to be interpreted carefully (ACARP 2008).

There is potential for NAG tests to result in an overestimate of the extent of oxidation relative to field conditions (Stewart et al. 2003). NAG tests are therefore most useful when used in combination with other static and kinetic geochemical test methods.

4.3.6 Sulfur and carbon speciation

Sulfur speciation test work can be conducted to overcome some of the limitations of ABA test work, such as the potential for overestimating acid-generation potential if a sample contains forms of sulfur other than pyrite, such as commonly occurring sulfate minerals (for example, anhydrite, gypsum, barite, jarosite, alunite, schwertmannite), native sulfur, non-acid-forming sulfides (for example, sphalerite, galena, covellite) or weakly acid-generating organic sulfur-bearing compounds. The reason for this overestimation is that first-stage calculation of the acid-generating capacity of a sample conservatively assumes that all sulfur present in the sample is in the form of pyrite, which will produce the maximum yield of acidity.

Sulfur speciation test work is being increasingly applied to the assessment of AMD potential, as it is particularly useful for:

- deeply weathered (that is, oxidised) deposits
- brownfield sites with old already partially or fully oxidised waste rock and tailings deposits containing significant amounts of jarosite and other acid-storing secondary minerals
- mineralisation that contains highly reactive sulfides (for example, mineral sand deposits).

Sulfur speciation test work can be used to distinguish various forms of sulfur:

- Total sulfur (STotal);
- chromium reducible sulfur or sulfide sulfur (S_{Cr}) (sulfide minerals) (Australian Standards 2008)
- adsorbed and water soluble sulfate sulfur (S_{KCl}) (for example, gypsum, melanterite—$FeSO_4$)
- acid or carbonate extractable soluble sulfate sulfur (S_{HCl}) (for example, gypsum, melanterite, jarosite, alunite)
- native sulfur.

Carbon speciation test work—including total carbon, inorganic and organic carbon—can be used to supplement ANC measurements (used in NAPP calculations) and in some cases provide a more reliable and more cost-effective alternative to standard ANC data for block model development.

4.3.7 Sample classification

The key aim of the static geochemical testing methods described above is to produce a sample classification scheme to underpin the development of block models showing the distribution of AMD risk through the waste, ore and surrounding materials.

At the most fundamental level, materials can be classified as potentially acid-forming (PAF), non-acid-forming (NAF) or uncertain using the findings from static tests. However, the application of this classification scheme alone can result in failure to identify materials at risk of producing NMD and/or SD and reduce opportunities for more efficient and cost-effective management of mined materials.

A set of criteria of the type represented by the compilation in Table 5 has often been, and continues to be, used for the preliminary categorisation of static test data produced early on in the AMD assessment process.

The use of both NAPP and NAG test results reduces the risk of misclassification by using NAPP data alone (see below and Figure 13). An 'uncertain' classification is used when there is an apparent conflict between the NAPP and NAG results (for example, when the NAPP is positive and NAG pH > 4.5, or when the NAPP is negative and NAG pH < 4.5).

Table 5: Preliminary screening criteria based on NAPP and NAG test data

PRIMARY GEOCHEMICAL MATERIAL TYPE	NAPP (KG H_2SO_4/T)	NAG PH
Potentially acid-forming (PAF)	> 10[a]	<4.5
Potentially acid-forming—low capacity (PAF-LC)	0 to 10[a]	*< 4.5*
Non-acid-forming (NAF)	Negative	≥ 4.5
Acid-consuming (ACM)	less than -100	≥ 4.5
Uncertain (UC)[b]	Positive	≥ 4.5
	Negative	<4.5

[a] Site-specific but typically in the range 5–20 kg H_2SO_4/t.
[b]Further testing needed to confirm material classification.
Source: Based on AMIRA (2002).

The 'uncertain' category is an especially important one that should be resolved by more extensive test work. If a substantial proportion of the material tested falls into the uncertain category, it is imperative that a program of kinetic testing be initiated as soon as possible.

A limitation of the classification scheme in Table 5 is that there is no recognition of the potential for material, especially in the NAF, ACM and UC categories, to produce NMD or SD. This additional level of classification should be done as part of a leading practice characterisation program (including kinetic testing) to determine the full extent of AMD risk posed by mined materials. Failure to do this could result in material that is misclassified as 'benign' being wrongly managed or being used inappropriately for construction or rehabilitation purposes.

Leading practice requires that further subdivision based on site-specific needs be applied to identify samples with varying acid-generating capacities, acid-neutralising capacities and NMD or SD potential so that the risk profiles of those materials can be identified and managed appropriately. For example, a geochemical risk classification scheme such as that shown in Table 6 enables the identification of a wider range of potential issues, although not all of the subcategories in column 4 may be needed. For a particular site, the subclassifications in columns 3 and 4 can typically be grouped into three or four major groups for more efficient and cost-effective management of the spectrum of AMD types.

Some mining operators use proprietary software to automate the classification process in order to provide unbiased, conservative and rigorously consistent mine material classifications. Parameters that are commonly used in AMD classification systems are produced by the static test methods described above.

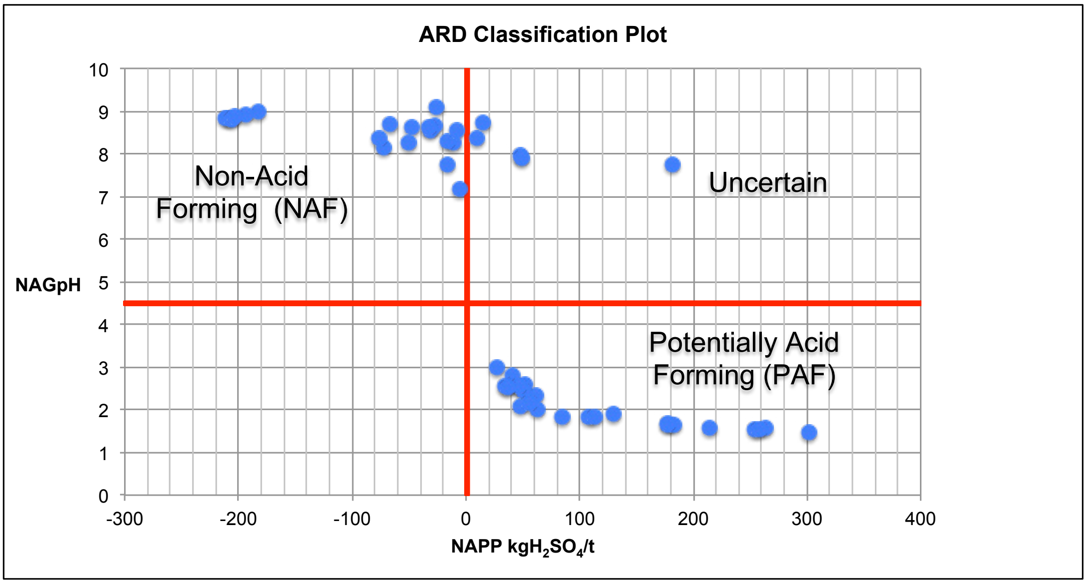
Individually, each of the test methods has limitations, but when a strategic combination of tests is applied (selected according to the nature of the deposit, the complexity of the geology, the reactivity of sulfides or the degree of weathering) the reliability of AMD classification is greatly enhanced. For example, NAG test and sulfur speciation results can be compared to NAPP calculations to clarify uncertainties or ambiguities in the results produced by each member of the test suite alone. An example of this is provided in Figure 13, which uses the same set of samples shown in Figure 12. Additional geochemical kinetic test work will be needed to assign an AMD risk classification to those samples that are labelled as 'uncertain'.

Table 6: AMD risk classification scheme

GENERAL AMD RISK CLASSIFICATION	DETAILED AMD RISK CLASSIFICATION		
	DESCRIPTION	AMD & NMD[1] RISK CLASSIFICATION	AMD & NMD & SALINITY RISK CLASSIFICATION
Potentially acid-forming (PAF)	High potential for acid generation (AG1)	AG1	AG1 Saline
	Moderate / high potential for acid generation (AG2)	*AG2*	AG2 Saline
	Moderate potential for acid generation (AG3)	AG3	AG3 Saline
			AG3 Non-Saline
	Low potential for acid generation (AG4)	AG4	AG4 Saline
			AG4 Non-Saline
Non-acid-forming (NAF)	Unlikely to be acid generating (UAG)	UAG	UAG Saline
			UAG Non-saline
		UAG NMD	UAG NMD Saline
			UAG NMD Non-saline
	Likely to be acid consuming (LAC)	LAC	LAC Saline
			LAC Non-Saline
		LAC NMD	LAC NMD Saline
			LAC NMD Non-Saline

NMD = pH neutral mine drainage (pH 6–8).

Figure 13: Comparison between ABA and NAG test results on the set of samples shown in Figure 12



Key points about AMD classification criteria

'Generic' AMD classification systems based on data from elsewhere that specify % sulfur cut-offs, NAPP values with threshold values of greater than or less than zero (for example, +/– 10) or low NAG value cut-offs should not be applied to final assessments of AMD risk without being substantiated by site-specific data.

Given the complexity of the issues involved, expert advice should be sought at an early stage of project development to assist site personnel in interpreting results, defining geochemical material types and developing an understanding of the implications of geochemical properties for AMD management during operations and after closure. Rigorous assessment should be undertaken before adopting classification values from other sites to confirm that they have sufficiently similar geology and geochemistry.

For a given site, it should be possible to establish the combination of parameters that best facilitates site-wide sample classification. Those parameters and their associated AMD classification criteria will be site specific and should only be used after completing a detailed review of the results of the full range of static geochemical tests, ideally coupled to kinetic data, and taking into account the mine plan and schedule. Examples of the development and implementation of site-specific classifications are provided in several of the case studies in this handbook.

4.4 AMD block modelling and materials scheduling

The likely geochemical behaviour or AMD classification of individual lithologies can be assessed from the results of the static test work programs outlined above. However, the likelihood of a site developing AMD issues and the effective implementation of options for managing them depend on understanding the quantity, distribution and timing of extraction of each mine material type exposed to oxygen by excavation or dewatering. This includes all mine material types defined in Table 6, not just acid-generating material.

The next level of interpretation involves applying the static geochemical test data to the project design or configuration under investigation to estimate the timing of exposure of PAF materials to oxidising conditions and to predict the potential extent of AMD generation. This requires:

- building a drillhole database that contains the geochemical characterisation data
- using the drillhole database to update the 3D geological model that defines the distribution of all the material types that will potentially be mined or exposed to oxygen
- constructing the AMD block model to assess the quantity, occurrence and distribution of different geochemical material types (AMD classifications) in the deposit
- developing the life-of-mine materials schedule for the disturbance or dewatering of those geochemical rock types, based on the AMD block model
- spatially modelling the waste rock dumping sequence by geochemical rock type to minimise future AMD risk, based on the AMD block model and mine material schedules
- predicting the likely spatial extent of oxidation within the dewatered zone (that is, the cone of depression) around the mine based on the physical characteristics of the materials, climatic conditions, the water balance, hydrogeology data and other parameters that influence AMD generation.

AMD block models are distinctly different from the geological block model used to design the initial sampling program (Scott et al. 1996), since they also incorporate the outcomes from the materials characterisation work in a risk context. Block modelling for AMD assessment purposes is similar to ore resource modelling, as it is used to calculate material volumes and hence tonnages (Downing & Giroux 2014).

Total sulfur is the most common parameter used for screening in the first phase of developing an AMD block model, before the results from the more detailed characterisation test work become available.

The initial availability of data to define the waste is commonly low relative to the data collected for initial definition of the ore body, but need not be. Rio Tinto, for example, typically completes sulfur (and carbon) analysis on most drill core samples whether they be ore or waste. Once calibrated using a geochemical test program on a subset of samples, this information can be used to initially populate an AMD block model.

In most cases, there remains a need to continually update the AMD block model with more extensive information about the distribution and volumes of waste types as mining progresses. Failure to do so is a common cause of unanticipated AMD management issues. Indeed, an initial assessment of low AMD risk could need to be revised as more information is obtained about the ore body and associated waste.

The spatial resolution of AMD block models (that is, block size) must be robust and practical to facilitate implementation by mine planners/managers. If the blocks are too small or too large, they may not be able to be segregated during mining and may not define correctly the distribution of the mine materials selected from the database.

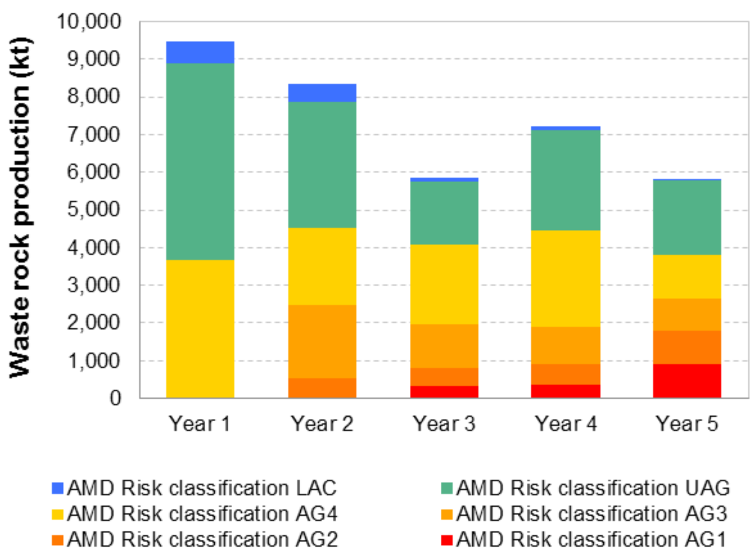
Block models and associated mine material schedules have significant implications for mine planning (such as the design of WRDs) and mine site operational management on a daily, weekly, monthly and yearly basis. Thus, there needs to be close coordination between the personnel who are developing the AMD block model and the mine planners. Shortcuts taken in waste deposition to achieve immediate cost savings may be substantially offset by subsequent long-term AMD and water-quality management costs. In particular, the timing of production of high-risk AMD material will determine when sufficient clean material needs to be available for encapsulation, so that the amount of time that the high-risk material remains exposed is minimised.

In addition to developing mining waste production schedules, it will also be necessary to define the geochemical material types and volumes exposed on final pit walls, and in porous backfill and cave zones in underground mines. This work is needed to assist in the prediction of post-closure AMD risk for final pit lakes and in underground workings.

An example of a waste rock production schedule (based on AMD classifications) that can be produced using AMD block modelling is shown in Figure 14. The AMD risk classifications in Table 6 were used to produce this schedule. The physical properties of each type of material should also be coded in the block model, since they affect the methods of construction and suitability using the materials for rehabilitation.

While the discussion above is focused on greenfield mine sites, the process for developing AMD block models can be applied to brownfield and legacy sites. However, existing exposed and already partly oxidised ex situ materials may be much more challenging to characterise and model than in situ fresh materials owing both to the variable extent of oxidation that has occurred and the typically heterogeneous distribution of the original lithotypes.

Figure 14: Example of annual production of waste rock, according to AMD classification



Case study 2: Use of characterisation data to direct appropriate management of PAF coalmining waste, Stockton coalmine, New Zealand

Context

The Stockton coalmine, owned by Solid Energy New Zealand Ltd, is the largest opencast coalmine in New Zealand. It is located between 500 m and 1,100 m above sea level within the Brunner Coal Measures (BCM) of the Buller coalfield on the west coast of the South Island. The climate is cool and wet, with rainfall of around 6,000 mm per year and an average temperature of 8°C (Solid Energy 2015).

A significant existing environmental liability is associated with historical AMD issues at the site, and the Stockton coalmine expects to be treating this in perpetuity. As with many sites, there is an increase in acidity load with increasing flow rates for streams impacted by AMD, indicating flushing of stored acidic oxidation products from the previously placed waste rock. The site currently treats about 6,000 to 10,000 tonnes of (H_2SO_4) acidity per year, although this acidity load is increasing as more acid-forming waste rock is disturbed.

This case study highlights the extensive program of geochemical characterisation that is being used to develop and implement improved management strategies for current and future PAF waste being produced from new mining areas.

Geological setting

The BCM formed during the Late Cretaceous and Early Tertiary marine transgression within an estuarine environment having a ready supply of sulfate derived from the periodic intrusion of seawater. The BCM are overlain by the Kaiata mudstone. Both units are composed of detrital quartz, albite, muscovite, and kaolinite. Pyrite is the most common sulfide in the coal measures and is present as both highly reactive framboidal (fine-grained) pyrite and coarser euhedral crystals. Since the BCM are typically carbonate deficient, the presence of pyrite nearly always leads to the formation of AMD when waste rock is exposed. The Kaiata mudstone has decreasing pyrite content and increasing carbonate content with distance from the BCM contact, such that higher in the stratigraphic column the Kaiata is NAF as per ABA classification techniques. However, the total volume of NAF at this site is very low compared to PAF waste rock, and this limits the options that involve selective placement and management of waste (for example, encapsulation of PAF waste rock with NAF waste rock).

Due to the prevalence of framboidal pyrite and the lack of neutralising minerals, disturbed rocks at Stockton can generate AMD within days. Geochemical sampling programs were therefore established to prioritise material that needed to be capped as soon as possible, to ensure the correct placement of PAF materials and to identify areas for further management measures such as the application of crushed limestone to limit acidity onset.

Brunner Coal Measures—ABA characterisation

By 2010, more than 220 samples had been analysed from the BCM for ABA characteristics. Of these, at least 100 had a complete suite of ABA analyses (paste pH, NAG, NAPP) and the remainder had only MPA and NAG data. This data from the BCM indicated that there is an excess of MPA (average = 22.5 kg H_2SO_4/t) compared to ANC (average = 2.9 kg H_2SO_4/t), reflecting a lack of carbonate materials in the BCM (Pope et al. 2010). Based on this dataset there was confidence that the BCM contained very little ANC. Hence, the test for ANC was discontinued and in future calculations it was assumed to be zero.

A daily tiphead sampling program was implemented to determine (total) sulfur, and thus pyrite trends, across the site (Weber et al. 2008). The results from the tiphead sampling program and the more detailed ABA characterisation process confirmed that mudstone and carbonaceous units of the BCM had a greater potential to generate acidity compared to coarser sandstone units of the BCM. In addition, the data indicated that geographically, the northern areas of the site had higher sulfur associated with a more estuarine palaeo-environmental depositional setting, compared to the south of the site, which was more fluvially (freshwater) dominated and typically had an MPA < 5 kg H_2SO_4/tonne. Such data enabled the development of an AMD management plan, where priority was given to capping material from the northern areas. The mudstones were preferentially buried in the core of WRDs; and active AMD treatment was initiated in the Mangatini Stream, which drained these northern areas.

Kaiata mudstone—ABA characterisation

A number of studies were initiated in 2006 to define the ABA characteristics of waste rock associated with the proposed new Cypress pit at the mine. Samples were taken from 12 holes drilled through the Kaiata mudstone into the underlying BCM across the expected footprint of the pit. The results indicated that the BCM formation from this location had very similar characteristics to what had been found previously (see above)—that is, there was little ANC—so all BCM units were classified as PAF.

However, NAF Kaiata mudstone was present within the Cypress pit, and additional geochemical investigations were undertaken to determine implications for AMD management. Results demonstrated that some samples classified as PAF by the NAG test (NAG pH < 4.5) had paste pH values greater than 6.0. For these materials (which contained moderate ANC) there would be a time lag to acid onset, as validated by kinetic NAG testing and field-based column testing (200 L drums). The results indicated that paste pH was a good indicator of the immediate acid-base reactivity of the waste rock. A classification system combining paste pH and NAG pH results was therefore proposed for the Cypress pit. This is summarised in Table 1, which also explains the management process for each type of waste rock.

As part of the stakeholder engagement process this methodology was reviewed by independent experts. It was suggested that the use of paste pH may not be suitable for fresh rock, and further work was recommended. Solid Energy recognised that such continued improvement of its AMD classification methodology and model was warranted as this would improve the management of waste rock and thus reduce AMD liabilities.

The additional work that was done is described below.

Table 1: Classification system based on NAG pH and paste pH for Kaiata mudstone

CLASSIFICATION	TEST CRITERIA	MANAGEMENT APPROACH
NAF	Paste pH > 6 NAG pH > 4.5	Possibly suitable for use as an engineering material (further testing required).
PAF: Lag to AMD	Paste pH > 6 NAG pH < 4.5	Encapsulation to reduce subsequent oxygen-water ingress prior to AMD formation.
PAF: Medium risk	Paste pH 4-6 NAG pH < 4.5	Neutralisation[1] required followed by immediate encapsulation.
PAF: High risk	Paste pH < 4 NAG pH <4.5	Neutralisation[1] required followed by immediate encapsulation.

[1]The site will apply neutralising materials to each lift of the WRD to minimise the amount of acidity generated by the PAF waste rock.

Operational refinement of the AMD characterisation program and waste rock management: Cypress pit

In 2015, as operations started at the Cypress pit, Solid Energy completed a new ABA study of waste rock, which included a review of the classification process (Table 1) and the determination of existing stored acidity within freshly blasted waste rock. This was used to help design the Cypress northern engineered landform (NELF) to minimise the oxidation of pyrite and the generation of AMD-impacted drainage from the site.

Results from ABA testing on 42 samples were reviewed against relevant resource consent criteria (regulatory classification requirements) and an improved classification process was implemented (Table 2) to develop an enhanced block model of waste rock to better define management options. Based on the number of samples in each classification grouping and the mean NAPP for that group, the potential acidity load for the site for each group was determined as a percentage of the total.

Assessment of the stored acidity within the Kaiata mudstone by methods proposed by the Australian acid sulfate soils guidelines (Ahern et al. 2004) indicated that the water soluble extractable titratable acidity was ~3 kg H_2SO_4/tonne 2–3 weeks after blasting, and prior to mining the blasted material, and ~7 kg H_2SO_4/tonne 5 weeks after blasting (Jarosite-type stored acidity was < 1 k H_2SO_4/tonne, but greater for older samples). This finding indicated the presence of very reactive pyrite. Such knowledge suggested that the proposed classification (Table 2) was fit for purpose and that the time from initial exposure by blasting to burial in the NELF needed to be minimised. This approach was incorporated into the AMD management plan for the site.

For the study area, the data showed that approximately 76% of waste rock samples were PAF or high acid forming (HAF), accounting for 94.5% of the potential acidity load. The analysis indicated that good waste rock management of the HAF group (representing 26% of the waste rock) would address about 55% of the total potential acidity load, representing an opportunity to substantially reduce overall acidity loads for the site. Using this data, the agreed management strategy for the site included the placement of all HAF within the centre of the NELF, which would be surrounded by PAF with low-risk and NAF materials on the outer surfaces of the NELF. This landform is being constructed in 5 m lifts to minimise advective and convective ingress of oxygen with the addition of neutralant to each lift and compaction.

Table 2: Classification of waste materials for the Cypress operations, Stockton coalmine

CLASSIFICATION	ABBREVIATION	CRITERIA	RELEVANT CONSENT CONDITION CRITERIA	NO. OF SAMPLES	% OF POTENTIAL ACIDITY LOAD [1]
		kg H_2SO_4/tonne	*kg H_2SO_4/tonne*		
Non-acid forming	NAF	NAPP < 0	NAG acidity = 0	*4*	*1.5%*
Low risk	LR	NAPP > 0 and ≤ 20	NAG acidity < 20	6	4.0%
Potentially acid forming	PAF	NAPP > 20 and ≤ 50	NAG acidity > 20	21	39.7%
High acid forming	HAF	NAPP > 50	n.a.	11	54.7%

n.a. = not applicable.
[1]The potential acidity load is derived from the sample population. The block model still requires development to confirm the true percentage.

Conclusions

Continued and ongoing refinement of the ABA characterisation process at Stockton has resulted in the development of a robust geochemical understanding and model for waste rock at site. For the BCM, tiphead sampling has confirmed the geographical distribution of high PAF waste rock, which has been prioritised for capping. For the Kaiata mudstone an improved classification scheme has been developed, which has been used to design the Cypress NELF to minimise oxygen ingress to HAF waste rock placed within the core of the WRD.

The ongoing geochemical characterisation program is a key step for directing management of high risk sulfidic waste at the site. The data is being used to more effectively contain the high-PAF material and thus reduce treatment costs. It has been shown at this site that effective ABA characterisation can help with strategic planning to minimise the production of AMD.

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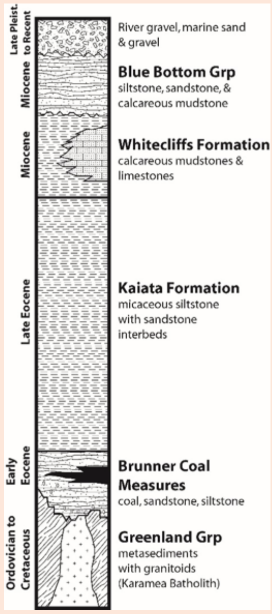

Figure 1: Stratigraphic column for the Stockton area

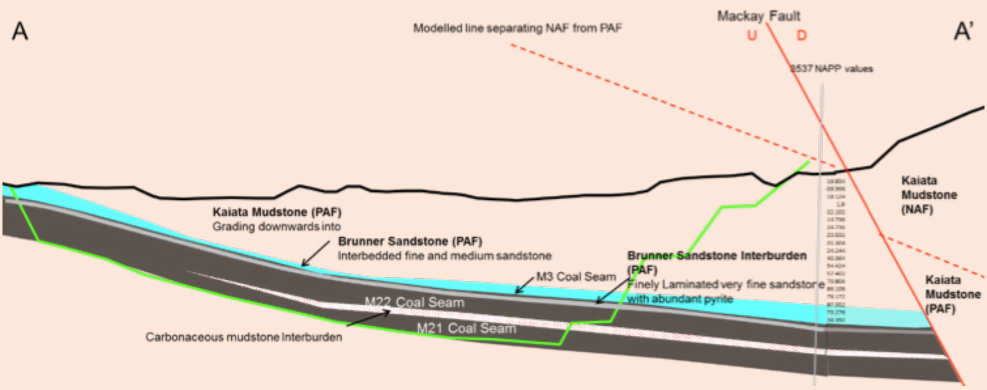

Figure 2: Cross-section through the Cypress north pit

4.5 Geochemical kinetic tests

While static test work assesses the likelihood and maximum capacity of specific lithologies to generate and buffer acidity, kinetic test procedures provide data on just how fast this acidity and/or buffering (and consequent load of AMD) can be generated, and the lag period before the onset of significant generation of AMD. Such tests reveal processes associated with the reactivity of various sulfide minerals, the kinetics of sulfide oxidation, neutralisation efficiency by carbonate and/or silicate (for example, chlorite) minerals, and the formation and solubility of secondary mineral phases and their dissolution characteristics. Kinetic tests typically involve controlled oxidation and some form of leaching, and can be carried out under conditions that approximate average site conditions, are typical of worst-case site conditions, or artificially accelerate pyrite oxidation rates relative to site conditions (Bourgeot et al. 2011; Pearce et al. 2015).

The key purpose of these tests is to evaluate:

- acidity generation rates for specific lithologies and eventually for broader domains (such as whole WRDs)
- the lag time before the onset of acidic conditions
- the duration of sulfide oxidation and acidity generation processes
- the potential for AMD
- the key elements of environmental concern in leachate
- the likely success of various AMD prevention and control strategies.

Pyrite oxidation rates obtained using traditional kinetic test work methods (for example, small-volume leach columns) may substantially differ from those found at mine sites due to a number of factors, including moisture content (which influences oxygen diffusion to reaction sites), particle size distribution (related to the surface area available to react with oxygen), sulfide mineralogy, oxygen concentrations in the vicinity of the particles, different boundary conditions and temperature.

Kinetic tests ranging from small columns to field-scale trials are used to develop relationships between the pyrite oxidation rate and one or more of the above variables to enable more reliable estimates of real-world acidity generation rates. For example, pyrite oxidation rates can be determined for various particle size fractions of waste rock samples or moisture contents. Those rates can be used together with knowledge of the particle size distribution of waste rock on site to predict acidity generation rates. The range of kinetic tests used to measure oxidation of sulfides, and hence AMD generation rates, is described below.

4.5.1 Column leach and humidity cell tests

Column leach tests and humidity cell tests are commonly used to assess the kinetic geochemical behaviour of mine materials. These tests typically involve subjecting a crushed sample of mine rock or mill tailings to sequential flushing and drying cycles. The tests can be conducted in the field at the mine site or in a laboratory. If they are conducted in a laboratory, it is important that the flushing process does not retard sulfide oxidation by inhibiting air entry to the sample, although some tests may be completed under saturated conditions if the mine waste is planned to be stored under a water cover, for example. The leaching regime should be selected to optimise oxidation but remain broadly consistent with site climatic conditions.

Column leach and humidity cell tests can be conducted on small (for example, ~2 kg of <4 mm size material) to large (for example, ~30–50 kg of <40 mm size) samples. Larger column sizes can be used, if required, to further examine scale-up factors. The larger the mass of the sample that is obtained, the greater the range of particle sizes that can potentially be tested. Multiple columns can also be established to assess various management scenarios (cover systems, blends and treatments).

Leach columns and humidity cells are good for identifying key elements of concern in leach waters and revealing changes in leach chemistry over time, as weathering progresses. However, it is difficult to directly relate or scale the changes in water chemistry and soluble metal concentrations in leachate to the future site water quality that will develop. While certain elements may report to leachate that will be produced in the field, their concentrations are unlikely to be well predicted by laboratory column tests. Therefore, while direct comparisons between water-quality guidelines and leachate from leach columns and humidity cells may provide some context, such comparisons should be treated with caution.

In situations in which no sulfides are present in rock samples, leach columns or humidity cells can still be used to assess leachate quality.

Kinetic leach tests generally need to operate for at least six months and in some cases 12–24 months before sufficient data is available for the effective interpretation of the AMD characteristics of a material. Longer time frames may be involved in hard rock mines, for example, for evaluating the performance of specific treatments or soil/rock type blends, or for samples that have a long lag time to the onset of AMD formation.

Other issues that need to be considered and/or addressed when interpreting data produced by these methods are as follows.

- Inferring pyrite oxidation rates from leach columns and humidity cells by measuring soluble sulfate release rates alone can be compromised in some cases by the potential for sulfur retention as secondary sulfate minerals (such as jarosite or gypsum) or for enhanced sulfur release from already existing soluble sulfate minerals (such as gypsum or melanterite) in the case of already partly oxidised material.
- Even if pyrite oxidation rates can be estimated from leach data, it is difficult to relate them directly to mine site conditions owing to the effects of variable saturation and the influence of micro- and macrostructural properties (for example, particle size distribution and the effect of the method of placement of the waste).
- The presence of acidophilic and/or iron-oxidising bacteria can substantially affect the rate of AMD production in the real environment. In some cases, consideration should be given to conducting a set of tests inoculated with bacteria.
- It can be challenging to control or fix sample moisture content for these tests, although it can be achieved (as shown in Figure 15). Hence, it is generally not straightforward to assess the relationship between AMD generation rate and sample moisture contents.
- Testing fine-grained materials can be challenging due to their low porosity, the slow rate of desaturation after sample irrigation or the intense desiccation that occurs if heat lamps are used to simulate sunshine and speed up the drying process.
- In some cases, it can be difficult to reconcile the need to simulate the climatic conditions of a site with the requirement to collect sufficient leachate for analysis. For example, for waste materials taken from a site located in an arid environment, the water application rate may need to be much higher than the equivalent annual rainfall to produce sufficient leachate for analysis in a time-constrained testing regime.

The alternative kinetic test work methods discussed below have been developed to address some of the limitations of column and humidity cell tests in estimating sulfide oxidation rates.

Figure 15: Example of moisture content control in a kinetic leach column test

Source: Robertson et al. (2015).

4.5.2 Oxygen consumption tests

The oxygen consumption method is based on the principle that reactive sulfides such as pyrite consume atmospheric oxygen as they oxidise. The rate of acidity production from pyrite decomposition is stoichiometrically related to oxygen consumption. Oxygen consumption methods can be applied directly within WRDs or tailings deposits or can be conducted in the laboratory.

In the laboratory, a known mass of pyrite-bearing material is hermetically sealed in a vessel of known volume. Once the test is initiated, oxygen in the vessel is consumed via pyrite oxidation (Anderson et al. 1999; Bennett et al. 2005; Bourgeot et al. 2011; Pearce et al. 2015). The internal oxygen concentration is automatically measured and logged (for example, every 10–60 minutes) over the measurement period (for example, 1–8 weeks). Since carbon dioxide gas may be generated as a result of carbonate neutralisation or bacterial carbon oxidation (for example, from acid sulfate soils), some test work methods also measure CO_2 to increase the accuracy of and enhance the information obtained through conventional oxygen consumption test work. Once the oxidation test work is complete, the sample can be flushed with deionised water and the leachate chemistry analysed to identify elements of potential concern.

Oxygen consumption techniques incorporating initial static ABA data and final leachate chemistry may offer a number of advantages over other laboratory-scale kinetic geochemical tests, including:

- much more rapid determination of sulfide oxidation rates (such as within days)
- the direct measurement of oxygen consumption, which can provide very accurate and unambiguous oxidation rates compared with other methods

- the ability to predict lag times in the shorter term before the complete depletion of acid-neutralising capacity
- potentially lower costs due to lower analytical costs and shorter test durations
- the ability to easily adapt test procedures to test the efficacy of site AMD management strategies, including water and soil covers, or various WRD designs.

Oxygen consumption methods are also a useful method for rapidly assessing the potential for spontaneous combustion of sulfidic mine materials, and are increasingly being used in metallurgical assessments of metal releases from block cave zones in underground mines and in stockpiles of ore and product concentrate.

4.5.3 Oxygen penetration tests

When advective or convective gas transport processes control sulfide oxidation, conventional humidity cell or oxygen consumption kinetic methods can be reasonably applied to estimate maximum oxidation rates. However, the transport of oxygen through very fine-grained unsaturated materials is often 'diffusion limited'. For example, within a tailings deposit, even if the depth of unsaturated tailings extends to several metres, only a small proportion of that depth is subject to oxidising conditions. This is due to oxygen consumption throughout the tailings profile at a rate that exceeds the resupply of atmospheric oxygen because of diffusion-controlled limitations. Measuring pyrite oxidation rates in diffusion-limited tailings can therefore be difficult, as the rates are likely to progressively decrease from the air–tailings interface to the final depth at which the oxygen concentration in the pore space is zero.

To enable the calculation of realistic acidity generation rates under these conditions, it is possible to quantify the quasi-steady-state depth of penetration of oxygen into a pile of fine-grained materials (such as tailings or mineral sands) and then quantify the corresponding oxygen flux (that is, the oxygen diffusion rate) per unit surface area. This has been achieved in the field and in the laboratory through the application of an oxygen diffusion profile test (Martin et al. 2006).

Fine-grained sulfidic materials are placed in open-topped cylinders and instrumented at multiple depths to measure oxygen and carbon dioxide concentrations and sometimes moisture content. The tests can be designed to simulate site conditions such as density, moisture content and average temperature, either during operations or after closure. After reaching a quasi-steady-state oxygen profile with depth (for example, in weeks or months), the cylinder is sealed and oxygen flux into the upper surface of the fine-grained material is logged over 1–5 days. In combination, this data permits the measurement of the early-stage thickness of the oxidation zone and the acidity generation rate as a function of surface area.

While oxidation rates in advection- or convection-controlled systems can be defined relative to the mass of oxidising material, the rates in diffusion-controlled systems are more appropriately defined relative to the exposed surface area of the material. Note that in some cases hardpan development on a tailings surface can influence the tailings moisture content and penetration of oxygen into the underlying tailings (Gilbert et al. 2003; McGregor & Blowes 2002; and Robertson et al. 2015).

Because oxygen diffusion profile cylinders do not need to be irrigated (unlike column leach tests), they can also be used to potentially more accurately simulate and test a range of closure strategies under essentially steady-state moisture conditions.

Computer models such as PYROX (Wunderly et al. 1996) can also be used to investigate the effects of saturation on rates of oxidation. PYROX is a numerical model that simulates one-dimensional, kinetically controlled diffusion of oxygen into the unsaturated surficial zone of mine tailings and the subsequent oxidation of sulfide minerals, such as pyrite.

Case study 3: Us static and kinetic test data in a waste management strategy for PAF material, Newcrest Cadia Valley Operations, New South Wales

Context

Cadia Valley Operations (CVO), which is owned and operated by Newcrest Mining Ltd, is one of Australia's largest goldmining operations. CVO is located in central western New South Wales, approximately 25 km from the city of Orange. The current phase of mining commenced after Newcrest discovered the Cadia gold–copper porphyry deposit in 1992. Over 8 million ounces of gold have been produced from CVO since commercial production commenced in 1999.

CVO currently has two operational mines—the Cadia East and Ridgeway underground mines. The original Cadia Hill open pit is under care and maintenance. The new Cadia East underground mine commenced commercial production on 1 January 2013. This case study documents the evolution of the waste characterisation and block modelling program for the Cadia pit, the learnings from which have been applied to the current operations. It is an update of a case study that appeared in the first edition of this series (Managing sulphidic mine wastes and acid drainage, Environment Australia, 1997).

Geological setting

The main sulfide species are pyrite and chalcopyrite, with lesser bornite, chalcocite and other accessory minerals, which are present in veins and as disseminations within the host rocks (Holliday et al. 2002). The proportion of waste to ore in the Cadia Hill pit over the life of the mine is approximately 1.4:1. It was estimated that approximately 430 million tonnes of waste rock would be generated during the life of the Cadia Hill pit (Cadia Holdings Pty Ltd 2009a).

AMD characterisation and classification at Cadia Hill

The challenge of accurately identifying PAF waste has been an evolving issue at CVO. Work on AMD characterisation commenced at Cadia as early as 1992 and has been refined through time with ongoing test work.

Between 1992 and 1995, early waste characterisation studies were carried out as part of the Cadia Project environmental impact statement (EIS). The early test work provided the parameters for initial waste modelling for the Cadia Hill pit. The first AMD model was produced in 1995, which estimated that 36% of waste at Cadia Hill would be PAF. This model was revised in 1996, with the estimate of PAF material increased to 38%. The parameters of the models were translated into sulfide abundance cut-offs, which were used for initial in-pit classification of waste at Cadia Hill. A cut-off of greater than 'trace' visually logged pyrite was classified PAF in the Cadia Hill pit, which corresponded to >0.35% pyrite. A summary of the process that was followed to produce the early block model was featured as a case study in the 1997 best practice *Managing sulphidic mine wastes and acid drainage* handbook.

After mining commenced, greater resolution was required for in-pit classification than could be provided by the existing AMD model. As a result, more extensive test work was completed to refine the classification scheme.

A waste characterisation program was completed to validate in-pit classification schemes. Three hundred samples from diamond drill holes were selected using visual logging and assessed by a range of static and kinetic tests. The influences of lithology and alteration on NAG capacity were also addressed. Results indicated a correlation between particular alteration assemblages and PAF classification. A subset of waste rock types was identified for more thorough testing. Those types identified as potential AMD risks included phyllic-altered (pyrite–sericite) faults, which commonly intersected the orebody.

A subsequent more refined program to optimise methods to be used for waste classification was completed in 2002 on over 200 samples collected from four blast patterns, which each contained material that had been classified PAF based on the visual logging cut-off. The program assessed various methods, including the potential use of a field NAG pH test, as a tool for short-term classification.

After considering the results from this program, the original sulfur cut-off of 0.35% was confirmed for in-pit classification of ARD. The cut-off was highly conservative and ensured that all PAF and ‘indeterminate’ samples would be captured by the classification system and managed appropriately. This meant that there was a low risk of PAF waste rock being incorrectly placed in the NAF waste rock dump.

Supporting the in-pit geochemical characterisation work, a kinetic test leach column program commenced in 1999. Over 50 columns were run over the life of the program, including 10 from the Ridgeway deposit. Many columns were run for up to 10 years. The column program confirmed that the phyllic faults were the most likely sources of PAF material at Cadia Hill, with acidic conditions developing within the columns that contained this material. As these fault zones are quite narrow, the need to capture this material as PAF, coupled with the size of the minimum block of waste that could be marked out for mining, meant that there was significant ‘dilution’ of the PAF material with clean waste rock. The column program also demonstrated that materials originally classified as NAF had been correctly identified.

Refining the waste classification scheme

A program of sampling of existing WRDs was completed by 2003 to forensically evaluate the geochemistry of each dump (Williams et al. 2003). The objective of this work was to determine if the classification in place was overly conservative and to provide additional data for refining the classification scheme. WRDs were drilled and samples were retrieved and analysed.

The program showed that ore control practices had been successful in keeping PAF material out of the NAF waste rock dump. However, as anticipated, the conservatism meant that much more NAF material than necessary was being placed on the PAF waste rock dump (figures 1 and 2). Subsequently, a new classification scheme was proposed, with a sulfur cut-off of 0.5% for PAF waste, which forms the basis of current site practice.

Operational classification and management of waste

Cadia Hill waste rock is classified into three different waste types depending on its mineralisation and sulfur content. The three classifications are Blue, Yellow/Green and Pink, representing NAF waste, low-grade ore/mineralised waste and PAF waste, respectively. Waste type is colour-coded to simplify the day-to-day operations.

One of the important operational outcomes from the forensic dump drilling program described above was the implementation of routine sulfur assaying of waste (ore and mineralised waste zones were already assayed for sulfur in addition to selected metals). This removed the imprecision associated with visual estimates of sulfur content and provided the data which could be used to generate a new AMD model. The cost of running the sulfur assays was far outweighed by the savings gained by reducing the amount of PAF waste tonnages (and the requirement for encapsulation) produced as a result of dilution by non-PAF material. Initial calculations suggested that reducing the tonnage of PAF material by a mere 2.5% would pay for all waste blastholes to be assayed (Williams et al. 2003).

Waste rock is sampled and classified on the basis of the sulfur content. A modelled 0.5% sulfur cut-off grade is used to identify PAF material, which was derived from static and kinetic test work, while taking into account the inherent ANC of the materials. The sulfur grade for each block is estimated by spatial modelling of the waste material, using a statistical process known as ordinary kriging. Ordinary kriging uses a weighted assay estimate based on a graph known as a geostatistical semi-variogram. Additionally, where there is a demonstrable geological cause (for example, a phyllic fault is identified), material with a lower sulfur cut-off may be conservatively classified as PAF.

Low-grade ore and mineralised waste (that is, Yellow and Green materials) are stockpiled in separate sections of the south waste rock dump for potential future processing. Blue waste rock is a resource for construction and is either used or stockpiled separately (Figure 3). Pink waste is encapsulated with a combination of a low permeability layer, covered by Blue waste material and topsoil. The low permeability layer is designed to minimise net percolation into the WRD and includes compacted clay layers (on the slopes) and high-density polyethylene (HDPE) on the benches (Cadia Holdings Pty Ltd 2010, 2013).

Between 2005 and 2012, approximately 39% of waste rock produced from the Cadia Hill pit was classified as PAF (Cadia Holdings Pty Ltd, Annual Environmental Management Reports 2006–12, http://www.newcrest.com.au/sustainability/environmental).

Conclusions

The investment in refining the original waste AMD classification scheme over a 10-year period and running routine sulfur assays to improve the block model for mined waste has paid major dividends at this site. The savings gained by reducing the amount of PAF waste tonnages (and the requirement for encapsulation) produced as a result of dilution by non-PAF material have far outweighed the cost of the characterisation program. The higher precision in classifying waste has enabled substantially more NAF material to be segregated for use in encapsulating PAF material and stockpiled for future use as a cover material.

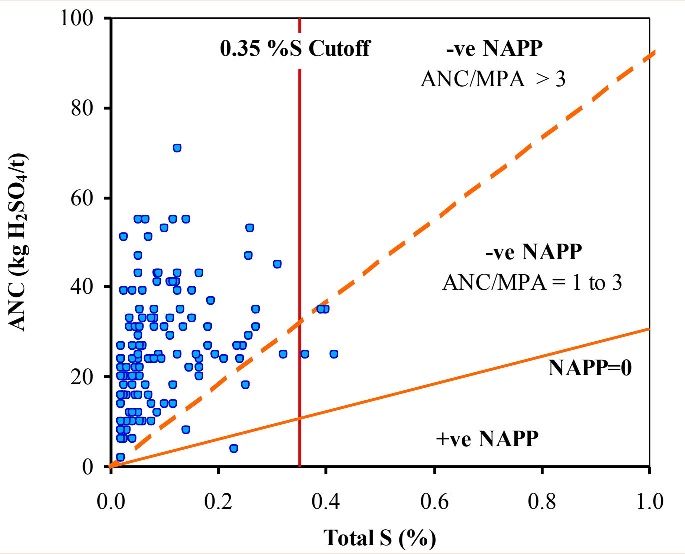

Figure 1: Acid-base accounting of the NAF (Blue) waste rock dump, showing that NAF waste was being correctly classified and placed (Williams et al. 2003)

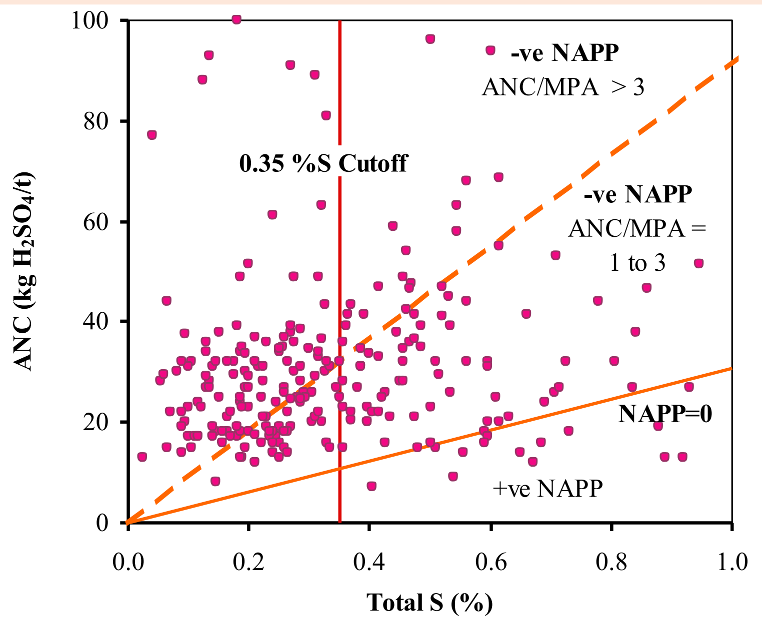

Figure 2: Acid-base accounting of PAF (Pink) waste rock dump, showing a significant proportion of NAF waste (Williams et al. 2003)

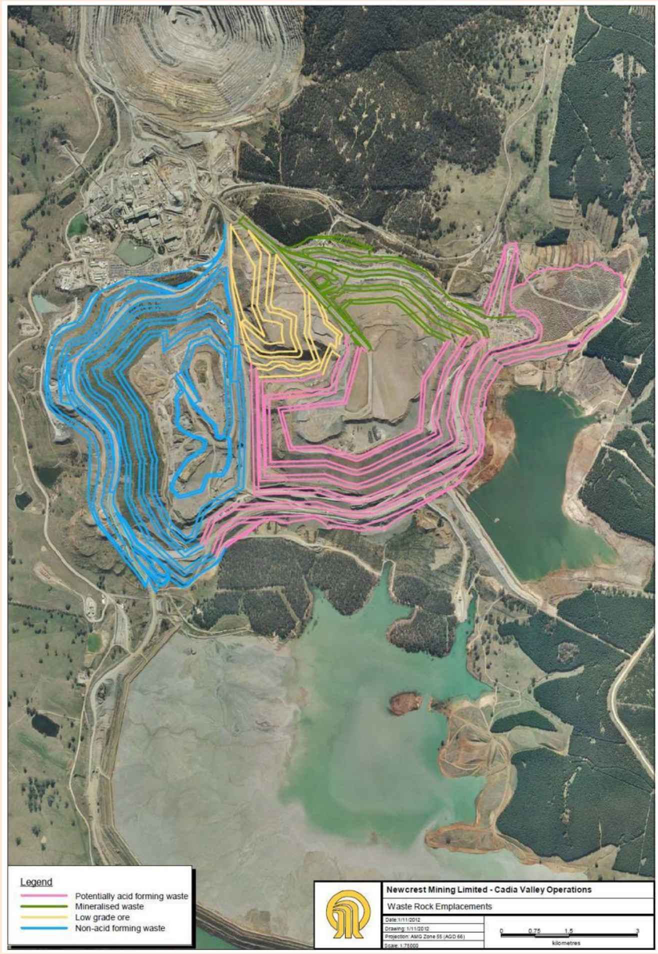

Figure 3: Waste rock dumps at Cadia Valley Operations, coloured by material type (Cadia Holdings Pty Ltd 2014)

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Williams, DJ, Jeffery, J, Gilbert, L, Wilson, GW, Panidis, C, Perry, B (2003). 'A review of the acid rock drainage potential and hydrological implications of selectively-placed waste rock at a gold mine in NSW, Australia', Proceedings of the Sixth International Conference on Acid Rock Drainage

4.6 Scaling-up of laboratory test results

4.6.1 Pilot-scale field trials

The main purpose of field trials is to scale up laboratory tests to better reflect site climatic conditions and particle size distributions and to overcome some of the limitations of laboratory-scale tests, such as preferential flow paths through kinetic leach columns. Field trials may also be used to evaluate the effectiveness of proposed mitigation options, in particular the selective placement of waste, encapsulation, reagent blends and cover systems.

The scale of typical field trials ranges from 1 to 3:

1. Barrel and crib scale leach test lysimeters (100–500 kg) (Figure 16)
2. Test pads (10 m x10 m x 3 m and typically 500–1,000 tonnes) (Figure 17)
3. Trial piles (typically 15 to 20 m pile height) instrumented for monitoring temperature, oxygen, hydrology and seepage chemistry.[28]

Figure 16: Leach test lysimeters at the Phu Kham copper–gold operation, Lao PDR, operated by Phu Bia Mining

Photo: Tim Rowles, Knight Piesold. See case study in Section 6.

28 An example is provided by the Diavik diamond mine in Canada (Blowes et al. 2006).

Figure 17: Test pads constructed at the Brukunga legacy site in South Australia

Photo: Earth Systems.

4.6.2 Large- to full-scale field trials

Well-designed and monitored large- to full-scale trials allow the effectiveness of AMD control measures to be more reliably determined, the need for any additional controls to be quantified, and compliance with regulatory requirements to be demonstrated. Full-scale trials are much less prone to complications from the 'edge effects' or preferential flow paths typically seen in laboratory or small-scale work. In addition, measurements at full scale can be used to test and refine models that incorporate parameters obtained from small-scale trials, leading to greater confidence in predictions of future behaviour based on the data produced by the much smaller scale tests.

In undertaking these trials, it is very important to be aware of the timescales associated with processes that occur at full scale. For example, the response time of a waste rock pile to changes in oxygen supply rate can be hours to days. On the other hand, the time needed for changes in pollutant generation rates in a pile to be detected in samples from a groundwater monitoring well can be years to decades.[29] Given this time context, field trials may need to be set up many years before the information is going to be needed for the design and implementation of, for example, closure strategies to mitigate the risk of AMD production in the long term.

Table 7 lists the most commonly used measurements that are made in full-scale field trials and briefly describes their value.[30]

29 A discussion of timescales associated with different AMD processes in full-scale piles is in Amos et al. (2015).

30 Examples of the deployment of such instrumentation and the use of the data obtained are in Andrina (2009) and Andrina et al. (2003, 2012) for the Batu Bersih pile at the Grasberg mine in Indonesia; (Blowes et al. 2006; Patterson et al. 2006; Ritchie & Bennett 2003; Pearce & Barteaux 2014).

Table 7: Typical measurements made in large- to full-scale field trials

MEASUREMENT	METHOD OF MEASUREMENT	INFORMATION OBTAINED	REMARKS
Pore gas oxygen concentration profile	Sampling tubes and portable gas analyser or online oxygen probes installed in drilled holes or during pile construction	Location of oxidising material; rate of oxidation; dominant gas transport mechanisms	Provides good qualitative and quantitative information; rapid response to change in conditions in pile.
Temperature profile	Thermistor strings installed in drilled holes or during pile construction	Location of oxidising material; rate of oxidation	Provides good qualitative information but difficult to quantify; slow response to change in conditions in pile.
Water infiltration rate	Lysimeters	Pollutant transport rate through the pile; effectiveness of cover systems	Debate continues over designs and data interpretation; may take years to collect meaningful data.
Chemical composition of drainage	Surface water sampling and groundwater piezometers	Pollutant concentration and loads released from the pile	Large amounts of data; very slow response to change in conditions in pile.

While the instrumentation and measurement protocols are relatively straightforward, the interpretation of the field data generally requires expert understanding of a complex set of interrelated physical and chemical processes.

4.7 Estimating and modelling pollutant generation and release rates

4.7.1 Overview

There are no simple methods for extrapolating laboratory test results to large-scale mine site waste containment facilities to predict the concentrations of solutes that will be produced (Pearce et al. 2015). While this issue of scaling may ultimately be addressed by the findings from large-scale trials of the type described above, such trials might take years to yield definitive results. In the absence of such scale-up data, there are two complementary approaches that can be followed to estimate the load of solutes (oxidation products/acidity load) that may be produced by mine materials.

The first approach uses empirical results from static and kinetic tests to infer the relative extents and rates of AMD generation from various domains across the mine site. This information can be vital for identifying high-risk mine material and for initially developing appropriate AMD management strategies.

The second approach uses computer models with varying degrees of sophistication to simulate the complex physical and chemical processes that take place in mine materials when they are exposed to atmospheric conditions. Process rates vary with position and over time in a waste containment facility. There are also feedback mechanisms between many of the processes, leading to counterintuitive system behaviour. Computer models therefore need to clearly state assumptions and may need to be very sophisticated to predict AMD behaviour with a level of confidence similar to that provided by the empirical approach. When field monitoring data is available, computer models should be calibrated and refined to produce a more accurate and reliable predictive output over time.

4.7.2 AMD prediction using empirical test results

A useful approach to estimating the rate of pollution generation from a site, and to prioritise the most significant likely sources of AMD for management, is to determine the maximum annual acidity generation rate or 'acidity load' (tonnes H_2SO_4 per unit time) for each waste unit or mine domain. The acidity load provides an overall measure of the scale of the AMD issue, as it includes both acid (H^+) and metal contaminant loads. It can be estimated directly based on tonnages and the results from static and kinetic geochemical tests of mined materials.

It is possible to estimate the maximum possible annual acidity generation rate of any bulk mine material (kg H_2SO_4/t/yr) using the pyrite oxidation rate applicable to the material's lithology or weathering type (normalised to the sulfur content, weight% FeS_2/yr) and the mass and average pyrite content of the bulk material.

For example, if 1 Mt of waste rock with an average pyrite content of 3 weight% FeS_2 and a pyrite oxidation rate of 0.5 weight% FeS_2/yr were completely exposed to atmospheric oxygen, the estimated maximum acidity generation rate would be approximately 250 t H_2SO_4/yr for all the waste rock, or 0.25 kg of H_2SO_4 per tonne of waste rock each year.

While the maximum rate of acidity *generation* is relatively straightforward to estimate in this way, predicting the rate of acidity release is more complicated, as only a proportion of the waste rock at the site may be exposed to atmospheric oxygen. The rate of release depends on factors such as:

- climatic conditions and the site water balance
- the hydrogeological characteristics of the mine materials and surrounding environment
- complex weathering and mineral–solution geochemical processes.

Rates for weathering and mineral–solution geochemical processes vary with position in a pile and over time. There are feedback mechanisms between many of the processes, leading to non-linear and sometimes counterintuitive system behaviour.

As noted above, the simplest measure of pollutant *release* rates is the acidity load (tonnes H_2SO_4 per unit time), which can be estimated based on maximum pollutant generation rates combined with climate, site water balance and hydrogeology data.

When the pollutant release rate is known, leachate chemistry data from kinetic geochemical test work can then be used to provide (back-calculate) indicative estimates of drainage/seepage water quality from various site domains.

4.7.3 AMD prediction using computer models

Various modelling approaches have been developed with the aim of simulating the complex range of physical and chemical processes driving oxidation and mineral–solution geochemical processes (Linklater et al. 2005; Linklater et al. 2006; Blowes & Frind 2003).

Reactive-transport models can help in investigating the complex behaviour of AMD generation, migration and evolution at mine sites. They need to take into account sulfide mineral oxidation, gas transport, heat transport, water and solute movement, and neutralisation processes involving the dissolution of carbonates and aluminosilicates. These models require site-specific input data, which is often sparse or poorly defined both spatially and temporally, at least in the early stages of the project's life. Such models are most useful as tools to:

- test hypotheses and run ‘what if’ scenarios to compare the relative, rather than absolute, effectiveness of different AMD control strategies, over the short and long terms
- determine the sensitivity of the system to particular input and design parameters
- help interpret field monitoring data
- provide time-dependent input to assess the ecological risk that may be posed by discharges to aquatic environments.

Many reactive-transport models have been developed, with varying capabilities and levels of complexity. Some take a sequential (non-coupled) approach in which the transport and reaction processes are solved separately, with or without iteration between the steps. Others take a one-step approach in which the equations describing physical transport and geochemical reactions are solved simultaneously.

The predictive accuracy of complex water-quality models is often limited by the quality of input data and the uncertainty or complexity associated with modelling complex geochemical processes. Nevertheless, some models have the potential to aid high-level AMD management decisions if appropriate sensitivity analysis is conducted, if they are combined with knowledge of pollutant generation and release rates estimated from the kinetic test methods described above, or both.

However, it may not be necessary to use a complex reactive-transport model to predict the likely effectiveness of different AMD control strategies, especially for fresh material that has not significantly oxidised. Because oxygen and water are the two exogenous reactants that have to be present for sulfide oxidation to occur, models that are capable of predicting the extent of oxygen and water ingress into waste rock and tailings may provide all that is needed to compare the efficacy of alternative management strategies. For example, in the case of fine-grained tailings, maintaining water saturation above 85% will largely eliminate oxidation (Bussière 2007).

Likewise, maintaining a similar level of saturation in the fine-grained clay layer that is part of a waste rock cover will prevent the ingress of oxygen and thus minimise the generation of AMD. Store-and-release covers function by minimising the amount of net infiltration and hence the amount of water available to transport reaction products. Therefore, a model that can predict the water balance under unsaturated flow conditions will enable the performance of such a cover system to be assessed.

Models that have been used to predict different aspects of the AMD generation process are documented in Section 5 of the GARD Guide (see Glossary). When selecting the model to use, three high-level questions and issues should be considered:

1. Has the model been widely used and tested/validated?
2. Does the regulator have a preference for a particular model or models, based on familiarity and in-house experience?
3. While proprietary modelling codes may be technically sound, they might not be accessible for review by regulators or independent third parties.

5.0 ASSESSING THE RISK POSED BY AMD

Key messages

- AMD can present a substantial risk of incurring potentially very large costs if it is not identified early and managed appropriately.
- Risks can include environmental, human health, financial, regulatory and reputational risks.
- A useful risk management strategy is to rank potential AMD hazards and then develop protocols to manage them.
- The preservation of agreed environmental values is a key principle driving the management of AMD.

The overall assessment of project risk is a much broader topic than can be addressed in this handbook, and is treated in detail elsewhere.[31] However, history and experience in the global mining industry have shown that AMD can be a substantial, high-priority risk with potential for reputational damage and large incurred costs if it is not identified early and managed appropriately.[32]

In Australia, as in other countries, corporate governance law requires a company to identify, evaluate and manage all significant risks that it faces. A prudent approach is to develop a focused AMD risk review program that starts by asking two key questions:

- What hazards does AMD pose to the mine operator?
- Is the operator managing those hazards to minimise environmental, human health, financial, regulatory and reputational risks?

Many mining companies assess the risks posed by AMD as part of project development and as part of closure planning. However, there are few published examples of processes used for those assessments.

Kinross Gold Corporation in the United States incorporates the results from AMD geochemical test work into its G4 mine planning process (Williams et al. 2015). The G4 process integrates geological, geochemical, geometallurgical and geotechnical data streams into resource block modelling to optimise the available reserves and to minimise AMD risk.

Rio Tinto uses the corporate AMD risk assessment approach described in Section 5.1 to identify risks (the case study in that section shows how), to assess existing projects and to help design new projects so that the next generation of mines has the best possible chance of managing AMD effectively and contributing to economically sustainable development.

31 See the *Risk management* leading practice handbook (DIIS 2016f).

32 Techniques to identify the potential for AMD and its likely magnitude are described in Section 4.

In describing the results of Rio Tinto's review of AMD risk, Richards et al. (2006) made the important point that compliance with government regulations and permit conditions does not necessarily guarantee that AMD is being managed in the most practical, robust and cost-effective way. Rio Tinto's review highlighted several types of issue that had typically been addressed inadequately, resulting in a level of risk higher than had previously been assumed. It is likely that those issues need more management attention throughout the mining industry. They are:

- the geochemical characterisation of materials
- the monitoring of potential groundwater impacts
- the management of groundwater impacts
- waste rock segregation
- cover design
- flooding of workings.

Given the broad range of issues that need to be addressed in assessing AMD risk, a mining company usually needs the services of experts in the field, at least for the first iteration, to ensure that all key aspects of risk are identified and properly considered.

5.1 Risk and liability—lessons from a corporate review

In 2003, Rio Tinto began an AMD risk review program covering all its operations. In 2006, Richards et al. (2006) described the method developed for the reviews and reported the main findings from its application over the first two years. The approach used and lessons learned from this program are still relevant and can be applied in mining companies of any size, leading to positive sustainable development outcomes.

Rio Tinto used a two-stage approach. The first involved the development of a hazard screening protocol to rank the potential AMD hazard posed by a given mining operation, based on the physical and chemical setting of the site. Broad issues examined as part of the assessment were assigned numerical values (weights) that were combined into a final hazard score. As shown in Table 8, those issues were geology (45%), incipient AMD risk (5%), scale of disturbance (25%), transport pathways (10%) and sensitivity of the receiving environment (15%).

The second stage involved applying a protocol to assess how effectively the operation was managing its AMD hazards and how it was reducing overall risks. To minimise future liabilities, the protocol identified latent as well as current issues. It paid particular attention to the long-term implications of management strategies and practices.

The risk review protocol examined 11 key performance areas that covered all aspects of successful AMD management (Table 9). As in the screening procedures, the individual elements were combined into a holistic approach to characterising and managing AMD. Such assessments should be applied to all sites and site materials, including waste rock, pit walls, tailings, and construction materials derived from sulfide-containing waste.

Table 8: Factors used in the Rio Tinto hazard screening protocol

BROAD ISSUE	FACTOR	WEIGHT
Geology	Ore deposit type	30%
	Host and country rock neutralisation potential	*10%*
	Known AMD issues on site	5%
Incipient AMD risk	Time since last major operational change	5%
Scale of disturbance	Total waste stored on site	15%
	Footprint of disturbed area	10%
Transport pathways	Water availability	7%
	Metal released to the environment[a]	3%
Receiving environment	Proximity of surface water bodies	5%
	Alkalinity of water body or groundwater	5%
	Proximity of protected or inhabited areas	5%

[a] The dissolved flux of metals discharged to the environment through approved permitted discharge points and approved operating practices.
Source: Richards et al. (2006).

Table 9: Key performance areas and elements used in the Rio Tinto AMD risk review protocol

KEY PERFORMANCE AREA	ELEMENT
Site baseline characterisation	Characterisation of existing mine wastes
	Climate
	Hydrology and hydrogeology
	Surface water and groundwater chemistry
	Ecosystem characterisation
Waste material and wall rock characterisation	Geologic setting
	Geochemical characterisation of rock masses and process wastes
	AMD geochemistry of pit walls and workings
	Physical characteristics of wastes
Materials management	Integration of AMD characteristics into mine planning
	Design of waste disposal facilities
	Waste material management
AMD generation processes	Sulfide oxidation
	Oxygen transport
	Oxidation products and in situ chemical reactions
	Infiltration and internal water movement

KEY PERFORMANCE AREA	ELEMENT
AMD migration pathways and fluxes	Surface water discharge and contaminant loading
	Groundwater flow and contaminant flux
Potential receiving environments	Assimilative capacity of the receiving environment
	Ecological sensitivity of the receiving environment
Integrated conceptual understanding	Conceptual models
	Numerical models
	Development of performance and closure criteria
AMD mitigation program	Mitigation strategy
	Implementation of the mitigation strategy
Monitoring and ongoing assessment	Monitoring strategy
	Data management and assessment
	Feedback mechanisms for management
Management skills and resources	Clear accountabilities and roles
	Institutionalised procedures and information management
	Resourcing
Stakeholder relationships	Stakeholder relationships

Source: Richards et al. (2006).

Rio Tinto's screening protocol has now been applied to all of its operations as well as all potential projects. It can also be applied to potential acquisitions to assess the level of contingent risk posed by AMD. Only those sites with an assessed low AMD are not reviewed in detail.

The protocol has now also been adapted for use at Rio Tinto's Pilbara operations. Green and Borden (2011) noted the potential for sulfidic material to be exposed by mining at those sites (as documented in Case study 4). For such operations, a four-stage process has been developed in which each of the two stages identified above has been divided in two to account for the specific circumstances at the sites. The initial hazard screening protocol involves a preliminary and detailed assessment, while the following risk assessment process involves assessments of technical and management strategy

Case study 4: Management of PAF shale waste based on the Rio Tinto AMD risk assessment framework, Rio Tinto Iron Ore, Pilbara region, Western Australia

Context

AMD, as well as spontaneous combustion (self-heating) problems, is known to be associated with the iron ore deposits that are mined in the Hamersley Province of Western Australia. Carbonaceous and sulfide-bearing wastes (pyritic black shale and lignite) pose both an AMD and self-heating risk.

Management of pyritic black shale and lignite at all Rio Tinto Iron Ore (RTIO) sites in the Hamersley Province, is carried out in accordance with a spontaneous combustion and acid rock drainage (SCARD) management plan. The plan's management strategy is broadly based on the following principles:

- identification of hazards and resultant risks to the environment
- identification of opportunities to manage or avoid AMD before mining an area has commenced
- provision of a rigorous basis for decision-making and planning
- evaluation and prioritisation of risks and identification of management measures to mitigate risks
- reduction of risk and operational expense
- enhancement of due diligence studies, governance, stakeholder relationships and business reputation
- minimisation of long-term post-closure risks, liabilities and environmental impacts.

AMD management strategy

RTIO has initiated a detailed AMD mitigation and management strategy that aims to preserve the environmental values of the regional water resources. The strategy includes the following aspects:

- quantification of background surface and groundwater quality and the potential release of contaminants into groundwater from each waste facility
- monitoring of groundwater and determination of groundwater flow patterns and mass transport
- geochemical characterisation of pyritic black shale, lignite and other mined lithologies
- waste rock pile and pit wall source term evaluation from both in situ and ex situ characterisation
- optimisation of cover design via modelling and monitoring of trial cover systems and future waste rock dumping strategies to minimise the overall risk of AMD.

Preservation of environmental values requires knowledge of the natural background variability of water resources, as well as how they are used regionally. AMD management programs aim to quantify the potential release of contaminants into ground or surface waters and implement mitigation strategies, if necessary, to reduce risk to environmental values.

RTIO has guidelines for assigning sulfide risk within the geological block model (Green and Borden 2011) . Five levels of risk are assigned. Risk category 0 indicates that the material poses no risk; category 1 waste material has a low risk for AMD; category 2 has low to moderate risk; category 3 has high risk for AMD and spontaneous combustion potential; and category 4 contains neutralising material. Pyritic black shale with a high sulfur content and lignite are assigned to category 3.

The AMD potential of the material is assessed from measurements of intrinsic oxidation rate, acid base accounting and kinetic column studies. At one site, a range of in situ measurements have also been carried out in waste rock piles. These measurements are used to quantify the potential for the generation and release of contaminants from each waste facility.

Groundwater models are developed for sites that mine below watertable. The models can be used to assess the influence of final pit voids on groundwater movement in regional aquifers and whether the mine site has the potential to influence water receptors (such as permanent surface water bodies and water extraction bores used for livestock) in the region.

Conclusions

Mineral waste management at the RTIO Pilbara operations involves an integrated approach. Accountabilities and actions for all groups that work with sulfidic wastes (black shale and lignite) are clearly described in the management plan. Implementation of the plan at each operation ranges from initial characterisation and modelling through project development, mine planning, production and closure. The objectives of the planning are not only to quantify risk to environmental values but to mitigate the potential for risks to occur.

Figure 1: Footwall zone (left) to Mount McRae (pyritic black) shale (right)

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5.2 AMD risks at brownfield sites

Brownfield sites are those sites that have been previously mined and are either being reactivated by the original owners after a period of care and maintenance, or are being economically reassessed or operated after having been acquired by a new owner. The AMD risk profile of such sites can be assessed by applying a protocol of the type described above. However, in the case of sites that have not been operated for many years, and where substantial oxidation of mined waste may have occurred, extra care needs to be taken in the geochemical characterisation methods (Section 4) that are used to quantify the extent of risk and the mitigation measures that may be required.

In addition to the technical issues that need to be resolved for the resumption of mining and mineral processing, regulators and the new operators often discuss how the past legacy is to be addressed. This occurs most often in the context of responsibility for managing existing waste and AMD impacts on the receiving environment. The resolution may be clear-cut when the new operator assumes full responsibility for the legacy, having accounted for the cost in its pre-acquisition feasibility assessment. However, in some circumstances a shared or cooperative approach to addressing the legacy may be judged to be the best outcome from a sustainable development perspective. Case study 5 is an example of how collaboration between the state and a mining company has allowed a profitable continuance of mining production, coupled with effective management of the AMD legacy, at a site that could not otherwise have been remediated with the resources available to the state.

Case study 5: Public–private partnership to facilitate ongoing economic operations and to improve closure prospects for a brownfield site, Savage River Mine, Tasmania

Context

The Savage River magnetite mine (Figure 1) and the Port Latta pelletising plant (70 km away on the coast) in Tasmania have been in almost continuous production since 1967.

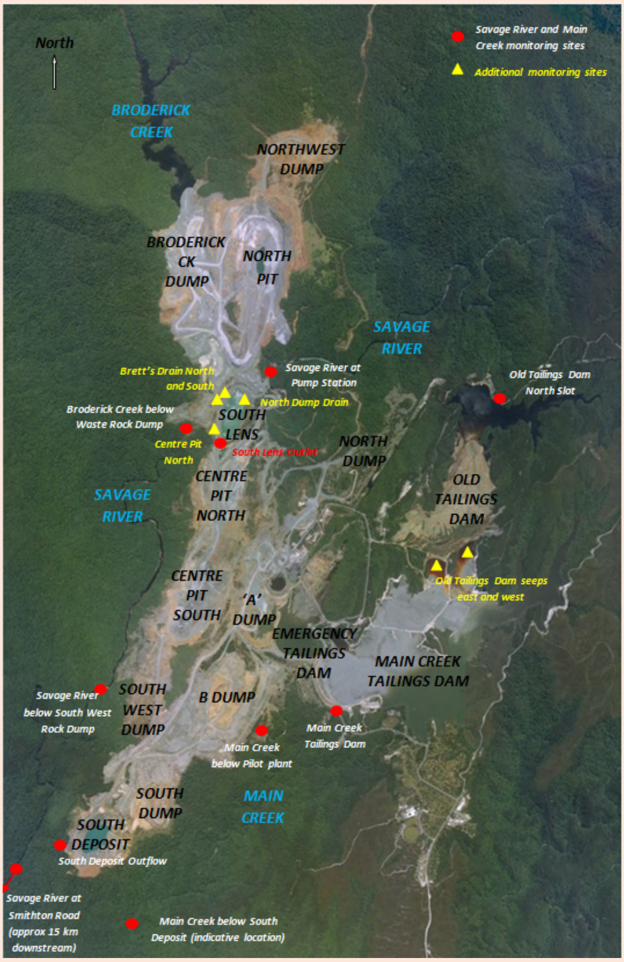

Figure 1: Site map showing key locations

The site was operated by Pickands Mather International from 1967 until the mine closed in early 1997. The mine was returned to the state government in March 1997, together with a rehabilitation bond of $11.4 million. It reopened in late 1997 under the management of Australian Bulk Minerals (ABM) and has operated continuously since, with ABM merging with Grange Resources Limited in January 2009.

Pickands Mather International left behind a significant environmental legacy. By the end of 1996, the two streams that traverse the mining lease, Savage River and Main Creek, were adversely impacted by mining. AMD from the historical WRDs and old tailings dam was the main source of the water-quality impacts on Savage River. Monitoring data showed high metal concentrations in Savage River for a distance of 30 km downstream of the mine site. The median copper concentration just downstream of the mine site at that time was over 25 times the ANZECC–ARMCANZ (1992) recommended value for soft waters, with maximum copper concentrations 3–5 times higher than the median concentrations.

The landmark Goldamere Act was enacted by the Tasmania Government to provide the new operators of Savage River Mine with an indemnity against the pre-existing AMD. The Goldamere Agreement set out the terms and conditions of the sale of the site (from the state government to ABM), with the agreed purchase price to be paid through the implementation of in-kind environmental remediation works to a value of $13 million, targeting legacy AMD. The remediation works to be undertaken were to be agreed to by both ABM and the Crown. The Savage River Rehabilitation Program (SRRP) was formed to oversee this process. The group overseeing the operation of the SRRP comprises representatives from Grange, the state Environment Protection Authority (EPA) and Mineral Resources Tasmania. The objective of the SRRP is:

> *Long-term remediation of environmental harm resulting from pre-1997 operations at the Savage River mine to the greatest extent possible, with a major focus on restoration of the aquatic environment downstream of the mine site.*

AMD management works on site

To achieve the SRRP objectives the following projects have been undertaken (Kent 2008):

- centralised passive water treatment, prior to discharge of the treated supernatant to Savage River, in the south lens pit (Figure 1, centre), using alkaline water produced in the active north pit from calcium chlorite schist passively crushed by haul trucks and leached with rainwater
- construction of a pipeline to divert and treat AMD
- placement of a water-shedding, compacted clay cover over B-dump to reduce the production of AMD run-off and seepage, with armouring of the sides of the dump with net alkaline-producing rock to generate additional alkalinity
- development of alkaline flow-through(s) (described below)
- passive treatment of acidic seepage using alkaline tailings.

The Broderick Creek WRD is the main active dump. Construction started in the mid-1980s. The dump is being built over an existing creek channel, which is being progressively converted to a flow-through, alkalinity-producing system prior to the placement of the PAF waste rock (D-type; Figure 2). This is achieved by tipping blocky, net alkalinity-producing waste rock into the creek and allowing the larger material to rill to the bottom. Creek water passing through this placed waste rock picks up alkalinity and delivers it to Savage River. Concentrations of alkalinity in the outflow have increased from about 70–250 mg/L as the dump has increased in size, with a concomitant increase in the path length of the flow-through system. Like the south lens, this alkalinity contributes to neutralising the remaining fugitive sources of AMD in Savage River.

Additional information about materials characterisation and the construction of the waste rock dumps and cover systems is provided in Hutchison and Brett (2006).

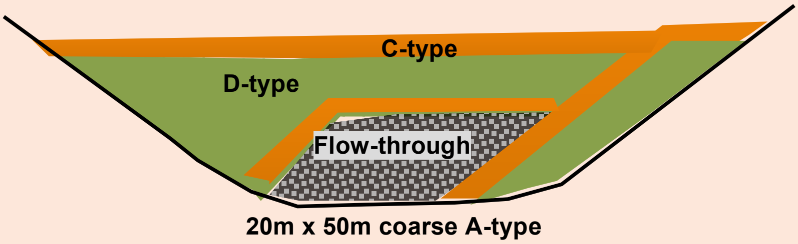

Figure 2: Broderick Creek flow-through: simplified cross-section showing A-type coarse-grained alkalinity-producing waste rock, C-type weathered clayey waste rock, and D-type encapsulated PAF waste rock

Performance monitoring

As shown in Figure 3, the remediation works carried out since 1997 have substantially reduced copper concentrations through time in the Savage River directly downstream of the mine (SRbSWRD in Figure 3 and Figure 4) and further downstream near the confluence of the Savage River with the Pieman River (SRaSR in Figure 3). Note that a logarithmic concentration axis has been used on Figure 3 so that the range of concentrations across four sites can be compared.

Most notably, the diversion of the north dump drain has seen a large reduction in copper 'upstream' of the mine (SRaPS in Figure 3).

In 2012–13, the median copper concentration in the Savage River directly downstream of the mine at the confluence of the Savage River with the Pieman River was 26 μg/L. This represents a five-fold reduction in copper concentration since the start of the remediation program, which is a substantial achievement. However, the median copper concentration is still five times the trigger value for the protection of aquatic species at the 99% level for hard water (ANZECC–ARMCANZ 2000), indicating the need for more work to be done.

Site-specific toxicological test work completed in 2001 and 2002 found that copper toxicity is reduced by the calcium and alkalinity present in Savage River water. The work established target copper levels over a range of calcium levels, and identified two toxicity targets—one that is sufficient to protect fish and other hardy aquatic fauna, and another that would protect more sensitive aquatic invertebrates. Since 2012, both these toxicity targets have been consistently met (Figure 4), demonstrating the effectiveness of the remediation works in reducing inputs of AMD to the river directly downstream of the mine site. The success in meeting these toxicity targets and the general improvements in the river have been further demonstrated through independent biological assessments of the Savage River.

Further details of the works that have been carried out to manage AMD issues on site and improve downstream water quality are provided in a recent comprehensive review (Williams et al. 2014).

Continuous improvement

The future construction of a new tailings facility in Main Creek, south of the current Main Creek tailings dam, will capture and neutralise seepage from B-dump and offers the SRRP the opportunity to continue the passive neutralisation of the acidity and metal load being produced from the old tailings dam for a further 20 years. The new tailings facility will also feature an alkaline filter face and flow-through system that will allow water to pass through the dam wall and pick up further alkalinity to add to Main Creek.

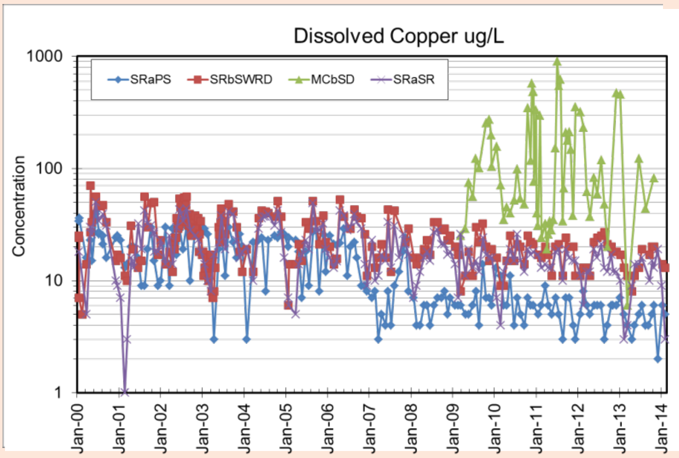

Figure 3: Copper concentrations at key sites since January 2000

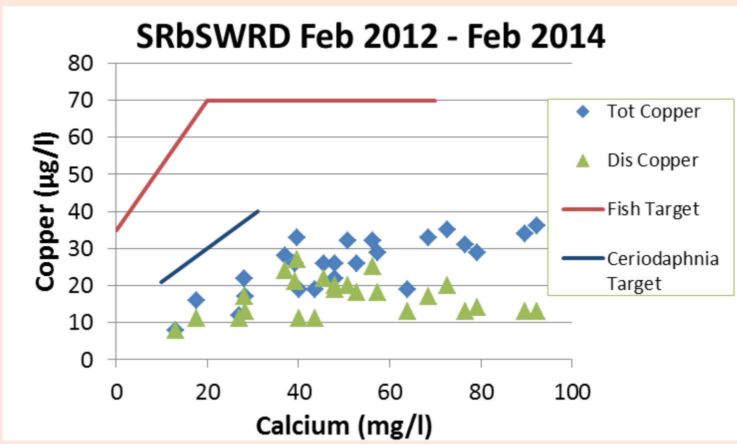

Figure 4: Comparison of total and dissolved copper concentrations in Savage River at south-west waste rock dump with site-specific toxicological targets. SRbSWRD = Savage River directly downstream of the mine

Conclusions

Following the implementation of the Goldamere Agreement, the relationship between the state and Grange (formally ABM) has not always been smooth. The early days provided significant challenges, with mistrust, conflict and differing value judgements coming between the parties. Grange had to earn trust by demonstrating itself to be a good environmental operator. In doing so, it has reaped the benefits of the Goldamere Agreement, has established a healthy long-term relationship with the government, and has made the introduction of equity partnerships and new business owners significantly easier as once potential environmental liabilities can be described as assets to the business. The operation of an environmentally sound operation also makes the recruitment of young professionals much easier in a human resource competitive world (Kent 2008).

The state has also benefited greatly. Of the original \$23.4 million remediation fund (including the \$13 million provided by Pickands Mather International), about \$20 million remains available for future works/treatment when the site eventually closes. In the meantime, a substantial portion of the legacy AMD has been treated or continues to be passively treated, raising the prospects that Savage River can ultimately be successfully rehabilitated and the site handed back to the state.

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5.3 Identifying and protecting key environmental values

5.3.1 Identifying values

Water is the focus of this section, as it is the medium most affected by offsite escapes of AMD. Environmental values for water are values that define the end use of the water resource. Determining those values for a mining project is the first stage in assessing the level of risk posed by AMD and the extent of management action that may be required to reduce the risk to an acceptable level. Irrespective of the jurisdiction, early and effective identification of those values that are of most concern to regulators and local communities, and the development of agreed strategies to address those concerns, reduce the subsequent possibility of reputational damage or challenge to the project.

For example, water values may be defined for the maintenance of a sustainable and functional ecosystem (for example, wildlife or fisheries values) or for public utility (such as irrigation) or human health (for example, potability). Groundwater uses may be more difficult to define, but should include consideration of the protection of stygofauna (if any), groundwater-dependent ecosystems and other surface expressions of water (such as springs), as well as potability and industrial uses if they are relevant.

The guiding principles relevant to the determination of water values and the consequent selection of water-quality criteria should be the preservation of the highest existing or agreed future values. Consequently, mining activities should not lead to degradation of surface water or groundwater quality that would prevent the maintenance of the highest water-quality value agreed upon with stakeholders or otherwise required for compliance (Batley et al. 2003). This does not mean that there should be no measurable changes in water-quality parameters, but rather that the extent of change should not result in degradation of water quality to the point where the existing environmental value is placed at significant risk. This principle is consistent with government requirements and other stakeholder expectations, as well as the sustainability aspirations of leading mining corporations. It also allows flexibility in applying or deriving water-quality guideline values based on the geochemical and social/environmental baseline conditions relevant for the particular mine site.

A number of environmental values have been specifically defined in the *Australian water quality guidelines* (ANZECC–ARMCANZ 2000a). They include aquatic ecosystems; primary industries (irrigation and general water uses, livestock watering, aquaculture and human consumption of aquatic foods); recreation and aesthetics; cultural and spiritual values of drinking water; and industrial uses, and guidelines for water-quality criteria for the protection of aquatic ecosystems, primary industries and aesthetic values are given. Further details about the derivation of water-quality guidelines and their application are provided below. Criteria needed to meet drinking water standards are given in the *Australian drinking water guidelines* (NHMRC–NRMMC 2011). Criteria for managing AMD-affected waters used for recreation, such as pit lakes, are in *Guidelines for managing risk in recreational water* (NHMRC 2008).

In general, guidelines for the protection of aquatic ecosystems are more stringent than those for drinking water, which in turn are more stringent than those for primary industries and recreation. No water-quality guidelines are provided for industrial uses of water, which should be determined on a case-by-case basis. Cultural and spiritual water-quality values are not defined either, and should be determined in consultation with relevant social groups.

In all cases, the values and consequent level of protection should be defined by discussion with relevant stakeholder groups before mining begins, so that the risk profile of the project can be fully defined. Mining operations of all types—at greenfield and brownfield sites as well as legacy sites—need to undertake this process.

5.3.2 Guidelines for the protection of environmental values

If aquatic ecosystem protection is the agreed highest value, then the aquatic ecosystem protection guidelines for preserving aquatic biota diversity and ecosystem function (ANZECC–ARMCANZ 2000a) are those that will most likely be initially applied by regulators throughout Australia and New Zealand to specify management or compliance objectives for waters that may be affected by AMD (Batley et al. 2003). However, because the ANZECC–ARMCANZ (2000a) framework is highly regarded internationally, it may be expected that the application of such an approach would be viewed favourably in other jurisdictions.

Table 3.4.1 in ANZECC–ARMCANZ (2000a) summarises the water-quality guidelines for freshwater and marine ecosystems. Guidelines for irrigation, general use, livestock drinking water, aquaculture, recreational use and aesthetics are in ANZECC–ARMCANZ (2000c).

The water-quality guidelines criteria listed in ANZECC–ARMCANZ (2000a) are presented in the context of a number of generic value categories, reflecting the level of protection needed to sustain those values. A framework is also provided for an innovative risk-based approach that can be applied to derive water-quality criteria to protect specific local values.

The concentration values listed in ANZECC–ARMCANZ (2000a) are defined as 'trigger values'. In practice, the trigger values are typically compared to concentrations of solutes measured in filtered water samples. They are designed to trigger further investigation and/or management actions if they are exceeded. An exceedance of a guideline value by a small margin does not, by itself, mean that an adverse impact has occurred, as there is a degree of conservatism built into the derivation of the values. If a trigger value is exceeded in a waterway that is affected by discharge of AMD, then the primary management action should be to stop that exceedance by preventing the discharge or treating the source water). The second and subsequent actions should be to determine whether an adverse impact has occurred, as this consequence is the one that will be of most interest to regulators and other stakeholders.

Since aquatic ecosystems are likely to represent the highest use value for waters affected by AMD in Australia, further discussion about how different levels of protection are derived and applied is warranted. Water-quality guideline values for the protection of aquatic ecosystems are often more stringent than those for many other end-use values, so if the aquatic ecosystem values are protected then other less demanding water-quality values are likely to also be protected.

The water-quality guidelines provide for three levels of aquatic ecosystem protection for sustained (chronic) exposures to potential toxicants:

- high conservation/ecological value ecosystems (99% of species to be protected)
- slightly–moderately disturbed ecosystems (95% of species to be protected)
- highly disturbed ecosystems (80–90% of species to be protected)

Guideline trigger values would be likely to be assigned at the 95% level for many greenfield mine sites because the ecosystem is likely to be slightly to moderately disturbed. Lower protection levels, such as 90% or even 80%, may be considered for sites previously disturbed, such as by grazing or historical mining, with already lowered water-quality values as a consequence. The level of protection to be assigned will need to be discussed and agreed with stakeholders, as this decides the water-quality guideline values that will be applied.

The guideline values are derived from the most robust and relevant biological dose–response data available. However, they are by necessity generic. Therefore, they may be overprotective (or even not protective enough) under some conditions. The ANZECC–ARMCANZ (2000a) framework provides tools to modify the generic values to take into account regional, local or even site-specific conditions (such as for discharges to individual rivers or lakes). For example, modifying factors (where available) can be applied to take into account local pH and water hardness. The extent of tolerance of local aquatic species to specific toxicants can also be taken into account. However, this can be quite a lengthy and costly process to implement. Specific examples of the derivation of local water-quality guidelines are provided in van Dam et al. (2014).

The guideline values have been derived from dose–response relationships for chronic exposures to toxicants. However, they do not implicitly address the effects of biomagnification, whereby the concentrations of a toxicant increase up the food chain. Thus, a water-quality guideline value that protects the health of organisms at lower trophic levels might not protect the health of organisms (aquatic or terrestrial) at higher trophic levels. Special consideration therefore needs to be given when assessing the risks posed by AMD containing one or more of the elements Hg, Se, Cd, and to whether an additional factor of safety needs to be applied (ANZECC–ARMCANZ 2000b).

Note also that the guideline values presented in ANZECC–ARMCANZ (2000b) for regional physico-chemical stressors (pH, salinity, temperature, turbidity, nutrient and oxygen dissolved concentrations) do not incorporate toxicity-test data at all. Rather, they represent the 80th percentile of a number of historical datasets obtained across the region. Therefore, it should not be assumed that adherence to those regionally derived values will be consistent with aquatic ecosystem protection at an agreed level for a project.

Surface water quality data obtained from baseline (reference) sites (sampled before disturbance and/or upgradient of mining influences) should be used to inform the setting of water-quality objectives and criteria for operational compliance and closure. Ideally, such reference sites should be monitored simultaneously with sites potentially influenced by mining activity to enable natural upstream variations in water quality to be distinguished from those caused by mining.

The maintenance of water-quality values must be considered not just for the operational mining phase but also in the long term following closure. Appropriate objectives and measures of performance should be defined in closure criteria. Guidance on the type of operational and closure monitoring required is provided in Section 9.

Surface waters and groundwaters of catchments containing mineral resources often show elevated baseline solute concentrations due to their geologic provenance. Thus, from an AMD risk assessment and management perspective, local water quality may be unique to the extent that site-specific water-quality guidelines may be needed (McCullough & Pearce 2014) to avoid the application, by default, of unrealistically low generic protection criteria. In this context, the first comparison will typically be made by stakeholders with the default water-quality guideline criteria applying to the agreed level of protection. Site-specific variations will only be able to be considered on the basis of the quality of available evidence (for example, a robust pre-mining water-quality baseline dataset, or abundance and diversity data for local aquatic organisms).

Alternative approaches to monitoring and managing water quality may be required where baseline data is not available and/or where specific types of aquatic ecosystems with unique characteristics necessitate the application of other than the generic guidelines to protect against the risk of AMD toxicity. In such cases, ecotoxicological assessments using relevant site waters and local species may be needed.

Leading practice assessment of the potential effects of discharges of AMD-affected water goes beyond just reporting and complying with default water-quality guidelines that have been agreed with stakeholders. It will include the development of, and reporting against, a multiple lines of evidence approach (including biological indicators where appropriate) to demonstrate lack of impact or significant risk.

5.4 Framework for assessing AMD risk to the environment

5.4.1 Defining the hazard posed by AMD

Assessment of AMD risk in Australia will most often be directed towards the protection of environmental receptors, as they are typically the most-sensitive receptors. However, engagements with stakeholders should be used as opportunities to discuss which site-specific aspects (for example, technical, environmental, legal, corporate, financial, regulatory and social) also need to be addressed in the risk assessment.[33]

Environmental risk assessment is a useful tool to understand and manage AMD risk, as it considers a contaminant–transport–receptor (CTR) model that defines the potential for AMD contamination to affect defined receptors (Figure 18). In the CTR model, environmental risk from AMD can be reduced by strategies that prevent contamination to begin with; by the absence of significant transport pathways from source to environmental receptor; or by the presence of only low-value (or tolerant) receptors.

Following the determination of the potential magnitude of AMD sources, potential environmental receptors should be identified and their values defined in consultation with stakeholders. Terrestrial animals may need to be considered as receptors for this analysis, as they can bioaccumulate and/or biomagnify elements (for example, Cd, Hg and Se) by consuming water, aquatic animals and plants. Likewise, soil contamination by AMD may also need to be assessed as a source for terrestrial flora and fauna.[34] The accumulation of AMD-derived elements in 'bush tucker' could also potentially affect the health of Indigenous consumers. Consequently, this contaminant pathway should be included in the risk assessment where relevant. Schedule B5 of the Contaminated Sites National Environment Protection Measure (NEPM), which deals with ecological risk assessments, sets out a process for assessing soil and water contamination that can be caused by the discharge of AMD at mine sites.[35]

Once potential receptors and their values are defined, existing and potential transport mechanisms should be critically examined. Workshopping potential pathways with a multidisciplinary panel comprising internal and external experts can be a very useful way of building a robust understanding of contaminant risk. The development by this team of a conceptual diagram to define contaminant sources, transport mechanisms and potential receptors for AMD for the entire site is an important step to frame further discussion about the potential magnitude of risk, and whether it might need to be mitigated.

33 See Section 10: Communicating AMD issues to stakeholders and investors.

34 See Appendix 1.

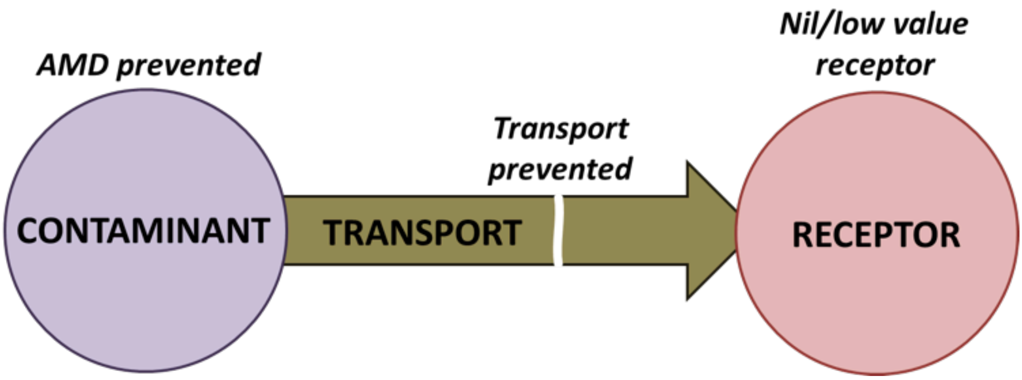
35 http://www.scew.gov.au/system/files/resources/93ae0e77-e697-e494-656f-afaaf9fb4277/files/7-ecological-risk-assessment.pdf.

The potential for transport processes should be considered for rainfall events of at least a 1:100 annual return interval during the operating life of the mine. A considerably longer return interval (such as probable maximum flood, or PMF) may need to be considered for the closure plan, given that the site will usually be required to maintain performance without active management intervention during this long time frame
(Logsdon 2013). Two types of processes need to be considered.

- immediate flushing by a weather event of AMD products stored on site (for example, in a pond)
- the delayed production of seepage after the event.

The second process is driven by percolation of infiltrated rainfall through waste containing soluble oxidation products. The delay in system response may mean that high-strength AMD can be delivered to a waterway as the water recedes without being substantially diluted.

Figure 18: Contaminant–transport–receptor model for the assessment of AMD risk



5.4.2 Assessing AMD risk

Risk assessment can be used to evaluate and rank the significance of specific AMD hazards. Generally, risk assessments are conducted in workshops using, as a minimum, internal staff with site and corporate level knowledge of the project and its socioenvironmental context. Staff from the environmental, mine planning, hydrogeology and community engagement sections should be involved. External advice from experts may often be required where specific expertise is not available in-house, where AMD risk is likely to be high, or where the issues involved are technically complex.

An example framework for AMD risk assessment is shown in Table 10. In addition to the usual risk components of likelihood and consequence, detail about the spatial extent and the likely duration of the hazard has been incorporated.

Table 10: Example risk component categories and final rating

WEIGHTING	LIKELIHOOD	CONSEQUENCE	EXTENT	DURATION	RATING	CLASSIFICATION
1	Rare	Insignificant	Immediate	Days	1-4	Very Low
2	Unlikely	Minor	Surrounds	Months	*5-36*	*Low*
3	Possible	Significant	Local	Years	37-144	Moderate
4	Likely	Major	Catchment	Decades	145-400	High
5	Almost certain	Catastrophic	Regional	Centuries	400-625	Extreme

In this example, weightings have been assigned on a scale from 1 to 5 (lowest to highest). However, any scale can be used as long as it is consistently and clearly defined before use. The overall risk rating is derived by the successive multiplication of each component category weighting. The maximum possible risk rating across the four columns in Table 10 would thus be 625 (that is, 5 x 5 x 5 x 5). Following mitigation, the aim is to reduce risk through decreases in the value of one or more of the component category values.

Risk assessment matrices can be used to assess the likely effectiveness of management strategies to reduce or even avoid risks of AMD to social and/or environmental receptors. Table 11 is a worked example using the weightings defined in Table 10.

Table 11: Example risk assessment, including initial derivation of the risk rating and the effect of a potential management strategy for closure on reducing the component contributions to the initial risk

ORIGINAL HAZARD RISK PROFILE: LEACHING OF AMD FROM PAF IN WRD				
LIKELIHOOD	**CONSEQUENCE**	**EXTENT**	**DURATION**	**RISK**
Likely (AMD seepage expected as poor capping material used)	Minor (no sensitive receptors likely affected)	Minor (no sensitive receptors likely affected)	Minor (no sensitive receptors likely affected)	Moderate
4	**2**	**2**	**5**	**80**

REVISED RISK PROFILE: MITIGATION BY DIRECT WRD SEEPAGE TO EVAPORATION BASINS DESIGNED FOR PMF EVENTS				
LIKELIHOOD	**CONSEQUENCE**	**EXTENT**	**DURATION**	**RESIDUAL RISK**
Likely (AMD seepage expected as poor capping material used)	Minor (no sensitive receptors likely affected)	Immediate (little transport away from site)	Long (centuries or more of AMD production possible)	Low
4	**1**	**1**	**5**	**20**

The potential for cumulative impacts from pre-existing sources of mine drainage located elsewhere in the catchment in which the mine is being developed should also be considered as part of the risk assessment. Regulators are becoming increasingly concerned about this issue in river catchments that contain multiple sources of inputs of solutes (Section 8.4).

6.0 MANAGING SULFIDIC MATERIALS TO PREVENT AMD

Key messages

- Proactive AMD source minimisation (that is, prevention) is preferred over control or treatment to manage AMD risk.
- The selection of optimal AMD minimisation and control strategies is site specific.
- The end-dumping method of construction has resulted in WRDs being the single largest AMD risk factor at many mine sites.
- Ongoing identification and segregation of AMD-generating waste, coupled with intermediate covering and compaction between lifts (ground-up construction), is an effective AMD prevention strategy for WRDs.
- Shortcuts taken in waste deposition to achieve immediate cost savings may be substantially offset by subsequent long-term AMD and water-quality management costs. There needs to be close coordination between the custodians of the AMD block model and mine planners to minimise this risk.
- The long-term minimisation of AMD risk for rock and tailings that remain above the land surface requires the construction of sustainable engineered covered systems to minimise oxygen ingress and/or to minimise the net percolation of rainfall.
- Leading practice design for the storage of sulfidic tailings should sustainably optimise the saturation of the tailings mass where possible, such that the geochemical integrity of the bulk of the material can be maintained over the long term.

Strategies for managing AMD fall into three general categories:

- minimising oxidation and the transport of oxidation products
- controls to reduce contaminant loads escaping to the environment
- active or passive treatment to allow water re-use or discharge.

The second and third categories also apply to all types of mined materials that may produce leachates that can adversely affect the receiving environment. From a risk reduction and sustainability viewpoint, minimising the amount of problematic waste produced and effective containment are preferred to control, which in turn is favoured over treatment. This section describes minimisation and control strategies for WRDs, TSFs, and pit lakes and comments on specific issues at brownfield and legacy sites. The treatment of AMD is addressed in Section 7.[36]

36 For a thought-provoking discussion of the broader issues addressed in this section, see Wilson (2008).

The identification of optimal minimisation and control strategies for a particular site will depend on climate; topography; the mining method; the material type (such as waste rock, tailings, wall rock and heap leach material); soil and rock types; mineralogy and available neutralisation resources; and inter-relationships between those factors (Miller 2014). The selective placement and encapsulation of waste materials based on their known physical and AMD-generating characteristics and risk profile is often the preferred AMD management practice during mine operations.

6.1 Management of waste rock dumps to minimise AMD

6.1.1 General considerations

The construction guidelines below refer to PAF (potentially acid-forming) material. This term applies to all waste rock types with a high AMD risk, including potential for NMD and SD, as a result of the oxidation of sulfides contained within the material. To ensure the long-term stability and overall performance of a WRD, geotechnical as well as geochemical properties need to be known for rock and other materials being used for construction. More information about the characterisation and properties of these materials is provided in the GARD Guide, in the cover design references cited in Section 6.3 and in case studies 6 and 7 in this section.

For surface WRDs, including those in valley-fill structures, PAF or high solute load potential waste rock should be identified and managed appropriately from the start of operations. This category of waste should be selectively placed and encapsulated with AMD-benign material (low AMD risk or NAF waste and/or waste rock with excess ANC) (see Figure 19). The rock that is proposed to be used for encapsulation must also be characterised for its potential to produce NMD and/or SD, since such material could present a significant AMD risk in its own right. The AMD risk presented by a waste type, and how it is managed, should be based on its solute load potential, rather than on predicted pH alone.

The entry of run-off or near-surface groundwater into the base of a WRD is a potential hazard that should be controlled. The most effective management strategy to limit the entry of surface flows into a WRD is to intercept clean rainfall run-off by diversion drains located upgradient of the WRD.

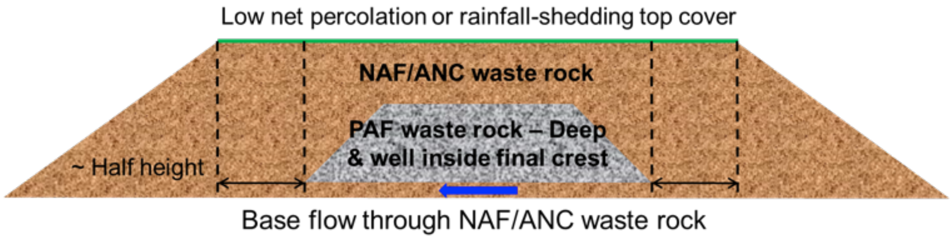
Depending on the topography of the footprint of the WRD, a pad of NAF and/or AC (acid-consuming) waste rock may also need to be placed first to provide a non-contaminating flow path for rainfall run-off from upstream of the pile.[37] Otherwise, the run-off may find its way along buried drainage channels and streams beneath the pile. There may also be a need to provide a sealing layer over buried drainage lines to limit the potential migration of contaminated seepage from the PAF waste rock placed above them. Alternatively, free-draining NAF and/or AC waste rock could be taken to full height above natural drainage lines, although this would require a large supply of non-contaminating waste rock.

Wherever possible, high AMD risk sulfidic waste rock should not be deposited so that it lies beneath the outer slopes of a WRD, as slopes are difficult to seal and more readily allow oxygen ingress and rainfall infiltration. A wide-side encapsulation (Figure 19) by NAF or AC waste rock is required so that any rainfall infiltration into the operational and final side slopes will not intersect PAF or high solute load potential waste rock. The tops of waste dumps undergoing construction, and with centrally located high AMD risk material, should be progressively covered between lifts by a compacted layer of NAF or AC waste rock. This should ideally be done before each wet season to limit the infiltration of rainfall into the PAF waste

37 See the Savage River case study in Section 5.

rock during operations and to reduce the time available for oxidation. At closure, a low net percolation or rainfall-shedding, non-contaminating cover is required on the final flat top of the WRD and on any residual benches.

Figure 19: Encapsulation of coarse-grained reactive waste



The ability to adequately encapsulate PAF or high solute load (NMD or SD) waste rock is a function of the proportion of PAF to NAF/AC waste rock and the change in that ratio as mining progresses. In general, mined rock that was located above the watertable will have oxidised over geological time, and any PAF waste rock will be likely to be extracted from below the watertable due to its lack of exposure to oxygen at that depth. Alkaline (AC) waste rock could occur both above and below the watertable. Oxidised waste rock will typically come out of the pit first and PAF waste rock later, although there are many examples where that has not been the case.

Open pits that are initially developed to the full extent of the planned pit shell and then extended to depth typically encounter NAF waste rock first and then any PAF waste rock present. Open pits that are advanced by a series of cut-backs encounter an alternating sequence of NAF and PAF waste rock with each cut-back. In both cases, different sequences of selective handling and placement are needed to ensure that maximum use is made of the NAF and AC material to encapsulate the higher risk material. For example, predominantly NAF or AC waste rock will be produced by the first case, and this may need to be deposited in a 'ring dyke' or 'doughnut' arrangement, leaving a central open core in which the subsequent predominantly PAF waste rock can be placed. Failure to optimise the use of NAF or ANC material when it is produced can seriously compromise the future effective minimisation of AMD risk as mining progresses.

To be able to effectively manage waste rock, it is essential to continuously record the tonnages of the different waste rock types and their locations in the evolving WRD and to regularly review waste rock placement to ensure that the waste rock management plan is being appropriately implemented. This information should also be used to produce an evolving 3D block model of the WRD, which will greatly facilitate closure planning by providing a robust basis for testing the efficacy of proposed designs to limit AMD risk. This testing can be undertaken using computer models.

6.1.2 Conventional end-dumped WRDs

Surface WRDs have conventionally been constructed by loose end-dumping from haul trucks and/or dozing over a tip-head. This results in the coarsest-grained particles travelling to the toe of the angle of repose slope to form a base rubble zone, and the formation of discontinuous, angle of repose layers within the pile that alternate fine and coarse-grained material (figures 20 and 21). The top surface of each lift of the pile becomes compacted by the loaded haul trucks.

The end-dumping method of construction has resulted in WRDs being the likely single largest AMD risk factor at many mine sites.

A conventional end-dumped WRD containing PAF waste rock is essentially an 'oxidation reactor', with a ready supply of oxygen provided through the base rubble zone and up the coarse-grained angle of repose layers driven by advection and convection, from which oxygen diffuses into the adjacent moist, fine-grained angle of repose layers. The fine-grained layers present a very high surface area to volume ratio, which promotes oxidation and the storage of moisture and the oxidation products. During the construction of a WRD containing PAF waste rock, it will be difficult to limit oxygen and rainfall ingress into such a pile.

For WRDs constructed by end-dumping, AMD can best be mitigated by limiting the flow of water into the pile and hence reducing the transport medium for the oxidation products. However, in many cases this is applying a retroactive management strategy that can only work by limiting the leaching of already formed oxidation products. In this scenario, the opportunity for limiting AMD risk by minimising the source term has largely been lost.

Figure 20: Base rubble zone (top); angle of repose and trafficked layers (bottom)

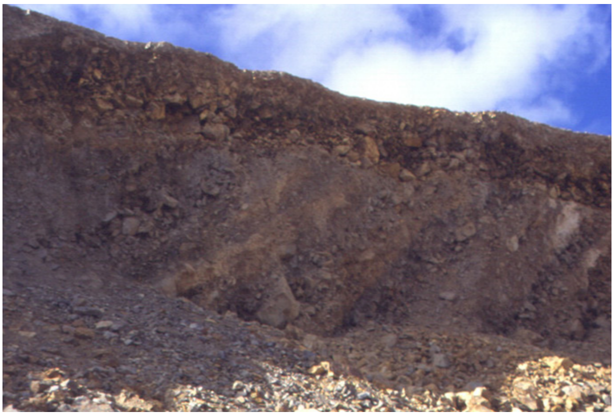
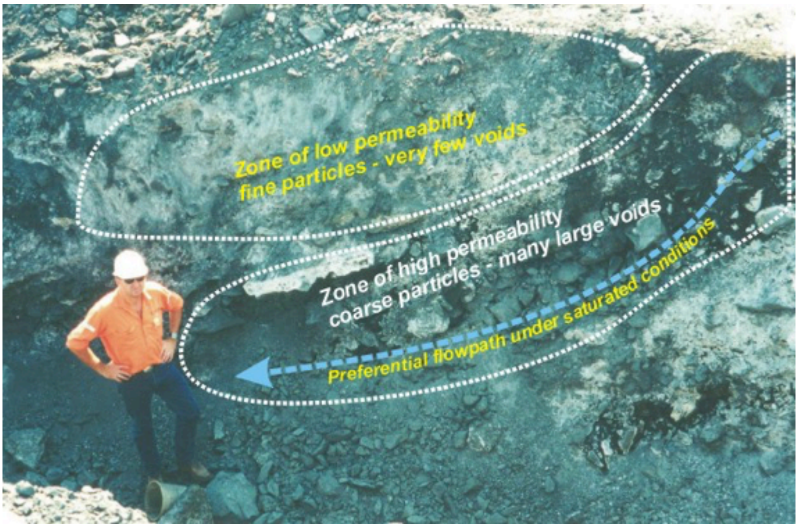


Figure 21: View of an opened up WRD showing zones of differing materials and particle sizes



Source: Team NT Toolkit 2004, courtesy M Fawcett.

Where PAF waste rock is encountered, advancing an open pit by a series of cut-backs is the most efficient means to manage the potential for AMD generation in a WRD being constructed by conventional end-dumping. This will facilitate the progressive encapsulation of PAF waste rock by NAF and/or AC waste rock, without the need for intermediate stockpiling and costly double-handling of the NAF/AC waste rock. Progressive cut-backs also better facilitate the in-pit dumping of waste rock (PAF waste rock, in particular) into the completed end of the pit. Where a number of pits are mined in sequence, mined-out pits may also be available for the effective containment and encapsulation of PAF waste rock

6.1.3 Oxidation rate and lag time to production of AMD

Tracking the locations of waste rock types within the WRD is especially important if sulfide oxidation rates are slow or lag times to production of AMD are protracted, as obvious AMD might not be released to the environment until many years after the site has closed. The length of the lag period to when AMD is seen to be emerging from a WRD is a function of the rate of oxidation, the presence of neutralising material, and the rate of wetting up of the dump leading to the transportation of the oxidation products to the external environment.

A high proportion of incident rainfall (of the order of 50–90%; Williams & Rohde 2008) infiltrates a fresh (uncovered) WRD, and much of the infiltration is initially stored within the pile. The pile will saturate only to the point at which the permeability of the waste rock is high enough to allow fingering flow along preferred pathways (from about 25% saturated for fresh waste rock to about 60% for weathered waste rock; Williams & Rohde 2007). Eventually, the fingers reach the base of the pile and either percolate into the foundation or flow along buried drainage channels and streams to emerge at low points along the toe of the pile. The more developed the wetting fingers, the lower the subsequent rainfall needed to generate seepage to the base of the pile, and the faster seepage will occur.

The rate at which a conventional end-dumped WRD wets-up due to rainfall infiltration is a function of the amount and intensity of rainfall over time, the rate of rise and height of the pile, the particle size distribution of the waste rock and the degradation of the waste rock over time. It will range from years for a low pile in a wet climate to decades (or longer) for a high pile in a dry climate. The time required for wetting-up determines when AMD starts to be released to the environment.

6.1.4 Construction methods for WRDs to minimise AMD production

In contrast to end-dumping, paddock dumping facilitates the construction of a WRD from the ground up. The term ‘bottom-up’ is also used to describe this process of constructing a WRD by depositing horizontal layers rather than by end-dumping from elevated tip-heads.

Ground-up construction of WRDs is a key component of leading practice for AMD control. This method limits the segregation of fine and coarse materials during placement and facilitates selective placement and compaction, thus restricting gas movement within the dump and the extent of potential oxidising zones.

Ground-up construction is the second most effective method (after a permanent water cover) of managing highly reactive sulfidic waste rock and substantially reducing AMD risk at open-pit mines with a dominance of PAF waste rock and insufficient NAF and/or AC waste rock for encapsulation.

The cost of this method of construction compared with end-dumping from high tip-heads is site specific, as it depends on mine scheduling, topography and haul distances. If ground-up construction is selected from the start as the waste rock management strategy, based on the outcome of the AMD risk assessment, the mine development schedule can be best optimised to accommodate the specific requirements of this method. However, if the waste dumping strategy has to be altered later on to address developing issues posed by reactive waste rock (including the need for treatment of AMD), costs are likely to be substantively higher and AMD control overall less effective than implementing a ground-up approach from the start.

Ground-up construction by paddock dumping in thinner layers, followed by spreading and compaction of waste rock, has major practical benefits:

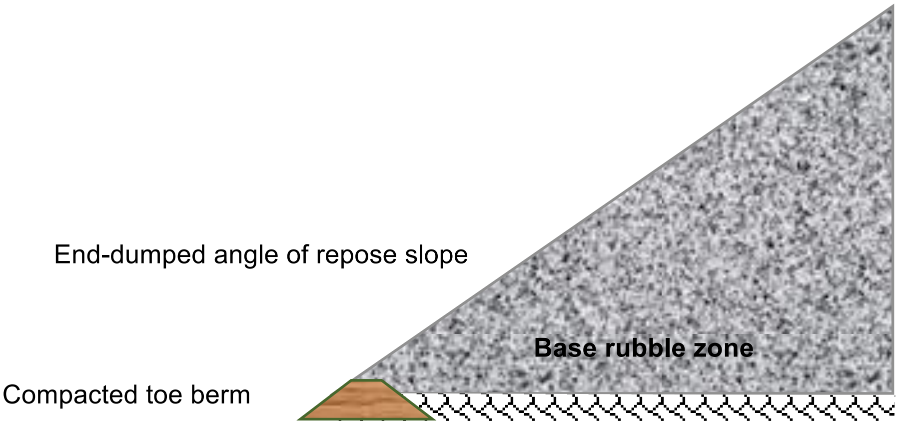
- A higher density mass compared with end-dumping off a tip-head reduces the storage volume requirement.
- Oxygen ingress and net percolation into PAF waste rock are greatly reduced, reducing the potential for spontaneous combustion and AMD.
- The time available to utilise the kinetically slow neutralising capacity of silicate rocks is maximised by greatly increasing the residence time of water.
- The greater bearing capacity of the waste rock enhances the potential for post-closure development.
- Greatly reduced post-construction settlement, including self-weight collapse on wetting-up and degradation-induced settlement, also improve the likelihood of post-closure success.

Sealing base rubble zone

The ground-up construction approach can also be used to seal the base rubble zone formed on end-dumping waste rock from a tip-head, and for sealing a flattened WRD slope or sealing up against an angle of repose WRD slope. The techniques described in this section (shown in figures 22 to 24) are similar to those routinely used for civil engineering construction of earth and rock fill dams. To date, their application to mine WRDs has been rare. However, they are relatively easily applied and could yield great benefits in reducing the oxidation of sulfidic waste rock and preventing AMD, provided sufficient clayey materials suitable for the construction of compacted sealing layers are available.

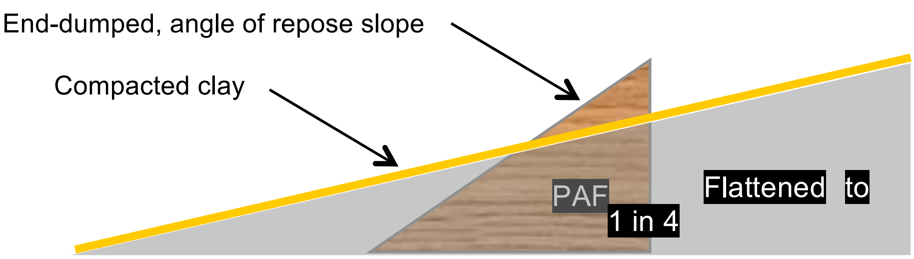
Paddock dumping of waste rock is often employed initially as a WRD footprint is being developed, before the start of end-dumping once a tip-head has been established. However, subsequent end-dumping from an established tip-head creates a base rubble zone and coarse-grained angle of repose oxygen pathways. If that has already happened, the base rubble zone could be sealed by constructing a compacted berm of NAF and/or ANC waste rock or clay around the toe of the pile (Figure 22). However, this needs to be extended as the pile expands and the berm is overtopped by waste rock cascading down the slope from above.

Figure 22: Schematic of compacted toe berm to seal base rubble zone



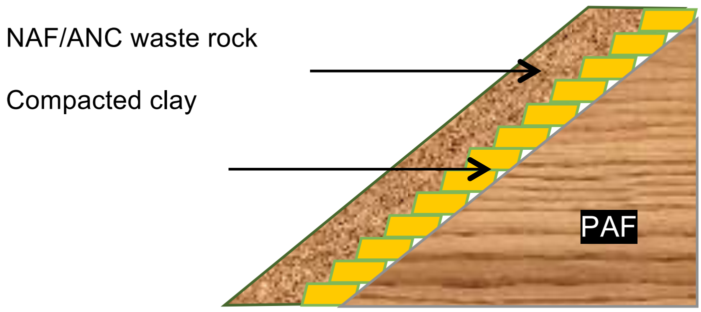
If suitable clayey material is available, a compacted clay layer could be constructed against a WRD slope containing PAF waste rock to seal it against oxygen ingress and rainfall infiltration. This can be achieved either by first flattening the angle of repose WRD slope and then constructing a compacted clay layer directly on the flattened slope (Figure 23), or by constructing, in horizontal lifts from the ground up, a compacted clay layer inclined against an angle of repose WRD slope containing PAF waste rock (Figure 24). While the latter method is routinely used for constructing clay cores in earth and rock fill water-supply dams, it has not as yet been generally adopted by the mining industry.

Figure 23: Schematic of compacted clay layer on flattened WRD slope containing PAF waste rock



In this case, the length of the inclined slope for an angle of repose WRD slope would be about three times less than the slope flattened to 1 in 4, reducing the ultimate footprint area of the WRD, potentially reducing the volume of clay required, and reducing the cost of the earthworks. In addition, raising the compacted clay layer in a series of horizontal lifts allows better compaction than can be achieved on a 1 in 4 slope, and it is easier to place material using this approach provided that suitable small equipment is used. The inclined compacted clay layer still has to be protected against erosion by the addition of a layer of NAF and/or ANC waste rock (Figure 24). This layer can be cost-effectively placed to any desired thickness by end-dumping from the crest. The additional weight of the protective layer compacts the inclined face of the clay layer.

Figure 24: Schematic of ground-up compacted clay layer inclined against an angle of repose WRD slope containing PAF waste rock



Note that slope flattening from the angle of repose (about 37°) to 1 in 4 (14°) makes the length (catchment) of the slope about three times greater, given that the height of the WRD is fixed. However, it will still be difficult to achieve good compaction of the clay on a slope that is relatively steep. Slope flattening also smooths the surface texture, increasing run-off and hence erosion potential. Therefore, the compacted clay layer on the flattened slope requires a sufficiently thick overlay of coarse NAF and/or ANC waste rock to be placed over it for erosion protection.

In both the cases (figures 23 and 24), a compacted clay layer should also be applied to the final flat top surface of the pile to reduce oxygen ingress and rainfall infiltration into the underlying PAF waste rock.

6.1.5 Minimising self-heating and AMD potential

Tracking the location of highly reactive sulfidic waste rock in the pit and its placement in the WRD is particularly important, since that material may have a potential for self-heating as well as the production of AMD. Great care should also be taken with the placement of highly reactive sulfidic tailings in a mass of unsaturated waste rock, as the high surface area to volume ratio of this material can exacerbate its self-heating and AMD generation potential.

The oxidation of sulfide is an exothermic (heat-generating) reaction. For highly reactive sulfidic waste rock, the rate of heating can be sufficient to cause self-ignition, with accompanying steam, SO^2 (which combines with moisture to form sulfurous acid) and either settlement (where carbonaceous material is consumed by combustion) or heave (due to the oxidation products occupying a greater volume). For exposed highly reactive materials, the potential for self-ignition is mitigated by atmospheric cooling, which is achieved by paddock dumping in thin layers. However, while mitigating the rise in temperature, this approach will enhance the production of AMD unless special placement methods are implemented.

For materials of this type, the potential for AMD is best mitigated by encapsulation in cells to limit both oxygen ingress and the net percolation of water, thereby reducing the generation and transport of the oxidation products. Self-heating and AMD production can be mitigated simultaneously by paddock dumping, spreading and compacting alternating layers of highly reactive sulfidic waste rock and NAF and/or AC waste rock within cells constructed out of NAF and/or AC waste rock (Landers & Usher 2015). This limits potential heating by limiting the thickness of each reactive layer and the potential for AMD by encapsulating the highly reactive sulfidic layers. In the absence of sufficient AC waste rock, it is prudent to add lime or crushed limestone to the NAF and/or PAF waste rock to provide some alkaline buffering capacity while the material is exposed.

Underwater disposal is the most effective means of limiting oxygen ingress but is difficult to achieve during operations, particularly in dry climates. One option is to place sulfidic waste rock in available completed pits, preferably where it will ultimately be below the recovered groundwater or pit lake level. Submergence will effectively stop the oxidation process. However, the PAF waste rock should be covered by water before it has had a chance to significantly oxidise, or else there is a risk of subsequent contamination of groundwater by dissolution of the oxidation products. Once again, a process of in-pit paddock dumping with interlayer covering by compacted NAF or especially ANC material reduces the risk of in situ formation of soluble oxidation products before the material is covered with water.

A second option is to construct a purpose-built waste rock impoundment within which the water level is raised as the depth of highly PAF waste rock increases during the mine's operating life.[38]

6.1.6 Minimising AMD risk at sites dominated by PAF waste rock

At open-pit mine sites with a high proportion of PAF or high solute load AMD waste rock overall or over a particular phase of mining, there may be insufficient NAF and/or AC waste rock to adequately encapsulate the PAF waste rock. In such cases, it is even more important to identify, early on, waste rock types with different degrees of acid-forming potential and to selectively place those types within the WRD or under water.

The most highly reactive sulfidic waste rock would best be placed under water, possibly within the TSF, provided that the TSF was initially designed or capable of being expanded to accommodate the amounts of both waste types that will be produced.[39] Ground-up construction methods involving paddock dumping, spreading and compaction, as described above, should be employed for the remainder of the highly PAF waste rock.

The second most reactive PAF waste rock should be contained within the centre of the WRD or used for buttressing TSF embankments, using an appropriate design for the containment of the material.[40] Successively less reactive PAF waste rock should then be placed in compacted layers from the ground up, with the least-reactive PAF waste rock being contained under the outer slope. While this strategy reduces the magnitude of the AMD issue at the site as much as possible, there is no guarantee that the amount of AMD that may still be produced will enable the site to meet its environmental management requirements without further mitigation, such as treatment.

38 See the Kelian mine case study in Section 3 for an example of the implementation of this process of progressive closure.

39 See the Phu Kam case study this section and the Kelian case study in Section 3.

40 See the Martha and Phu Kham mine case studies this section.

Case study 6: Long-term monitoring demonstrates success of ground-up construction to contain waste rock, Martha Mine, New Zealand

Context

The Martha Mine produces gold and 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 °C and 19 °C and the monthly rainfall varies between 120 and 240 mm throughout the year.

Mining produces PAF and NAF waste rock. Between 1988 and 1999, waste rock was disposed of in a specially engineered embankment integrated into the TSF known as Storage 2. The key performance objectives for the Storage 2 embankment were reducing the rate of oxidation of sulfide-bearing waste rock and production of acid and sulfate in leachate, the neutralisation of produced acid and the control of the quantity and quality of water leaving the waste rock.

This study documents the design and construction of the embankment to minimise AMD production from the start. The findings from more than 10 years of monitoring data indicate that the strategy has been a success. Further details and references to original data sources are contained in Garvie et al. (2012).

Waste rock characteristics

Although initial geochemical investigations were conducted between 1984 and 1986 prior to the start of mining, subsequent characterisation work and site experience identified the need to reduce the rate of formation of AMD in the PAF waste rock and the release of AMD leachate from the mine. Additional geochemical test work commenced in 1993 and continued until Storage 2 was completed in 1999.

The partially oxidised and unoxidised waste rock contains variable amounts of sulfide sulfur (mostly pyrite) with a range of 0.5–3 weight% and an average value of 1.5 weight% S. The amount of soluble sulfate in the waste rock prior to mining was low. The partially oxidised rocks were nearly entirely lacking in sulfide sulfur and were classified as NAF. The following criteria were developed and applied for characterising the AMD potential of the waste:

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

Material initially classified as uncertain was assumed to be PAF, pending the findings from additional test work.

AMD control

AMD control was undertaken using four approaches:

- hydrological control—placement of low permeability layers to reduce infiltration, and installation of water management structures to control the release rate
- water treatment—modification of the drainage chemistry

- geochemical control—addition of neutralising materials to control pH and oxidation rates
- oxidation control—reducing oxygen flux to reactive sulfides.

Embankment design and placement of waste rock

The embankment of Storage 2 was constructed from waste rock excavated during mining. To achieve the requirements of structural strength, tailings retention and control of AMD, the design of the embankment structure incorporated various zones with particular properties. Zone D contains the majority of waste rock (Figure 1). Approximately 60% of the waste rock in zone D was unoxidised and 40% was oxidised. Zone E comprised soft and wet oxidised and unoxidised rock that was placed as subcompartments of zone D.

Special handling procedures were implemented for zone D material, to reduce sulfide mineral oxidation and acid production. For example, the surface of PAF material directly exposed to the atmosphere was limited to compartments that were no greater than 4 ha in area and no greater than 5 m in height. An intermediate 0.25 m thick encapsulating compacted layer of low permeability fill was placed on the surface if it was to be exposed for longer than 3 months. 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 done 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 F is the transition layer between the coarse rock in zone D and the fine-grained soil in zone G. It consists of both oxidised and unoxidised waste rock that was compacted in lift heights of less than 0.25 m. Zone G is a 1.5 m thick cover layer of fine-grained oxidised NAF waste rock, the function of which is to limit transport of atmospheric oxygen into the waste rock. It was placed in compacted lifts no greater than 0.25 m to achieve a saturated hydraulic conductivity of less than 1×10^{-8} m/s, with measured saturation 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. Zone C is a structural zone of oxidised and unoxidised rock.

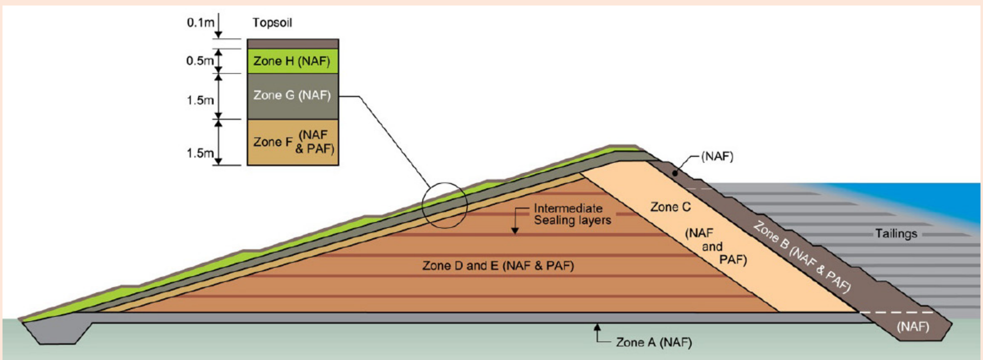


Figure 1: Cross-section design of embankment of storage 2, designed to contacin tailings

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 installed in Storage 2 to collect water from the tailings, zone D, the subsoil beneath the embankment and the natural groundwater are shown in 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.

Geochemical control

Crushed limestone was applied to the surface of zone D, to extend the lag time for production of AMD from the PAF rock, for a time greater than that required to place the zone G layer. Application of crushed limestone, and sealing by compaction and placement of zones G and H, began in May 1994. The facility was fully operational in 1995. The pH values measured in two surface water ponds confirm that this application rate was sufficient to achieve circum-neutral to alkaline surface waters most of the time (Figure 3). The addition of lime enabled direct discharge of some stormwater from collection ponds since July 2014, with significant savings on water treatment costs.

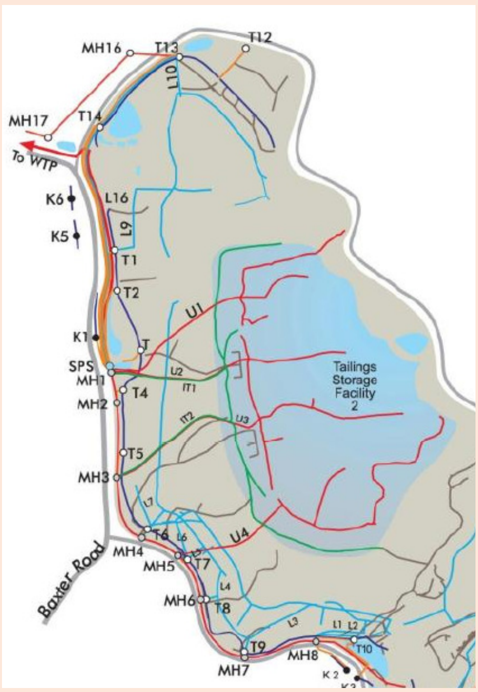


Figure 2: Storage 2 drainage network

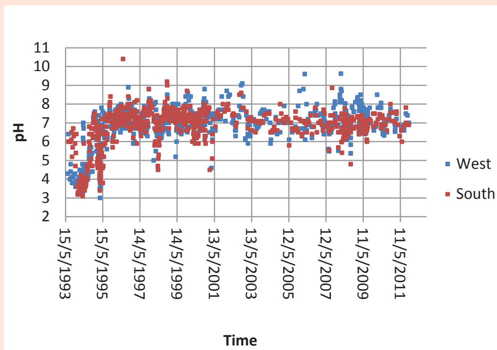


Figure 3: pH values in the south and west collection ponds

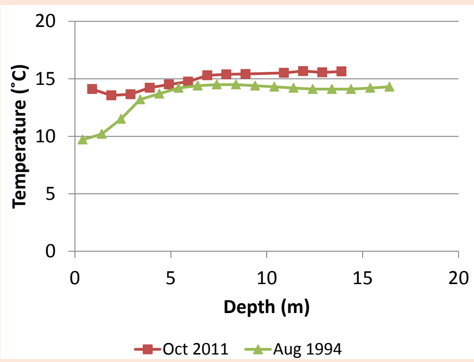
Oxidation control

The surface cover for the emnkmant of Storage 2 was designed to reduce oxygen transport by diffusion, convection and advection from the surface of the embankment to the interior.

Waihi Gold undertook a rigorous quality assurance program up to November 1998; subsequently an independent testing agency undertook a quality assurance program to confirm that material characteristics specified for limiting oxygen transport were consistently achieved during construction. The construction specification required the 10-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 m^3 of zone G and 20 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%) and to be placed with a water content that would allow optimum compaction.

Measured hydraulic conductivity values were generally in the range of 1×10^{-9}m/s to 1×10^{-8} m/s, and the degree of saturation was around 0.9, indicating that both of these key design parameters were achieved. The degree of saturation was consistent with values of the oxygen diffusion coefficient of zone G measured in situ, which had a representative value of 1.5×10^{-8} m^2/s.

Ten vertical probe holes installed in Storage 2 in 1994 were used to measure vertical oxygen concentration and temperature profiles. These measurements were repeated in 2011. Selected results are presented in Figure 4. Oxygen concentrations measured in 2011 were similar to those measured in 1994. Typically, oxygen concentrations decreased rapidly with distance from the surface concentration (21 volume%). At some locations the oxygen concentration was less than or equal to 1 volume% within 5 m of the surface and within 3 m of the base of zone G. The temperature gradient was zero below about 3 m in depth, indicating that the oxidation rate was approximately zero. Similar results were obtained for other probe holes.



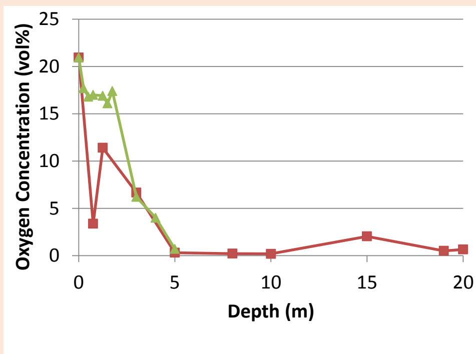


Figure 4: Temperature and oxygen concentration measurements at probe hole WI08

Modelled acid sulfate generation rates indicated that zone G reduced the rates by more than 95% compared to the case without a cover. The concentrations of sulfate in leachate collected by 14 drains in zone D ranged between 100 and 5,500 mg/L and were all much less than the maximum concentration predicted for the pore water before application of the cover, assuming no sulfate solubility controls.

Conclusions

Early materials characterisation, the development and implementation of a staged construction plan with clear AMD management objectives and adherence to a rigorous materials quality control program during embankment construction have been the key to maintaining low acid sulfate production rates from the contained PAF waste rock at Martha Mine for more than 17 years. The use of intermediate covers between lifts of PAF material limited acid and sulfate production during embankment construction, and the addition of limestone buffered the acid produced until placement of the final cover, such that direct discharge of some stormwater is possible. The final compacted cover layer of oxidised waste rock on the embankment of Storage 2 has reduced the rate of acid and sulfate generation from underlying sulfide-bearing waste rock by more than 95% compared to uncovered PAF waste. The cover layer was effective because it had a low oxygen diffusion coefficient of about 1.5×10^{-8} m^2/s^{-1}.

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Garvie, A, Miller, S, Fransen, P, Ruddock, J, Haymont, R (2012). 'Early and prolonged limitation of AMD production via waste landform construction', in Price, WA, Hogan, C, Tremblay, G (eds), Proceedings of the 9th International Conference on Acid Rock Drainage, 20–26 May 2012, Ottawa, Ontario, Canada.

Case study 7: Construction of WRD with progressive compaction between lifts, PanAust mine, Laos

Context

Phu Bia Mining Limited (90% owned by PanAust Limited and 10% by the Government of Lao) commissioned the Phu Kham Copper–Gold Operation in 2008. The site is approximately 120 km north of Vientiane and is located within the Nam Mo River catchment. The average rainfall is approximately 3,000 mm/year, with a distinct wet and dry season.

The Phu Kham open pit is mined as a conventional truck and shovel operation. Approximately 200 Mt of waste rock will be mined over the current mine life to 2021. Ore is processed by flotation to produce a copper–gold concentrate.

Evaluation of the AMD risk associated with the development of the Phu Kham deposit began prior to mining, with the inclusion of sulfur assays as an integral component of the geological and project investigations and modelling. The pre-mining data indicated that the flotation tailings would be PAF with a high pyritic sulfur content of about 7% S. The average sulfur grade for waste rock within the pit shell was indicated to be 3.2% S, with 20% of the projected waste rock having a sulfur grade of more than 5% S. The ANC was generally low and less than 15 kg H_2SO_4/t.

Based on the pre-mining data, block modelling indicated that about 75% of the waste rock was likely to be PAF. As a result, the Phu Kham deposit was recognised as being one of the highest AMD risk sites worldwide. This situation highlighted the need for integrated management of tailings and waste rock to mitigate AMD risks within the life of mine plan.

The term ARD (rather than AMD) is used for the remainder of this case study since the initial characterisation work showed the high acid rock drainage potential of mine materials.

ARD management plan

A multidisciplinary team of representatives from geology, mining, processing and environment departments in conjunction with external consultants was established prior to mine start-up to develop an ARD management plan for the life of mine. The team evaluated various options, including ex-pit WRDs and placement of all PAF waste below water in the tailings storage facility (TSF). The final preferred strategy was to encapsulate lower capacity PAF rock in the downstream section of the TSF embankment and place higher capacity PAF rock in the TSF, where it would be progressively inundated with TSF water. The team developed detailed operational guidelines that allow the integration of ARD management practices into the daily operating activities, with the overall objective of preventing any geochemical legacy from waste rock and tailings at Phu Kham. An ARD committee was established to ensure implementation of the plan, with an ongoing role to provide regular review and recommendations throughout the operating phase and into closure.

The ARD management plan is based on the fundamental strategy of isolating sulfidic mine waste from atmospheric oxygen. Engineering options for achieving isolation from atmospheric oxygen include placing sulfidic material under a permanent water cover or construction of an engineered seal that limits oxygen transfer to geological rates. At Phu Kham, both strategies have been adopted, with the higher sulfidic acid generating waste reporting to the tailings impoundment and the lower sulfidic acid generating waste isolated in cells and zones (PAF cells) within the downstream portion of the TSF.

A simple 'traffic light' colour coding system is used to map and segregate waste into the four main types, labelled blue, green, amber and red. Green and blue are non-acid forming (NAF); blue is also acid consuming, with a high acid-neutralising capacity (ANC). Amber and red are both PAF, with the sulfur cut-off for amber set to ensure that material placed in the PAF cells meets the design specifications for sulfur grade. Further details of the development of the classification scheme are provided in Miller et al. (2012).

The current criteria used for waste rock grade control are as follows:

- green (NAF): sulfur ≤0.3% S
- amber (PAF): sulfur 0.3 to 5% S
- red (PAF): total sulfur >5% S
- blue (acid-consuming material)
  - limestone; OR
  - %S ≤ 0.5% S and ANC ≥ 100 kg/t; OR
  - %S > 0.5% S and ANC/MPA > 4.

Grade control drilling undertaken in advance of mining drives the short-term planning for both ore and waste mining. Waste blocks are characterised, marked up in pit, and then scheduled into the TSF embankment construction (green and amber) or deposited within the TSF. All red and amber material in excess of TSF construction requirements is managed within the TSF. The mining operation uses real-time, GPS-based equipment management software to allow the waste (and ore) blocks to be directed to the correct destination on a truck-by-truck basis.

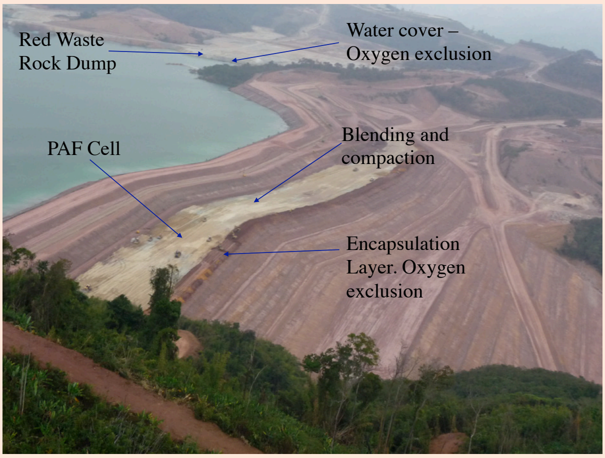
PAF cell design

The PAF cells are a mix of green, amber and material extracted from borrow pits within and around the final TSF footprint. The sulfur cut-off for amber classification is set to ensure the blend meets the following design specifications for sulfur grade within the PAF cells:

- monthly average total sulfur content not more than 1.5% S
- 90% of monthly samples less than 2.5% S.

At mine start-up, the design specifications were set at an average S grade of 1% S and 90th percentile of 2% S. These values were progressively increased based on the results of performance monitoring of the PAF cells.

Figure 1 is an aerial photograph showing the general arrangement of the PAF cells within the embankment and the location of the WRD within the TSF. Initially the WRD received only red waste, but as mining has progressed with increasing quantities of green waste needed for embankment construction, most amber waste now also reports to the WRD within the TSF. The dump is referred to as the red road waste dump (RRWD).



embankment showing location of PAF cells in the downstream section of

Figure 1: TSF embankment showing location of PAF cells in the downstream section of the dam wall and the Red Road Waste Dump located within the TSF catchment

Figure 2 is a schematic cross-section showing the PAF cell design and encapsulation layers. Also shown is the location and design of oxygen-monitoring arrays that are part of the extensive operational monitoring program.

Figure 3 shows active construction of a PAF cell and the encapsulation layer. The encapsulation layer provides a 6 m horizontal and 2 m perpendicular cover over PAF material. The amber, green and borrow material placed in the PAF cell is run-out from dump trucks and dozer-mixed. It is placed in 300 mm layers to facilitate blending and machine-compacted as shown in Figure 3. The encapsulation zone is also constructed in 300 mm layers, but only green and amber material types are used and it is compacted to tight specifications, including a minimum density of 98% of the standard maximum at a moisture content equivalent to a degree of saturation of not less than 90%.

The degree of saturation at which the material is placed and compacted is critical for the effectiveness of the encapsulation layers to reduce the flux of oxygen to the PAF waste. Constructing both the PAF zone and encapsulation layers at high moisture content substantially reduces the ability for oxygen to diffuse through unsaturated pore spaces.

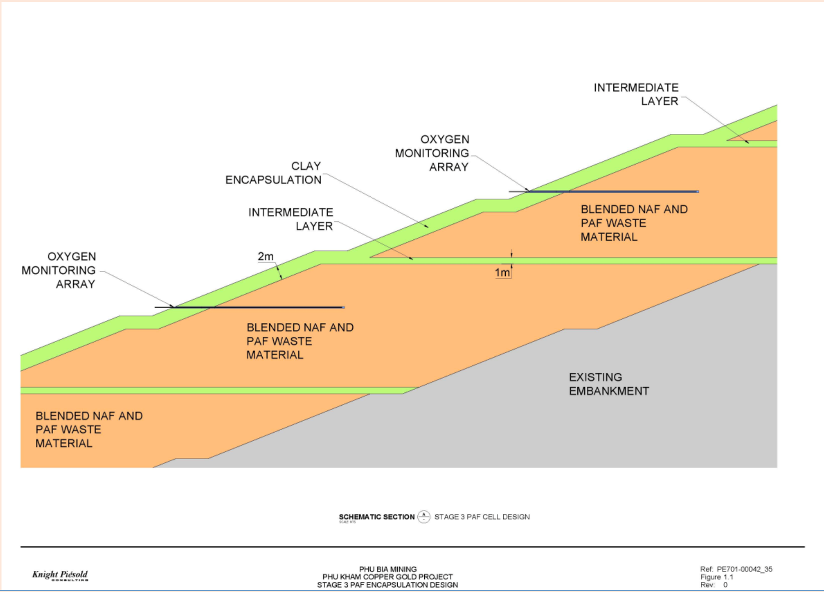


Figure 2: PAF cell encapsulation design showing general positioning of oxygen-monitoring arrays

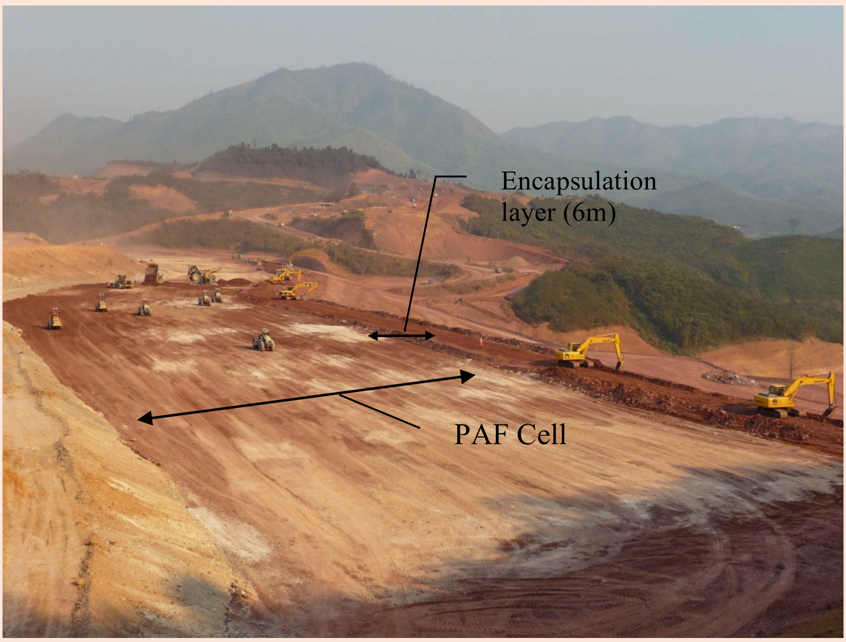


Figure 3: PAF cell construction on the downstream face of the TSF embankment

To evaluate the performance of the PAF cells, pore gas oxygen monitoring arrays are installed horizontally as the cells are constructed. The general positioning of the horizontal arrays is shown in Figure 2; Figure 4 shows active installation. The oxygen-monitoring tubes are located at 2 m intervals through the first 10 m of the PAF waste, then at 15 m and 20 m into the PAF cells. A bentonite grout seal is located across the contact between the encapsulation layer and PAF cell, and compacted low-permeability fill is used between each sampling port within the PAF cell.

Vibrating wire piezometers are installed to monitor pore pressures in the embankment for geotechnical purposes. They also provide a continuous record of temperature at the monitoring points, which is then used to assess any historical or ongoing oxidation of sulfides in the PAF cells.

Figure 4: Installation of horizontal pore gas monitoring system

Red Waste Dump

The Red Waste Dump (RWD) is located within the tailings impoundment (see Figure 1) and is progressively inundated as the TSF water level rises. Acid and soluble metals (including Fe, Al and Cu) are generated and released from the PAF rock while, above, the water level mixes with the alkaline TSF pond water, where the acid is neutralised and metals are precipitated.

The RWD is constructed by a push dumping method, with trucks tipping short and the material dozed over the face. Due to the fine-grained nature of much of the waste rock and the low height above the TSF water level, this method of placement has prevented the development of convective/advective gas transfer within the dump, limiting oxidation to the immediate surface layer. Paste pH determinations in test pit investigations within the RRWD show lower pH and oxidation occurring in the immediate surface layer only. Material at depth shows no evidence of in situ oxidation.

The alkalinity of tailings liquor discharged to the dam and the return water are monitored daily to ensure that the dissolved alkalinity within the dam is targeted to be above 30 mg $CaCO_3$/L. Additional lime is added at the mill to the tailings discharge to provide a level of protective alkalinity to buffer acid inputs from the exposed rock during construction of the RRWD as well as any acid inputs from pit dewatering.

At closure, the RWD will be permanently under water to prevent ongoing oxidation and acid generation.

Quality assurance and performance monitoring

In-pit waste classification

Routine monthly mine bench samples are assayed for total S, ANC, C, Cu and NAG. The results are used to confirm the in-pit classification and mapping of waste rock types (green, blue, amber and red). All data is compiled, evaluated and presented in monthly reports, and an ongoing database is maintained with sample locations and logged details. This data is used to update the ARD rock type model of the mine pit to facilitate short-term mine planning for selective mining and segregation of ARD rock types for placement in the TSF embankment and RRWD.

Geotechnical monitoring of PAF cells and encapsulation layers

Geotechnical monitoring is completed on all encapsulation material placed in accordance with the testing requirement. Test results are reported to the construction team upon completion to provide feedback on construction quality. At the end of each month all data is compiled and documented into a report to record the construction history.

A geotechnical construction specification has been produced for all zones within the TSF embankment. This specification provides particle size limits, Atterberg limits, and compaction and moisture content requirements for the materials to be placed within the PAF cells and material which can be used for the construction of encapsulation layers.

The testing data for the material placed within the PAF cells indicates that a high degree of compaction has been achieved. The average compaction for all tests equated to a density of 99.9% of the standard maximum dry density, which is above the specified minimum of 98%.

To limit the oxidation of material placed within the PAF cells, an encapsulation zone is constructed at the outer limits of the PAF cells. Field data for the encapsulation material indicates that a high degree of compaction (99.8% of the standard maximum dry density) and a high degree of saturation (average degree of saturation of 93.6%) have been achieved.

Geochemical monitoring of PAF cells

Each 1 m raise in PAF cells is sampled on a 50 m grid to a depth of 1 m. The samples are assayed for total sulfur, ANC and NAG (NAG pH and NAG capacity kg/t).

Figure 5 is a plot of the mean and 90th percentile of the monthly sulfur-monitoring data for the PAF cells for construction stages 3, 4 and 5. Note that construction of these cells occurs during the dry season only and each cell is completed prior to the wet season. Each month approximates a 1–2 m rise in the PAF cells, and these percentage S values are used to evaluate performance. The results show that for stage 5, the average monthly sulfur grade for the first 3 months was close to or slightly exceeded the target values. This highlighted an issue with material segregation within the pit. The issue was resolved and resulted in the last 3 months falling well within target.

It was concluded that this short period of higher sulfur waste was unlikely to have compromised PAF cell performance due to the high degree of compaction and saturation achieved. Moreover, this section of the embankment will be covered by subsequent construction stages. These time series data demonstrate the need for diligent quality control and regular review/audit systems to ensure the design targets for all components of the ARD management plan are being met on an ongoing basis.

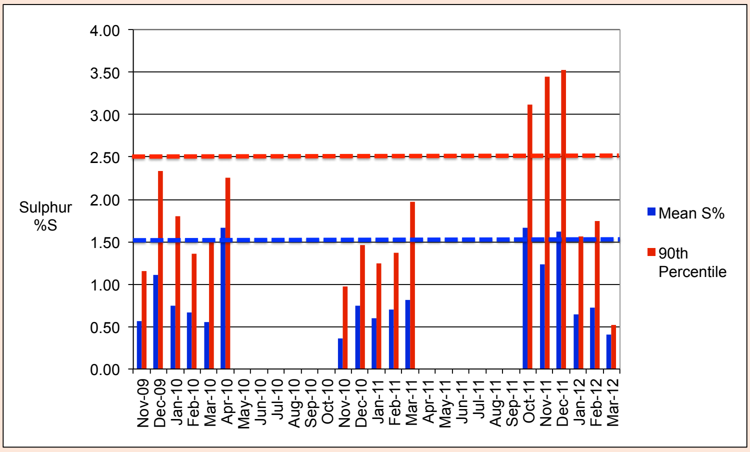


Figure 5: Monthly average and 90th percentile sulfur grades for PAF cell samples

(dashed lines show mean and 90th percentile targets)

The oxygen concentration profiles and trends through time are shown in Figure 6 for one of the oxygen-monitoring arrays (OMA 10) from stage 4 of construction. Thirty-five arrays have been installed to date, and each location shows a similar trend, with oxygen concentrations deceasing to zero within 1–4 months. The rate of oxygen consumption indicates that the PAF cell material has intrinsic oxidation rates ranging from 1.0 x 10^{-8} to 4 x 10^{-7} $kgO_2/m^3/sec$. These rates are typical of reactive PAF rock and confirm the need for focused and diligent management of the waste.

Vibrating wire piezometers provide continuous temperature records. The trends in temperature through time are consistent with the annual average daily temperature at Phu Kham. No other sources of heat in the PAF cells—such as oxidation of pyrite, which is an exothermic reaction—are evident.

This supports the findings from the oxygen-monitoring data that indicate that oxidation is being effectively controlled within the PAF cells.

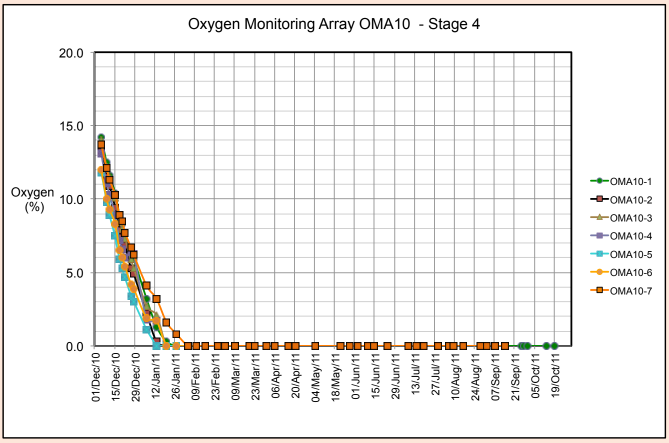


Figure 6: PAF cell pore gas oxygen concentration trends (port 1 (OMA10-1) furthest from sealing layer and port 7 adjacent to sealing layer)

Drainage from the TSF embankment is directed through a V notch weir where the flow is continually monitored and a water sample is collected weekly. Seepage from the TSF has maintained circum-neutral pH (median 6.6) with high total alkalinity (mean 127 mg$CaCO_3$/L) and low sulfate (mean 63 mg/L).

The results show that the discharge guidelines are being met at the 99th percentile level. The dominant dissolved constituents in the seepage are Ca, Mg and sulfate. Due to the low redox potential, the seepage contains some dissolved iron (average 0.33 mg/L) and manganese (average 3.3mg/L). Mn was originally highlighted as the main element of concern for discharge to the Nam Mo. To address this issue, PBM installed a wetland treatment system comprising a rock-filled flow-through cell for oxidative Mn removal.

Conclusions

The Phu Kham site presents significant challenges for managing waste, with a high proportion of PAF material in a steep, high-rainfall and sensitive environment. Early identification of the risks prior to mining and integration of the geochemical requirements for containment with the mine plan are enabling PBM to manage ARD during operations without any significant adverse events. Key to the success of the management plan has been company-wide awareness of the risks associated with the high proportion of PAF waste and diligent operational management with regular auditing and technical review through a formalised committee structure.

Comprehensive performance monitoring demonstrates that with good design, focused monitoring and day-to-day management, ARD risk can be managed in a cost-effective manner from the start of operations. The management strategies that have been implemented during operations will continue to work post-closure to minimise future ARD risk.

REFERENCES

Miller, SD, Rowles, T, Millgate, J, Morris, L, Pellicer, J, Gaunt, J (2012). 'Integrated acid rock drainage management at the Phu Kham Copper Gold Operation in Lao PDR', *Proceedings of the Ninth International Conference on Acid Rock Drainage* (ICARD), 20–26 May 2012, Ottawa ON, Canada.

6.2 Management of tailings to minimise AMD

6.2.1 Overview

No single solution exists for the secure management of tailings, given the large range of site-specific environmental conditions, ore types, geochemistry, topography and other constraints. However, any proposed tailings management option should minimise interactions between the tailings and the local environment to prevent acid generation, solute leaching and potential impacts to surface water and groundwater during operations and after closure.

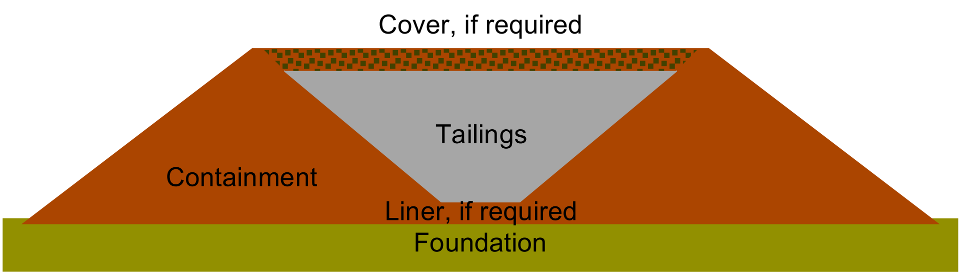
Full details of the issues involved in the deposition and storage of tailings and the closure of TSFs are in the *Tailings management* handbook in this series (DIIS 2016e). This section focuses on specific aspects of tailings management that minimise the future AMD risk posed by tailings.

Tailings associated with gold and base metals production have conventionally been deposited in a wet (saturated) state during the operating life of the mine, and often at an alkaline pH owing to the addition of lime to facilitate the flotation and/or extraction of the target mineral. Thus, apart from the surface layer on tailings beaches exposed to the atmosphere, there is only a low risk of oxidation of the gangue sulfide minerals in the bulk of the tailings mass and the consequent production of acidic seepage during the operational life of the facility. While there are moves towards depositing tailings with lower initial water content (for example, thickened tailings or paste tailings) and to central deposition to achieve greater stability and better consolidation, the water content of such tailings is still quite high.

The oxidation potential of reactive tailings is driven mainly by the diffusion of oxygen through the desiccated surface of the tailings. The oxidation products may then be transported by tailings water and/or rainfall run-off from the tailings surface or seepage through the tailings.

Because tailings are conventionally deposited in slurry form (at various solids concentrations), their surface storage requires some containment or encapsulation (Figure 25). However, the form of encapsulation varies. A base liner may or may not be required, depending on the ground conditions and the risk posed by the tailings water.

Figure 25: Encapsulation of reactive tailings



In the early phases of a mine's life, the containment wall generally comprises borrow material or run-of-mine weathered rock. Later, it may involve the use of rehandled dry tailings, with an outer protection of benign waste rock and/or soil. In some cases, where excess NAF and/or AC waste rock is available and the TSF is in close proximity to the pit, the perimeter of the TSF may be encapsulated by a thick layer of this rock. This has the added advantage of providing a buffer against the possible future loss of encapsulation through erosion.

The tailings should be deposited at as high a percentage of solids as possible to limit the amount of pore water available to report as seepage. This is facilitated by the use of high-rate thickeners or paste tailings technology. Evaporative drying by cycling tailings deposition between cells, potentially combined with dry stacking, should be taken advantage of where possible to limit seepage during operations. However, this strategy may also expose sulfidic tailings to oxidation.

The impact of ongoing rainfall run-off following the closure of a TSF needs to be considered. For example, this may require the installation of a low net percolation surface cover (see below) to minimise infiltration and a spillway for the release of water to relieve the head of water that would otherwise drive vertical seepage of contaminated pore water or cause the catastrophic collapse of the containment walls.

As noted above, tailings are conventionally deposited in a saturated state, which contrasts with waste rock, which is deposited unsaturated, thereby increasing the rate of sulfide oxidation. However, following the cessation of tailings deposition, the phreatic surface falls in the tailings mass, and the extent of drain-down is greatest closest to the containment walls where the coarsest material has often been deposited. This desaturation process can expose the tailings to oxygen and oxidation, and the oxidation products are flushed downwards during successive wet periods. TSFs can thus substantially lag WRDs in the production of an obvious AMD signature in seepage from the base.

Several strategies are available for reducing the future potential for tailings to generate AMD, depending on whether the tailings have been deposited in an above-ground constructed TSF or below ground in a mined-out pit. The latter option is most typically available at a multi-pit operation, in which the first pit to be mined out can receive the tailings produced from subsequent mined ore bodies. However, the relocation of tailings from the TSF to the pit at the end of processing may still be the most appropriate option for ensuring the long-term mitigation of high AMD risk tailings at a single-pit mine.

Note that a commitment to relocate tailings from the TSF to the pit at the end of processing should not be made lightly. If the open-pit operation progresses to an underground operation, there may be insufficient capacity for storage of the additional tailings in the open pit. In this case, there still needs to be an above-ground TSF, albeit of reduced size, which has to be rehabilitated and which may require significant ongoing management.

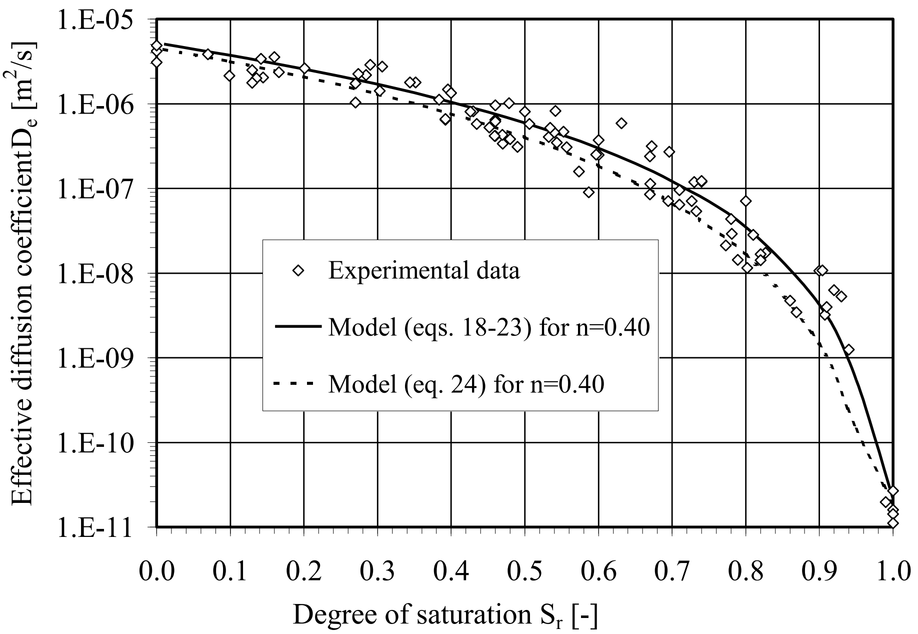
6.2.2 Water covers for tailings

The tailings may be able to be maintained under a water cover in perpetuity. A strong relationship exists between the degree of saturation of fine-grained materials (such as tailings) and the rate of sulfide oxidation (Dagenais et al. 2006; Hutchinson & Brett 2006; Bussière 2007). The rate of diffusion of oxygen through water is approximately 10,000 times slower than through air. This explains why a water cover is so effective at reducing the rate of sulfide oxidation.

Figure 26 illustrates the relationship between the coefficient of oxygen diffusion and the degree of saturation for soils or porous media. Oxygen diffusion rapidly decreases by three or four orders of magnitude, comparable to the oxygen diffusion coefficient in water, as the degree of saturation increases above 85%. Thus, if the degree of saturation in tailings can be maintained above 85%, the oxidation rate can be reduced to an extent similar to the rate that can be achieved using a free water cover over the tailings.

If a water cover strategy is to be considered for an above-ground TSF, the operation must be in a climate zone with a favourable positive annual water balance, or else the topography must be such that water can be harvested from local catchments to maintain the water cover. This normally requires a net positive water balance climate, generally limiting its application in Australia to Victoria, Tasmania and the wet tropics. Water covers over reactive tailings have been used in Canada (Ludgate et al. 2003), in the equatorial tropics and in temperate Australia.

Figure 26: Coefficient of diffusion versus degree of saturation for porous media



Source: from Bussière (2007).

Backfilling a pit with tailings, whereby the sulfidic material is maintained below the recovered watertable level and the rest of the pit is filled with benign material, has the following intrinsic advantages over the free water surface option:

- The tailings will be covered by the rebounding groundwater level.
- There are unlikely to be problems with the long-term stability of the containment structure.
- This strategy can be used over a much broader climate range.

In contrast with the backfilled pit option, the embankments around the TSF must have been constructed to provide a sufficient factor of safety to reduce the future risk of embankment failure to acceptable levels.[41]

To be effective, water covers require the topography and rainfall to provide a minimum water depth of 1.5–2 m, preferably more, depending on the potential for the resuspension of fine-grained reactive tailings by surface wave action and currents (Catalan & Yanful 2002). The greater the annual flushing of the water cover by through-flow of water from the surroundings, the better. The feasibility of this long-term water-quality management strategy is determined by site topography and climate.

Water covers have become established as industry leading practice primarily for unoxidised tailings material when the geochemistry, climate, catchment and net water balance permit. This strategy is not recommended for sulfidic waste (tailings or waste rock) that has appreciably oxidised and contains a large load of readily soluble oxidation products. For oxidised material, an infiltration-limiting cover is a more suitable option.

Water covers can be a relatively uncomplicated and cost-effective solutions for tailings dams that have been constructed as engineered water-containing structures. However, the vast majority of tailings impoundments are constructed as solids storage facilities that do not have sustained water-holding capacity. Water covers over such structures can be engineered, but require considerably more effort and expense and possibly maintenance than dams that were constructed as water-containing structures with an appropriate factor of safety. Failure to go to that effort and expense can result in catastrophic embankment failure, such as occurred in August 2014 at the Mt Polley copper and gold mine in Canada (Government of British Columbia 2015).

A hybrid cover system—a central permanent pond with a peripheral annulus of 'dry' cover—is an option that can be engineered to ensure that the phreatic surface is kept sufficiently low in proximity to the dam wall, which may not have been constructed to long-term water-holding standards, to provide a sufficient factor of geotechnical safety. The cover system that has been installed at the Henty mine in Tasmania is an example of this approach.[42]

An elevated watertable (relative to the surrounding ground surface) maintained within a closed-out TSF can also be a viable alternative to a surface water cover if specific conditions are satisfied. The lower watertable facilitates a sufficient geotechnical factor of safety for the structure. For this option to be viable, the residual ANC in the column of tailings beneath the watertable must be sufficient to more than neutralise all of the potential acid-generating capacity in the layer of unsaturated tailings remaining above the watertable.

6.2.3 Covers for tailings

If a total or partial water cover cannot be maintained, the placement of a cover system to limit the infiltration of rainwater and/or diffusion of oxygen will be required. However, the partially consolidated tailings mass that is typically present at the end of the operating period poses major challenges for the design and placement of a cover system.[43]

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41 See the Benambra case study in the *Tailings management* leading practice handbook (DIIS 2016e).

42 See the case study in the *Tailings management* handbook (DIIS 2006e).

43 These challenges and ways of overcoming them during the operational life of the TSF are described in the *Tailings management* handbook (DIIS 2006e).

In many cases, fine-grained material sourced from elsewhere is proposed as one of the components of a cover system for tailings and waste rock. Presuming that such material is available, this generally requires the opening of a borrow pit to extract often quite large volumes, and the consequent need to rehabilitate the footprint of the borrow area. Costs for sourcing this material can be very high if substantial haulage distances are involved.

However, tailings are by their nature fine-grained and may have many of the properties, apart from elevated sulfide content, required for an effective infiltration and oxygen diffusion limiting cover layer. This realisation has led to the initial trialling and subsequent full-scale application of desulfurised tailings—that is, the intentional removal of a large proportion of the sulfides (usually by froth flotation, although gravity separation may be an option in some cases) to produce a product that can be used for progressive containment/rehabilitation (for example, in a multi-celled TSF) during operations. Indeed, a slurry tailings system provides the ideal hydraulic method for conveying and placing such a cover layer (Duckett & O'Kane 2006).

Desulfurised tailings have been used for cover placement at the Inco Copper Cliff operations in Sudbury, Canada (Hanton-Fong et al. 1997), and desulfurised tailings have also been used as the final cover layer at the Renison Bell tin mine in Tasmania.

The application of a desulfurised tailings cover at Renison was strongly supported by detailed geochemical and reactive-transport modelling (Romano et al. 2002) and appeared in a case study in the first edition of this handbook (Environment Australia 1997). The mine is now owned and operated by Bluestone Mines Tasmania Joint Venture Pty Ltd. The long-term water-quality monitoring record for the main seepage point from two tailings dams has provided good evidence for the effectiveness of this cover system, the application of which started in the late 1990s. Compared with 1994, there has been a rise in the average annual pH from 4.1 to 5.7, a 10-fold reduction in the acidity (mg $CaCO_3$/L) and concentrations of Fe and Zn, a threefold reduction in Mn and a 24-fold reduction of Al in the seepage. Given that the discharge rate of the seepage is now about half that in 1994, there have been very large reductions in the annual loads of metals and acidity.[44]

The desulfurisation of tailings may also be applied more broadly to reducing the AMD risk of this material to the environment. Case study 8 illustrates how effective this has been in reducing the future environmental legacy of a riverine tailings disposal system in Papua New Guinea.

Irrespective of the final closure option that is selected, leading practice TSF design for sulfidic tailings must take account of the need to sustainably optimise the saturation (that is, the water content) of the tailings mass, such that the geochemical integrity of the bulk of the material and the geotechnical integrity of the containment structure can be maintained over the long term.

44 Further information about tailings rehabilitation at Renison is in a case study in a recent leading practice booklet produced by the Minerals Council of Australia (MCA 2015b).

Case study 8: Removal of sulfides from tailings to improve closure prospects, Ok Tedi mine, Papua New Guinea

Context

Ok Tedi Mining Limited (OTML) operates a large copper and gold mine in the Star Mountains of the Western Province, Papua New Guinea. The management of tailings at the Ok Tedi mine is complicated by the inability to construct a conventional TSF adjacent to the mine. As a consequence, the mine has practised riverine disposal of mill tailings into the Ok Tedi (Ok means 'river' in the language of the local people), which then meets the Fly River at D'Albertis Junction before ultimately discharging through its delta into the Gulf of Papua. Figure 1 shows the location of the mine.

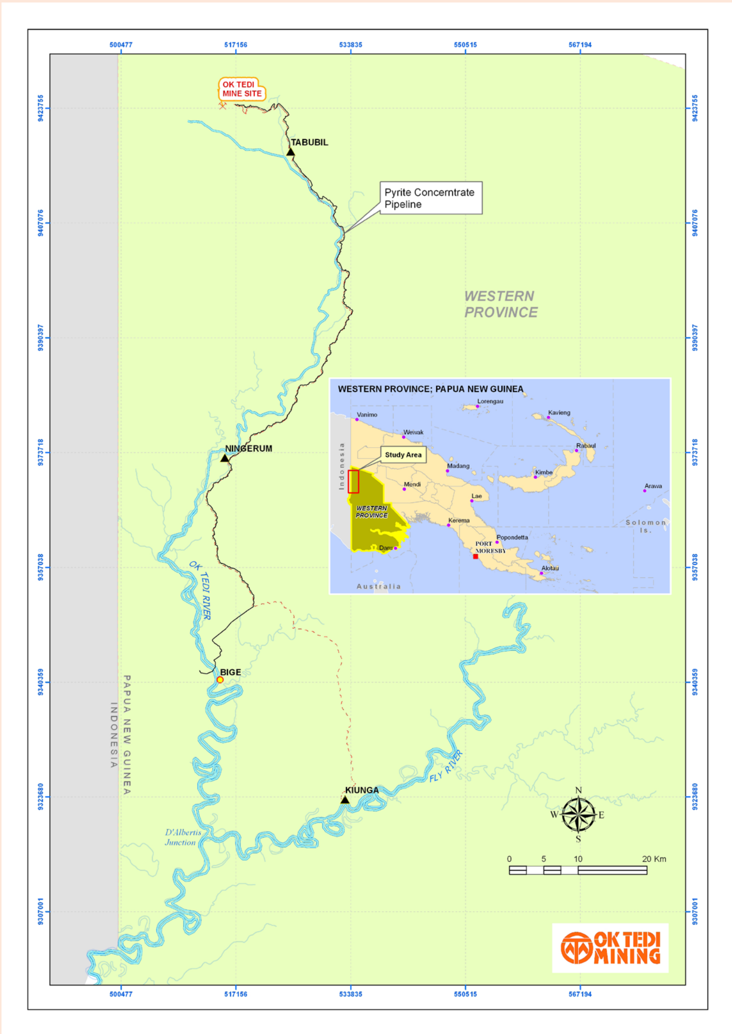


Figure 1: Location of Ok Tedi mine site, Bige dredge and pyrite concentrate (PCon) containment facility

Mitigation of AMD risk of riverine tailings

To mitigate the effect of the riverine tailings disposal, OTML has operated a dredge at Bige, approximately 100 km downriver of the mine, since 1998. The dredge captures the sand-sized fraction of the mill tailings (estimated to be approximately 70% of the tailings) in addition to the sand that is derived from waste rock dumps (WRDs), natural erosion and pre-existing mine derived sediments. In 2014, 18.8 Mt of sand was dredged from the river and placed in engineered stockpiles on either side of the Ok Tedi. Figure 2 shows some of the arrangements in the area.

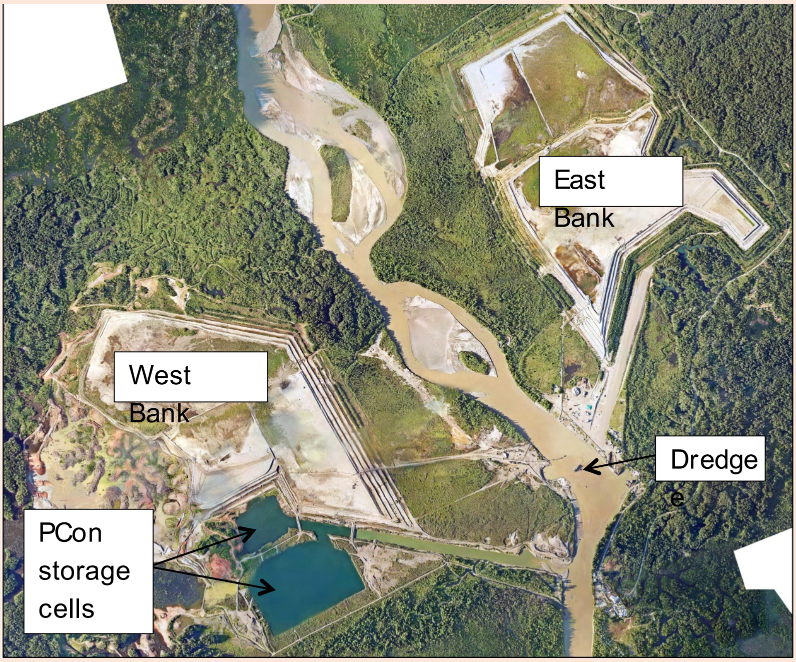


Figure 2: Aerial photo of the Bige dredge location. The Ok Tedi flows from top to bottom. There are stockpiles on both sides of the river, with pyrite concentrate (PCon) storage cells on the west bank

Riverine waste disposal poses a risk of AMD developing in the overbank deposited sediments and in dredged material unless control measures are put in place. It is fortunate that the Ok Tedi mineral deposit is located in an environment dominated by limestone. More limestone is added to the WRDs, resulting in a relatively high ANC in the waste rock derived sediments in the river.

In the early 2000s, OTML management recognised from its ore block model that ore feed sulfur grades were scheduled to increase to 8% towards the projected end-of-mine life (then in 2013, now in 2026). The effect of this increase would have been to have raised the existing AMD risk to an unacceptable level. To control the developing AMD risk, OTML implemented the mine waste tailings project (MWTP) in 2008.

The MWTP consisted of a tailings processing plant which treats the copper concentrator tailings to remove pyrite as a concentrate (PCon) and produce a depyritised tailings stream with a target mean sulfur content of less than 1% that is discharged to the river. The tailings processing plant (Figure 3) is designed to process in excess of 20 Mt/a of tailings that will meet this target for mean sulfur feed grades of up to 8%. The total capital and operating costs of the MWTP was greater than US$510 million as at the end of 2014.

Figure 3: Tailings processing plant

The PCon is sent via a 130 km pipeline to Bige (Figure 1), where the material is stored subaqueously below the natural water table in dredged pits on the west bank of the Ok Tedi (Figure 2). As the PCon pits are filled, they are covered with at least 12 m of dredged NAF sand. The thick sand cover raises the level of the natural watertable in the cells via the process of groundwater mounding. The level of the watertable was modelled under a variety of rainfall scenarios and was found to be higher than the surface of the deposited PCon in the driest of years. This permanently saturated state will ensure long-term isolation of the PCon from atmospheric oxygen and thus prevent its oxidation.

Performance Monitoring

Since the implementation of the MWTP, there has been a marked decrease in the sulfur content of the discharged tailings, with the 1% target having been consistently met on a mean annual basis since 2010 (Figure 4).

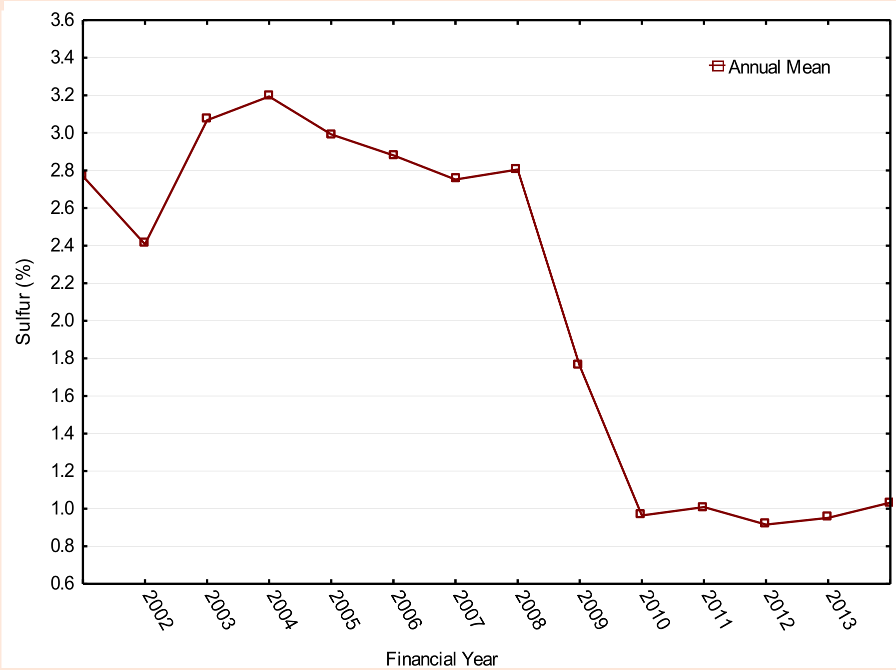


Figure 4: Time series sulfur concentration in tailing by financial year 2000–01 to 2013–14

More importantly, the pyrite content of sediments in the river system has decreased since the implementation of the MWTP. The dredged sand stockpiles at Bige were a particular potential AMD risk as the material is relatively coarse and free draining with elevated levels of sulfur (3% prior to the implementation of the MWTP and projected to continue rising towards 8%). From a level of 3% prior to the MWTP, the mean value in the dredged sediment has reduced to 1.5% for 2013–14 (Figure 5). It is notable that the sulfur concentration at Bige has taken several years to decline following the inception of release of 1% sulfur tailings from the mine. The reason for the lagged response is that the material dredged at Bige is made up of a combination of freshly released tailings and material with much higher sulfur concentrations that had been deposited into the 130 km stretch of river upstream of Bige, prior to the MWTP.

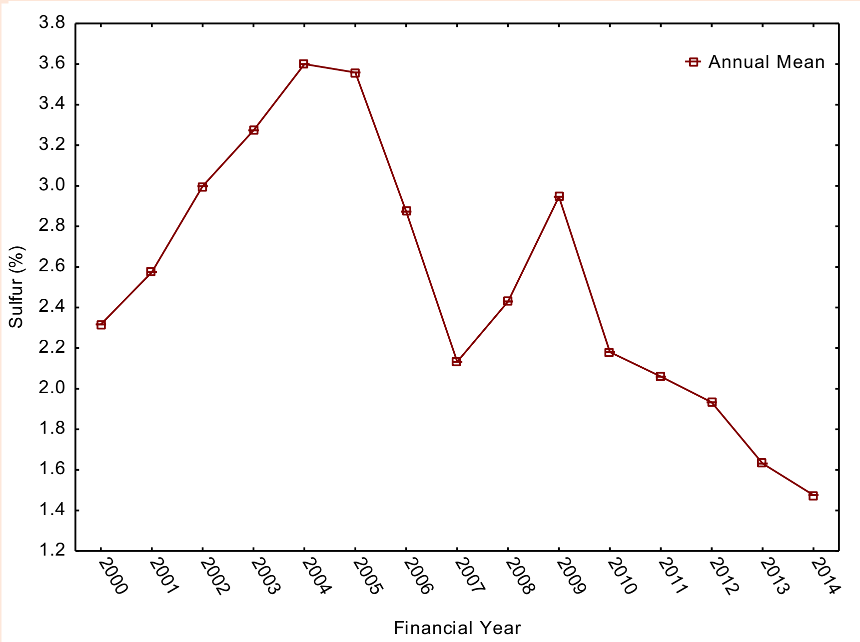


Figure 5: Variation in sulfur concentration in dredged sediment by financial year 2000–01 to 2013–14

In addition to removing pyrite from the tailings, OTML adds limestone (and hence ANC) to the river system. This is achieved through two pathways:

- addition of limestone to the failing WRDs on the steep mountainsides of the upper Ok Tedi system
- adding (since 2011) approximately 4,000 tonnes of additional limestone per day through the milling circuit at the concentrator.

The rationale for adding limestone through the mill is that it directly adds ANC to the finer sediment fraction in the river system, including the dredged material at Bige and the sediments transported further downstream to the Ok Tedi floodplain.

A measure of the risk of AMD is the ratio of ANC to the MPA calculated from the pyrite content of the sediment. An ANC:MPA ratio less than 1 indicates a high probability of acid generation, whereas a ratio greater than 1 indicates a lower probability of acid generation. The higher the ANC:MPA ratio, the greater the confidence that net acid generation will not occur in the future. In the case of the sand fraction (for example, cover material at Bige) sourced from Ok Tedi ore and waste, the risk of AMD generation becomes negligible above an ANC:MPA ratio of 1.5.

The combination of the MWTP and the limestone addition has seen a marked improvement in the ANC:MPA ratio in dredged material since 2004 (Figure 6).

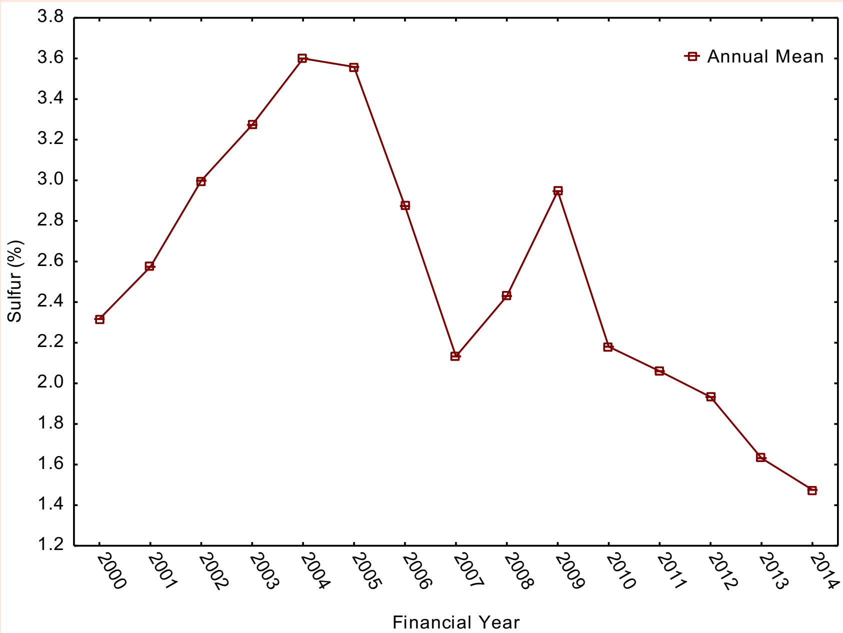


Figure 6: Time series ANC:MPA ratio in dredged sediment at Bige by financial year since 2000

It can be seen that the ANC:MPA ratio has increased from a low of 0.6 in 2004–05 to 2.3 in 2013–14. This means that the dredged material currently being produced at Bige can be used as a NAF cover for placement over older PAF material and that OTML can rehabilitate the stockpiles with relatively little risk of acid hotspots developing on their covered surfaces.

Conclusions

While the Ok Tedi experience is atypical due to the use of riverine disposal for tailings, it can be seen that, through the implementation of the MWTP, in combination with the addition of extra limestone to the WRDs and the milled ore, a marked decrease has been achieved in the AMD risk associated with the mine's operation and ultimate closure. The use of tailings desulfurisation to both reduce downstream AMD risk and produce a suitable NAF cover material is an example of a leading practice implementation of this technology.

6.3 Soil cover systems for waste rock and tailings

Most mine waste storage facilities are located on the surface, resulting in an elevated landform. Different treatments are needed for the flat top surface and for the sloping sides of these storage facilities.

6.3.1 Covers on flat tops

Covers on flat surfaces comprise one or more layers of geochemically benign materials with specific physical properties intended to limit the percolation of rainfall and/or the ingress of oxygen into stored reactive wastes. Further information about required material properties is in this section and in the cited references.

Soil covers must maintain an acceptably low risk of harm to society and the environment from AMD over a very long period. They must also be resistant to breakthrough by erosion, plant roots or burrowing animals. The components listed below (in sequence from the surface) can make up a cover system, but note that the range of design elements (and their thicknesses) that are ultimately included depends on the site-specific climate, the geochemical and physical characteristics of the waste, material availability and performance requirements. For example, at most coalmines in Australia, available sedimentary rock types are insufficiently competent to construct and maintain capillary break layers. At those operations, the application of capillary breaks within a cover system would not only be very expensive, but also likely to be subject to failure through differential settlement and the migration of fines into the capillary break layers over the longer term.

The possible components of a soil cover system include:

- topsoil—normally a key component, requiring a high water storage capacity, a reasonable nutrient cycling capacity and sufficient depth (>0.5 m) for plant roots
- upper capillary break—durable, benign fresh rock with minimal fines, if required to limit root penetration into the underlying seal; this layer requires a low air-entry value (desaturation pressure), low water storage capacity and a thickness of at least twice the height of capillary rise
- sealing layer—a key component comprising compacted clay, if available, compacted mine wastes or a geosynthetic composite clay liner; this layer requires a low hydraulic conductivity ($<10^{-8}$ m/s) to hold up rainfall infiltration and a high air-entry value to maintain saturation to reduce the diffusion coefficient of oxygen
- lower capillary break—if the mine wastes are saline and/or potentially acid-forming, to limit the upward migration of contaminants into the cover.

In wet climates, where it is difficult to stop rainfall percolation, the main function of a soil cover is to limit oxygen ingress and so limit the oxidation of the stored reactive wastes and the production of AMD.

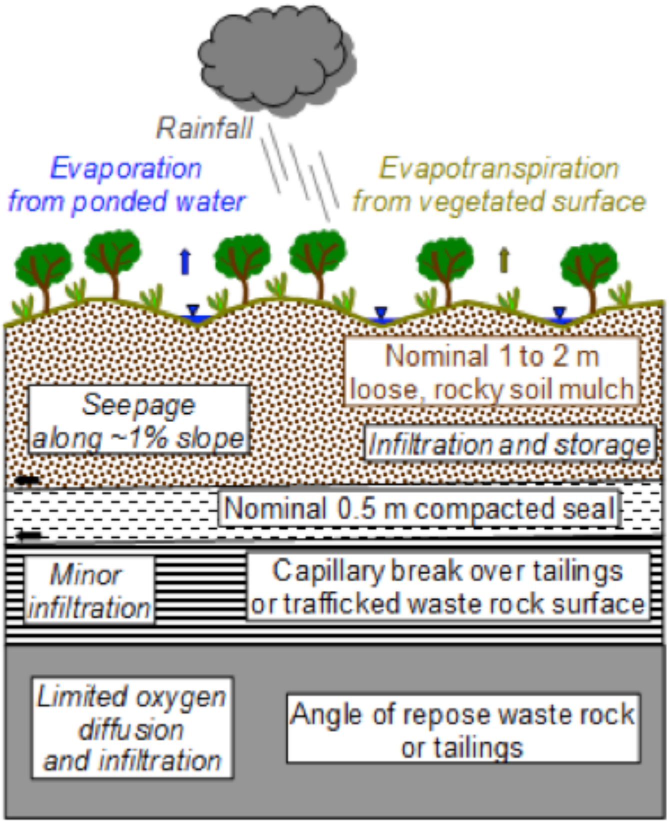
A shedding or barrier cover typically comprises a compacted clayey soil seal about 0.5 m thick, overlain by a growth medium that may be as thin as 0.3 m. This thickness of growth medium may support grasses but is inadequate for most shrubs and trees. It is also too thin for all but the wettest of climates, as extended dry periods will result in the desiccation of the underlying clay layer, leading to cracking and loss of performance. For example, cover performance modelling has indicated that a top layer thickness of at least 1 m and possibly over 2 m is needed to maintain saturation of the clay layer in the wet–dry tropics.

Covers containing a clay barrier layer are not recommended for use on soft tailings. A compacted clay sealing layer placed on initially unconsolidated tailings is bound to fail due to shear stresses caused by the ongoing differential settlement of the tailings mass.

Shedding or barrier covers have been used with reasonable success on both flat surfaces and steep slopes at a number of wet climate mine sites, including the Savage River iron ore mine in north-western Tasmania (see case study 5 in this handbook) and in wet climates overseas. A well-vegetated cover can handle the high rainfall while limiting excessive erosion. In dryer or seasonally wet–dry climates, where it is difficult to maintain a soil cover in a saturated state, the main function of the soil cover is to limit the percolation of rainfall into the wastes (Figure 27). In order to construct a cover system over tailings, it may be necessary to first develop a construction platform over the tailings. This could also serve as a capillary break, if practical and required.

Recognising the potential shortcomings of a shedding or barrier cover at dry-climate mine sites, a store-and-release cover system incorporating a single capillary break layer was developed during the mid-1990s for covering WRDs at Kidston goldmines in north Queensland (Williams et al. 2006). The system is designed to store wet-season rainfall without shedding it, since excessive run-off would lead to the erosion of the cover, and to release the stored water through the dry season through evapotranspiration, with no net wetting up or drying out of the cover from year to year. A store-and-release cover can significantly limit the percolation of the average annual rainfall (Williams et al. 2006), thereby reducing ongoing wetting of the underlying mine wastes and the production of seepage. Other references to the effectiveness of store-and-release covers are provided in O'Kane and Ayres (2012).

Figure 27: Schematic of store/release cover



Source: Williams et al. (1997).

Factors crucial to the success of a store/release cover are:

- a hummocked / truck-dumped top surface to prevent run-off
- a sufficient thickness of loose, rocky soil to store water from prolonged heavy rainfall
- a sealing layer of sufficiently low hydraulic conductivity to hold up most of the rainfall infiltration stored in the loose rocky soil layer
- a sustainable vegetative cover to transpire stored rainfall that is not evaporated.

The Kidston cover system was a success from the point of view of limiting net infiltration (Williams et al. 2006). However, the cover was placed on the top of the WRD towards closure following many years of exposure to oxygen and rainfall infiltration, which continued to draw down and produce seepage of poor quality, which emerged at low points around the toe of the facility. This occurrence illustrates the difficulty of retroactively controlling the production of contaminated AMD seepage by the installation of a cover after significant oxidation and rainfall infiltration into the uncovered WRD have occurred.

More information about the design of cover systems and the characterisation and properties of materials used for their construction is in the GARD Guide and in a comprehensive four-volume report series published by the Canadian MEND program (MEND 2004).

6.3.2 Treatment of outer slopes

The outer slopes of mine waste storages are necessarily steep, and their stability is often problematic over the long term due such practices as:

- reshaping (flattening) slopes by dozing—this reduces the 'roughness' of the surface by crushing and burying coarse-grained materials, resulting in increased run-off and reduced erosion resistance
- increasing slope lengths by flattening slopes of a given height—this increases their catchment and the potential erosion for a given surface treatment
- concentrating rainfall run-off in contour and downslope drains—this increases the potential for tunnelling and gully erosion
- constructing drainage structures that are inadequate due to underlying settlement, particularly on contour benches and at connections
- placing fine-grained and/or dispersive growth media on steep slopes, which are particularly prone to erosion.

WRD and TSF outer embankment slopes generally have adequate geotechnical and erosional stability. However, the conventional rehabilitation of such slopes can result in a final slope with adequate geotechnical stability but inadequate erosional stability. Alternative approaches to creating stable final slopes, drawing upon surrounding natural analogues, offer the potential to produce sustainable slopes of high geotechnical and erosional stability and improved aesthetics. Natural slopes are generally concave and are armoured with rock, cemented cap rock and vegetation.

The methods used to stabilise mine waste storage outer slopes vary greatly depending on the climate and surface materials. Heavy vegetative cover may be highly successful in reducing erosion in some areas, whereas areas with seasonal, arid or semi-arid climates may not support sufficient vegetative cover to control erosion. Such areas require other erosion protection strategies, including the limitation of slope catchment or the placement of a surface cover of coarse-grained benign waste rock. Rock may be mixed with underlying material or some fines may be added to the mix to enhance water retention and the potential for some revegetation.

While contour and downslope drains have a poor performance history, substantial rock-filled gullies could be constructed to handle excessive rainfall run-off. Angle of repose final slopes, which limit the cost of slope construction, may be possible on the upper part of the slope, provided that some profiling of the slope is conducted (that is, incorporating concave slope profiles). Concave slope profiles, which mimic natural slopes, limit the loss of sediment from the slope. Monitored trials are generally needed to develop the most appropriate slope treatments for the particular mine site.

6.3.3 Cover design and performance

The key performance-related features that need to be considered during the design of cover systems to minimise or prevent AMD are:

1. minimisation of oxygen ingress
2. minimisation of net infiltration (that is, the proportion of incident rainfall that penetrates to the reactive material)
3. long-term stability (for example, compacted clay layer, capillary break, drainage layers) of the components used to construct the cover system
4. long-term resistance to surface erosion
5. sustainability of a vegetation cover when required for aesthetic and/or functional reasons (for example, store-and-release covers).

Numerical modelling can be applied to the design of covers to ensure that they will minimise the generation and escape of sulfide oxidation products over the long term. The objective is to limit the ingress of oxygen and/or water, which are the two reactants necessary to drive the formation of AMD. Limiting water ingress also minimises the transport of oxidation products.

A number of suitable computer programs are available for this purpose. They are commonly based on the finite element method and incorporate unsaturated soil mechanics parameters. Most analyses are now two-dimensional. They are driven in the first instance by historical climatic data for the site. However, the potential effect of climate change on input parameters also needs to be addressed, given that performance needs to be maintained over the long term.

When designing soil covers using models, it is necessary to fully understand the geochemical and geophysical characteristics of the material to be covered and also the materials available to construct the covers. Every soil and rock type behaves differently, and laboratory testing is necessary to provide the input parameters for the models. For a more detailed listing of available models and indications of their applicability, refer to the GARD Guide.[45]

Depending on the availability of soil materials and their variability, many tests may be needed to adequately characterise them. Monitored trials are generally required to verify the cover design and to select the most appropriate vegetation species for the particular mine site. The installation of monitoring equipment and the interpretation of the collected data will be required to 'demonstrate' the cover's performance and also to recalibrate the model based on performance monitoring data. Since covers are dynamic systems that are crucially dependent on the vegetation cover and its ability to cope with any climatic variations, long-term monitoring is essential to demonstrate sustainable performance.

45 http://www.gardguide.com.

Rainfall-shedding or barrier covers are likely to perform poorly in seasonal climates and to fail in semi-arid and arid climates, where the vegetation cover is poor and the sealing layer is prone to cracking, root penetration and erosion. While compacted clays may initially provide a hydraulic conductivity of less than 10^{-8} m/s or 300 mm/year, cracking that occurs during dry periods increases it by about a hundred-fold and they no longer seal as well.[46]

The results from monitoring of cover performance over the intermediate term (one to two decades) are discussed in Taylor et al. (2003), O'Kane Consultants Inc. (2003) and Williams et al. (2006). Wilson et al. (2003) attempted to place in perspective the short- and long-term integrity of various cover systems, ranging from simple vegetative covers to complex composite covers used for landfills, and costing from $10,000 to $400,000/ha. Among the low- to mid-cost range covers typically used at mine sites, compacted clay barrier-type covers costing about $35,000/ha are known to (generally) perform poorly over the longer term, while store-and-release covers costing about $50,000/ha (for example, at the Kidston goldmines) have performed far better (Wilson et al. 2003). In this context, it should be noted that these quoted per hectare costs are from before 2003, so the cost to build similar covers today will be substantially higher.

The likelihood of significant deterioration in the performance of covers through time must be factored into the overall design of the waste containment facility, in the context of developing sustainable post-closure performance objectives and criteria.

6.4 Blending and co-disposal of wastes

Blending of materials has not been routinely adopted in the Australian mining industry, mainly due to the logistical problems and costs associated with scheduling, delivering and mixing significant volumes of mine wastes.

Blending and co-disposal for backfill does occur in underground mines, and the pumped co-disposal of coal washery wastes is reasonably common in the coal industry. PAF material is sometimes mixed with cement or a mixture of cement and tailings and placed in underground voids as backfill. In addition to its role as a strength-conferring binder, the cement matrix has significant neutralising capacity.

The blending of PAF waste rock material with carbonate rock or co-disposal with carbonate-bearing waste rock material has had a mixed history of success because of inadequate mixing, the addition of an insufficient excess of neutralising capacity and the use of too coarse a size range for the neutralant. The armouring of carbonate grains with precipitates of AMD neutralisation products significantly inhibits the dissolution of carbonate minerals, so a substantial excess of neutralising capacity is needed if this strategy is to work over the longer term. The use of neutralant materials at sites where substantial oxidation of waste has already occurred must also take into account the storage of acidity in secondary minerals such as jarosite.

Control of AMD through the use of high-carbonate mine waste (blending/covers) has been demonstrated at Ok Tedi (Papua New Guinea), Grasberg (Indonesia) and Savage River (Australia). At both Ok Tedi and Grasberg, a substantial factor of safety (ANC/MPA >2) in the reactive fraction of the waste rock and limestone blend is needed to provide assurance of performance over the longer term (Miller 2014).

In general, the addition of crushed limestone during the construction of WRDs from the ground up using

46 A comprehensive review of these issues, together with relevant case studies, is in MEND (2004).

thinner vertical layers (as described above) or by using crusher/stacker or conveyor methods for placement have a better chance of achieving the effective use of neutralant than the end-dumping of run-of-mine carbonate material down the faces of active dumps.

6.5 Underground mines

Many historical underground mines in mountainous or hilly terrain were carefully located and designed to permit drainage adits to passively dewater as much as possible of the country rock surrounding mineral deposits. This approach readily overcame dewatering problems but effectively maximised the potential for AMD generation and discharge.

Despite such lessons from the past, little thought is often given to the post-closure environmental implications of locating and designing mine access portals at current underground mines, with the consequence that many mines discharge AMD-affected water from their portals. This poor-quality drainage is generated from wall rock surrounding the mined area, as well as from backfilled sulfidic mine wastes. In addition, modern mines that use block cave methods also generate very large volumes of highly fractured sulfidic ore that can remain exposed underground for years, in contact with air and water, and thus generate poor-quality drainage during operations.

The AMD challenges associated with underground mines are best addressed during the early stages of mine planning, with a focus on prevention and minimisation strategies. Leading practice strategies include the following:

- Avoid the need to employ (high-risk) pressure bulkheads after closure by planning to position the mine portal at a topographic high point, ensuring that all unsaturated wall rock and any sulfidic backfill will be passively inundated by natural post-closure groundwater rebound.
- Avoid developing a post-closure mine spill point if possible (depending on local hydrogeology).
- Strategically align decline and development drives to minimise the likelihood of intercepting PAF material during mining. For example, mineralisation types such as volcanic-hosted massive sulfide deposits can have a mineralised footwall and a relatively barren hanging wall. Targeting the latter for development drives could substantially lower PAF waste rock production and therefore AMD risk, provided that the hanging wall is geotechnically competent.
- Segregate waste so that PAF rock can be retained underground for mine backfilling and eventual inundation below groundwater.
- Avoid storing PAF waste rock and tailings above the final passive minimum mine water spill level if post-closure mine spilling is difficult to prevent. All sulfidic backfill wastes may need to be under a minimum 2 m permanent cover by groundwater (Oxley et al. 2008).

The prevention and minimisation of AMD remains a significant issue for underground mines, particularly at brownfield sites, and emerging technologies such as inert atmosphere systems (Taylor & Waring 2001) are currently being tested to address such challenges (Section 11).

6.6 Pit lakes

6.6.1 Background

Pit lakes continue to be of major interest in the development of closure planning strategies, and are becoming an increasing feature at completed mine sites around Australia (Kumar et al. 2013). However, such lakes can potentially represent significant public health, environmental and financial liabilities for many jurisdictions, particularly as the full effects of acidity and solute concentrations may not become apparent for several decades after mine closure. Pit lakes may also become important wildlife habitats where some solutes can potentially enter local food webs.

These closure landforms may substantially modify future land-use options and mine closure risks at the landscape scale (McCullough and Van Etten 2011) in areas where many large pit lakes are expected (such as the Pilbara and Collie Coal Basin regions in Western Australia and the Bowen Basin in Queensland). Thus the cumulative effect on, for example, the groundwater level and quality of another nearby pit lake may also need to be addressed as part of closure planning.

Anticipating the ultimate end use for a mine pit at the start of a mining project is not necessarily a clear-cut task, especially since the boundaries of the economically recoverable resource may change throughout the life of the project. At the end of the open-cut phase of operations, whether or not to allow the pit to fill with water may need to be considered in the context of maintaining the potential for future underground access. The dewatering of a pit that has been allowed to fill can be a very costly exercise if the water requires chemical treatment before re-use or discharge. For these reasons, planning the end use for a mine pit against a backdrop of an initially indeterminate configuration and mine life is likely to be one of the most challenging aspects of the close-out and rehabilitation of the mine.

6.6.2 Predicting water quality in pit lakes

Advances over the past decade in the prediction of water quality have been made possible by some well-monitored international examples of full-scale field implementation and by the evolving capacity of coupled hydrodynamic and solution geochemical models. However, pit lakes as closure options can still present a substantial potential for AMD legacy risk and liability (McCullough & Lund 2006), given the long periods of time that these large bodies of water will be interacting with their surrounding environment. In this context, the distinction needs to be made between those pit lakes that have essentially developed without any planning (legacy sites) and those that have been produced as a well-developed component of a strategic closure plan, with defined water-quality objectives. While the focus of this section is on the latter, the approach that is described is also applicable to legacy pit lakes.

Understanding the evolution of pit lake water quality over the long term is critical for assessing and managing the closure risk presented by this option (Castendyk & Eary 2009). The complexity and extent of interactions between hydrological and geochemical factors means that the prediction of pit water quality is a highly site-specific exercise. It should be done by following a risk and outcomes-based approach using robust scientific methodology.

At the simplest level, a prognosis for long-term pit lake water quality may be able to be obtained by comparing the proposed lake with nearby pit lakes or pit lakes elsewhere that have formed in historical workings with a similar geological context. The predictive ability can be improved by incorporating site-specific findings from basic geochemical assessments, such as static and kinetic tests (Section 4). If initial assessment indicates a high likelihood of poor water quality, irrespective of possible mitigation measures (such as, catchment management or rapid flooding), it is unlikely that further investment in prediction will be warranted (Hannam & Green 2014). Conversely, if the initial assessment indicates that higher water quality may be able to be achieved, opening up the possibility for higher level uses, then more detailed prediction, using the full range of modelling tools available, could be warranted as part of closure planning.

The selection and application of predictive models should be underpinned by a well-defined conceptual model that accounts for all key assumptions and processes that need to be addressed by the model. The conceptual model should ideally be developed in a workshop involving operations staff and technical specialists from relevant disciplines. Critically, the model should evolve in response to changes in the mine plan and findings from subsequent modelling, monitoring and validation exercises.

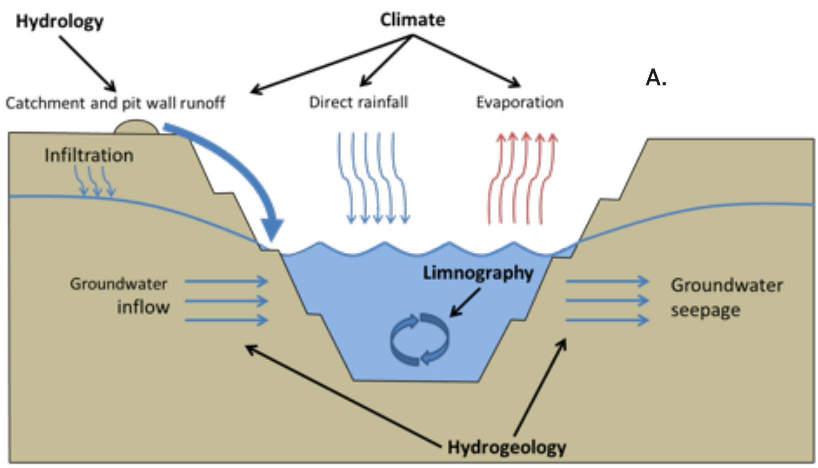
Key model conclusions should be reassessed as new data becomes available. For example, this should be done as mine development progresses from exploration and resource definition through to operations. Equally, initially identified data limitations should iteratively drive data collection by specifically designed monitoring programs. Uncertainty in data input does not necessarily limit the robust use of models, but it must always be recognised as a limitation to the certainty (precision) of model output.

Although robust prediction might be the primary goal of predictive modelling, models can (and should) also be used to explore the extent of contributions made to water quality by various processes. This process contributes to a better understanding of risk uncertainty in informing management decisions. Such sensitivity analysis can also highlight model limitations and weaknesses and identify opportunities to improve accuracy.

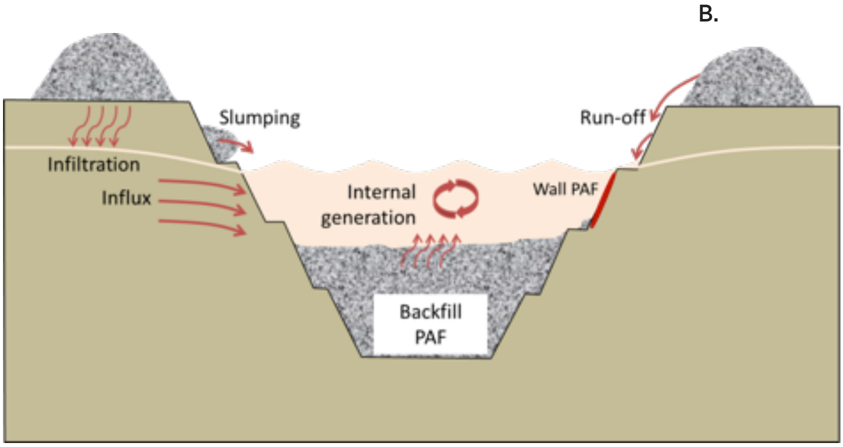
Predicting pit lake water quality requires a coupled understanding of the lake's water balance[47] and of sources of AMD and other solutes that will affect water quality (Figure 28). However, there is no universally accepted approach to doing this. Although there are some similarities between natural lakes and pit lakes, there are fundamental differences that must be addressed. In particular, pit lakes are more akin to constructed reservoirs in that they are typically deeper relative to their surface area, with implications for the effect of water column stratification on water chemistry (Geller et al. 2013).

The risk of periods of meromixis (ongoing stratification), punctuated by periodic mixing of the water column, in particular, can have major implications for the ongoing maintenance of water quality. Pit lakes also differ from natural lakes in that their catchments often contain reactive (sulfidic) materials that have been exposed or disturbed during mining. Lakes forming in pits may be influenced by the hydraulic properties and weathering products of waste rock and tailings that form aquifers discharging into the system. Those properties may change over time, and the hydrology and water quality of the final void may evolve for many years before approaching a steady state.

Figure 28: Conceptual models for processes for determining long-term water balance (A) and water quality (B) that pit lake water-quality prediction should explicitly consider



47 See the *Water stewardship* leading practice handbook for guidance on factors affecting water balance (DIIS 2016g).



Note: 'Limnography' refers to the physical processes occurring in a freshwater body of water.

Whether a pit lake will be a terminal sink (that is, it will never discharge via surface or groundwater), be part of a groundwater flow-through regime, or ultimately discharge by overtopping will be defined by the water balance. A pit lake's water balance can be broadly defined using tools ranging from simple spreadsheets through to more complex probabilistic modelling software. Most fundamentally, the water balance will need to indicate whether the pit lake will ultimately be a flow-through regime or a terminal sink for water and solutes (McCullough et al. 2013). The risk presented by surface decant and groundwater of pit waters can then be assessed.

The processes that control the volume of water in a pit are direct rainfall; evaporation; surface water and groundwater inflows; and surface water and groundwater outflows. Unlike many natural lakes and reservoirs, mine pits often lack a surface water outflow. In some cases, a filling pit may lose water only through evaporation. Variability in rainfall, especially in areas affected by tropical rain depressions, complicates water management and the prediction of water levels. At such locations, large changes in pit water storage can be expected to occur, and the available freeboard to prevent uncontrolled overflow and exposure of reactive strata becomes critical.[48]

Solute inputs into the developing lake can be defined using the outputs from static or, preferably, kinetic test work on wall rock, waste or tailings placed in the base of the pit, waste in dumps located in the catchment of the pit and from water-quality data on surface waters and groundwaters that will input to the pit. This data can be used in mixing models or more complex proprietary geochemical models for lakes. Ideally, all inputs used for predictive modelling should be obtained using samples and test durations directly relevant to the final pit lake context and timescale.

A number of modelling tools are available, as described by Vandenberg et al. (2011). Model selection should focus on:

- what approach and package best meets the requirements of the modelling question
- what data is available
- the level of technical expertise and experience available either in-house or from external consultants to run the model and to interpret its outputs.

48 See the Mount Morgan case study in Section 7.

The expectations of stakeholders and the previous experience and confidence of regulators with certain modelling packages may also be important factors in selecting the model. For instance, open source models or models that are broadly accepted and used should be preferred, as they are more able to be independently tested and peer-reviewed.

Generally, the quality and extent of input data and the assumptions made in developing the model will be the biggest limitations on the reliability of prediction, given the sophistication of modern lake modelling software. The production of a robust pit lake water balance is often limited by:

- available hydrogeological data being restricted to that acquired for pit dewatering operations
- inadequate time-series rainfall and other relevant climate data
- limited baseline surface water and groundwater quality data
- limited broader regional understanding of preferential flow features, such as fractures and palaeochannels.

The potential effect of climate variability on rainfall and evaporation should also be considered, given the long time frames involved. Similarly, geochemical assessments should provide information on the reactivity of materials (for example, wall rock, backfilled rock or tailings) that are expected to be exposed and/or submerged in the pit lake. This may involve column or batch testing samples of the materials under saturated oxic and/or saturated anoxic conditions (to simulate efforts of stratification) spanning several months or even years.

It is especially difficult to validate the predicted evolution of water quality for pits that may take decades or longer to fill following the cessation of dewatering (Castendyk & Eary 2009). For this reason, predictions of pit lake water quality should be very carefully assessed, and the use of probabilistic models incorporating consideration of stochastic variability should be favoured over deterministic approaches for the final analysis. In this context, studies have shown that the predicted trajectory of pit water quality through time for those lakes for which ecological or other beneficial uses are envisaged has often been overoptimistic (Kuipers et al. 2006).

The following factors have been found to be the most important for establishing and maintaining values in a pit lake at risk from influence of AMD to achieve the broadest range of closure objectives (McCullough 2011; Geller et al. 2013):

1. **rapid filling to minimise the exposure time of reactive sulfides in wall rock or in sulfidic waste that has been placed in the pit**
2. **cutting back of the pit wall and/or covering exposed PAF materials with NAF rock if sulfidic wall rock will still be exposed at maximum fill level**
3. **minimising the amount of AMD in surface water and groundwater catchments reporting to the lake**
4. **maintaining stratification of the water column**
5. **establishing a surface flow-through regime to flush accumulating solutes from the lake**
6. **reducing catchment erosion and potential for slumping of the pit walls**
7. **facilitating the development of biological productivity in the euphotic (light penetration) zone.**

Case study 9 illustrates the benefits to be gained by implementing a surface flow-through regime for a pit lake in the Collie coalmining district of Western Australia.

Case study 9: Flooded pit lake with flow-through regime, Lake Kepwari, Collie coal fields, Western Australia

Context

The Collie Coal Basin is the centre of the coalmining industry in Western Australia and is located approximately 160 km south-south-east of Perth. Over 100 years of coalmining in Collie has resulted in the formation of at least 13 pit lakes with differing water quality and extent of rehabilitation. All are acidic due to moderate AMD production from oxidation of the regional low-sulfide lithologies.

Mining began in the Lake Kepwari (WO5B) pit, with diversion of the Collie River South Branch (CRSB) around the western void margin (Figure 1). The closure aims for the lake were to provide a recreation resource for water skiing, diving and other aquatic recreational pursuits. When mining ceased in 1997, most reactive overburden dumps and exposed coal seams were covered with waste rock to cover PAF sources. Lake edges were backfilled and graded in some areas to form beaches and an island, and then extensively revegetated with endemic flora by direct seeding. The diversion channel was then maintained to permanently divert the CRSB around the pit void.

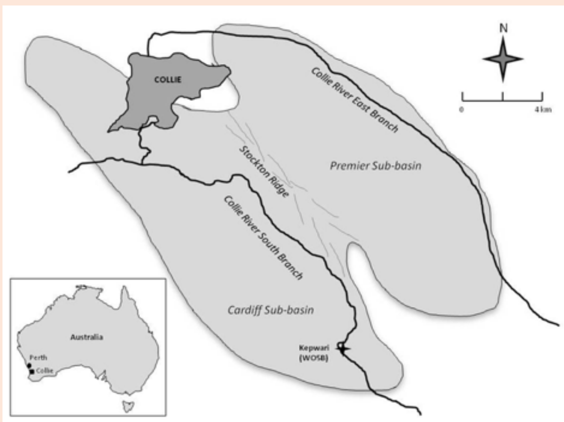


Figure 1: Location and plan schematic of Lake Kepwari in Western Australia

Evolution of pit lake water quality

Lake Kepwari has a volume of around 32 × 106 m^3, with a maximum depth of around 65 m and surface area of 1.0 km^2 (McCullough et al. 2010). To further reduce wall exposure and rates of resulting acid production, the lake was rapid-filled by seasonal diversions from the CRSB over the winters between 2002 and 2008. CRSB water quality is typically brackish and highly tannin stained, with moderate eutrophication potential. Although the input of CRSB water initially raised the pH to above pH 5, the pH fell below 4 once river inflows ceased. This low pH and associated elevated concentrations of some metals and metalloids reduced water-quality values and restricted end-use opportunities.

Heavy rainfall in August 2011 led to the CRSB overtopping the diversion channel along the south branch wall that separates it from Lake Kepwari. Approximately 30 m of wall failed, and 2 GL of CRSB water flowed through Lake Kepwari, decanting through culverts in the north-east and north-west sides of the lake (McCullough et al. 2013).

This unintentional flushing had the effect of substantially improving the water quality and ecosystem values (McCullough et al. 2012), indicating that maintaining the lake as a flow-through system would probably provide a leading practice closure strategy for the lake. Regular seasonal flow-through was expected to yield a seasonal thermally stratified brackish lake of circum-neutral pH (Figure 2).

Following broad stakeholder engagement, including community presentations, and approval by state regulators, a flow-through trial was proposed to assess the benefits of this leading practice management option. The trial was run between 2012 and 2014 with over two years of flow and one year of no flow due to low rainfall. Regular quarterly assessment of biota and chemistry of both Lake Kepwari and the CRSB, and annual reporting of the findings to regulators and community was a key component of the trial.

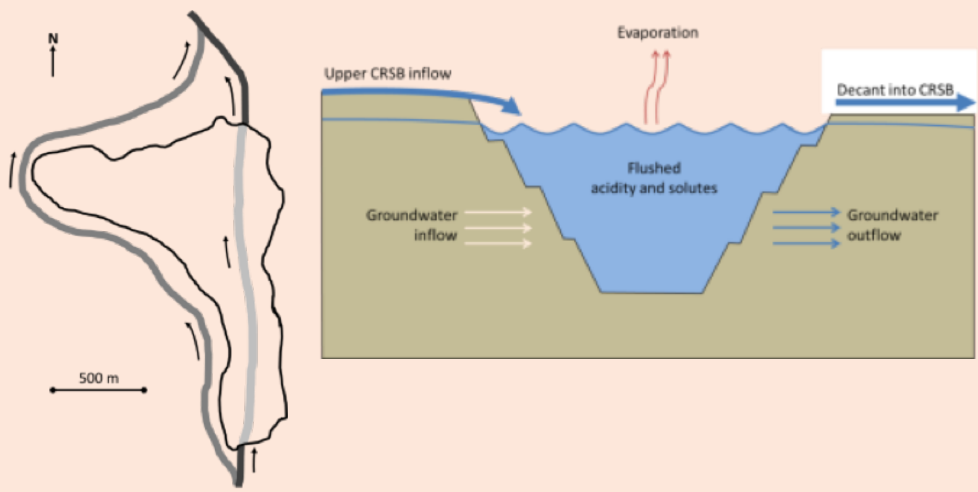


Figure 2: Conceptual model of (A) Lake Kepwari flow-through design showing the historical CRSB channel in black, previous river diversion in dark grey and presumed lake passage in light grey; and (B) AMD groundwater source shown as pink arrows, sinks and water balance during flow-through

Monitoring of water quality

The monitoring results showed that downstream river water quality is not significantly degraded by flow-through, since the greatest decant volumes occur predominantly during high CRSB flow with consequent high lake water dilution. Further, river water quality may actually be improved relative to upstream through the trapping of sediment, including particulate forms of nutrients, by the lake benthos and the removal of soluble forms of phosphorus by co-precipitation with dissolved aluminium species present in the lake (McCullough and Schultze 2015). Monitoring indicates that, following implementation of the flow-through regime, the lake pH is higher than historical levels and, importantly, is on an upward trend (Figure 3).

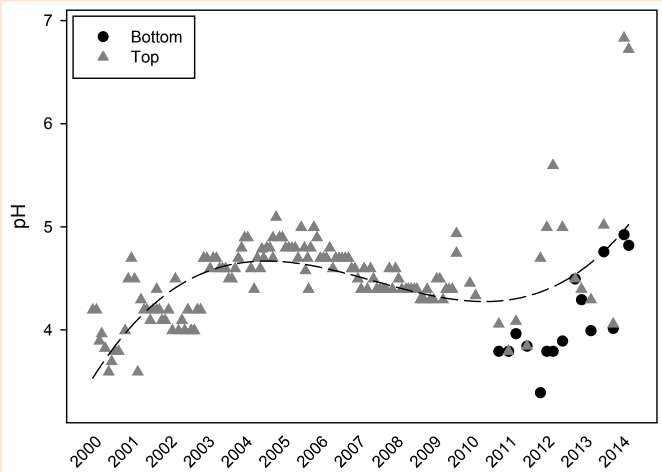


Figure 3: Time-series graph of Lake Kepwari pH historically, during and after flow-through began (after McCullough et al. 2010; McCullough et al. 2012). Dotted regression line indicates surface water pH trend over time

Conclusions

The lake flow-through strategy has worked particularly well in this situation because (a) the river channel was able to be maintained in its historical course and (b) water quality was already degraded by upstream land uses, thus reducing the risk to downstream river values posed by the addition of AMD. The findings from a literature review prior to the beginning of the project, coupled with early and regular engagement, provided confidence to stakeholders that the approach was leading international practice. However, it should be noted that flow-through may not be leading practice for all pit lake closures, particularly those with good river water quality downstream and associated high-value end-uses. Each case needs to be assessed within its site-specific context.

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6.7 Brownfield mine sites

Brownfield mining projects occur on or very close to a previously existing or current mining operation, as distinct from greenfield mining projects, which are developed in previously unmined areas. Brownfield sites present opportunities to apply leading practices to older sites, where past practices may have resulted in poorer management of AMD risks. The resumption of mining can also provide improved socioeconomic value from the site in concert with a higher standard of environmental protection.[49]

The expansion of an already approved and operating mining project at an existing site is not the focus of this section. The focus here is on sites where there has been a step-change in regulatory and/or community expectations between when the former site ceased operations and when a new operation starts. This change of ownership or restarting after a period of dormancy can pose both opportunities and risks for the assessment and management of AMD and for designing for closure.

Opportunities:

- Considerable information may have already been collected on the site geology and geochemistry. This may include data collected during mining operations, in addition to exploration. In contrast, a greenfield site has to rely on exploration data or, if an existing operation in a similar geological region exists, data from that site may allow a broader understanding of the likely AMD risks.
- Time may have allowed for the assessment of the weathering and evolution of the properties of waste materials.
- Waste lithologies are accessible above ground for additional sampling and analysis to gain a greater understanding of AMD characteristics and to model long-term weathering products to inform waste management, landform design, water management, revegetation plans and closure design.
- If available, long-term water monitoring datasets can be re-evaluated to inform the development of AMD control strategies.
- Leading practice AMD management systems may be created by building on the foundational knowledge gained from past activities.
- Current leading practice methods can be implemented to minimise AMD from existing waste on site, in parallel with the effective management of newly produced waste to reduce the overall waste management liabilities of the site.

Risks:

- due diligence assessment[50] undertaken as part of the site acquisition process substantially underestimates the AMD liabilities for the new project by not adequately accounting for:
  - the extent of AMD risk owing to insufficient or inappropriate waste characterisation and/or water-quality monitoring data from previous mining operations
  - historically poor waste management practices, such as inadequate segregation or encapsulation of waste
- limited or no care and maintenance activities to manage water (surface run-off, seepage to groundwater, filling to overflow of open pits or underground workings) on the site during extended dormancy periods
- limited or no water-monitoring data (surface water or groundwater) from which to gather an understanding of long-term trends in AMD
- lack of trust and community opposition to the new mining project due to the past legacy

49 See the Savage River case study in Section 5.

50 See Section 5.3

- delays in project approvals owing to the need to conduct time-consuming and expensive investigations because of a lack of prior robust data on AMD risks.

The key to the successful AMD management of brownfield sites is to understand their differences from greenfield sites by addressing the following issues:

- Use targeted AMD characterisation to understand how waste materials have changed since being mined and placed in WRDs or TSFs.
- Design and implement operational waste management and closure plans to take into account the characteristics of existing waste facilities.
- Understand hydrological interactions between wastes and surface waters and shallow and deep aquifers.
- Characterise the receiving environment by determining both current and potential (that is, what values could be there in the absence of impacts from existing operations) environmental values.

6.8 Abandoned/legacy mine sites

Legacy sites[51] pose many of the same challenges as brownfield sites except that they are not under the ownership of a mining company with responsibility for care or maintenance of the legacy and they will most likely never be reopened (Pepper et al. 2014). This means that in whatever state the site has been left there will need to be systematic site assessments and planning to manage the AMD legacy. Sulfide oxidation can often be at an advanced stage, resulting in extensive downstream impacts that need to be quantified and mitigated.

By default, many of these sites are now the responsibility of governments and landholders. Legacy site programs are required to identify and prioritise sites on the basis of environmental and socioeconomic risks and opportunities in order to bring the greatest value from intervention activities. The Australian strategic framework for managing abandoned mines (MCMPR–MCA 2010) encourages a multidisciplinary approach to ensure that the values of these sites are recognised, data is collected and managed, risk assessment is undertaken and the sites are managed in partnership with other organisations wherever practicable.

Alternative land uses should also be evaluated to explore economic opportunities. For example, AMD&Art[52] in Vintondale, USA, is a park that uses the characteristics of AMD (from coalmining) through a combination of art and science to create renewal in an otherwise degraded landscape. Large open spaces for recreation, relaxation and public events have been created at several rehabilitated uranium mining AMD legacy sites in the former East Germany.

The keys to success in managing AMD at these sites are similar to those for brownfield sites. However, the following issues also need to be addressed:

- Set closure objectives.
- Identify the key current and future risks and liabilities posed by the site and develop management/ remediation plans based on them.
- Assemble a suitably qualified team with multidisciplinary and project management skills.
- Develop a complete understanding of the site by collating historical information in a spatial database and knowledge management system, and undertake new investigations where required.

51 See also Section 8.6.

52 http://www.amdandart.info/.

- Understand and address stakeholder expectations, including for indigenous and other cultural and industrial heritage values.
- Consider innovative ways to address specific issues; for example, public–private partnerships to fund and manage sites such as the one at the Britannia mine in British Columbia.

The current national partnership agreement between the Australian and Northern Territory governments for the rehabilitation of the legacy Rum Jungle copper and uranium mine site is a leading practice example of the application of the above principles (Case study 10).

CASE STUDY 10: Rehabilitation planning for a legacy site at Run Jungle, Northern Territory

Context

The former Rum Jungle mine site is located near Batchelor in the Northern Territory, approximately 105 km south of Darwin by road. Between 1954 and 1971, Rum Jungle produced 3,530 tonnes of uranium oxide and 20,000 tonnes of copper concentrate. Full details of the history and a comprehensive listing of historic and current technical reports can be found at www.nt.gov.au/d/rumjungle/.

As shown in Figure 1, the current 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 overlain with contaminated soil and rock—Dysons.

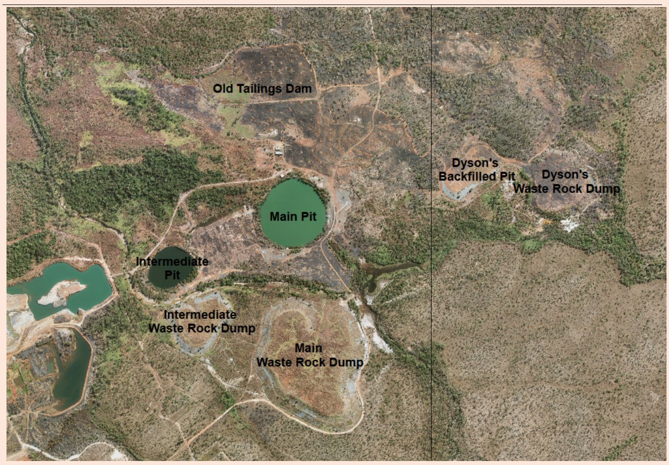

Figure 1: Site map showing key locations

The methods of placement used for waste rock and process tailings during this time resulted in ongoing oxidation of sulfide minerals in the waste and production of substantial volumes of AMD containing acid and high concentrations of copper and other heavy metals. These oxidation products were flushed into the adjacent east branch of the Finniss River during the wet season, resulting in depletion of aquatic life for many kilometres downstream over a period of several decades.

The site was initially rehabilitated between 1982 and 1986, to address these issues, and this was followed by a 12-year performance monitoring program (Northern Territory Government 2013). After the rehabilitation works were completed, a comprehensive technical assessment determined that the engineering and environmental criteria set for the rehabilitation project had been successfully met (Pidsley 2002). Indeed, this standard of performance has continued to be met to the current day.

Although the mid-1980s works represented international leading practice at the time, they did not result in a final condition for the site that would meet contemporary water quality or contaminated site clean-up standards. Advances in the understanding of cover design have shown that the multilayer covers that were placed over the sulfidic waste rock were too thin to accommodate the wetting–drying cycles imposed by the wet–dry monsoonal climate, with cracking of the compacted clay layer occurring (Taylor et al. 2003). In addition to the technical issues, the original works were completed without any input from the traditional Aboriginal owners of the site.

Since 2009, the Northern Territory (through the Department of Mines and Energy) and Australian governments have been working under a national partnership agreement to improve site maintenance and environmental monitoring, and to develop an improved rehabilitation strategy that is consistent with the views and interests of the traditional owners (Laurencont & Rider 2014).

Geochemical characterisation of waste rock

A key consideration for developing rehabilitation scenarios was to understand the geochemical properties of the waste material in the existing WRDs and backfilled pit. The geochemical processes occurring and the extent to which the sulfidic material has already oxidised needed to be known to assess the likely effectiveness of different potential future management strategies. This information will be used to prioritise waste material in the context of the most effective control strategy—for example, relocating as much as possible of the higher residual sulfide containing material below the groundwater table in the existing water-filled pits to prevent further oxidation and acid generation. In this context, the total capacity of the Main and Intermediate open pits represents about 50% of the total volume of waste rock and contaminated soils contained in the WRDs and Dysons (backfilled) open pit.

Two campaigns of sampling have been completed over the past four years to characterise the current status of the materials contained in the WRDs and the backfilled pit. An initial investigation into the current geochemical characteristics of the waste contained in the three WRDs and in the Dysons backfilled pit was completed in 2012 (SRK Consulting 2012). Four trenches were dug in each WRD to provide a reasonable spatial coverage of the surface of each dump, with the locations of these trenches informed by historical information about the construction of the WRDs. This investigation concluded that the waste materials were very heterogeneous, with a generally poor relationship between field pH and EC. Oxidation was found to be very incomplete in the main and intermediate WRDs, with a greater extent of oxidation for Dysons WRD.

It was initially anticipated that the extent of oxidation and lithological provenance of the waste might be able to be inferred from visual observations. This approach would have reduced the laboratory characterisation work, with fewer samples needing to be subjected to more detailed analysis to provide a comprehensive picture of the geochemical characteristics of the wastes. Unfortunately, geochemical characteristics such as the extent of oxidation, acid potential and leachable metal content did not correlate with the visual characteristics.

However, it was found that there was generally good correlation between NAG pH values and the findings from ABA (that is, NAPP positive materials produced an acidic NAG pH).

SRK Consulting (2012) identified that, with the exception of Dysons WRD, all structures contain substantial residual sulfides (incipient acidity) with the potential to continue to oxidise and to generate AMD conditions into the future. By far the highest residual sulfide content was present in the Intermediate WRD.

The sample trenching that was done for this initial work provided a valuable insight into the nature of waste present in the WRDs, but it was limited to a maximum 10 m depth in the WRDs. This meant that only the top 50–60% of the waste profile in the WRDs was sampled. In addition, the amounts of existing acidity stored in the waste were not quantified by this characterisation program. Existing acidity consists of two components:

- directly titratable acidity comprising sulfuric acid and mineral acidity
- acidity present in the form of poorly soluble secondary minerals such as jarosite and alunite.

Information on the amount of existing acidity is required to estimate the potential for this material to leach solutes, so that the most appropriate strategies for its future management can be developed. The importance of determining the amount of existing acidity in partially oxidised waste rock at brownfield and legacy sites is emphasised in Chapter 4.

Additional investigations were undertaken in 2014 to both characterise the complete profiles of waste from the WRD surface down to the original ground surface and better define the extent of oxidation and the presence of secondary acid sulfate minerals such as jarosite. Test pits were excavated using a 35 tonne excavator and extended from the top surface of the WRD to the depth of natural ground (up to 21 m). This excavation process was a small-scale re-mining activity, with access to depth requiring substantial lateral cutbacks with intermediate benching. The footprint of the deepest excavation measured 100 m x 50 m (Figure 2).

Figure 2: Excavated sampling pit (15 m bench level) in the Main WRD. Scale is provided by a 2 m person standing on the right-hand side

Samples were collected at 1 m vertical intervals or where a change was observed in lithology. The excavator was equipped with a 7.5 cm mesh sieve bucket in which the bulk sample was collected. The bulk sample was brought to the surface and the bucket shaken to separate out the <7.5 cm fraction, with the >7.5 cm material being put to one side. Twenty-four kilogram subsamples from the <7.5 cm pile were then taken for characterisation. By this means, a good indication was obtained of the particle size distribution of rock at each 1 m vertical interval. This information is needed to inform materials handling, for the design of proposed new waste storage materials and to scale geochemical characterisation information to the bulk mass of material.

Findings from geochemical static tests

The incipient acidity data (expressed as kg/t H_2SO_4 equivalents) obtained from this program is summarised in Figure 3. Incipient acidity (acidification potential, AP) refers to the acidity that could be produced if all of the sulfide remaining in the partially oxidised material were to oxidise. Intermediate WRD (INT in Figure 3) contains by far the highest incipient acidity.

Figure 4 shows the distribution of existing acidity between jarosite acidity and directly titratable acidity in the Intermediate WRD. It is clear from this data that failure to account for the jarosite component of acidity would result in a gross underestimate of the existing acidity load of this material, and hence the amount of neutralant that would need to be added to account for the total existing acidity.

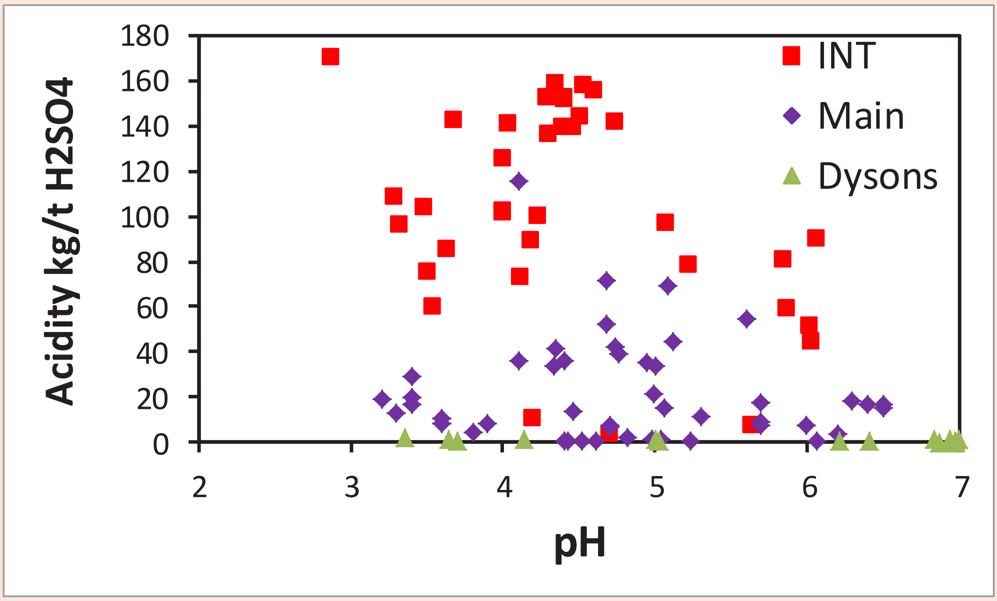

Figure 3: Comparison of incipient acidity in the three WRDs

INT = Intermediate

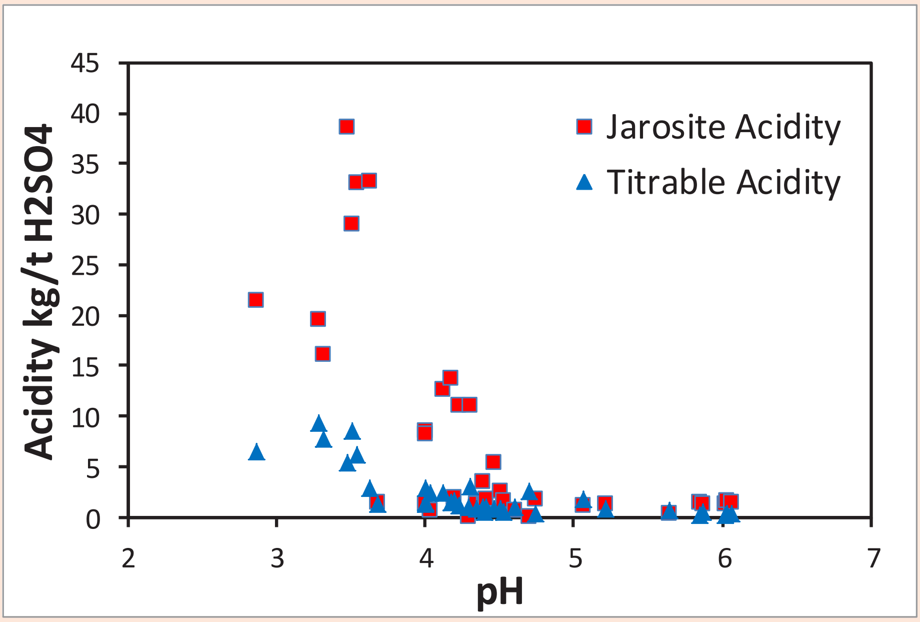

Figure 4: Comparison between titratable acidity and jarosite acidity in the Intermediate WRD

Proposed rehabilitation strategy

The preferred strategy for rehabilitation of the site involves backfilling the main open pit with the highest sulfide content (incipient acidity) waste rock (all of the Intermediate and part of the Main WRDs), with consolidation of the remaining waste rock and other contaminated material on site into a new purpose-designed above grade waste storage facility. This preferred option was identified using the multiple accounts analysis tool (Northern Territory Government 2013). The multiple accounts analysis process was structured around four key issues:

- environmental
- technical feasibility
- cultural
- financial.

Each individual component of these issues was scored on its relative projected performance against the rehabilitation objectives, with a consensus weighting applied to derive the final assigned value.

Both of the waste containment strategies require the geochemical nature of the materials to be understood and addressed. In the case of the material that will be placed in the pit, residual sulfide content will not be an issue since placement under the groundwater table will prevent any future oxidation of residual sulfides. However, for this material, the existing acidity may need to be neutralised to reduce the potential for groundwater contamination by soluble acidity and metals.

Laboratory titration test work has indicated that pH 7 is the optimum pH target for metal removal in the event that partially oxidised material needs to be neutralised for placement in the new waste containment facilities.

The need for partial or complete neutralisation of material to be placed in the pit will be assessed by sensitivity analysis of the outputs from hydrogeochemical transport modelling of the in-pit containment.

The design of the new above grade waste storage facility will need to minimise both the future potential for oxidation of residual sulfides and the amount of infiltration of water (transport medium for oxidation products) through the surface of the facility. It is currently planned to add sufficient neutralant to the material as it is placed to neutralise the existing acidity.

Conclusions

Rum Jungle is an example of a site that was initially abandoned with essentially no rehabilitation of reactive sulfidic material. The consequence was major contamination of a river that received the AMD. A first pass at rehabilitation was undertaken in the mid-1980s with a then substantial investment of approximately $20 million (1980 $). Many aspects of that work were world-leading practice at the time, and the performance targets (percentage reduction in infiltration into the surface of the WRDs and percentage reduction in downstream contaminant loads) were achieved and maintained. However, the end result does not meet contemporary water-quality criteria for environmental protection. In addition, the cover design was not well suited to the wet–dry monsoonal climate, and the performance of the cover system has been steadily deteriorating. These factors, coupled with the fact that the site's traditional Aboriginal owners were not consulted about the original rehabilitation works, has led to the need to fundamentally revisit the rehabilitation objectives and strategy for the site. It is now estimated that it will cost in excess of $200 million to implement the preferred rehabilitation strategy.

The primary learning from this case study is that failure to prevent the occurrence of AMD from the start can lead to very costly remedial action needing to be undertaken in the future. The secondary learning is that any such remedial action needs to be very well thought out and scoped (including social and cultural aspects) so as to avoid having to come back again at even greater future cost.

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7.0 TREATMENT OF AMD

Key messages

- AMD treatment can be a costly part of mining operations and potentially an even more costly post-closure liability if the propensity for sulfidic materials to produce AMD is not recognised and managed appropriately from the start of mining operations.
- No treatment approach can provide a total 'walk-away' solution, as all systems require long-term monitoring and maintenance.
- Active treatment using calcium-based neutralising reagents is likely to remain the choice for some time for primary (first-stage) treatment of medium- to high-strength (low pH) AMD, and for treating those systems where the flow rate of acidic water varies over a large range.
- Passive treatment systems are restricted to low acidity load situations where the flow rate of the water to be treated is relatively constant through time.
- Wetland systems offer an attractive option for final treatment of pre-neutralised water and for circumstances in which the pH is above pH 4.5.
- In situ treatment of ponds or pits containing AMD requires specialist knowledge of the physical and chemical behaviour of those systems to succeed.

7.1 Why and when do we need to treat?

It makes good business sense, in addition to being leading practice, to avoid and minimise the production of AMD (using the methods described in Section 6), and to treat AMD only as a last resort if other approaches have failed.

AMD treatment needs to be considered not only for protection of environmental values of waterways but also for cases where:

- the re-use of mine or process waters is required in areas where the available water supply is limited
- process or other critical equipment requires protection from corrosion or from fouling by scaling
- water in pits or underground workings must be removed to regain access to an ore resource (this is an especially important factor in the context of resource sterilisation)
- commercial metal recovery is a possibility
- groundwater is contaminated by a plume of AMD and the plume needs to be remediated.

AMD produced during operations can often be managed at a relatively lower cost than after closure—for example, by storing the AMD in process or pond water circuits, or by co-disposal with alkaline tailings (a hidden part of the cost of production). At closure, these operational management options are no longer available.

It is possible that the requirement for post-closure management of AMD may not have been recognised during operations because the extent of the future issue was hidden by long lag times, compounded by lack of appropriate monitoring data. This situation is most likely to occur if there has been insufficient static and kinetic testing of the waste.

The contamination of groundwater by mining operations has historically received significantly less attention in Australia than the contamination of surface water. This is probably largely due to many mines being remote from competing users of groundwater. However, in some locations in Australia, and especially in the European Union and in the United States, groundwater contamination by mining has required very costly remedial action (see, for example, Brown et al. 2009).

Preventing the further spread of solutes in groundwater by containment and recovery ('pump and treat') can be a difficult, costly and very long-term proposition. A general rule-of-thumb is that one year of groundwater contamination requires many years of pump and treat to remediate the plume. Consequently, the emphasis in the design and location of WRDs, ore stockpiles and TSFs at new mine sites should be on the prevention or minimisation of future groundwater impacts.

No single treatment approach can provide a total 'walk-away' solution, as all systems require a degree of long-term monitoring and maintenance. The selection of the appropriate AMD treatment method (or combination of methods) invariably depends on site-specific conditions, including water composition and treatment targets.

The whole treatment process (including sludge settling and disposal) needs to be systematically assessed before the most cost-effective option can be identified, and is likely to require the expertise of water treatment specialists. AMD treatment technologies described in this section are generally applicable to both surface water and extracted groundwater. Technologies that are specific to in the situ treatment of groundwater are identified in Section 7.3.3.

7.2 General considerations for the selection of treatment systems

The key factors that need to be addressed when selecting the most appropriate water treatment method are as follows:

- *Water chemistry.* Metals/metalloids and pH (that is, acidity) are the most common targets for the treatment of AMD, but the removal of major ions, such as magnesium and sulfate, may also be required.
- *Water volume (or flow rate).* The cost of water treatment is a function of both the flow rate to be treated and the composition of the water. In many cases, the flow rate is the primary driver for sizing a treatment system, whether active or passive. Efforts should be made to constrain the volume/flow rate requiring treatment, both during operations and after closure.
- *Treatment targets.* Targets for treated water quality will be site-specific and depend on a number of factors, including issues relating to the protection of plant and equipment from corrosion and the protection of the environmental values of receiving waters.

The derivation of treatment targets requires consideration of the risk assessment framework detailed in ANZECC–ARMCANZ (2000a).[53] Decision trees and computer software to assist with the selection of AMD

53 As described in Section 5.3. The Mount Morgan case study in this section demonstrates an application of this approach. Refer also to the *Risk management* leading practice handbook (DIIS 2016f) for a broader coverage of this topic.

treatment methods and cost estimates for various technologies are described in Taylor et al. (2005), in the guidance provided by the Interstate Technology and Regulatory Council (ITRC)[54] in the US and by the AMDTreat software package (see Glossary) available from the US Office of Surface Mining Reclamation and Enforcement (OSMRE).[55] AMDTreat can assist a user in estimating costs to treat AMD using a wide variety of active and passive treatment types.

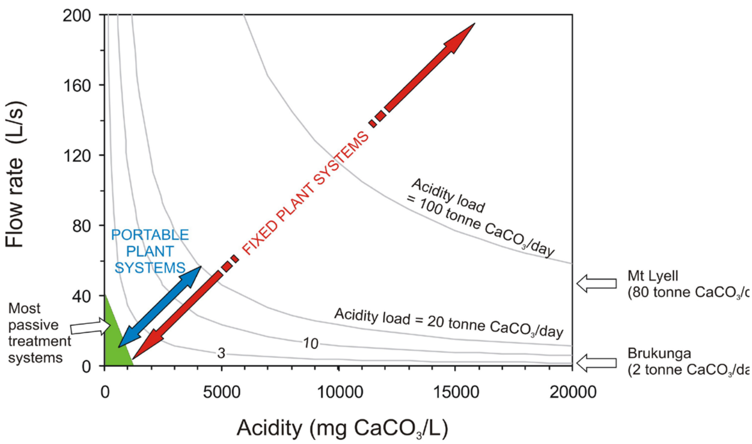
7.3 Treatment technologies—active or passive?

7.3.1 Overview

AMD treatment systems can be categorised as either active or passive. The common attributes of a passive treatment system are no or minimal requirements for active (electric or diesel) pumping, and no requirement for the remotely powered addition of chemical reagents.

Whether an active or passive method is suited to a given AMD application can be determined by assessing the acidity load of the influent AMD. Passive treatment approaches can be economically attractive in the right circumstances, but have some significant limitations. They are best suited to the treatment of waters with low acidity (<800 mg $CaCO_3$/L) and low acidity loads (100–150 kg $CaCO_3$ per day), and with steady flow rates. Passive treatment solutions are not suited to treatment tasks requiring in excess of approximately 150 kg $CaCO_3$ per day equivalent. There have been many examples in which this rule has not been followed in the design and implementation of passive treatment systems, and the inevitable consequence has been overload and failure to meet treatment targets. Figure 29 can be used to infer the applicability of different AMD treatment systems based on the acidity load of influent AMD.

Figure 29: The selection of appropriate treatment approaches can be initially based on daily acidity loads



Source: Taylor et al. (2005).

A passive wetland treatment system that failed is shown in Figure 30. The wetland was initially designed to treat pH 5 water. It operated successfully for three years but was overwhelmed when large unanticipated volumes of strongly acidic seepage began to discharge from an adjacent covered WRD. Despite the

54 http://www.itrcweb.org/miningwaste-guidance.

55 http://amd.osmre.gov/.

installation of engineered open limestone drains upstream of the wetland, there was insufficient neutralising capacity to treat the increased acidity load.

Figure 30: Wetland constructed to polish seepage from a covered WRD, during initial years of neutral drainage (left), and after it was overwhelmed and rendered ineffective by breakthrough of highly acidic and metal-rich seepage (right).

In the event of slightly acidic to near-neutral mine drainage, such as pH 5–8, very large flows (within limits defined by required residence time and available area) can be directly treated by wetland systems at lower cost and with potentially better output water quality than can be achieved by active water treatment.

7.3.2 Active treatment systems

Centralised treatment

The conventional approach to centralised active AMD treatment involves the use of a fixed water treatment plant (WTP) to which acidic water is pumped or directed from one or more mine water bodies. A series of processes is used to neutralise the acidic water and then separate the solids (metalliferous precipitates) from the water. Many process control aids are available at WTPs (feed flow rate, residence time, mixing speed, mixing time, aeration rate, reagent addition rate, flocculent addition rate, sludge recycle rate and so on). They enable fine-tuning of the WTP operation to achieve a constant stream of effluent treated to a prescribed standard.

The advantage of centralised WTPs is that they can be engineered to accommodate a wide range of acidity loads (Figure 29). However, the selection of an appropriate treatment technology or combination of technologies that will provide robust and economically viable service very much depends on the composition of the source water and the required treatment targets.

The four basic types of active treatment technologies are:

- precipitation of metal hydroxides by addition of neutralising agents to raise pH, or precipitation of metal sulfides
- ion exchange—a resin bed is used to take out metals in positively or negatively charged forms
- membrane separation (reverse osmosis, electrodialysis)—removes both major ion salts and metals/metalloids to low levels
- bioreactor systems for the removal of metals/metalloids and sulfate.

Note that membrane separation is a secondary treatment step that follows first-stage neutralisation of acidity by pH adjustment. Rigorous pre-treatment is needed to remove solutes (especially, iron, manganese, and calcium sulfate and carbonate) that can rapidly and irreversibly foul the costly membranes.

By far the most common and generally lowest cost form of active primary treatment is chemical neutralisation using centralised WTPs. This is also the case for portable equipment for in situ treatment, which can become a viable option when the cost of collecting and pumping AMD to a centralised WTP exceeds the cost of building and operating a smaller, portable plant (see Taylor et al. 2005 and below). Most metals/metalloids of potential concern can be removed by raising the pH to the required level. However, mercury (Hg), molybdenum (Mo), chromium-VI (chromate), arsenic-III (arsenite) and selenium cannot be managed by pH control alone, and require a second (or more) stage of treatment. Designing a plant to treat AMD requires the calculation of the neutralant demand of the water and screening test work to determine which of a range of potentially available neutralants will be the most cost-effective to achieve the required treatment target.

Two components of acidity need to be considered: acid (H^+) and mineral (latent) acidity.[56] Total acidity values can be determined from soluble metal concentrations and pH values using tools such as ABATES shareware (Waters et al. 2014; see Glossary).

Choosing the most appropriate neutralising agent for a given application requires consideration of:

- the pH needed to meet waterquality targets
- the cost (cost of supply plus cost of operational use)
- the rate and extent of pH increase
- occupational health and safety
- the extent of preparation (such as grinding) and the delivery system needed
- required dosage rates (that is, mass of neutralant per m^3 of water)
- reaction time, including the need for oxidation
- the ease of settling, volume and chemical properties of the sludge produced (note that the cost of sludge disposal may be comparable to the initial treatment cost).

The most commonly used neutralising agents for large-scale treatment of AMD are lime (quicklime, hydrated lime), caustic soda (sodium hydroxide), magnesium hydroxide, magnesium oxide and limestone. This is due to the ready commercial availability of these reagents, their non-proprietary nature, the existence of well-proven mixing and dosing technologies for their use, their cost-effectiveness and their manageable occupational health and safety properties for large-scale application.

The most important factor in neutralant selection is the target pH required to meet water-quality discharge objectives. While limestone is the lowest cost reagent, the maximum pH that it can achieve is around 7. This is often not high enough to remove metals such as manganese, nickel, zinc, cobalt and cadmium to acceptable levels, so hydrated lime or one of the other neutralants listed above may also be needed.

Engineered sulfate-reducing bacterial systems have been developed by BioteQ[57] (BioSulfide®) and Paques[58] (THIOPAQ®). Sulfate-reducing bacteria contained in a high-rate bioreactor reduce sulfate to

56 Defined in Section 2.2.

57 http://www.bioteq.ca.

58 http://www.paques.nl.

sulfide and sulfur. This process can produce water containing <300 mg/L sulfate and also removes metals/metalloids that form insoluble sulfides (copper, cadmium, nickel and lead, as well as arsenic, selenium and molybdenum). The technology has been in operation at full scale since the mid-1990s, and several plants have been installed. It is best suited to situations in which high levels of control can be provided and in which commercial metal/metalloid recovery is possible.

A more recent variant on the use of biogenically produced hydrogen sulfide for the treatment of metal-containing water uses elemental sulfur as the starting point in a stand-alone hydrogen sulfide generator (Bratty et al. 2006). The amount of acetic acid or sucrose needed to reduce sulfur to sulfide is only one-quarter of that needed to reduce sulfate to sulfide, considerably reducing operating costs compared with using sulfate as a source of the sulfide. At least five commercial-scale plants using this technology have been commissioned in the United States and Canada.

While the removal of metals by precipitation as sulfides may have been the initial focus of active sulfate reduction treatment processes, the removal of sulfate in its own right (via a number of different types of treatment process) is now becoming a focus in many overseas jurisdictions. Reasons for this increased requirement for lowering sulfate concentrations in discharged mine waters include:

- its contribution to salinity in water discharged to catchments
- its indirect toxicant effect as a result of microbial reduction of sulfate to hydrogen sulfide in the pore water of downstream sediments containing sufficient organic carbon[59]
- the triggering of eutrophication caused by the release of phosphorus from sediments in response to the conversion of ferric hydroxide phases (strong binders of phosphate) to much more stable iron sulfide phases.

Case studies 11 and 12 examine active treatment. Case study 11 documents the operating history of a large active lime dosing WTP installed at the Mount Morgan legacy site in Queensland.[60] Case study 12 describes a world leading practice WTP in South Africa that is producing large volumes of potable water for municipal consumption from treatment of the AMD being generated by four underground and three open-cut coalmines.

59 While sulfate per se is of low toxicity (ANZECC–ARMCANZ 2000), hydrogen sulfide is more toxic than hydrogen cyanide.

60 This is an update of a case study in the last edition of this handbook documenting the lead-up to the construction of the plant.

Case study 11: Active treatment of pit water at a major legacy site before discharge of treated water into a river, Mount Morgan legacy site, Queensland

Context

The Mount Morgan legacy mine site is located approximately 40 km south-west of Rockhampton in Central Queensland. It comprises 270 ha of un-rehabilitated land, including a large flooded open-cut pit (OCP) which contained approximately 9,000 ML of highly acidic and metal-rich water in 2003. In the absence of a spill from the OCP, the primary AMD inputs to the adjacent Dee River from the site are made by rainfall run-off and lateral seepage. In 2003, it was determined that an OCP spill would substantially increase the annual load of acidity and metals to the Dee River. It was estimated that there was a 50% probability of an uncontrolled spill from the OCP in future years in the absence of active management of the pit water level. Active water treatment was the only practicable way of reducing the future probability of a spill.

The development work underpinning the implementation of lime-dosing water treatment of OCP water was presented as a case study in the 2007 edition of this handbook (DITR 2007). Full technical details are provided in Jones et al (2003). Quicklime (CaO) was the chosen neutralant because of its high ANC, local availability and cost-effectiveness. Quicklime slaked to hydrated lime ($CaOH_2$) is added to the pit water, raising the pH to a treatment target of 7.5. Treating acidic and metal-rich OCP water to a standard suitable for release into the Dee River had the objective of lowering the pit water level from a 50% spill probability (pre 2006) to <1% spill probability for any given year based on historical annual rainfall records. The principal metal contaminants targeted for removal were aluminium, iron, copper, cadmium and zinc.

Description of treatment plant

The treatment plant was designed to treat pit water that contains contaminants at the levels shown in Table 1. A single train operating mode plant was built initially to treat water at a rate of 37.5 L/sec with capacity built in to expand to a twin train operating mode in the future. The single train plant was commissioned in 2006 at a cost of $3.4 million. Upgrades completed in 2013 resulted in a treatment plant capable of treating OCP water at a maximum rate of 75 L/sec. The total capital cost for the water treatment plant, including upgrades, was $5.71 million.

The water treatment plant (Figure 1) is now a continuous, automated operation with high-density sludge capacity. Treatment with slaked quicklime results in the precipitation and removal of >99% of target metals (Table 1). The very low residual concentrations of metal levels in the treated water are much lower than in the AMD that discharges to the river in site run-off during rainfall events, the AMD that bypasses the seepage interception system or the AMD that could potentially be subject to uncontrolled release during pit overtopping due to extreme rainfall events.

Figure 1: Mount Morgan mine lime-dosing water treatment plant—neutralisation tanks

The plant currently operates an average 65 L/sec feed rate (5.6 ML/day) from the pit with a yield of 48% (2.3 ML/day) of treated water for discharge. Average feed rate takes into consideration scheduled maintenance and operational downtime. Sludge underflow has a density of around 3%.

Table 1: Mount Morgan Mine OCP water quality—raw and treated

	DISCHARGE TO RIVER TARGETS[1] (MG/L) UNLESS STATED	RAW PIT WATER		TREATED WATER (65L/S RAW PIT FEED AND ONE CLARIFIER MODULE)	
Key discharge targets		Pre-2006	March 2014	Plant design target	2014
pH	6.5-8.5	2.68	3.82	7	7.06
TDS	11500	18736	10300	14400	7860
Aluminium[2]	1.1	714	1134-1320	0.25	LDL-0.91
Iron[2]	N/A	231	10.4(15.7)	0.04	LDL
Copper[2]	1	35	66.6(79.3)	0.016	0.02-0.07
Cadmium[2]	0.02	0.15	0.25(0.28)	0.016	0.04
Zinc[2]	0.16	33.1-72.6	44.3(51.8)	0.04	0.15
Secondary discharge targets					
Magnesium	1,500	1290	2350	-	1800
Sulfate	9500	12805	16680	9120	9870
EC (µS/cm)	11000	11703	15209	N/A	12128
Calcium	1,000	536	523	-	529

LDL = less than detection limit; N?A = not available

[1]Targets are 'key trigger targets' (arbitrary desirable targets based on down-river toxicology risk assessment and water quality for stock guidelines).

[2]Dissolved concentrations (<0.45µm), with total concentrations in brackets.

The sludge waste (mainly gypsum and metal hydroxides) holding more than 99% of the metal contaminants is pumped back into the OCP, with the treated water supernatant discharged directly into the adjacent Dee River.

Performance of Treatment Plant

At the current lime-dosing rates, concentrations of the key contaminant metal elements of Al, Fe, Cu and Zn are successfully lowered to within a desired target range for release and the pH is successfully neutralised (Table 1). Desired target levels for EC, sulfate, cadmium and magnesium are variably achieved. An increase in the lime-dosing rate would be required to raise the pH further to reduce discharge levels for these metals to the desired target level at all times. However, a very significant cost penalty in lime usage, coupled with increased sludge production, would be incurred to achieve this. Plant automation provides for alert signals for pH and turbidity to be sent by SMS to the plant operator, who can remotely interrogate control functions for plant operation and manually adjust the settings if needed. Should the pH of treated water fail to meet the required target, the plant will automatically shut down.

Reduction in the OCP water level to meet the forward target for spill probability (that is, less than 1% annual) was achieved briefly in 2008. However, a subsequent succession of historically high rainfall events, culminating with rain following Tropical Cyclone Oswald in 2013 (equivalent to a 1:2,000 year rainfall event), was beyond the range considered in the original pit water balance modelling. Prior to 2013, in response to increasing OCP water inventory, water treatment capacity upgrades were completed in 2012 and four mist evaporators with the combined capacity to treat 1.9 ML of pit water daily were installed and commissioned. Unfortunately, in January 2013, this combined capacity was not enough and the massive influx of water from Cyclone Oswald culminated in an uncontrolled discharge of OCP water into the Dee River.

Current operating capacities for OCP water treatment and evaporation are considered sufficient to lower pit water level to a <1% spill probability over the next 2–5 years, depending on the occurrence of extreme (well above average) rainfall events. The focus will be on site water management in response to impacts from extreme rainfall events. This includes options for diverting freshwater away from the OCP and increasing emergency freshwater storage capacity above the OCP.

Conclusions

The water treatment plant at Mount Morgan was installed to reduce the level of AMD in the OCP to ensure sufficient freeboard to prevent uncontrolled discharge to the Dee River occurring in response to major rainfall events. The plant has met waterquality objectives for the discharge of treated water to the river and has operated at a high level of availability. However, there has been an instance of uncontrolled discharge of AMD from the OCP. This occurrence illustrates the challenges involved in sizing a treatment plant to accommodate extreme weather events that occur soon after the installation of the plant and before there has been time for sufficient initial drawdown of a storage. It is not only the size of the event but also the timing of the event that is important. The treatment capacity of the plant was doubled in 2013 to address this issue.

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Case study 12: Active treatment of mine water for site use and for municipal drinking water, eMalahleni, South Africa

Context

Reliable access to water, its management and its disposal is a fundamental requirement for all current and future Anglo American Coal South Africa operations. In addition, there is a need to demonstrate, and be recognised for, responsible and sustainable water management, both in the short and long terms, to enhance the company's social licence to operate.

Water management cannot be restricted to within-mine boundaries but must also consider stakeholders and processes that occur outside, as these can have significant implications for how the business is conducted within geographical, legal and social frameworks. Coal South Africa's water strategy is driven by:

- its commitment to continued investment in treatment and technology innovation
- the use of infrastructure to benefit communities
- ensuring that quality and supply are not compromised
- driving efficiency to minimise the company's footprint
- partnering with stakeholders to find mutually beneficial solutions to shared water challenges.

Anglo American's Coal South Africa underground workings located in the vicinity of the eMalahleni municipality in the Mpumalanga Highveld (100 km from Pretoria) currently contain approximately 100,000 ML of water (rising by over 9,000 ML/y) compared with about 104,000 ML in the Witbank Dam, the area's main source of potable water. Whilst the local community is faced with a critical shortage of water to support residential, commercial and industrial growth, mining is impacted by a surplus of contaminated water in the underground workings.

Water treatment process

A decade of research and development into mine water treatment technology has resulted in the implementation of desalination technology for the production of potable water from the otherwise unusable mine water. In parallel there has been a reduction of safety risks between connected mine workings and the prevention of environmental impacts that would have resulted from uncontrolled discharges of AMD at the surface.

The eMalahleni Water Reclamation Scheme, commissioned in 2007, is a joint venture between Anglo Coal South Africa and BHP Billiton Energy Coal South Africa (BECSA) and was designed to treat 25–30 ML per day of the water from four Anglo operations and the water from a nearby, defunct mine owned by BECSA (see Gunther et al. (2006) and Hutton et al. (2009) for details of the treatment process). Figure 1 shows an aerial photo of the plant and the process flowsheet is shown in Figure 2. The compositions of the feed and treated water are summarised in Table 1.

Benefits of water treatment

Anglo Coal entered into a bulk water supply agreement with the water-stressed eMalahleni Local Municipality and currently supplies approximately 12% of the town's daily potable water requirements through the provision of 16 ML of water per day, decreasing the percentage of people without drinking water from 14% to 2%. Since inception, the eMalahleni water reclamation plant (EWRP) has treated in excess of 50 billion litres of mine-impacted water to potable water standards and has supplied 35 billion litres to the municipality.

The operation of the plant has also made several surrounding mining operations self-sufficient with respect to water requirements, thereby reducing pressure on an already constrained municipal water supply system. The remainder of the treated water is released into the local catchment, thereby alleviating some of the impacts of pollution in the river system.

Conclusions

The EWRP is a world-class sustainable development project that has turned a major liability into a valuable asset and created far-reaching benefits for the environment, the local community, and its feeder collieries. The project has earned several international awards and is the only mining initiative to be endorsed by the United Nations Framework Convention on Climate Change's Momentum for Change program (http://unfccc.int/secretariat/momentum_for_change/items/6634.php).

In July 2011, Anglo Coal approved an additional capital investment to increase the current treatment capacity to 50 ML a day, with a peak capacity of 60 ML a day, to address the water management needs of contributing mines over their remaining 20–25-year life. The second phase is currently under construction and is expected to be operational before the end of 2015. This development includes the construction of a 23-kilometre pipeline to cater for the company's 4.5 million tonne per annum Kromdraai Mine and its post-closure environmental obligations. The expansion represents a US$75 million investment, and brings Anglo American's total expenditure on mine water purification technology in eMalahleni alone to about US$140 million.

Figure 1: Phase 1 of the water reclamation plant

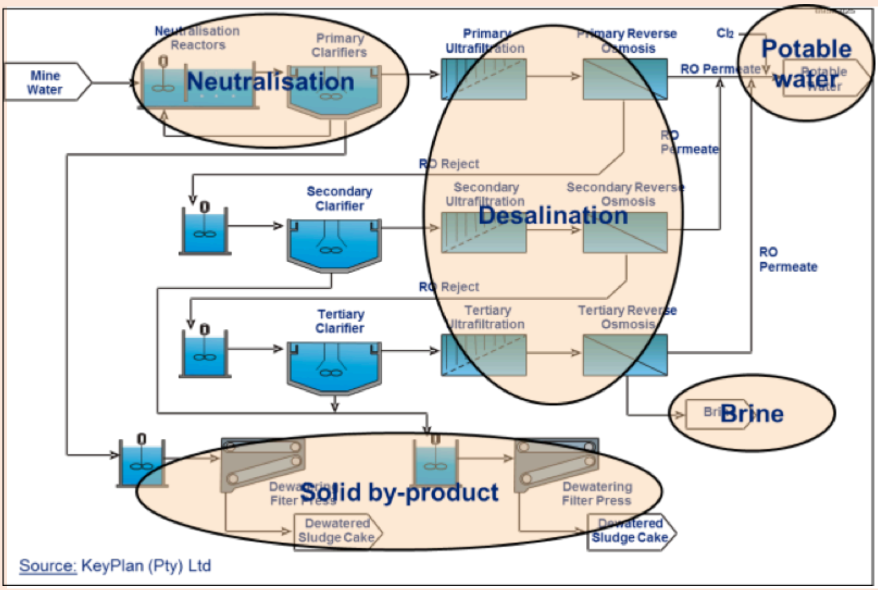

Figure 2: Process flow diagram

Table 1: Design feed and treated water qualities

WATER QUALITY PARAMETER	FEED WATER (95TH PERCENTILE)	TREATED WATER
pH	2.7	6.0 – 9.0
Total dissolved solids (mg/L)	4930	< 450
Calcium, Ca (mg/L)	660	< 80
Magnesium, Mg (mg/L)	230	< 30
Sulfate, SO4 (mg/L)	3090	< 200
Iron, Fe (mg/L)	210	< 0.01
Manganese, Mn (mg/L)	35	< 0.05
Aluminium, Al (mg/L)	40	< 0.15

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In situ or 'at source' treatment

In situ or 'at source' treatment becomes a more logical choice than centralised WTPs for active AMD treatment (for one or multiple water bodies) when the cost of pumping and piping polluted water to a centralised WTP exceeds the cost of mobilising a dosing system and neutralisation reagent to a polluted water body (Taylor et al. 2005). In situ treatment is conducted within the target water body via the direct application of reagent. Hence, the treatment equipment needs to be portable and possibly mobile (figures 31 and 32).

Figure 31: Generator-powered portable mixing and dosing system for remote sites

Figure 32: Truck-mounted, diesel-powered mobile mixing and dosing unit

All of the normal treatment processes (mixing, neutralisation, aeration and solids settling) occur within the water body. Although considerably cheaper than a centralised WTP, in situ treatment does not have the benefit of in-line process control. As a result, semicontinuous multiparameter water-quality monitoring before, during and after reagent dosing is vital for controlling the rate of treatment and achieving the desired water quality. Without accurate and comprehensive water-quality monitoring, underdosing or overdosing can result. Nevertheless, in situ treatment methods are highly flexible, low cost and easy and rapid to implement, and are normally able to treat far larger water volumes per unit time than fixed WTPs, as they can dispense 20–50 tonnes of reagent per day. They are particularly well suited to dealing with event-based and emergency response scenarios.

In situ treatment can be conducted on pit lakes, tailings dam ponds, process water ponds, stormwater ponds, rivers and creeks. It can be conducted as routine treatment, supplementary treatment, continuous treatment or batch treatment or as an emergency response.

Methods for the in situ treatment of pit water are described in Castendyk & Eary (2009), McCullough (2011) and Geller & Schultze (2013). Considerable care needs to be taken with the planning and execution of in situ treatment, as mixing dynamics (and hence the effectiveness of the treatment method) can be complex in these typically deep bodies of water. There are many examples in which in situ dosing of pit lakes has not yielded the desired outcome. There are several reasons why this may have occurred:

- Inexperienced operators attempted complex chemical dosing tasks without adequate knowledge or experience.
- In the case of in situ biological manipulation (for example, stimulating sulfate reduction below the hypolimnion by the addition of organic compounds or biomass), the interplay between the physical dynamics (for example, seasonal mixing) and the biological processes is intrinsically complex and difficult to predict.
- In some cases, the initial in situ treatment was successful, but ongoing inputs of acidity to the pit (from wall rock or from run-off or seepage from elsewhere) were not prevented, resulting in the water quality again deteriorating to an unacceptable level for either the establishment of biological function or for pumped discharge to a surface catchment.

Successful in situ treatment of AMD therefore requires specialist knowledge of the physical and chemical behaviour of the affected water body and the geochemical behaviour of materials disposed to the pit, the pit wall rock and the pit catchment.

7.3.3 Passive treatment systems

Surface water

There three basic classes of passive treatment system for surface waters affected by AMD are:

- oxic and anoxic limestone drains or riffle channels to neutralise low-pH water
- assisted chemical neutralisation (the use of solar or water power to drive reagent dispensing systems)
- wetlands (surface and subsurface flow through an organic matter substrate, with or without added limestone).

Historically, the use of passive systems to treat AMD has had mixed success, largely as a result of attempts to apply this technology to unrealistically high acidity and/or metal load situations. However, if passive treatment systems are designed and operated within their chemical and physical load limitations, they can provide a very effective and low-cost treatment alternative. While they cannot be regarded as walk-away solutions, correct implementation will minimise their maintenance and maximise their life expectancy.

Single-stage active or passive treatment systems that use chemical neutralants alone may have difficulty meeting stringent targets for aquatic ecosystem protection, depending on the range of metals and other solutes in the source water. This is where a second-stage passive biological polishing system (such as a wetland) can provide a distinct advantage by achieving the required water quality without the large capital and operating costs of secondary and tertiary active treatment technologies.

However, wetlands cannot rapidly adjust to a sudden deterioration in water quality or to a major short-term increase in flow rate. They work best at pH values greater than 4.5 under steady-state conditions, with a residence time of 10–15 days. They require a relatively constant inflow rate from a pond in which the mine water is initially collected (and pre-neutralised, if necessary) and must be protected from storm events using a split-weir diversion system to divert higher, more dilute flows into catchment flow lines.

The design lifetime of a passive treatment system is a key issue. In some cases, significant volumes of mine water requiring treatment may only be produced during the operations phase, before the rehabilitation of source material (such as waste rock piles) or the cessation of dewatering. In such cases, there is obviously less emphasis on long-term (post-closure) sustainability. The need for self-sustaining systems becomes much more critical following site decommissioning. Passive treatment systems accumulate toxic metals/metalloids, and the resultant long-term implications for closure planning should be addressed when this type of system is considered.

Further details about the design and application of wetland systems for AMD treatment are in guidance produced by the US Office of Surface Mining Reclamation and Enforcement (OSMRE)[61] and by the Piramid consortium (2003).[62]

Groundwater—permeable reactive barriers and natural attenuation

Permeable reactive barriers

Given the difficulties associated with the traditional pump-and-treat approach for remediating contaminated groundwater—especially for low-yielding fractured rock aquifers—a new generation of passive in situ treatment methods has been developed, tested and implemented at full scale over the past 20 years (Vidic 2001; Wright & Conca 2006; Naidu & Birke 2014). The treatment process may involve abiotic and biological processes acting individually or in synergistic combination. These passive systems are called 'permeable reactive barriers' (PRBs, since the groundwater is treated as it passes through the reactive zone. These systems are 'barriers' to the target solutes and not to the flow of water.

For AMD treatment applications, this class of technologies comprises barriers in which:

- sulfate is reduced to sulfide and metals are precipitated as insoluble sulfides (because the barrier is located below the watertable, there is minimal risk of future reoxidation of the precipitated metal sulfides)
- metals are precipitated as insoluble forms (for example, using crushed apatite to precipitate metal phosphates)
- inorganic (for example, Cr (VI), Se (IV), U(VI)) ions are reduced to insoluble forms
- nitrate (from blasting residues) is degraded by microbial or inorganic (for example, reduction by metallic iron) processes.

Finely divided particles of iron (so-called zero-valent iron, ZVI), often combined with an organic-rich 'spacer' substrate, are becoming the material of choice to be used in PRBs for the treatment of water containing a wide variety redox-sensitive metals and metalloids, such as Co, Cr, Cu, Ni, As and Se, provided the pH is not acidic. Importantly, all of these metals can be soluble at pH values typical of NMD and are not easily removed to environmentally acceptable levels by low-cost conventional methods of treatment. In the case of an acidic plume, a neutralant (for example, fine-grained limestone alone or in combination with organic matter to drive sulfate reduction) needs to be added in combination with the iron.

PRBs can be used to treat an existing source of AMD in groundwater or they can be installed as sentinel barriers that provide a long-term backup to a primary control of AMD provided by an engineered containment facility (such as a cover and/or liner combination) for sulfidic waste. PRBs have been applied to plumes originating from both WRDs and tailings dams.

61 http://amd.osmre.gov/.

62 http://www.imwa.info/piramid/files/PIRAMIDGuidelinesv10.pdf.

PRBs are subject to limitations similar to those applying to passive treatment systems for surface water; that is, they are best suited to relatively low acidity and metal load but long-duration applications, such as the downgradient interception of a broadly dispersed plume.

As for all types of treatment system, the longevity of a PRB depends on the loading of solutes and acidity in the plume being treated relative to the amount of treatment substrate that has been initially installed in the PRB. It may be necessary to excavate and replenish the treatment substrate if the initial charge becomes exhausted, as revealed by downgradient performance monitoring, before the ultimate treatment objectives can be met.

Natural attenuation

Natural attenuation can potentially be used, where applicable, to control the contamination of groundwater by downward seepage of AMD. This relies on the action of natural adsorption and precipitation to remove most solutes from the groundwater (but is less effective for the major ions—calcium, magnesium and sulfate) before receptors (groundwater users, groundwater-dependent ecosystems) are affected.

For example, natural attenuation could be considered where a WRD is underlain by a thick horizon of carbonate-containing rock with sufficient neutralising capacity to account for the acidity load to be produced over the long term by the oxidation of sulfide waste in the dump. The carbonate layer would effectively be a horizontal PRB. This natural attenuation process would be an adjunct to the function of a cover system installed to reduce infiltration and/or oxygen ingress.

Making use of the possibility presented by natural attenuation is not a 'do nothing' management option, but relies on robust evidence from field investigations, modelling and monitoring being provided to regulators to demonstrate that the predicted removal processes are taking place and that there will be an (acceptably) low risk of adverse impact on the receiving environment. The United States EPA and US state regulatory authorities have produced guidance for using natural attenuation to manage metals and radionuclides in groundwater, and that guidance should be used as a reference by those who are considering the potential for its application.[63]

In the Australian context, natural attenuation, by neutralisation by the thick underlying layer of Andamooka limestone of downward percolating acidic leachate from the tailings dams at Olympic Dam, has been accepted to be an effective process (South Australian Government 2011).

Pit lakes

International experience with the application of passive treatment of pit water quality by microbiological processes has been mixed (Geller & Schultze 2013).

Despite the considerable promise that has been shown by field-scale work to date, much more remains to be done to develop a technical understanding needed so that the proposed methodology can be relied upon for the long-term maintenance of water quality for a given pit lake. There have been several examples in which attempts to create stably stratified systems with anoxic hypolimnions of appropriate composition have not worked as expected (Fisher & Lawrence 2006; Park et al. 2006), necessitating subsequent remedial action.

63 http://www.itrcweb.org/Guidance/GetDocument?documentID=5.

In particular, excess hydrogen sulfide gas has been produced in anoxic hypolimnions created by either adding organic matter (such as at Island Copper in British Columbia; Wilton & Lawrence 1998) or stimulating biological productivity by adding nutrients. This problem can occur in systems in which there is an excess of soluble sulfate relative to dissolved metals (Fe, Cu, Zn) available for precipitation as metal sulfides, and hence excess free H_2S gas can be generated. Because the oxidation of pyrite commonly involves the early physical separation of iron from the drainage by precipitation of ferric hydroxide, it is common to have a significant stoichiometric excess of sulfate relative to iron in AMD.

At best, the passive treatment of an established pit lake must be viewed as a very long term strategy that will require substantial ongoing investment in monitoring and research.

8.0 REGULATORY FRAMEWORKS FOR ASSESSING AND MANAGING AMD

Key messages

- Basic compliance with legislation is the minimum standard that should be achieved when managing environmental impacts of mining activities. Leading practice goes beyond this.
- Leading practice recognises that there are obligations to stakeholders over and above those strictly required by legislation.
- For Australian mining companies operating overseas it may be necessary to consider international guidelines for those cases where national legislation does not provide specific guidelines for mining practices, or where the existing requirements do not meet contemporary or internal corporate standards of practice.

8.1 Introduction

Both the Australian Government and the state and territory governments have responsibilities to protect human health and the environment from the harmful effects of mining, including by mitigating potential adverse impacts from AMD. Those responsibilities commence during the exploration phase of a mining project and typically persist long after mine closure.

To discharge those responsibilities, Australian jurisdictions have enacted legislation for the assessment and management of mining. In line with international environmental law, the key principles that underlie that legislation generally include the following aspects:

- *The Precautionary Principle.* Where there are threats of serious or irreversible environmental harm, the lack of full scientific certainty should not be used as a reason to approve an activity or to postpone implementing control measures.
- *The Principle of Intergenerational Equity.* The current generation should ensure that the health, diversity and productivity of the environment are protected and will continue for the benefit of future generations.
- *Improved valuation, pricing and incentive mechanisms and the Polluter Pays Principle*. Environmental, economic and social equity considerations should all be factored into decisions made on potentially polluting activities such as mining. The full costs of preventing or remediating pollution associated with the activity should generally be borne by the polluter.
- *Public participation in decision-making.* Members of the public (including indigenous communities) should be able to participate in the environmental decision-making processes of governments.

This section illustrates how legislation and statutory instruments are typically used to direct proponents to assess and manage the effects of AMD in Australia, and examines the different roles that the state, territory and Australian governments have in regulating the management of AMD. Jurisdiction-specific requirements need to be determined by dialogue with the relevant authorities.

8.2 The role of state and territory governments

Although the administration of environmental protection legislation in Australia is facilitated through a number of bilateral agreements with the Australian Government, in practice much of the administrative work of assessing and managing mining projects and their environmental impacts is done by the state and territory governments.

Under Australian law, the ownership of minerals within each state or territory is vested in the state, and the extraction of minerals is authorised under provisions of the specific mining legislation that applies within that jurisdiction. As a consequence, government agencies in each state or territory have a responsibility to ensure that mineral exploration, mining and mine closure take place in accordance with the relevant environmental protection legislation. The detail of environmental legislation that applies to mines varies considerably between jurisdictions (as do the administrative arrangements between government agencies within the jurisdiction), but the legislation generally uses the following statutory instruments to ensure that AMD risks are adequately managed:

- *Environmental impact assessment/statement (EIA/EIS).* These are interdisciplinary and multistep procedures to ensure that all potential environmental issues associated with the mining project are adequately assessed and appropriate management responses are identified before the project is approved and commences operation. EIAs and EISs usually have a considerable amount of public involvement and form much of the basis of the 'social licence to operate' needed before the operation can proceed. As part of the EIA process, proponents are generally required to undertake sufficient sampling and geochemical test work to determine whether there is a significant risk that mining will cause AMD. If AMD is identified as a significant risk, proponents are also required to indicate how AMD risks would be managed during operations and following closure if the mining project were to proceed.
- *Mine site infrastructure construction approvals.* Proponents may need geotechnical and environmental approvals from government for the construction of infrastructure such as water storages, TSFs and waste rock facilities. As part of the approvals process, they generally need to demonstrate how PAF materials can be segregated from other waste materials and managed in a manner that will prevent the future generation of AMD.
- *Environmental discharge and dewatering approvals.* Proponents may need an environmental licence for the operation of TSFs and wastewater management ponds. Approvals may also be required for dewatering mine pits or underground workings and for disposing of the dewatering effluent to the environment. As part of any licence requirements for managing these systems, proponents may be required to undertake ongoing water-quality monitoring to demonstrate that groundwater has not been contaminated by seepage from TSFs or other waste facilities. If chemical constituents indicative of AMD are detected in groundwater near TSFs or other wastewater management infrastructure, proponents are usually required to inform the relevant government agencies and to put in place measures to investigate the cause of the seepage and ensure that it will not create adverse environmental impacts.
- *Mine closure plans.* Proponents may be required to start developing a mine closure plan soon after (or, in Western Australia, before) the mine has received environmental approvals. In order to develop such plans, proponents generally need information about the long-term geochemical behaviour of waste materials. This typically requires extensive geochemical test work (especially long-duration kinetic testing) and geochemical modelling. A mine closure plan is meant to be a living document that changes and becomes progressively more detailed as more is learned about the geology and geochemistry of the mine site. This means that geochemical testing of exposed faces (such as pit walls), mined waste and process residues should continue during the mine's life to ensure that adequate information is available to inform both the ongoing operational management of the waste and the final version of the closure plan before mining ceases.

- *Financial provisioning for closure.* Financial assurance may be required to be lodged by the proponent before the start of operations. The quantum of assurance funds is generally calculated based on the liability created by the mining operation and the level of risk and associated mitigation that is required, and may include a contingency. In some jurisdictions, financial assurance is accrued via a resources levy imposed on the industry as a whole.
- *Contaminated sites reporting and management.* Where monitoring has indicated that groundwater or surface water run-off from the mine site has shown evidence of contamination due to AMD, proponents are generally required to report the issue under provisions of the contaminated sites legislation that applies in the jurisdiction. As part of the reporting requirements, they are usually required to determine the source, extent and severity of the contamination and the extent to which the environment has been or is likely to be affected by it. Depending on the level of environmental risk posed, proponents are then usually required to put in place measures to mitigate the environmental impacts.

8.3 The role of the Australian Government

The Australian Government has an important role in ensuring that the regulation of mining activities by each state or territory is carried out in a consistent manner according to nationally recognised guidelines and principles that meet Australia's obligations under international treaties.

The principal piece of Australian Government legislation that is used to assess mining projects is the *Environment Protection and Biodiversity Conservation Act 1999* (the EPBC Act). Under the Act, the federal government can require an EIA to be undertaken where there is the potential for a mining project to have impacts on one or more of nine matters of national environmental significance. The only mineral resource that is specifically listed for full automatic assessment under the EPBC Act is uranium. The Act was amended in 2013, making water resources a matter of national environmental significance in relation to coal seam gas and large coalmining operations. This has direct relevance for the involvement of the Australian Government in assessing the potential for AMD at coalmines as part of the project approvals process.

The Australian Government has a specific role in the administration of the *Aboriginal Land Rights (Northern Territory) Act 1976* and in the matter of determining Indigenous land ownership rights. Both of these issues can potentially affect the identification of the groups that need to be consulted when determining the environmental values that underpin the AMD risk assessment undertaken by proponents.

By producing guidelines for water quality and soil contamination, the Australian Government also has an important role in maintaining a nationally consistent approach to assessing and managing contamination caused by AMD. Those guidelines are applied in all states and territories.

The *Australian water quality guidelines* (ANZECC–ARMCANZ 2000a,b,c) are arguably the single most important instrument for assessing the risk posed by the impact of AMD on the aquatic environment. The use of the guidelines framework to identify environmental values and to assign water-quality guidelines values as the basis for assessing the risk posed by AMD is described in detail in Section 5.3.

The National Environment Protection Council was established under the *National Environment Protection Council Act 1994* to maintain a consistent approach to managing environmental issues by developing National Environment Protection Measures (NEPMs). The key NEPMs for assessing, managing and reporting on environmental issues associated with AMD are the Assessment of Site Contamination and the National Pollutant Inventory NEPMs.

The National Environment Protection (Assessment of Site Contamination) Measure was established to develop a consistent approach to dealing with site contamination through the development of technical guidelines and policies that are implemented through legislation in each state and territory.[64] The NEPM was recently updated and now contains expanded guidance on how to undertake risk assessments that include accounting for bioavailability of metals in soils contaminated by mining wastes or AMD.

Similarly, the National Environment Protection (National Pollutant Inventory) Measure was developed to collect information on emissions and transfers of potential pollutants and disseminate that information in a readily understandable form to a wide range of stakeholders.[65]

8.4 Cumulative impacts of AMD

Although current regulatory measures for managing the cumulative impacts of mining in Australia do not specifically address AMD, regulators are becoming increasingly concerned about the cumulative effects of land uses such as mining on sensitive environmental receptors. For example, the Western Australian EPA released a position paper in 2014 to provide strategic advice to government on the cumulative impacts of mining development in the Pilbara region (WA EPA 2014). Similarly, recent revisions to the EPBC Act require proponents and regulators to consider the cumulative impacts of coalmines and the extraction of coal seam gas on water resources.

The Centre for Social Responsibility in Mining at the University of Queensland has produced guidance for addressing the cumulative impacts of coalmining to meet the requirements of state and federal legislation (Franks et al. 2010). The concepts in that document are directly applicable to other forms of mining. The multiple land-use framework (SCER 2013) has been designed to assist in identifying and resolving the cumulative impacts of mining in the context of other land uses in mining regions.

The key implication for regulation is that environmental criteria mandated for assessing new mine sites may in future vary both spatially within a catchment and over time where land uses (including mines) are discharging to the environment.

8.5 Non-statutory considerations for mining operators

The concept of the 'social licence to operate' was first developed by the mining industry in Canada when the industry recognised that communities potentially affected by mining activities had a right to be involved in decisions about how a mine would be developed and managed. From a purely business perspective, it was also found that mining companies that encouraged community participation in decision-making usually faced less opposition to opening and operating a new mine and fewer administrative hurdles than companies that allowed more limited public participation.

More importantly, it has also become increasingly apparent that mining companies that make a long-term commitment to being part of the local community (for example, to involving affected communities in planning and decision-making) generally encounter less opposition than companies that do not make such a commitment. Communities generally assess how a company has performed in complying with its social licence by the extent and nature of its communication with them (for example, is communication

64 Further information on this NEPM is available at the SCEW website, http://www.scew.gov.au/nepms/assessment-site-contamination.

65 Further information is available at the NPI website, http://www.npi.gov.au/.

transparent?), by the extent to its work has been in accordance with leading practice, and by the extent to which it has kept its promises and fulfilled its commitments.

To a large extent, those factors are also requirements of legislation for the regulation of mine sites in Australia. However, mine operators should view basic compliance with legislation as a minimum standard to be achieved when managing environmental impacts of mining activities. Leading practice recognises that there are obligations to stakeholders over and above those strictly required by legislation.

8.6 Abandoned or legacy mine sites

A strategic framework for managing abandoned mines has been developed for Australia (MCMPR–MCA 2010). The framework encourages jurisdictions to strategically manage mining legacies and to apply common approaches to:

- site inventories and site data management
- improved understanding of liability and risk relating to abandoned mines
- improved performance reporting
- the standardisation of processes and methodologies
- knowledge and skill sharing across jurisdictions.

However, neither the Australian Government's EPBC Act nor state nor territory environmental protection legislation specifically addresses environmental impacts from abandoned mines, many of which have AMD issues. Consequently, in the absence of abandoned-mine legislation, ambiguity remains about responsibilities, standards and processes to achieve the adequate rehabilitation of such sites. Each abandoned site is typically dealt with on a case-by-case basis.

However, leading practice is being demonstrated by aspects of abandoned/legacy mine programs in the states and territories.

Western Australia's detailed abandoned mine inventory was released in 2002 to help to mitigate the immediate human health and safety impacts of those sites and to provide a basis for planning remediation. In other jurisdictions, the amount of data on legacy sites is more variable, as the information has been extracted from existing mining databases.

Since 2010, the Western Australian Department of Mines and Petroleum and Environmental Protection Authority have been implementing significant reforms to the regulatory and policy frameworks relating to mine closure in that state (DMP–EPA 2015), including drafting policy to manage and rehabilitate mine sites once they are abandoned. The rehabilitation of abandoned mines is enabled through the Mining Rehabilitation Fund, which is a pooled fund contributed to by Western Australian mining operators. The state's *Mining Rehabilitation Fund Act 2012* provides the framework for the fund.

All tenement holders operating on Western Australian *Mining Act 1978* tenures (with the exception of tenements covered by state agreements not listed in the regulations) are required to report disturbance data and contribute annually to the fund. Money in the fund is available to rehabilitate abandoned mines across the state where the tenement holder/operator fails to meet rehabilitation obligations and every other effort has been used to recover funds from the operator. For legacy abandoned mines, rehabilitation occurs using the interest earned on the fund contributions.

A similar levy-based approach has been implemented in the Northern Territory. In October 2013 the Northern Territory *Mining Management Act* was amended to require operators to pay in cash an annual levy of 1% of their security bond. To offset this impact of the levy on operators, the amount of the security bond required to be held was reduced by 10%. The purpose of the levy is defined by the Act as being 'for the effective administration of this Act in relation to minimising or rectifying environmental harm caused by mining activities.'[66]

The amendment requires a 1% levy on the total calculated rehabilitation cost applied to each mining operation authorised under the Mining Management Act for all new mines. Unlike mines in Western Australia, all mines in the Northern Territory are also required to have a 100% bond for mine closure (based on third-party costs), and to contribute the non-refundable Mining Rehabilitation Fund levy. As the security calculation for the bond includes a 15% contingency component, the 10% discount means that the requirement for 100% security remains effective.

The initial returns from the Mining Rehabilitation Fund have been used to create the Legacy Mine Unit (Woollard 2014), which is working towards a whole-of-jurisdiction approach to identifying, prioritising and managing the remediation of legacy mines in the Northern Territory.

In New South Wales, the Auditor-General has recommended that seven large AMD legacy mine sites (Conrad, Woods Reef, Captains Flat, Sunny Corner, Ottery, Cowarra Gold and SCA Cobar) be notified to the state's Environment Protection Authority by the Department of Trade and Investment, Regional Infrastructure and Services under section 60 of the *Contaminated Land Management Act 1997.* The Auditor-General noted that currently some abandoned mines may not yet have relevant legislative frameworks for management (NSW Auditor-General 2014).

The Australian Government has provided funding for the rehabilitation of abandoned uranium mine legacy sites on Aboriginal lands in the Northern Territory. The work that is currently of most relevance to this handbook is the development of a rehabilitation strategy for the AMD legacy at the former Rum Jungle uranium mine site (NTDME 2013). This activity is being undertaken under the terms of a national partnership agreement between the Australian and Northern Territory governments.[67]

8.7 International mining operations

Australian mining companies operating overseas must comply, as a minimum, with the legislation of the host country. However, it may also be necessary to consider international guidelines such as those produced by the World Bank and World Health Organization if national legislation does not provide specific water-quality guidelines for mining operations or if the existing requirements do not meet contemporary or internal corporate standards of practice.

66 http://www.nt.gov.au/d/Minerals_Energy/index.cfm?newscat1=&newscat2=Indigenous%20Engagement%20Unit&header=Amendments%20to%20the%20Mining%20Management%20Act.

67 http://www.nt.gov.au/d/rumjungle/; see Rum Jungle case study in Section 6 of this handbook.

The World Bank's International Finance Corporation has developed environmental, health and safety guidelines for mining that are relevant to water quality associated with AMD (IFC 2007). In particular, the guidelines state:

> Management of potentially acid generating material, AMD and metal containing leachate should extend for as long as there is a need to maintain effluent quality to the levels required to protect the local environment, including where necessary, into the decommissioning, closure, and post-closure phases of the mine.

Globally, increasing community expectations will drive the application of leading practice AMD management strategies in both developing and developed countries.

An additional complicating factor for managing AMD risk in river catchments in developing countries is the potential for artisanal mining. Multiple small-scale mining activities (whether legal or illegal) can create cumulative environmental impacts adjacent to active mines and at legacy mine sites. ICMM (2009) provides guidance on how industry can work together with small-scale miners.

Further guidance on key aspects of international engagement by mining companies is provided in *Social responsibility in the mining and metals sector in developing countries* (DRET 2012).

9.0 PERFORMANCE EVALUATION AND MONITORING

Key messages

- At the outset of the AMD evaluation process, establish specific, clear and achievable performance criteria framed in terms of the overall site environmental management objectives.
- A robust conceptual site model of AMD processes should first be developed to identify those parameters that need to be monitored and the locations for monitoring.
- Because the oxidation of sulfides begins immediately upon contact with oxygen and water, the post-baseline monitoring program should start immediately after the emplacement of sulfide-bearing wastes.
- No matter how well executed the monitoring program might be, collecting inappropriate data will not assist performance evaluation and could even fail to identify the development of an AMD management issue.
- Companies operating sites at risk of post-rehabilitation AMD should conduct a robust long-term post-rehabilitation monitoring program, with appropriate financial and technical provisioning, to demonstrate that closure criteria have been met.

9.1 Introduction

Performance evaluation and monitoring are established practices in the mining industry. They are needed to aid the development of good mine management practices and to demonstrate effective performance to stakeholders and regulators against agreed environmental management objectives. The demonstration of performance is achieved by identifying what needs to be evaluated and then determining what should be monitored to best inform the evaluation.

The objectives of this section are to provide an outline of the principles of performance evaluation and monitoring and some examples of parameters that are monitored for AMD evaluation. Comprehensive guidance on the topic of evaluating performance and monitoring is in the *Evaluating performance: monitoring and auditing* leading practice handbook (DIIS 2016c).

9.2 Performance evaluation

Performance evaluation assesses the response of a system by comparison with specific criteria. It typically involves quantifying the change that has taken place and often involves demonstrating that the change occurred in response to particular conditions and that the effort needed to produce the change was cost-effective.

At least two sets of performance criteria need to be established: first, those that provide early warning of developing trends and trigger site management actions to address AMD issues; second, those that relate to the reporting of site environmental performance to regulators and stakeholders.[68]

Early warning triggers need to be set conservatively enough to allow management action to be taken before there is a risk of noncompliance with external regulatory triggers.

In the context of AMD, the performance evaluation may include:

- assessing the development of AMD as a result of mining activities
- identifying the need for AMD management strategies
- quantifying the effectiveness of AMD management strategies (that is, whether or not an AMD management strategy achieved its stated objectives)
- establishing whether agreed requirements have been met (for example, the construction specifications of a WRD cover or regulatory compliance).

The AMD performance evaluation program should include:

- specific, clear and achievable objectives framed in terms of the overall AMD management objectives and relevant quantifiable variables
- criteria against which the performance will be evaluated (this could require the establishment of control sites and/or baseline measurements for water quality)
- the capacity to link the causes of AMD generation to their effects on AMD characteristics
- a site-specific conceptual model of AMD processes and site characteristics.

The 'relevant quantifiable variables' mentioned in the first point above should ideally be directly measurable, but in some cases quantification will rely on monitoring of a related variable. The relationship between the variables should be as direct as possible. For example, in situ measurements of sulfide oxidation rates in a WRD are difficult, and in practice it is likely to be more effective to measure oxygen concentration and temperature distributions.

9.3 Conceptual site model of AMD processes

Developing a site-specific conceptual model of processes that will influence the characteristics of AMD improves the likelihood of identifying the causes of changes that lead to AMD and therefore assists in identifying variables that should be monitored. A poorly conceived model may lead to monitoring the wrong sets of variables or measuring at the wrong times or in the wrong places. No matter how rigorous the monitoring program, collecting inappropriate data will not assist performance evaluation and may even fail to identify the development of a major AMD management issue. The collection of inappropriate data may also delay the issuing of permits or approvals, since such data will fail to convince regulators.

A comprehensive conceptual model upon which to base a performance evaluation program for AMD will include, among other things:

- boundary conditions, such as those represented by flows into and out of the area of interest
- initial conditions for various processes, including site conditions before mining
- site topography and climate
- the location of sulfide and carbonate minerals

68 The process for setting the latter criteria is described in Section 5.

- mechanisms that determine the rate of oxygen supply that supports sulfide oxidation
- geochemical controls (such as the addition of lime during WRD construction)
- hydrological and hydrogeological paths and controls
- locations and types of impacts
- the existence of water treatment
- initial estimates of the rates of relevant processes (for example, sulfide oxidation) and associated parameter values.

For effective performance evaluation, the conceptual model should be developed to a level of detail that includes consideration of the lag time for AMD development, the potential duration of the process, the times at which there may be opportunities to establish management strategies for the control of AMD, and the times at which those opportunities may no longer exist. The model will assist in setting priorities for the monitoring needed for the performance evaluation.

A well-conceived site conceptual model provides the foundation for the design of an effective AMD monitoring program to evaluate performance. It should be developed at the earliest possible phase, ideally at feasibility, but certainly before the site becomes operational.

9.4 Monitoring

Monitoring is the making of an intermittent series of observations to quantify change over time. The conceptual model developed for the site using the techniques described in the previous sections in this handbook should be used to identify:

- which variables will be monitored and at what frequency
- when particular components of the monitoring program should be established
- suitable locations and times for monitoring of control and baseline sites
- the duration over which monitoring should occur
- where monitoring is to occur (that is, the requirements of monitoring locations and their spatial distribution)
- the range of values a monitored variable might take over the monitoring period.

The outcome after consideration of the specific site objectives and the AMD conceptual model will be a monitoring program designed to capture the required information practically and cost-effectively. It is likely that the program as a whole will be phased in over time and split into a set of smaller, more focused components. Table A.1 in Appendix 1 provides a detailed reference guide for parameters that might be monitored, locations and times of monitoring and potential performance evaluation criteria. This compilation is not intended to be prescriptive and should be adapted for use as required.

The number of variables to be monitored depends largely on the conditions and the objective of the monitoring program, as well the level of AMD risk at the site. Consequently, it may be necessary to monitor a number of independent variables that could potentially affect the dependent variable of interest so as to increase the likelihood of determining cause and effect when evaluating critical processes or the performance of AMD management.

Resources required for monitoring include the personnel to run and to interpret the results from the monitoring program and the equipment that is used for monitoring and data storage. The distribution between the two depends on costs, reliability, technical feasibility, availability, the required continuity of operation, and capacity to respond to extreme events (for example, staff are often unavailable to respond during prolonged intense rainfall).

Careful consideration should be given to using existing resources or working in conjunction with studies undertaken for other purposes. Such opportunities have the potential to reduce costs. For example, the logging and sampling of drillholes installed primarily for exploration, geotechnical characterisation and dewatering purposes may assist in the identification of the potential for AMD and avoid the need for drilling holes dedicated to AMD sampling. However, such holes are unlikely to be suitable for water-quality monitoring, and it should be noted that there is often competition for access to drill core samples between disciplines.

The use of monitoring facilities installed for other purposes might not provide the information required. For example, monitoring groundwater quality using an existing water-level monitoring bore at the edge of a lease to determine whether a TSF or WRD is producing AMD may be inappropriate. Although the borehole might be available at no additional cost and water samples might be easily collected during regular visits to determine water levels, the groundwater flow directions or flow rates may be such that the AMD released from the TSF will not arrive at the monitoring location for decades.[69]

An appropriate choice of measurement equipment has the potential to improve the likelihood of data accuracy, frequency and continuity while reducing labour costs and addressing access issues such as safety and potential damage to sensitive areas from repeated traffic movements. Because the diversity and capabilities of monitoring equipment, data logging and telemetry continues to evolve, efficiencies in the setting up of the program are likely to come from early consultation with equipment suppliers and installers.

9.4.1 Examples of parameters to monitor on site

Sulfide oxidation

Without oxygen, sulfide minerals will not oxidise to produce AMD, although secondary minerals such as jarosite can continue to release acidity under anoxic conditions if some oxidation has already occurred. Prompt control of oxygen supply is thus the mine operator's best opportunity to control acid and sulfate production rates and should be at the forefront of consideration when developing AMD management strategies.

Because the oxidation of sulfides begins immediately upon contact with oxygen and water, the best short- and long-term benefits will be achieved by limiting oxygen supply immediately upon the emplacement of sulfide-bearing wastes.[70]

As oxygen supply management strategies are most effective if implemented soon after initial exposure and placement of the waste, a monitoring program that provides feedback within that time frame is required. Only with short-term feedback will it be possible to identify performance issues in time to apply corrective actions.

Given the potentially long times for infiltrating water to transport oxidation products from a waste landform to surface water or groundwater, the monitoring of seepage will not provide feedback within the time frame needed to demonstrate that strategies to limit the supply of oxygen have been effective.

69 Further information about groundwater monitoring is in the *Evaluating performance: monitoring and auditing l*eading practice handbook (DIIS 2016c).

70 A detailed discussion of the theory of oxidation and gas transport is in Blowes and Jambor (1994) and in Blowes et al. (2003).

A practical and proven method of evaluating the performance of oxygen supply management strategies is based on measuring oxygen concentration and temperature distributions within the waste (Harries & Ritchie 1985; Garvie & Taylor 2000). Interpretation of the data can produce estimates of the overall rates of oxidation, rates of acid and sulfate production, an assessment of the dominant oxygen supply mechanism and an assessment of the need for a management strategy or the effectiveness of an existing management strategy. Both oxygen and temperature distributions should be measured to make it possible to provide a unique explanation of the observed data.[71]

Water quality

The quality of surface water and groundwater is often considered to be the primary focus of an AMD monitoring program. However, water quality is a lagging indicator of the development of an AMD issue, since it is typically measured at the exit seepage points of a mine waste management facility. Earlier indications of developing seepage water quality can be obtained from the sampling of lysimeters (of appropriate scale) installed as construction is progressing or from monitoring bores installed and extended with the rise in level of the facility. However, great care needs to be taken in the design and installation of such monitoring systems to minimise the potential for physical damage and erroneous monitoring data.

The initial water-quality parameters to be measured will have been defined by the findings from the geochemical static and kinetic tests described in Section 4. However, this starting range of parameters should be periodically revisited and revised if required, based on evolving operational experience.

Waste facility water balance

A common objective for leading practice design of waste storage facilities is to minimise the interaction of infiltrating water with at-risk AMD material. Hence, monitoring the net water infiltration into waste storage facilities is important, and the extent of reduction in net water infiltration is a key performance indicator for most cover systems.

The net infiltration of water into a waste storage facility can be inferred by monitoring results produced by a number of different approaches, each of which has advantages and disadvantages (O'Kane 2011; Schneider et al. 2010). Ideally, several of those methods should be used in parallel to ensure a high degree of confidence in the result that is obtained.

9.5 Data storage, evaluation and reporting

The amount of effort required to store, manage and interpret the many data streams that are part of an AMD monitoring program should not be underestimated. External assistance may often need to be engaged to supplement the resources that are available on site.

The data interpretation stage of performance evaluation is much faster and easier if monitoring data is stored in a safe place in a logical order with all of the critical associated metadata and in a format that is easily retrieved. Establishing processes that automatically organise data and files can save many hours of work at the data processing stage. Well-organised storage may also reduce the likelihood of introducing errors and ease the process of auditing.

71 The two case studies in Section 6 illustrates the use of oxygen concentration and temperature measurements to assess the effectiveness of a soil cover over sulfide-bearing waste rock, combined with a ground-up construction method using shallow lifts.

Existing data storage systems may be both suitable for and a highly valuable means of storing data. For example, incorporating static test results in the mine's established geological model can provide a relatively easy method of comparing static test results with other rock type data.

Monitoring data should be regularly reviewed, interpreted and evaluated against the performance criteria. A pragmatic approach is to undertake the evaluation at defined times as the data becomes available. This has the potential benefits of identifying:

- faulty monitoring equipment
- the need to reconsider the conceptual model and monitor additional variables to obtain a clearer evaluation
- trends indicating a developing AMD issue.

Communicating monitoring results, both internally and externally, is crucial to the overall success of the AMD management strategy. It is only with adequate communication that appropriate changes will be made during the operations phase and prevent many of the long-term issues and difficulties associated with AMD that have to be faced at mine closure. The effective communication of monitoring results and of the outcomes of performance evaluations with stakeholders and regulators is one of the pillars of leading practice management of AMD issues.

The monitoring data acquired during the operations phase will provide the basis for revisiting, and updating if required, the site closure plan in consultation with relevant stakeholders. If the monitoring data shows that effective minimisation or control strategies have been implemented during operations, there will be much greater confidence that closure will be successful.[72]

72 Further details about monitoring and evaluating closure performance are in the Mine closure (DIIS 2016a) and *Evaluating performance: monitoring and auditing* (DIIS 2016c) leading practice handbooks.

10.0 COMMUNICATING AMD ISSUES TO STAKEHOLDERS AND INVESTORS

Key messages

- Effective engagement with stakeholders in the early stages of projects is essential to identify and agree the values that need to be protected.
- A transparent, clearly communicating and accountable operation will be viewed favourably for its commitment to sustainability and to maintaining its social licence to operate.
- Communication in a form that is understandable and relates directly to demonstrating the protection of agreed values is likely to be the most fruitful approach.
- A key part of the process for obtaining investment for the development of a new mineral resource is estimating its potential value.
- Both the JORC and VALMIN codes for estimating and reporting the extent and value of a resource require aspects that can materially affect the value of the project (such as AMD risk and waste management costs) to be implicitly addressed.

10.1 Overview

The previous sections of this handbook focus on how a leading practice approach to the technical aspects of the mining life cycle should be used to minimise AMD risk. However, mining invariably also takes place in a social and regulatory landscape comprising potentially affected communities, other stakeholder groups, government agencies and project investors. The way these interactions are addressed and managed by the mining company profoundly affects mining approvals, regulation and mine closure and the way in which the mining industry is viewed by society in general.[73]

Given that more than 60% of Australian mining operations have neighbouring Indigenous communities, particular consideration must be given to Indigenous Australians as key stakeholders or, as some would argue, 'rights-holders', in recognition of Indigenous rights and interests and the special connections of Indigenous people to land and water.

Identifying potential AMD issues during the exploration and feasibility phases at greenfield sites is critical, as those phases feed in to:

- the consultation that should occur as part of the EIA component of the project approvals process
- the economic valuation of the resource reported to the share market and to potential project investors (see below).

Effective engagement in these early stages with potentially affected communities and other key parties,

73 The focus in this section is on the communication of AMD-related issues. For a broader discussion of strategies for leading practice communication and engagement, see the *Community engagement and development* (DIIS 2016b) and *Working with Indigenous communities* (DIIS 2016d) leading practice handbooks.

including regulators, is essential to identify and agree on the values that need to be protected, and which underpin the specification of project environmental performance criteria.

The acquisition of quality technical data from a well-targeted performance monitoring program is essential for effective environmental management by the mining company. However, it is the communication of this information in a form that is understandable and relates directly to demonstrating the protection of agreed values that is likely to be the most fruitful approach. A transparent, clearly communicating and accountable operation will be viewed favourably for its commitment to sustainability and to maintaining its social licence to operate. Leading practice involves recognising that there are obligations to stakeholders over and above those strictly required by legislation.

Communities generally assess how the company has performed in maintaining its social licence by:

- the extent and nature of its communication (for example, is it transparent?)
- the extent to which its work has been in accordance with leading practice
- the extent to which it keeps its promises and fulfils its commitments.

If performance issues subsequently arise, the potentially affected communities and regulators alike are more likely to listen to and to reasonably discuss the issue and proposed management actions than if periodic and open communication has not previously occurred. In particular, communicating the management strategies and systems being put in place to manage AMD risk provides important context and demonstrates the operation's serious intention to minimise the risk of impacts. Public participatory water-quality monitoring programs have also been used to mitigate and manage conflict associated with mining by enabling the public to make more informed decisions (CAO 2008).

The regular collection and analysis of data, followed by the public release of both raw and interpreted data, including assessments of long-term trends and how they compare with the level of protection required to maintain environmental values, along with community consultation and engagement, are integral to responsible leading practice AMD management. While the collection and publication of data may be a legislative requirement, what is currently openly communicated often falls well short of leading practice. At present, there is very limited public reporting of AMD-related data by mining companies, particularly on how much sulfidic material is being produced and how effectively it is being managed.

10.2 Reporting frameworks

10.2.1 Environment and sustainability reporting

Potential frameworks for reporting the range of aspects related to AMD include:

- annual company sustainability reports—information on social, economic and environmental aspects of the mining operation (or the company as a whole)
- water monitoring plans and water-quality data interpreted in the context of the values to be protected, including data on seasonal and long-term trends
- rehabilitation and closure plans (for example, in line with DMP–EPA 2015) that include strategies for the identification, management and remediation of AMD
- community and stakeholder consultation/engagement—reports, fact sheets, information kits, presentations, meetings and dedicated websites[74]

74 Refer to the *Community engagement and development* leading practice handbook (DIIS 2016b) and resources from the ICMM.

- the Global Reporting Initiative (GRI)[75]—a global framework for corporate sustainability reporting—and the *Enduring value* framework for Australian mining companies (MCA 2005a).

10.2.2 Resource valuation reporting

A key part of the process for obtaining investment for the development of a new mineral resource is estimating its potential value using a standardised procedure. The most widely recognised standards for doing this are the 2012 Joint Ore Reserves Committee Code (JORC Code) and the 2005 Code for the Technical Assessment and Valuation of Mineral and Petroleum Assets and Securities for Independent Expert Reports (VALMIN Code). Both codes require that public reports prepared for the purpose of informing investors on exploration results, mineral resources or reserves and their associated valuation consider reclamation and rehabilitation liabilities in addition to other factors that can materially affect the economic value of the resource.

JORC Code

The JORC Code is a professional code of practice that sets minimum standards for the public reporting of minerals exploration results, mineral resources and ore reserves. The need to estimate deleterious co-occurring elements in the resource or other non-grade variables of economic significance (such as sulfur for AMD characterisation) and to characterise wastes are included. Environmental and social aspects are included under 'modifying factors'. Clause 12 of the code discusses how the modifying factors work with details provided in Table 3 in Section 4. Specifically, a 'competent person' (defined in the code) is required to address:

> the status of studies of potential environmental impacts of the mining and processing operation. Details of waste rock characterisation and the consideration of potential sites, status of design options considered and, where applicable, the status of approvals for process residue storage and WRDs should be reported.

VALMIN Code

The VALMIN Code requires tenement reports made by an independent expert to provide details of potential reclamation and rehabilitation liabilities/costs as well as geological risks and project uncertainties. Waste and tailings treatment/management is specifically addressed in the capex and opex provisions of Clause 92.

10.3 Mine closure and rehabilitation

Regulators and affected stakeholders should be involved as early as possible in the development of mine closure objectives and criteria for sites at risk of AMD. The production of agreed criteria is essential since, to a large extent, those criteria will determine the magnitude and cost of necessary engineering and other rehabilitation works. Failure to develop these criteria far enough in advance will invariably lead to protracted (and costly) delays in the ability to finalise rehabilitation works (Dowd 2005).

Many communities hold legitimate concerns about the long-term capacity of engineered structures to isolate sulfidic materials and minimise AMD. Extrapolation into the future is a very difficult technical issue.

75 http://www.globalreporting.org.

Consultation can help to identify locally appropriate strategies to manage AMD issues and ensure that economic, technical and regulatory constraints are satisfactorily addressed.

The reactivation of a brownfield site often requires re-engagement with the community on new project concepts to demonstrate how leading practice AMD management methods will be applied. A particular area of risk or conflict is original commitments that were made by the first owner of the project, since it is uncommon (unless the project has a very short life) for the company that first opens up a mine to also be there to close it at the end of its life.

New projects need to quantify and report on AMD risks from the past in order to demonstrate to communities and regulators how it will be possible to produce better outcomes from further mining, processing and/or water treatment activities. Similar considerations apply to developing and implementing rehabilitation plans for legacy sites.

11.0 CLOSING REMARKS AND FUTURE OPPORTUNITIES

Key messages

- The true economic value of a mineral resources project will only be realised by identifying AMD risk early and by preventing AMD formation before, during and after operations.
- Most of the knowledge that is needed to prevent AMD already exists. It is the implementation of that knowledge that is lagging. Leading practice principles for the management of AMD risks are not currently universally understood or applied.
- If the oxidation of at-risk sulfidic mine materials can be prevented from the start, there is a much higher likelihood of successfully and sustainably meeting operational water-quality and mine closure performance objectives.
- Improvements in approaches to the management of sulfidic waste and in the treatment of AMD will continue to be made, but this should not delay the implementation of current leading practice to prevent future long-term impacts.
- The mining company's social licence to operate is significantly influenced by impacts on water and soil quality during operations and after closure. AMD is the most significant environmental legacy and impact in the public's perception.
- Monitoring records need to be compiled and publicly communicated by the industry and regulators to robustly demonstrate the successful mitigation of AMD risk over the long term.

11.1 The big picture

The true value of a mineral resource in a sustainable economic development context will only be realised by preventing AMD before, during and after mining operations. If this is not properly addressed, there will continue to be large unprovisioned costs for closure towards the end of the project's life, along with ongoing risks to the environment and to the reputation of the industry.[76]

In addition to direct impacts on project finances, the cost to the industry of withdrawal of its social licence to operate as a result of perceptions about past performance can be substantial, including lengthy delays in project approvals or rejections of proposed developments.

11.2 Prevention and mitigation technologies

A very large amount of work has been done over the past three decades to understand the nature of AMD, to predict its occurrence and to develop practical solutions to manage its generation and release. Many of those technologies are described in this handbook, as they are now being implemented at full scale. There

76 Examples of direct costs incurred for retroactive AMD management are in Section 1

is now a substantial body of performance monitoring data demonstrating that the best way to prevent AMD is to implement an effective mine waste management strategy from the start of operations. In particular, if the oxidation of at-risk sulfidic material can be prevented, there is a much higher likelihood of meeting operational water-quality objectives and subsequently meeting closure performance criteria, both of which significantly reduce ongoing expenditure.

Much, if not most, of the knowledge that is needed to prevent AMD already exists. It is now a case of diligently and effectively using it, from the characterisation through to the placement and containment of sulfidic mine material. In the case of AMD, the proverb ‘a stitch in time saves nine’ is especially apt. As noted in various places in this handbook, leading practice principles for the management of AMD risks are currently not universally understood or applied.

Incremental improvements in assessing AMD risk will continue to be made as prediction methods are further refined and a better understanding is obtained of scaling from laboratory measurements to the field scale. Lessons will also continue to be learned about the design, installation, maintenance and long-term performance of cover systems.

While there is no single solution that will prevent AMD at all sites, it is clear that minimising the ingress of oxygen into unoxidised waste rock by ground-up methods of construction, combined with interlayer compaction to limit water flux, is probably the closest to a ‘universal’ solution for this AMD source. In the case of sulfidic tailings, maintaining saturation throughout the tailings mass is the most effective means for preventing oxidation over the long term. In drier climates, control of infiltration by store-and-release covers is likely be the most practical and effective means of limiting releases of oxidation products into the environment.

The desulfurisation of tailings has been shown to be a very effective technique, not only for reducing the legacy AMD risk from TSFs, but also for producing fine-grained low-sulfur material for use in cover systems. In many locations, the availability of sufficient fine-grained NAF material is one of the key factors limiting the effective management of AMD risk. Indeed, the acquisition of enough fine-grained borrow material from the area surrounding the mine can be curtailed or prevented on environmental grounds owing to the substantial collateral damage to the borrow area and the associated requirement for additional areas to be rehabilitated. There is scope for the much wider application of the desulfurisation of tailings by the industry.

Pit lakes are an increasing feature of the mine closure landscape for all mineral commodity types, in Australia and elsewhere. In common with the other areas of AMD risk management, knowledge of the behaviour of these water bodies has advanced substantially, particularly over the past decade. However, owing to the physical, chemical and biological complexity of such systems, much more work is needed to improve confidence in the long-term prediction of water quality for those cases where a self-sustaining, functional and regionally appropriate ecological system is required. Once again, this area would benefit from a greater number of well-monitored sites where this closure option has been proposed or implemented.

In underground mines, there are remaining issues involving the ongoing oxidation of exposed sulfides that lie above the recovered watertable. This applies especially to mines in elevated terrain with unsealed horizontal or inclined drives that express at the surface. In those cases, maintaining the underground atmosphere at very low oxygen concentrations has the potential to prevent sulfide oxidation and the discharge of poor-quality drainage. This approach (inert atmospheric systems) involves sealing off underground mine workings and developing a slight gas overpressure to prevent oxygen entry and minimise sulfide oxidation.

Emerging technologies and future opportunities that offer significant potential for AMD management, pending successful large-scale field trials, include:

- co-disposal of tailings with waste rock (paste rock), in which the fine-grained tailings fill up the void space in waste rock (Longo & Wilson 2007)
- reducing the acidity load in drainage water from WRDs by applying alkalinity-releasing (surface) covers that generate sufficient alkalinity to passivate the surfaces of the underlying sulfides
- geochemical engineering (see Glossary) of WRDs to reduce the formation of secondary acid-forming minerals such as jarosite and facilitate the formation of more stable mineral phases
- strategic addition of clays, when required, to the final 1–2 m of deposited tailings to significantly lower air-entry values
- identification and use of low-sulfur content NAF tailings or waste rock to be used as oxygen-scavenging barrier layers in the construction of waste containment facilities
- lower cost water treatment technologies (Lorax Environmental 2003) that remove sulfate at high loading rates to sufficiently low concentrations to meet increasingly stringent concentration discharge standards
- better use of in-pit blasting patterns and blast efficiencies to control the particle size and hence the surface area of reactive sulfidic waste and alkalinity-generating waste to maximise the effectiveness of geochemical controls within WRDs.

11.3 Monitoring, reporting and the social licence to operate

It is essential that more comprehensive operational and post-closure monitoring records are produced by the industry and regulators to robustly demonstrate the successful management of AMD at-risk sites over the longer term. Unfortunately, there are very few publicly available examples to demonstrate that mining practice is sustainable in this context. This situation is increasingly contributing to the lack of trust, and hence withdrawal of the social licence to operate, being exhibited by communities to proposals towards new mining projects in Australia and overseas. It is further compounded by the existence of many highly visible legacy sites that fuel the connection between poor AMD mitigation and management in the past and current social opposition to mining.

The types of monitoring needed are described in Section 9. As noted in Section 3, companies operating sites at risk of post-rehabilitation AMD should be prepared to conduct robust long-term post-rehabilitation monitoring programs, with appropriate financial and technical provisioning, and to undertake remedial works if and as required. The findings and implications from the monitoring programs need to be clearly communicated, as described in Section 10.

Even when mitigation measures have been incorporated into project operations, those measures have often not achieved their predicted (desired) performance objectives. For example, a comparison between predictions in EIS documents and performance has found that 64% of failure modes for predicted versus actual water quality at hard rock mines were due to a failure to meet performance objectives (Kuipers et al. 2006). This result indicates that there is still some way to go to achieving industry-wide leading practice management of AMD risk.

APPENDIX 1

Table A.1—Elements of an AMD monitoring program

Facility	Component	Parameters	Frequency[a]		Performance evaluation criteria
			Exploration / feasibility phase[b]	Operations phase	
General	Meteorology	Rainfall, evaporation, temperature etc.	Baseline: hourly	Hourly	n.a.
	Rock type	Lithology, weathering state, sulfide and carbonate content	Log all drill core, including waste volumes	Log all drill core	n.a.
	Hydrology—up-stream and downstream of site[cd]	Flow rate	Baseline: daily	Hourly	n.a.
	Surface water quality—upstream and downstream of site	General water quality parameters (field)	Baseline: quarterly	Daily/weekly: event-based	State/national water quality guidelines for ambient surface water (e.g. ANZECC–ARMCANZ 2000a). Baseline and upstream.
		Total suspended solids, acidity / alkalinity, major ions and ligands, metals (laboratory)	event-based	Weekly / monthly: event-based	
	Hydrogeology—upgradient and downgradient of site[cd]	Groundwater levels, flow rate and direction	Baseline: quarterly	Weekly/monthly	n.a.
		General water-quality parameters (field), including total suspended solids, acidity/alkalinity, major ions and ligands, metals/metalloids (laboratory)	Baseline: monthly/quarterly	Weekly/monthly	State/national water quality guidelines for groundwater. Baseline and upgradient.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	Baseline: quarterly/yearly	Monthly/quarterly	
	Social and cultural (e.g. downstream water use)	Downstream water uses (e.g. drinking, fishing/ aquaculture, irrigation/ farming, livestock, washing, bathing, small-scale mining, hydropower, recreation, cultural significance etc.)	Baseline: yearly	Quarterly/yearly	Baseline data on downstream water uses.

Facility	Component	Parameters	Frequency[a]		Performance evaluation criteria
			Exploration / feasibility phase[b]	Operations phase	
General	Vegetation (e.g. vegetated waste rock pile covers, other rehabilitated areas, natural vegetation adjacent to site)	Extent of vegetation cover, dieback or bare patches (if any), diversity of flora and avifauna	Baseline: quarterly/yearly	Monthly/quarterly	Baseline data on natural vegetation or rehabilitated areas.
	Aquatic fauna—upstream and downstream of site	Algae, macroinvertebrates, fish, larger vertebrates etc.	Baseline: half-yearly (seasonal) / yearly	Quarterly / half- yearly (seasonal) / yearly; event-based	Baseline data on downstream aquatic fauna.
	Site water balance and acidity balance	Flow rates/pump rates, acidity loads	Baseline	Daily	Sufficient but not excessive volume of water on site.
		Water levels and volumes in storage facilities	n.a.	Daily	
	Discharge points	Flow rates	n.a.	Daily: event-based	n.a.
		General water-quality parameters (field)	n.a.	Daily: event-based	State/national/ international water-quality guidelines for discharge water (e.g. IFC 2004). Mixing zones important.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.	Monthly/quarterly: event-based	
	Production geochemistry	Geochemical classification of soil/rock (static tests)	n.a.	As required for operational control (e.g. blast holes, face samples)	Modelled predictions.
		Geochemistry of mill tailings (static tests)	n.a.	As required	Modelled predictions.
WRDs and ore stockpiles	Waste rock and ore material	Waste rock and ore production rates, mass/volume of waste rock piles and ore stockpiles	Modelled predictions	Daily	Modelled data.
		Geochemical characterisation of lithologies (static and kinetic tests)	Baseline: as required	As required	n.a.
	WRDs (during dump construction or in brownfield dumps)	Pore-space oxygen concentrations and temperature (in situ)	Monthly	Monthly	Modelled data.
	Hydrology (surface water run-off and surface seepage)	Flow rates (surface water run-off, surface seepage)	n.a.	Weekly	n.a.

Facility	Component	Parameters	Frequency[a]		Performance evaluation criteria
			Exploration / feasibility phase[b]	Operations phase	
WRDs and ore stockpiles	Water quality (surface water run-off and surface seepage)	General water-quality parameters (field)	n.a.	Weekly	Baseline and upstream data. Predicted water quality.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.	Monthly	
	Hydrogeology (water in waste rock piles; groundwater upgradient, beneath and downgradient of piles)	Infiltration rates in waste rock piles (pore pressure / hydraulic / lysimeter data)	n.a.	Quarterly	Target/design infiltration rates.
		Water levels; volume of pore water in waste rock piles	n.a.	Monthly	n.a.
		Geophysical survey (e.g. electromagnetic, resistivity) to map subsurface conductivity and seepage flow pathways	As required	As required	n.a.
		General water-quality parameters (field)	n.a.	Monthly	State/national water-quality guidelines for groundwater. Baseline and upgradient.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.	Quarterly	

Facility	Component	Parameters	Frequency[a]		Performance evaluation criteria
			Exploration / feasibility phase[b]	Operations phase	
TSFs, tailings dams	Tailings material	Milling and tailing production rates, mass/volume transferred to TSFs	Modelled predictions	Weekly	Modelled data.
		Geochemical characterisation (static and kinetic tests)	Baseline: as required	As required	n.a.
	Hydrology (supernatant water)	Volume, water level, flow rate of tailings into facility, flow rate of decant pumps, spillway flow rates	n.a.	Daily	n.a.
	Hydrology (surface seepage)	Flow rate	n.a.	Weekly/monthly	n.a.
	Water quality (supernatant water and surface seepage)	General water-quality parameters (field)	n.a.	Weekly	Site-specific water-quality criteria (for onsite use) or discharge water-quality guidelines (e.g. IFC 2004).
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.	Monthly	
	Hydrogeology (pore water in tailings; groundwater upgradient, beneath and downgradient of TSFs)	Water levels	Baseline	Monthly	n.a.
		Geophysical survey (e.g. electromagnetic, resistivity) to map sub-surface conductivity and seepage flow pathways	As required	As required	
		General water-quality parameters (field)	Baseline	Monthly	State/national water-quality guidelines for groundwater. Baseline and upgradient data.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	Baseline	Quarterly	
	Unsaturated tailings	Pore-space oxygen concentrations and temperature (in situ)	Monthly	Monthly	n.a.

Facility	Component	Parameters	Frequency[a]		Performance evaluation criteria
			Exploration / feasibility phase[b]	Operations phase	
Pits / open cuts	Pit-wall material (groundwater cone of depression)	Lithology, talus mass/ volume of material exposed to oxygen	Modelled predictions	As required	Modelled data.
		Geochemical characterisation of lithologies (static and kinetic tests)	Baseline: as required	As required	n.a.
	Pit hydrology / stormwater	Dewatering pump flow rates	n.a.	Daily	n.a.
	Pit water quality	General water-quality parameters (field)	n.a.	Weekly	Water-quality criteria (for onsite use) or discharge water-quality guidelines (e.g. IFC 2004).
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.	Monthly	
	Pit hydrogeology (groundwater cone of depression)	Groundwater levels, flow rates (e.g. dewatering bores)	Modelled predictions	Weekly	Modelled data.
		General water-quality parameters (field)	Baseline	Weekly	Water-quality criteria (onsite use) or discharge water-quality guidelines (e.g. IFC 2004).
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	Baseline	Monthly	
Under-ground mines	Dewatered material (cone of depression)	Mass/volume of material exposed to oxygen	Modelled predictions	Monthly	Modelled data.
		Geochemical characterisation of lithologies (static and kinetic tests)	Baseline: as required	As required	n.a.
	Hydrogeology (groundwater cone of depression, groundwater quality)	Groundwater levels and flow rates	Baseline	Weekly	n.a.
		(dewatering bores)	Baseline	Weekly	Water-quality criteria (for onsite use) or discharge water-quality guidelines (e.g. IFC 2004).
		General water-quality parameters (field)	Baseline	Monthly	

Facility	Component	Parameters	Frequency[a]		Performance evaluation criteria
			Exploration / feasibility phase[b]	Operations phase	
Heap and dump leach piles	Ore material	Ore production rates, mass/ volume of ore in leach pad	Modelled predictions	Daily	Modelled data.
		Geochemical characterisation of lithologies (static and kinetic tests)	Baseline: as required	As required	n.a.
	Hydrology (surface water run-off and surface seepage)	Flow rates	n.a.	Daily	n.a.
	Water quality (surface water runoff and surface seepage)	General water-quality parameters (field)	n.a.	Weekly	Water-quality criteria (for onsite use). Predicted water quality.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.	Monthly	
	Hydrogeology (groundwater upgradient, beneath and downgradient of heap leach pad / leach piles)	Groundwater levels	Baseline	Weekly	n.a.
		Geophysical survey (e.g. electromagnetic) to map subsurface flow	As required	As required	n.a.
		General water-quality parameters (field)	Baseline	Weekly	State/national water-quality guidelines for groundwater. Baseline and upgradient.
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	Baseline	Monthly	
	Sulfide heaps (at end of life)	Pore-space oxygen concentrations and temperature (in situ)	Monthly	Monthly	n.a.
Other facilities	Hydrology (water storages, sediment basins etc.)	Flow rates	n.a.	Event-based: as required	n.a.
	Water quality (water storages, sediment basins)	General water-quality parameters (field)	n.a.	Event-based: as required	Water-quality criteria (onsite use) or discharge water-quality guidelines (e.g. IFC 2004).
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.		
	Road run-off / surface seepage water quality (e.g. haul roads, exploration roads)	General water-quality parameters (field)	n.a.	Event-based: as required	
		Total suspended solids, acidity/alkalinity, major ions and ligands, metals/ metalloids (laboratory)	n.a.		

n.a. = not applicable.

[a] Monitoring frequency for some locations may need to be higher during the wet season (and high-flow periods) and lower during the dry season (and low-flow periods). A higher frequency will also be required before/during offsite discharge (e.g. in the case of downstream surface water monitoring).

[b] Monitoring frequency during the exploration/feasibility phase will depend on the expected time before the start of operations.

[c] DIIS (2016g).

[d] MCA (1997).

GLOSSARY

ABATES	A software tool to assist with mine site water-quality management. It was developed to assist mining companies with acid base accounting and water-quality assessment and is available for free download at the Earth Systems website.[77]
Acid	A measure of hydrogen ion (H^+) concentration, generally expressed as pH. Acid is not equivalent to acidity *(see Acidity)*.
Acid base account (ABA)	An ABA evaluates the balance between acid-generating processes (oxidation of sulfide minerals) and acid-neutralising processes. It involves determining the acid-producing potential (APP) and the inherent acid-neutralising capacity (ANC) to determine the net acid-producing potential (NAPP) of a mined material. Each of these terms is defined below.
Acid drainage	A form of AMD characterised by low pH, elevated metal concentrations, high sulfate concentrations and high salinity.
Acidity	A measure of hydrogen ion (H^+) concentration and mineral (latent) acidity, generally expressed as mg/L $CaCO_3$ equivalent. Measured by titration in a laboratory or estimated from pH and water-quality data.
Acidity load	The product of acidity and flow rate, generally expressed as mass $CaCO_3$ equivalent per unit time.
Acidity load balance	For a mine site, takes into account water volumes and flow rates as well as acidity and incorporates all mine facilities that are potential sources of AMD, such as waste rock piles, ore stockpiles, TSFs, pits, underground workings, heap leach piles and mine construction materials.
Active treatment	A process in which chemicals or natural materials are added to AMD to improve water quality. Operator control can vary from relatively simple batch treatment to a sophisticated computerised treatment plant with multiple additives and detailed process monitoring and control.[78] Active treatment involves regular reagent and labour inputs for continued operation, compared with passive treatment (see below) that only requires occasional maintenance. Active treatment systems can be engineered to deal with any acidity, flow rate and acidity load.
ADTI	Acid Drainage Technology Initiative.[79]

77 http://www.earthsystems.com.au/tools.htm.

78 http://www.inap.com.au.

79 http://www.unr.edu/mines/adti/.

Alkaline cover	A soil cover (for example, a water-shedding or store-and-release cover) that has an alkalinity-generating (acid-neutralising) component deployed above, within or at the base of the cover. The aim is to minimise infiltration and ensure that any water that migrates through the cover contains substantial alkalinity.
AMD	Acid and metalliferous drainage (see detailed definition in Section 2.1).
AMDTreat	Software produced by the Office of Surface Mining Reclamation and Enforcement, US Department of the Interior, that can be used to predict and model AMD treatment costs. The software provides many different treatment options for passive and active treatment systems. Available for free download.[80]
AMIRA	AMIRA International Limited.[81]
ANC	Acid-neutralising capacity, expressed as kg H_2SO_4 equivalent per tonne.
ANSTO	Australian Nuclear Science and Technology Organisation.
APP	Acid-producing potential, expressed as kg H_2SO_4 per tonne. Also referred to as acid-generating potential (AGP).)
Blending	Mixing of potentially acid-generating mine wastes with sufficient quantities of alkaline materials to create a composite material in which any acid produced is consumed in situ by surrounding alkaline materials.
Block model	A three-dimensional cellular model of the distribution of ore and waste materials with different geochemical properties. See also *Grid/layer model.*
Chlorite	$(Mg,Fe)_3(Si,Al)_4O_{10}(OH)_2 \bullet (Mg,Fe)_3(OH)_6$—is a member of the phyllosilicate family of minerals.
Co-disposal	Combined disposal of coarse-grained (waste/rejects) and fine-grained (tailings) waste streams; used extensively in the Australian coal industry.
Domains	Land management units within a mine site, usually with similar geophysical characteristics, such as mine pit, WRD, TSF, process plant area, ore stockpile.
Environmental impact assessment (EIA)	In this handbook, also refers to environmental impact statement (EIS), environment effects statement etc.

80 http://www.amdtreat.osmre.gov.

81 http://www.amira.com.au.

Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act)	The Australian Government's central piece of environmental legislation. It provides a legal framework to protect and manage nationally and internationally important flora, fauna, ecological communities and heritage places—defined in the EPBC Act as matters of national environmental significance.[82]
Foot wall	For a dipping fault, the block positioned under the fault.
GARD Guide	The Global Acid Rock Drainage Guide (2009).[83]
Geochemical engineering	Engineering using geochemistry to contribute to key design elements of aspects of waste rock dumps (for example, limestone blending and optimal moisture content of layers to minimise the rate of sulfide oxidation).
GRI	Global Reporting Initiative.[84]
Grid/layer model	A two-dimensional model of the distribution of ore and waste materials with different geochemical properties. Seen also *Block model.*
Hanging wall	For a dipping fault, the block positioned over the fault.
Heap leach spent ore	Material remaining after the recovery of metals and some soluble constituents through heap leaching and heap rinsing of ores.
International Council on Mining and Metals (ICMM)	Founded in 2001 to improve sustainable development performance in the mining and metals industry.[85]
International Finance Corporation (IFC)	A member of the World Bank Group; the largest global development institution focused exclusively on the private sector in developing countries.[86]
International Network for Acid Prevention (INAP)	An international consortium of mining industry partners focused on improving the management of AMD-generating waste.[87]

82 http://www.environment.gov.au/epbc.

83 http://www.gardguide.com/images/5/5f/TheGlobalAcidRockDrainageGuide.pdf.

84 http://www.globalreporting.org/Home.

85 http://www.icmm.com.

86 http://www.ifc.org.

87 http://www.inap.com.au/.

Kinetic test	Procedure used to measure the magnitude and/or effects of dynamic processes, including reaction rates (such as sulfide oxidation and acid generation), material alteration and drainage chemistry and loadings that result from weathering. Unlike static tests, kinetic tests measure the behaviour of a sample over time.
Lag time	Time delay between the disturbance or exposure of acid-generating materials and the onset of acidic drainage.
Lithology	A description of a rock's physical characteristics visible at outcrop, in hand or core samples or with low magnification microscopy, such as colour, texture, grain size and composition.
Low-grade ore stockpile	Material that has been mined and stockpiled, with sufficient value to warrant processing, either when blended with higher grade rock or after higher grade ore is exhausted, but that is often left as 'waste' at the end of a mine's operating life.
Metalliferous drainage	A form of AMD characterised by near-neutral pH, elevated heavy metal concentrations and high sulfate salinity.
Minerals Council of Australia (MCA)	The peak industry body of Australia's exploration, mining and minerals processing industry.[88]
Mine Environment Neutral Drainage Program (MEND)	Program funded by the Canadian Government and industry partners.[89]
Muscovite	$KAl_2(AlSi_3)O_{10}(OH)_2$—a member of the mica group of the phyllosilicate family of minerals.
Net acid-generation (NAG) test	Also referred to as 'single addition NAG test'. Uses hydrogen peroxide to oxidise any sulfides in a sample, then any acid generated during oxidation that is consumed by neutralising components in the sample. Any remaining acidity is expressed as kg H_2SO_4 per tonne. A 'sequential NAG test' involves a series of NAG tests on a sample. This may be required if the sample cannot be fully oxidised using the standard single addition NAG test. The kinetic NAG test monitors the temperature and pH of the NAG solution during reaction with hydrogen peroxide and can provide information on the lag time preceding acid generation in the field (AMIRA 2002).
Net acid-producing potential (NAPP)	Calculated by subtracting acid-neutralising capacity (ANC) from acid-producing potential (APP); expressed as kg H_2SO_4 per tonne.
NEPM	National Environment Protection Measure.
NPI	National Pollutant Inventory.

88 http://www.minerals.org.au.

89 http://www.nrcan.gc.ca/ms/canmet-mtb/mmsl-lmsm/mend/default_e.htm.

PADRE	Partnership for Acid Drainage Remediation in Europe.[90]
Potentially acid-forming (PAF) material	Material identified by the results from static and kinetic tests.
Passive treatment	Treatment using systems that require little input of external energy. Typically include both biological and non-biological components staged to treat AMD. Examples include wetlands and permeable reactive barriers. Best suited to AMD with low acid loadings. Also see *Active treatment.*
Phyllic alteration	An alteration zone in a permeable rock that has been affected by circulation of hydrothermal fluids. Commonly seen in copper porphyry ore deposits in calc-alkaline rocks. Characterised by the assemblage of quartz + sericite + pyrite.
Precautionary principle	Where the scientific evidence is uncertain about the likelihood of adverse impact, decision-makers should take action to limit continued environmental damage and should err on the side of caution when evaluating proposals that may have a serious or irreversible impact on the environment (See EPBC Act, Schedule 3).
Saline drainage	type of AMD characterised by high sulfate salinity but near-neutral pH and low concentrations of heavy metals.
Soil cover	One or more layers of soil-like materials intended to limit the percolation of rainfall or the ingress of oxygen, or both, into AMD-generating materials.
Static test	Procedure for characterising the physical, chemical, or biological status of a sample at one point in time. Includes measurements of mineral and chemical composition and the analyses required for acid base accounts.
Store-and-release cover	Cover system designed to minimise the infiltration of water to underlying materials by incorporating materials with high water storage capacity and plants with high rates of evapotranspiration.
Tailings	Finely ground materials from which the desired minerals have been largely extracted. Approximately 98% of the material mined for processing at metal mines is discharged as tailings. At coalmines, tailings comprise the coarse and fine rejects from the coal washery.
Tailings dam	Facility designed for the storage of saturated tailings material and supernatant water produced during ore processing. Tailings dams, unlike tailings storage facilities, are designed as competent water-holding structures.
Tailings storage facility (TSF)	Facility designed for the storage of unsaturated tailings material produced during ore processing. Unlike tailings dams, TSFs are not suitable for storing supernatant water.

90 http://www.padre.imwa.info/.

Waste rock	Material such as soils, barren or uneconomic mineralised rock that surrounds a mineral or coal ore body and must be removed in order to mine the ore. Generally referred to as waste rock in metalliferous mines and overburden, interburden, interseam or spoil in coalmines.
Waste rock dump (WRD)	Facility constructed to store waste rock.
Water cover	Layer of surface water (for example, in a tailings storage facility or pit) or groundwater (for example, in a backfilled pit) intended to limit the ingress of oxygen into AMD-generating materials.

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