

Forecasting long term water quality at closure for current mining operations

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Abstract

There are many operational mine sites worldwide undertaking investigations to determine water quality at closure where site waters are impacted by acid and metalliferous drainage (AMD). Estimating final water quality and flow rate from Waste Rock Dumps (WRDs) and Tailings Storage Facilities (TSF) is critical for closure planning. Such data needs to be compared with: regulatory limits for closure; stakeholder expectations; and, any anticipated treatment options, treatment longevity, and treatment costs.

There are often numerous seeps and streams impacted by AMD, which can be used to determine typical water quality, solubility constraints, flow rates, contaminant loads, and thus source terms for WRD/TSF drainage. This generally requires the development of a conceptual model for the catchment based on flow rates and water quality, which is then coupled with geochemical modelling.

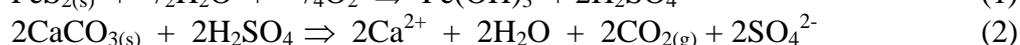
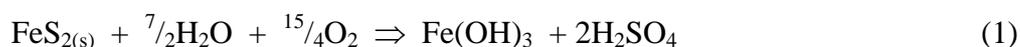
Often WRD/TSF drainage water quality is solubility controlled, particularly for older structures associated with the formation of melanterite-type soluble acidity; jarosite-type sparingly-soluble acidity; and other secondary phases such as gypsum or metal salts. Geochemical test work can confirm the presence of these minerals, which are then incorporated into the modelling process.

This paper focuses on understanding water quality at closure and into the long term. Three key phases of water quality after closure need to be considered: the draindown water quality phase; the transition water quality phase; and, the long-term water quality phase. During the draindown phase, which occurs after cover system installation, the water quality is generally assumed to be comparable to the current derived source term for the WRD. The duration of the draindown phase is determined by numerical modelling as the system equilibrates to a lower net percolation rate. The transition phase water quality is due to the mobilisation of stored oxidation products; these products being the reservoir for ongoing acidity and metals load. Long term water quality can be determined by forward reaction path modelling, or by a first-principles approach using key mineral dissolution and oxidation kinetics. These three phases then provide a model for estimating long term water quality after closure, which can be applied to closure planning.

Keywords: Acid and metalliferous drainage, AMD, water quality, mine closure.

Introduction

Acid and metalliferous drainage (AMD) is the result of sulfide mineral oxidation, typically pyrite, where the pyrite reacts with oxygen and water during the mining process to produce acidity and metal precipitates such as Fe oxyhydroxides (Equation 1). This can lead to low pH (< pH 4.5) in streams impacted by acid mine drainage. The acidity generated by pyrite oxidation can be neutralised by materials such as limestone (CaCO₃) as shown in Equation 2.



Waste rock may contain acid forming sulfides (*e.g.*, pyrite) and acid neutralising carbonate minerals. If the carbonate content is sufficient and readily available to neutralise acidity, low

pH drainage (< 4.5) may not occur. In circum-neutral drainage, certain metals may still remain in solution and produce metalliferous drainage.

To obtain regulatory consent for a mining project to proceed, or as part of the mine's AMD management plan, there is often a requirement for geochemical classification of the overburden potential to produce AMD. Geochemical classification of waste rock overburden involves a variety of acid-base accounting (ABA) techniques. The industry standard approach is to determine the net acid production potential (NAPP) and the Net Acid Generation (NAG) pH and acidity of the sample (*e.g.*, AMIRA, 2002). These tests focus on the potential acidity, which is often associated with unoxidised sulfides. Comparative testing (*e.g.* NAPP vs NAG pH) is displayed graphically, although often there are uncertain samples and other classification processes may need to be considered (*e.g.*, Olds *et al.*, 2015). A number of other tests are commonly used for AMD geochemical classification, including paste pH, column leach tests, humidity cell trials, and a variety of field trials (from kilogram scale to thousands of tonnes) to help confirm classification processes and metalliferous drainage chemistry.

For operating mine sites, legacy sites, or sites with substantial oxidised/weathered overburden, additional tests that determine the quantity of stored acidic/metalliferous oxidation products should be undertaken. These tests include quantification of the soluble stored acidity/metals by 1M KCl digestion and 4M HCl digestion to determine sparingly soluble acidity/metals (*e.g.*, Ahern *et al.*, 2004). Key geochemical indicators of stored acidity from conventional ABA testing is low paste pH values < 5.5, negative ANC values, high salinity (*e.g.*, as measured by EC), and the observation of secondary minerals in laboratory tests.

Understanding the proportion of stored and potential acidity/metalliferous oxidation products is a key aspect of understanding the longevity of AMD impacted waters at closure, in the longer term, and also for treatment of AMD. Such acidity/metalliferous oxidation products create the reservoir of contaminants that lead to poor water quality. Exclusion of oxygen ingress will mostly prevent the ongoing oxidation of sulfide minerals. Exclusion of water, the transport medium, will prevent the mobilisation of stored acidity/metalliferous oxidation products created from previous sulfide oxidation.

Geochemical conceptual models

For many operational mine sites a key objective for closure planning is determination of water quality at closure and over the longer term associated with geochemical maturity of the site. Waste Rock Dumps (WRDs) and Tailings Storage Facilities (TSFs) are typically important components to consider. These data need to be compared with regulatory limits for closure; stakeholder expectations; and be considered in regards to anticipated treatment options, treatment longevity, and treatment costs. Even for greenfield sites, given the nature of the resources industry, it is likely that mine plans will change during the operational phase from those originally permitted, resulting in modifications to the closure plan. Analysis of site-based empirical data (such as water quality monitoring databases) to derive water quality models for closure and beyond can assist in developing closure plans and management strategies.

For operational sites with AMD issues there are often numerous seeps and streams impacted by AMD, which can be used to determine typical water quality, solubility constraints, flow rates, and contaminant loads from their parent structures (*e.g.*, the WRD, TSF, etc.). This

generally requires the development of a conceptual model for the catchment to determine source terms (water quality) from the various site components (WRD, TSF, etc.), which are based on flow rates and water quality. Geochemical processes can then be incorporated into the model to determine long term water quality trends.

This paper presents an approach that considers three key phases of closure water quality including the draindown water quality phase; the transition water quality phase; and the long-term water quality phase. The process for estimating post closure water quality is site specific. The quantity of flow rate and water quality data collected across a site also affects modelling methodologies.

Phase 1: Draindown

Water quality during the draindown period is expected to be similar to that at mine closure, although there may be small changes in flow regimes. Source terms for modelling the draindown period are therefore derived from site flow rates and water quality monitoring databases in the years preceding closure. In many instances a waterway, or seep will be the product of a number of water source types such as groundwater, clean surface waters, and AMD point sources (underground workings, TSFs, and WRDs). To determine the water quality from a discrete component of a mixed water system (that cannot be measured directly) requires inverse geochemical modelling using the conceptual model as the blue print.

Draindown analysis

The duration of the draindown phase and net percolation rates are determined by numerical modelling (*e.g.*, SEEP/W). Draindown occurs as a function of a change from a higher permeability dump surface to a lower permeability surface after cover installation as the system equilibrates to the new net percolation rate. Draindown duration may be affected by changes in the water table, for instance, due to restoration of streams or pit lake flooding. The modelled flow regimes at the end of the draindown period are generally expected to remain constant over the long term, *i.e.*, the site is hydrologically stable at the end of the draindown period.

Inverse geochemical modelling

Inverse geochemical modelling provides the water quality (source term) for the component of interest (*e.g.*, the WRD) that is missing from the conceptual model and is completed in a step-wise manner using empirical data where possible. This process essentially fills data gaps in site monitoring databases or models that would otherwise be managed using assumptions or simplification. Inverse geochemical modelling involves:

- (1) Assign flows to all components of the conceptual model based on measured or calculated values.
- (2) Assign water quality to each component of the model based on measured values or estimated values where appropriate.
- (3) Determine loads for each component of the model and by inverse calculation allocate remaining load to the missing component, *i.e.*, the WRD net percolation (NP), and based on NP volumes, derive water quality.

- (4) Complete geochemical modelling of the derived water quality to estimate likely solubility constraints and compare to any mineralogical data (to validate the model outputs) and determine the final source term for the WRD.

Often the derived source terms are considered constant for modelling purposes during the draindown phase. A constant concentration assumption can be validated by analysis of flow rate variability from the component (WRD or TSF) and water quality.

Phase 2: Transition

The transition period is defined as the time it takes for the WRD or TSF to reach geochemical maturity. Contaminant loads over this period are derived largely from mobilisation of stored acidity/metalliferous oxidation products. Understanding the proportions of stored versus potential acidity/metalliferous oxidation products is critical for understanding the longevity of AMD impacted waters in the medium and longer term as it provides the reservoir of contaminants that can be released. This reservoir of oxidation products can, of course, be added to by ongoing oxygen ingress and subsequent sulfide mineral oxidation, and should be included in long term models when necessary.

Mineralogical analysis: Stored and potential acidic/metalliferous oxidation products

A key parameter for determining water quality after closure is quantifying mineralogy type and abundance within the WRD or TSF. Specifically this includes potential sources of acidity and metals from sulfide minerals that will become available due to oxidation (oxygen ingress) and stored acidity and metalliferous oxidation products that will become mobilised with net percolation (flow).

Derivation of potential sulfide and stored acidity/metalliferous oxidation products is best considered on a kg/tonne basis. Furthermore, the proportions of stored and potential acidity/metalliferous oxidation products dictate closure AMD management methodologies. For instance, components with significant concentrations of stored oxidation products should be managed by limiting net percolation to reduce loads to the receiving environment.

Flow dominated contaminant loads

Often WRD or TSF drainage water quality is solubility controlled, particularly for an older WRD or TSF where water quality is associated with the formation of stored oxidation products including melanterite-type soluble acidity; jarosite-type sparingly soluble acidity; and other secondary phases such as gypsum and metal salts. The solubility limits control the rate that stored acidity and metalliferous oxidation products are mobilised from the WRD and TSF, and therefore define the duration of the transition water quality phase.

Solubility controlled systems are easily identified by assessment of flow rate versus contaminant load. A solubility controlled system will typically have an increasing contaminant load with increasing flow rate, which means that with more flow (*i.e.*, net percolation) there are more stored oxidation products flushed from the system. Solubility controlled systems are observed at both acid-forming sites and non-acid forming metalliferous drainage sites.

Examples of solubility controlled systems are shown in Fig. 1. Fanny Creek (Figs. 1a, b) drains a side-cast open coal operation (Island Block Coal Mine). There is generally a constant acidity concentration with increasing flow, which is represented by an increasing acidity load with flow rate. This confirms the acidity load to the receiving environment is constrained by NP through the waste rock dump. Acidity load could therefore be reduced by minimising NP.

Data presented for the St Patrick's Stream, Stockton Coal Mine also demonstrates an increasing acidity load with increasing flow rate (Fig. 1d). It is interesting to note the logarithmic nature of concentration (mg/L) in the stream (Fig. 1c), with the acidity concentration decreasing with increasing flow rate. This is likely to be a function of increasing uncontaminated surface runoff associated with increasing rainfall during any storm event, *i.e.*, simple dilution. Inverse geochemical modelling of components upstream of the St Patrick's Stream may identify the source of the contaminant load. If the source is a seep it is likely to have a linear acidity load to flow rate correlation.

A number of underdrains have been installed at the Globe-Progress Gold Mine beneath the Devil's WRD and the Fossickers TSF. These systems collect drainage (net percolation) that has passed through the waste rock or tailings and present a strong correlation between contaminant load and flow rate (Fig. 1f). In this example, sulfate concentrations are reasonably constant for each component (Fig. 1e) and each site appears to have a relatively constant flow rate, although the Devils Waste Rock Stack (WRS) has the greatest range in flow (2-3 orders of magnitude). Analysis of flow rate versus sulfate load for all these structures provides a robust correlation, which becomes an excellent tool for subsequent analysis and closure planning.

The strong correlations presented in Fig. 1 provide confidence in the data for each component and evidence that the system is solubility controlled. Data indicate contaminant loads are a function of NP through waste rock or tailings and are likely to persist until the stored acidity/metalliferous oxidation products have been flushed from the system. Such datasets also provide confidence in the generation of source terms derived from inverse geochemical modelling using seepage water quality and flow monitoring data. For instance, concentration data shown for Globe Progress Mine underdrains (Fig. 1e), can be used to provide a robust source term based on median or average data for WRD and TSF seepage water quality.

The duration of poor water quality from a WRD or TSF is often unknown unless the mine site understands the mineralogy and ABA characteristics of its waste rock and tailings and has developed a conceptual flow model for the site. By adopting the derived source term for WRD or TSF seepage water quality (as validated by assessment of load versus flow rates) and then forecasting future flow rates through the structure (using modelled or measured net percolation rates) a model can be created for water quality over the transition phase. At some point in the future the solubility controlled water quality condition should change to a more geochemically stable water quality condition when the mobile stored oxidation products have been flushed by NP from the system. For this to occur the concentration of contaminants (mg/L) in drainage waters must decrease as the structure achieves geochemical maturity.

In a situation where there is minimal accumulation of stored acidity/metalliferous oxidation products, at the end of the draindown period the contaminant load could be controlled by oxygen flux and therefore oxidation of sulfides. The transition phase would therefore be bypassed with the site moving straight to the long term water quality phase associated with geochemical maturity of the site.

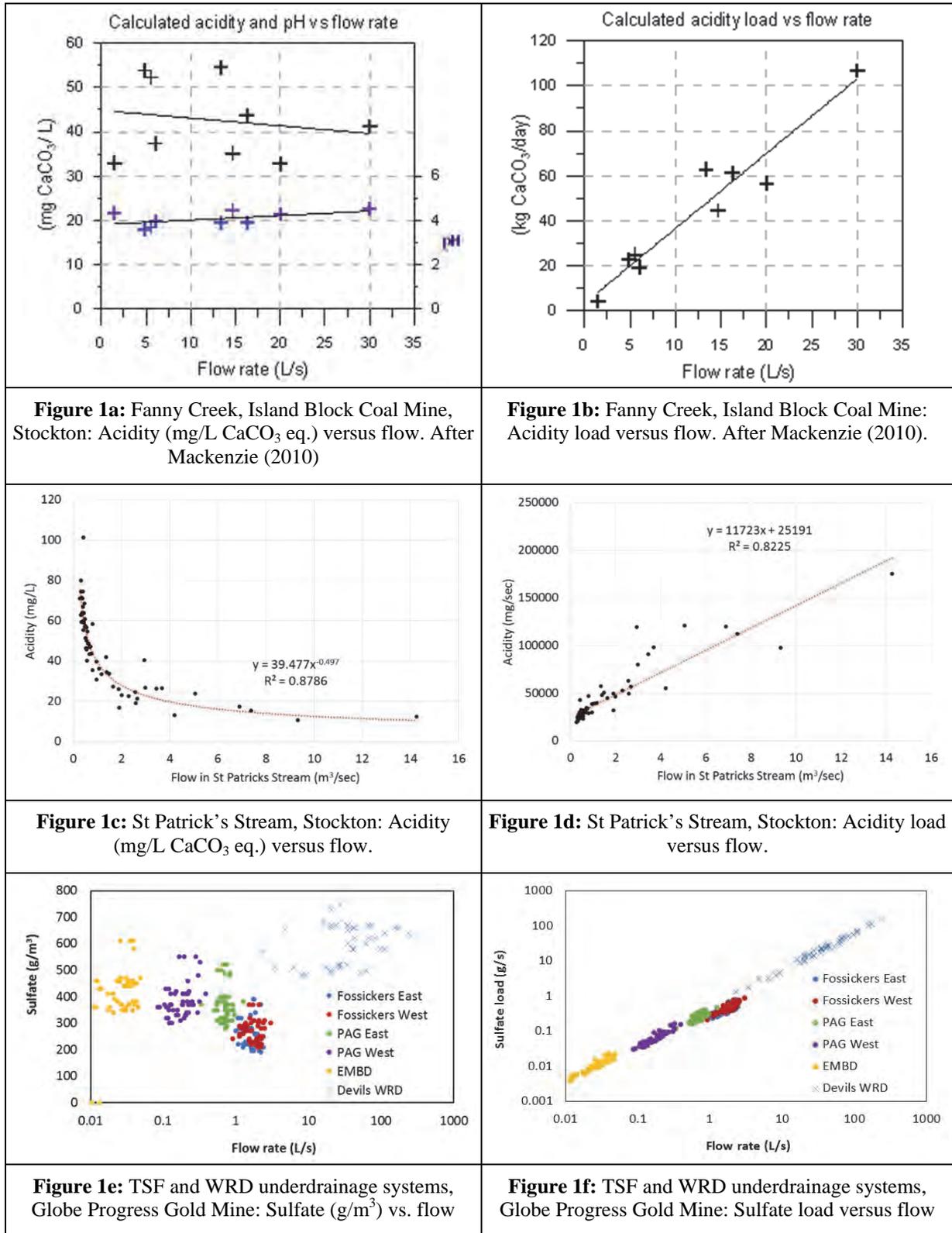


Figure 1: Examples of solubility controlled systems.

Phase 3: Long term

For the purposes of this paper the long term water quality is defined as the point in time when the system achieves geochemical maturity. At this stage oxygen ingress rates will have

stabilised and most stored oxidation products will have been flushed from the system. It could be suggested that geochemical maturity represents a balance where the contaminant load created due to oxygen flux into the system is approximately equal to the contaminant load discharging from the structure and reporting to the receiving environment. Long term water quality can be determined by forward reaction path modelling or by a first-principles approach using key mineral dissolution kinetics.

Water quality modelling

Geochemical computer modelling may utilise software such as Geochemist's WorkBench (GWB) where key reactive mineralogy is included in the model with dissolution/oxidation rates assigned from peer reviewed literature. Geochemical modelling should consider aspects such as oxygen flux, net percolation, and the abundances of sulfide minerals, carbonate minerals, sparingly soluble minerals (jarosite and alunite), and reactive silicate minerals. Such techniques are relatively common for determining long term water quality.

A first principles approach can also be undertaken as another method to estimate long-term water quality. For AMD sites where the oxygen flux is the key driver to poor water quality a numerical calculation can be undertaken to determine oxygen flux, and therefore sulfide oxidation rates and acidity generation assuming 100% consumption of oxygen by equation 1. Metal and acidity loads can then be compared against mineralogical and ABA data to estimate key water quality parameters such as annual acidity/metal loads and acidity/metal concentrations.

In these modelling processes soluble minerals such as melanterite and other soluble salts/metals, which can be difficult to determine analytically and quantify, can be managed by simple numerical analysis as part of the long term water model having been quantified by ABA methods (*e.g.*, Ahern *et al.*, 2004). Sparingly-soluble minerals such as alunite and jarosite can be included in the geochemical computer model to determine effects on the long term water quality.

Example

The following section provides an example for how a simplistic long term water quality model can be developed based on an understanding of the potential and stored acidity load; drainage water quality and flow rates, and long term water quality forecasts after geochemical maturity. The same process can be applied to a metalliferous drainage site.

As this is a simple example, the water quality only considers Fe, Al, and pH, which are the main constituents of acidity load; trace metals and other dissolved parameters have not been considered but could be incorporated in a similar manner.

Modelling example

Site A has a 5 Mt WRD with a mean MPA value of 30 kg H₂SO₄ eq./tonne based on total S content, which is assumed to be present as pyrite. MPA based on sulfide S content is 24 kg H₂SO₄ eq./tonne, which is the potential acidity of the WRD. The balance of the total sulfur is in the form of soluble acidity represented by melanterite-type acidity of 6 kg H₂SO₄ eq./tonne. This means that there is 120,000 t H₂SO₄ eq. of potential acidity and 30,000 t H₂SO₄ eq. of

stored acidity within the WRD. Such data would be reasonable for a waste rock having fine grained reactive pyrite and no ANC. ABA data is summarised in Table 1.

Table 1. Acid Base Accounting data for the Site A Waste Rock Dump.

Parameter	Quantity	Units
Waste rock	5,000,000	Tonnes
Total S	1	wt%
MPA (based on Total S)	30	kg H ₂ SO ₄ /tonne
MPA (based on Sulfide S)	24	kg H ₂ SO ₄ /tonne
Stored soluble acidity	6	kg H ₂ SO ₄ /tonne
ANC	0	kg H ₂ SO ₄ /tonne
Total potential acidity	120,000	Tonnes
Total stored acidity	30,000	Tonnes

Based on the presence of significant unreacted potential acidity the site should plan to restrict oxygen ingress to prevent ongoing pyrite oxidation by installing a cover system. This will reduce acidity generation in the longer term. However, acidity will continue to report to the receiving environment as a function of net percolation due to the stored acidic oxidation products. For this example, following installation of the cover system, it is expected the net percolation will reduce from 40 L/s to 30 L/s; it is also assumed that oxygen ingress thereafter is zero.

Draindown phase

Numerical modelling is undertaken typically using computer software such as SEEP/W to determine the draindown period. For this example a draindown period of 20 years was used. At mine closure water quality is poor being Fe (50 mg/L); Al (25 mg/L), a pH of 2.5, and a calculated acidity of 430 mg/L H₂SO₄ eq. Based on a flow rate of 40 L/s this equates to an annual acidity load of ~550 t H₂SO₄ eq./yr. For modelling purposes it is assumed that this acidity load will remain for almost the entire draindown phase. The cover system was assumed to completely eliminate oxygen flux, so the entire ~550 t H₂SO₄ eq./yr acidity load is derived from stored oxidation products.

Long term water quality phase

The long term water quality is assumed to have a pH of 5.5 and an acidity < 1 mg H₂SO₄ eq./L. Thus, the concentration of Lewis acids (Fe and Al) will be low. In practise, geochemical modelling to derive actual long term water quality would use calculated oxygen flux rates, mineralogy, etc. but has not been done for this example.

Transition phase

No model can remove more acidity/metal load than is present as stored acidity/metalliferous oxidation products within the WRD or TSF, this being the reservoir. Such oxidation products have to be present to be mobilised and transported by net percolation. Furthermore, it is generally expected that not all these oxidation products will be removed from the WRD or TSF. Generally 20 - 40% of stored acidity/metals might not report in the transition phase as a

contaminant load, but rather over the very long term such that their contribution to water quality is negligible.

In this example there is 30,000 t H₂SO₄ eq. of stored acidity; of this only 23,000 t H₂SO₄ eq. will be removed by net percolation during the transition phase (~75%). This means approximately 25% of the stored acidity/metals is considered immobile. The model is thus constrained to a reservoir of 23,000 t H₂SO₄ eq. Furthermore, the maximum annual acid load the model can use is 550 t H₂SO₄ eq./yr, due to the closure acidity concentration of 430 mg H₂SO₄ eq./L equivalent.

Forecasting a model using a constant acidity yield of 550 t H₂SO₄ eq./yr would see a step change in acidity load at approximately 40 years after closure and a step change in concentration from 430 to 0 mg H₂SO₄ eq./L following total exhaustion of the stored acidity. Such dramatic changes are not natural, and thus this is not a reasonable model. However, it does provide a minimum time frame for the model.

A gradual improvement in water quality during the transition phase is more likely. This will be a function of flow path variability, flow path flushing, and a decreasing quantity of acidic salts being dissolved during the transition phase. Decreases in acidity (mg H₂SO₄ eq./L) have been reported as being up to 3.3% per year over a 30+ year period for underground coal mines (*e.g.*, Mack *et al.*, (2010) and references therein) and such models could be applied to WRD's and TSF. However, this would also result in a step change in acidity load at approximately 60 years from ~130 to 0 mg H₂SO₄ eq./L as the reservoir of oxidation products are consumed. This is also not natural and would not occur.

Generally, a reasonable geochemical model can be developed using the minimum timeframe for removal of the stored acidity (being ~40 years in this example) and the maximum stored acidity load that can be removed (being 23,000 t H₂SO₄ eq. in this example) integrated with industry accepted geochemical processes such as the inclusion of buffering plateau due to Lewis acids (Fe and Al in this model), all of which consume acidity and restrains the system longevity.

Example summary

To provide a graphical summary for the Site A example a model was constructed using the data (Fig. 2), which shows the three key phases of long term water quality after closure. The draindown phase lasts for 20 years, which is followed by a transition phase of 70 years until pH 5.5 and acidity being < 1 mg CaCO₃ eq./L is achieved.

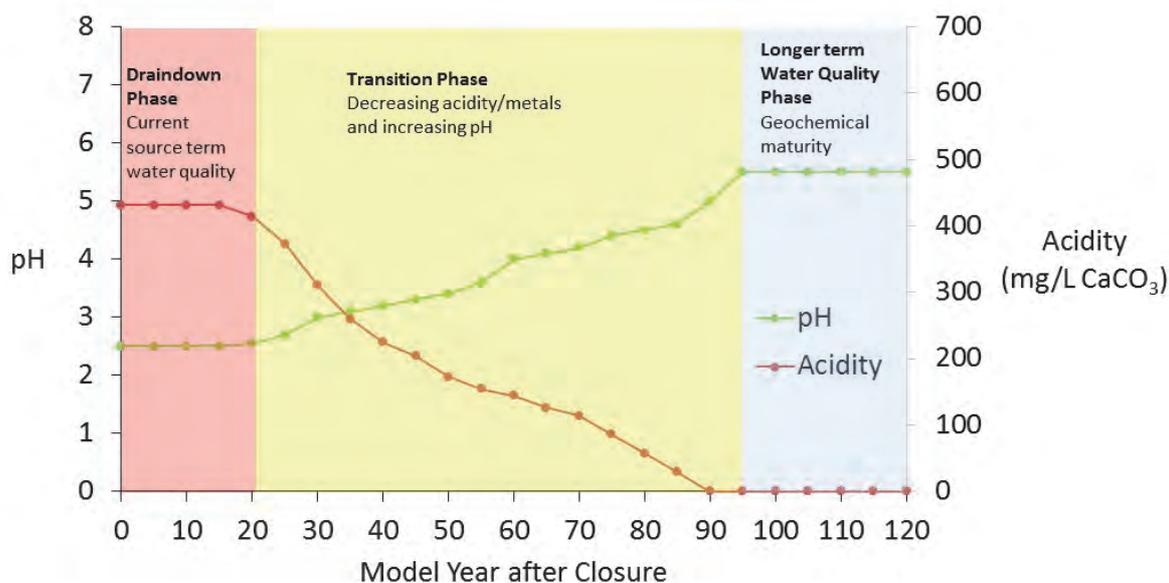


Figure 2. Example long term water quality model. Model commences at closure after cover installation.

Conclusions

This paper has presented a process to create a conceptual model for a mine site to determine water quality for WRDs and TSFs at mine closure and over the longer term based on empirical monitoring data. There are many ways to model long term water quality and all have merits. This approach, based on an estimation of stored oxidation products, inverse geochemical modelling to develop source terms for drainage waters, and numerical modelling provides a simple approach that is reasonable given the many uncertainties in forecasting long term water trends for mine sites after closure.

As shown by examples presented in this paper, stored acidity/metalliferous oxidation products at both acid and non-acid forming mine sites report to the receiving environment as a function of flow rate (net percolation) demonstrating solubility controls within the WRD or TSF. Understanding and incorporating such data is critical in developing a long term water quality model. Geochemical ABA testing provides an indication of oxidation product abundance.

In a solubility controlled system it is reasonable to assume that current water quality will persist until the end of the draindown phase. Following that a dilution effect will occur as the remaining stored oxidation products are flushed from the system and clean water pathways increase in proportion. At the end of this transition phase contaminant loads will have stabilised and remain constant over the long term water quality phase.

Such models can be used to determine if, and what, water treatment options are required at closure and for how long. This allows the operator to forecast acidity loads post closure and determine when a transition from active to passive treatment technologies is likely to occur. Such data is invaluable for net present analysis of closure options.

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