NEW ZEALAND MINERALS SECTOR ENVIRONMENTAL FRAMEWORK

Appendices

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Appendix A: INTRODUCTION

The following section provides an overview of regulatory requirements associated with mining operations in New Zealand, particularly as they relate to potential environmental impacts on streams, and to the West Coast and Southland. This information is not intended to replace regional council or Department of Conservation (DOC) advice, and all potential applicants are advised to also contact the relevant council and DOC to discuss their specific application. Further, the information provided in this document is correct as at September 2014, and users are advised to check with relevant councils and DOC for any updated requirements.

Mining is primarily regulated by requirements under the Crown Minerals Act (1991) and the Resource Management Act (1991) and amendments (2009). The key agencies responsible for regulating mining activities are NZ Petroleum and Minerals, DOC, and regional and district councils. Three types of permits are required prior to mining operations proceeding:

- A permit granted under the Crown Minerals Act
- An access arrangement negotiated with all landowners and occupiers; for land administered by DOC, an access arrangement is required
- Resource consents (e.g. use of land and water, discharges to water, air), which are granted by district and regional councils.

While permits or licenses granted under the Crown Minerals Act are largely related to the economic aspects of mining, granting of access (by DOC) and resource consents are often concerned with the potential for environmental impacts from the proposed operation, and thus are the focus of the following discussion.

A.1 Mining permit

NZ PAM (under the Ministry of Business, Innovation and Employment) manages the Government’s oil, gas, coal and minerals. Anyone wanting to prospect, explore or mine the Government’s resources must obtain a permit from NZ PAM under the Crown Minerals Act 1991. Further information is available on its website http://www.nzpam.govt.nz/cms.

A.2 Access arrangements

Mineral-related activities (including exploration, prospecting, mining and conducting minimum impact activities) are authorised under the Crown Minerals Act. Anyone wishing to undertake mineral-related activities requires a permit from Crown Minerals and permission from the landowner for land access. The access arrangement requirements of private landholders will vary. In the case of public conservation lands, the permit holder requires DOC permission to access the land.

Permission for minimum impact activities is granted by way of a consent, while an access arrangement is required for all non-minimum-impact activities (mining, non-minimum-impact exploration and prospecting).

Minimum impact activities generally involve surveying or initial soil sampling undertaken by hand or hand-held methods only. These activities are generally undertaken on a prospecting permit or the first stages of an exploration permit.

Non-minimum-impact exploration, prospecting and mining activities include a wide range of activities involved with locating and extracting minerals. A comprehensive application for the required access arrangement is necessary and will need to include an assessment of environmental effects (AEE). This Framework is intended to assist in completing and assessing the information provided in the AEE. Specifically it is intended to assist with the sections on water management and disturbance to the environment (effects and monitoring of effects). In addition to onsite management to minimise and mitigate effects, additional safeguards such as insurances, compensation and bonds (to ensure restoration of the site and cover for potential risks) will be required as part of any access arrangement granted.

There are different categories of public conservation land, ranging from stewardship land to national parks. Each conservancy has a conservation management strategy (CMS), which is a management plan for land managed within the conservancy and describes the inherent values and purpose of each category of land managed. The conservation management strategy also outlines the policies associated with granting access arrangements.

Under Section 61(1A) of the Crown Minerals Act applications for access are unable to be accepted in
some cases, primarily dependent on the status of the land (e.g. in wilderness areas and most areas of national parks). Access to the minerals (and consequently land containing the minerals) under a minimum-impact-activity consent or access arrangement can only be granted to land within a minerals permit. For any mining-related activities on public conservation land but outside the minerals permit, such as access roads, buildings or processing facilities, a concession may be required. It is recommended that you contact DOC staff in the relevant conservancy prior to commencing the application process, although DOC is developing a national standard operating procedure for assessing applications. This SOP will outline information requirements and the process each consideration of an application is required to follow.

For further information on access for mineral-related activities on public conservation lands, and the relevant application forms and guidelines, visit the DOC website (www.doc.govt.nz) or contact the relevant conservancy office.

A.3 Resource consents

Resource consents will be required for mining activities. The Resource Management Act (RMA) sets out the responsibilities of regional and local authorities and applicants. Regional councils issue four types of resource consent, covering:

- Land use (including disturbance of riverbeds)
- Water use
- Discharges to water, land, air
- Any activity in the coastal marine area.

The appropriate regional council will be the primary agency responsible for issuing resource consents for mining operations. A building consent from the regional council may be required for any dam construction. However, district councils issue resource consents for land use activities and building consents, which could be required for some mining operations.

A key component of resource consent is the AEE (assessment of environmental effects). General guidelines for assessing environmental effects are provided in Schedule 4 of the RMA. This framework is intended to assist in completing and assessing the information provided in the AEE, specifically those sections that relate to assessment of the potential effects on the environment, discharge to the environment (including sensitivity of the receiving environment, alternatives for discharge, and mitigation options) and monitoring requirements.

Individual councils provide information on their websites about obtaining resource consents, and may indicate specific information requirements. For example, the West Coast Regional Council (WCRC) requires an annual work programme to be completed as a condition of the consent. Templates for coal and gold mining are provided on the WCRC website (http://www.wcrc.govt.nz/consents/mining.htm), and require information on the general operation of the site, rehabilitation, and water management.

Environment Southland currently outlines the general process for resource consent applications, but does not have anything specific for mining (http://www.es.govt.nz/Departments/Consents/index.asp?sm=g_a). Consultation with the relevant council will provide specific information on requirements, templates for applications, and any changes to plans. As with access arrangements, applicants should discuss their application with the relevant regional council to ascertain current requirements.

Bonds will be required to ensure compliance with the resource consent conditions, and will form part of the negotiations during resource consenting.

A.4 Regional plans

Individual regional councils may have regional plans containing rules that apply to mining operations and their effect on the environment. These rules must be taken into account in a resource consent application. Existing regional plans that are of potential relevance to mining activities on the West Coast or in Southland are identified below. Regional plans for the management of water will also typically outline the classification of different aquatic systems according to the water quality classes set out in Schedule 3 of the RMA (2009 Amendment).

West Coast Regional Council

The Proposed Water Management Plan (WCRC June 2007), the Proposed Regional Land and Riverbed Management Plan (WCRC July 2009) and the Discharge to Land Plan (WCRC 2002) may be relevant to mining activities. While the water management and land and riverbed plans are not yet operative, all appeals on the policies and rules relating to acid mine drainage and mining are resolved, and thus are treated as if operative in accordance with Section 19(1) of the RMA. Further, there is an intention to merge these three plans (pers. comm., Chris Ingle, CEO, West Coast Regional Council).

The Proposed Water Management Plan identifies swimming areas, and Policy 7.4.1 of the plan states that the council will manage those areas for contact recreation purposes (Class CR) and all other surface
water bodies in the region for aquatic ecosystem purposes (Class AE). This plan also identifies rivers that have acid drainage issues, and has a specific policy for their management (7.4.2). Under the plan, the discharge of mine waters to a stream is a discretionary activity. There are also rules for abstraction (e.g. taking of water for dust suppression/coal washing) and diversion of water. The information required in resource consent applications is outlined, including specific requirements for the discharge of water or contaminants and acid mine drainage outlined in sections 14.3.4 and 14.3.5.

The Proposed Regional Land and Riverbed Management Plan (WCRC 2009) should also be taken into account if disturbance of vegetation or riverbeds is expected to occur or when earthworks or construction are planned.

The Discharge to Land Plan (WCRC 2002) should be taken into account, particularly in relation to the stockpiling of waste rock (Objective 6.3.1, Policy 6.4, Rules 1 and 17).

Both plans also provide information on circumstances and purposes for which financial contributions (e.g. bonds) may be imposed and used, and the assessment criteria for determining the amount of the financial contribution.

Environment Southland

Environment Southland’s Regional Water Plan (Environment Southland 2010) is the most relevant regional plan for mining activities. While there are no specific rules covering mining activities, the plan includes rules for abstraction (e.g. mine dewatering, taking water for dust suppression/coal washing), discharge, or diversion of water. The plan also outlines the information required to be submitted with a resource consent application. The plan has water quality standards for different characteristics of surface water bodies (Policy 5.2.1), which will potentially also influence discharge consents. The plan also provides information on circumstances and purposes for which financial contributions (e.g. bonds) may be imposed and used; the manner in which the amount of the contribution will be determined is also set out.

Large-scale mining and processing of minerals, or the flaring of gases, may also require a resource consent under the Regional Air Quality Plan (Environment Southland 1999).

The Regional Solid Waste Management Plan (Environment Southland 2007) may be relevant if it is considered that the mining operation constitutes a ‘Refuse Disposal Facility’ (Rule 4.5.2). This may occur in co-disposal of overburden with other material, e.g. sludge from any treatment plants or ash from boilers that consume the coal (pers. comm. Stephen West, Senior Resource Planner, Environment Southland).

A.5 Conservation orders

There may also be conservation orders in place that set water quality standards for certain rivers, and it may be applicable to consider these when assessing the potential impacts of proposed mining.


A.6 References


Appendix B:  SUSPENDED SOLIDS MANAGEMENT

B.1  Treatment systems

Active treatment systems

Total Suspended Solids is typically reduced through sedimentation with the solids recovered as a slurry or sludge. Active treatment typically comprises five steps: equalisation, coagulation/flocculation, gravity-assisted separation, filtration, and sludge dewatering and disposal, described in more detail below.

Which processes are used is partly dependent on the flow rate of the mine drainage and the specific gravity of the suspended solid load. For proposed operations, specific gravity can be determined on pulverised waste rock material, which may form the bulk of the TSS for the site. If flow rates are highly variable, equalisation will be required prior to the addition of coagulants or flocculants. Removal of particulates via gravity-assisted separation typically occurs via one of two commonly used techniques: clarification or dissolved air flotation. If the specific gravity of the TSS is greater than water, clarification is typically used. If it is similar to water, dissolved air flotation can be used. If the TSS load is unacceptably high after gravity-assisted separation, an additional filtration step may be required. Dewatering and disposal of the sludge are the final steps in active treatment of TSS-contaminated water. The sludge produced can often be disposed of on site in mine pits if it is non-hazardous. For existing mine sites, specific gravity of TSS is determined by laboratory analysis (method described in Appendix IV of U.S. Army Corps of Engineers 1986).

Equalisation

Equalisation is the process of collecting waste streams with variable flow in a tank for a certain period to obtain a stable waste stream that is easier to treat. These holding tanks can minimise the variability in volume and contaminant concentrations of incoming wastes or consolidate smaller volumes for batch treatment. Agitators or aerators are typically installed to prevent build-up of suspended solids.

Coagulation/flocculation

Coagulation and flocculation are separate processes involving addition of chemicals to promote settling of suspended particulates (Bratby 1980; Skousen et al. 1993, 2000). Both processes enhance sedimentation performance by increasing particle size resulting in increased settling rates.

Coagulation refers to the addition of chemicals to reduce the net electrical repulsive forces at particle surfaces, promoting consolidation of particles. The most commonly used coagulants are aluminium sulphate ($\text{Al}_2\text{(SO}_4)_3$), ferric sulphate ($\text{Fe}_2\text{(SO}_4)_3$), and ferrous sulphate ($\text{FeSO}_4$).

Flocculation refers to the addition of chemicals to join particles by bridging the spaces between suspended particles. Flocculants consist of polymer chemicals that adsorb suspended particles onto polymer segments. The most commonly used flocculants are synthetic anionic (negatively charged surfaces) and cationic (positively charged surfaces) chemicals and polyampholytes (both positive and negative charges on surface based on pH).

Coagulants and/or flocculants are usually added in a rapid-mix tank (or with an in-line mixer and then the wastewater flows to a flocculation basin or tank where slow mixing allows the particles to agglomerate in a more settleable solid. The treated water then flows either into a clarifier or a settling pond for removal of TSS. Bench-scale laboratory experiments are recommended to determine appropriate type and dosage of coagulant/flocculant.

Gravity-assisted separation

Two forms of gravity-assisted separation are commonly used: clarification and dissolved air flotation (DAF). In clarification, the wastewater flows into the centre of a clarifier tank and then flows slowly and uniformly outward toward the tank walls. The water leaves the clarifier over a weir at the top allowing solids to settle to the bottom of the clarifier. The solids are periodically removed, dewatered and disposed.

In DAF, removal of solids is achieved by dissolving air in the wastewater under pressure and then releasing the air at atmospheric pressure in a flotation tank or basin (Kiuru and Vahala 2001; Wang et al. 2004). The released air forms tiny bubbles which adhere to the suspended matter causing it to float to the surface of the water where it may then be removed by a skimming device. DAF is effective in removing suspended particles that have a specific gravity close to that of water. The major components of a conventional DAF unit are a centrifugal pump, retention tank, air compressor and flotation tank. Wastewater treatment textbooks generally recommend a bubble size of 100 micrometres. The most practical way to establish the proper rise rate is to conduct experiments at various air pressures.
**Filtration**

Three types of filtration technologies are commonly used to lower TSS: sand filtration, multimedia filtration and membrane filtration.

Sand filtration consists of either a fixed or moving bed of sand that traps and removes TSS from the wastewater. In the fixed sand filters water is either forced through the filter under pressure or flows under gravity downward through the filter. In the moving bed filters, the filter moves downward while the water flows upward through the filter.

Multimedia filtration involves using two or more layers of different granular material, such as garnet, sand or anthracite coal. The finer, denser medium is typically at the bottom and the coarser, less dense medium at the top. The solids are retained in the voids between the media particles while the wastewater passes through the bed. Flow pattern is usually top-to-bottom.

In membrane filtration a semi-permeable microporous membrane and a pressure differential are used to remove TSS from wastewater. The primary design consideration is the membrane selection. A membrane pore size is chosen based on the size of the contaminant particles targeted for removal. Other design parameters to be considered are the solids concentration, viscosity and temperature of the feed stream, pressure differential, and membrane permeability and thickness. The rate at which a membrane fouls is also an important design consideration.

**Sludge dewatering and disposal**

Sludge typically contains between 1 and 5% solids. If disposed of to landfill, dewatering of sludge will be required, although dewatering may not be required for on-site disposal. If disposed of to landfill, sludge dewatering and disposal can be a significant cost.

The most common dewatering methods are plate and frame pressure filtration, belt pressure filtration and vacuum filtration. A plate and frame filter press can produce the driest filter cake, followed by the belt press, and lastly, the vacuum filter.

A plate and frame filter press consists of a number of recessed filter plates connected to a frame and pressed together between a fixed end and a moving end. Each plate contains a drainage surface and filter cloth is mounted on the face. Sludge is pumped under pressure into the chambers between the plates. Water passes through the media and solids are retained in the cavities of the filter press. As it is a batch process, when the filtrate flow stops, the plates are separated and the filter cake drops into a hopper below the press.

In belt pressure filtration the sludge stream is fed onto the lower of two moving cloth filter belts. As the belts move closer together, the sludge is compressed between them and water drains through the lower belt. A series of rollers compress the sludge cake completing the dewatering, and at the end of the roller series the belts move apart and the cake drops off.

Vacuum filtration uses vacuum to extract solids from sludge. In a typical configuration, a continuous belt of filter fabric is wound around a horizontal rotating drum and rollers. The drum is perforated and when connected to a vacuum is partially immersed in a shallow tank containing the sludge. As the drum rotates, the vacuum that is applied to the inside of the drum draws the sludge onto the filter fabric. The water from the sludge passes through the filter and into the drum, where it exits via a discharge port. As the fabric leaves the drum and passes over the roller, the vacuum is released and the filter cake drops off of the belt.

Disposal of filter cake is usually to an off-site landfill, incineration facility or appropriate area for land application. Leaching of trace elements from the sludge may influence disposal in a landfill and leach tests should be performed on the sludge. If the sludge is non-toxic, filter cake from the active treatment process can sometimes be disposed of on-site, or as mentioned above, pumped directly into deep mines with little to no dewatering.

**Passive treatment**

Passive treatment for mine drainage containing unacceptable concentrations of TSS consists of settling ponds or sedimentation ponds. With sufficient residence time of the wastewater in the settling pond, suspended solids settle out of solution and accumulate at the base of the pond. Depending on the rate of accumulation of solids, sludge may be removed periodically from the pond to maintain the necessary residence time.

For passive treatment systems with high suspended solid loads surface channels may be preferable to pipelines for transportation of sediment-rich water to the treatment system. Channels offer easier access for monitoring and cleaning. Conversely, because flow velocities tend to be higher in pipes, the likelihood of settlement of suspended solids is lower in pipes, and therefore maintenance requirements are less frequent. Also, depending on construction materials, channels tend to be more expensive to install.

The following are some general guidelines for designing setting ponds (Hill 1996):
All structures in the sedimentation pond system should be designed, as a minimum, to withstand a 1 in 200-year flood event.

The design flow for removal of suspended solids in sedimentation ponds should correspond to the 10-year, 24-hour flood flow.

Sedimentation ponds should either last the lifetime of the mine without requiring removal of accumulated sediment, or should have provision for easy removal of sediment at regular intervals. Normally a pond is allowed to fill with sediment up to 50% of its effective depth, with 1.5 m (minimum) depth of pond liquid above the sediment (DIAND 1986).

Ideally, a smaller pond should be located upstream from the main sedimentation pond to remove the coarse fraction of the sediment. This smaller pond should be designed to have easy removal of sediment.

The inlet section of the pond should have some type of energy dissipater (such as barriers, baffles, etc.) to spread out the flow and reduce the velocity of the incoming water.

The discharge section of the pond should be at the opposite end to the inlet section and should have a spillway (or decant or discharge riser) designed to maintain a minimum 0.5 m of freeboard on the embankment in a 1 in 200-year flood event. The spillway must be armoured to prevent erosion of the channel.

Sedimentation ponds should be provided with means of draining or dewatering.

The preferred shape of sedimentation ponds is generally rectangular with a length-to-width ratio of about 5:1 (DIAND 1986). Such ponds tend to prevent short-circuiting, and facilitate removal of accumulated sediment. The need for additional pond capacity and retention time due to accumulated sediment volume, turbulence and currents in the pond should be considered (Hazen and Sawyer 1975; DIAND 1986). If the length-to-width ratio is less than 5:1, baffles may need to be constructed to ensure the water travels a distance three to five times the pond width.

Appropriate retention time and therefore required size of the settling pond can be determined by three methods:

- Sedimentation rates can be measured using simulated samples or representative samples collected from the mine site.

- If the size distribution of the influent TSS is known, the settling velocity can be derived using the Stokes’ Law formula \( V_s = \frac{g \mu}{18} \times \left( \frac{5 - 1}{D^2} \right) \). A 20-100% correction should be applied due to the non-spherical shape of actual mineral particles when their diameters are measured by a method that does not utilise settling.

- Particle size can be assumed to be 5–10 microns and the settling velocity can be assumed to be \( 2 \times 10^{-5} \) to \( 5 \times 10^{-5} \) m/s.

Other details for ponds such as preliminary investigations, estimation of storm runoff, excavation of ponds, sealing of ponds, and operation and maintenance can be found in USDA (2000).

In some situations a combination of passive and active treatment for removal of TSS may be appropriate. This may include the use of settling ponds with a mechanism to add settling aids such as coagulants or flocculants (Bratby 1980).

### B.2 Monitoring of treatment systems

For all systems designed to reduce high suspended solid loads, the main parameters of interest in both inlet and outlet water will be:

- Flow rate
- Turbidity (TSS)
- pH

These parameters should be monitored regularly, and it may be feasible to have them monitored continuously using a data-logger with regular downloads using telemetry. It may also be appropriate to monitor additional parameters, such as As, Mn, Fe and Al or other metals or metalloids, if these have been determined to be of concern in the mine discharge during baseline and ongoing monitoring.

### B.3 References


Hazen and Sawyer 1975. Process design manual for suspended solids removal. EPA 625/1-75-003a. United
States Environmental Protection Agency (USEPA) Office of Technology Transfer, Washington, DC, USA.


Appendix C: GEOCHEMISTRY

C.1 Background geological information – West Coast

Greenland Group Rocks

Greenland Group Rocks are Cambrian to Ordovician in age (c. 400-500 million years old) and form the basement rocks for large parts of Westland. The Greenland Group includes alternating mudstones and sandstones that have been metamorphosed to a hard-rock sequence. These rocks host hard-rock gold deposits in quartz veins and shear zones, particularly in the Reefton area, where a new mine was opened in 2007. Greenland Group rocks have been intruded by granites and contain scattered carbonate minerals, particularly calcite; alteration zones around gold deposits contain calcite and iron-bearing dolomite. Sulfide minerals, especially pyrite arsenopyrite, and stibnite, occur in and adjacent to gold deposits (Christie and Braithwaite 2003).

At a regional scale the Greenland Group rocks are not acid forming because of the ubiquitous but low carbonate content. At the mineral-deposit scale acid-forming rocks are present but uncommon.

Paparoa Coal Measures

Paparoa Coal Measures (Figure C1) are Cretaceous to Paleocene in age (c. 100-50 million years old) and were deposited in a river and lake setting in a fault-controlled basin that was inland from the coast at the time. Thick accumulations of clean peat developed in these narrow fault depressions (Newman and Newman 1992; Moore et al. 2006). Paparoa Coal Measures were deposited on thick gravels of the Cretaceous Pororari Group or older basement rocks (Greenland Group greywackes, and granites). Sediments accompanying the coal contain abundant rock fragments from the underlying basement rocks. Greenland Group rocks contain scattered carbonate minerals (Christie and Braithwaite 2003), which can contribute to acid neutralisation capacity (ANC). Groundwater alteration of the Paparoa Coal Measures include clay minerals (kaolinite and illite), carbonates (siderite > magnesite > calcite) and silica overgrowths (Boyd 1993; Boyd and Lewis 1995).

Data collected to date indicate low acid-producing potential (MPA) in the Paparoa Coal Measures at the regional scale. This is interpreted to relate to low sulphur in the depositional environment because the sediments formed in an environment removed from the sea (Pope et al. in press).

Brunner Coal Measures

Brunner Coal Measures (Figure C1) are Eocene in age (c. 40 million years old) (Pocknall 1992) and were deposited in a river or estuarine environment close to the sea, at a time when sea levels were rising and covering the coals after they formed (Nathan 1986; Leask 1993; Flores and Sykes 1996). Brunner Coal Measures were deposited on top of Greenland Group basement rocks, granite, and locally, the older Paparoa Coal Measures (see above) (Nathan et al. 2002). The Brunner Coal Measures are overlain by the Kaiata Formation, which was deposited in the sea. The Kaiata Formation is mainly mudstone (generally called Kaiata Mudstone) with some sandstone layers (called Island Sandstone locally). The Kaiata Formation sediments are dominated by quartz (Newman 1988) with some feldspar (Titheridge 1992) that was derived from Greenland Group and granite basement rocks.
Calcite fossils occur throughout the Kaiata Formation, increasing in numbers with distance above the Brunner Coal Measures. The abundant rock fragments of the Paparoa Coal Measures are not present in the Brunner Coal Measures and Kaiata Formation. Groundwater alteration of Brunner Coal Measures includes abundant kaolinite formation. Pyrite is common as scattered grains (sometimes very fine grained; Weisener and...
Weber in press) or as a cement. Rare calcite and dolomite nodules occur in the Garvey Creek Coalfield (Newman 1988). The relative proportions of pyrite and calcite in the Kaiata Formation immediately above the Brunner Coal Measures and their effects on acid-base accounting (ABA) are described by Hughes et al. (2007).

At a regional scale the Brunner Coal Measures are moderately to highly acid forming. They were deposited in a sulphur-rich tidal environment and are overlain by marine sediments that also contain sulphur, and that supplied additional sulphur to the coal measures during burial. The overlying Kaiata Formation has acid-forming and acid-neutralising characteristics and at Stockton these characteristics have been stratigraphically and geochemically defined (Hughes et al. 2004, 2007; Weber et al. 2006). However, it is unclear if the stratigraphic controls on acid-forming compared with acid-neutralising rocks that are used for management at Stockton will have regional significance throughout the West Coast. Additional analyses of the Kaiata Formation are required.

Rotokohu Coal Measures
Rotokohu Coal Measures are early Miocene rocks (c. 20 million years old) that form part of the Bluebottom Group. They are underlain by marine mudstones and overlain by non-marine sediments. Insufficient data have been collected to generalise about the acid-forming potential of the Rotokohu Coal Measures. However, acid mine drainage (AMD) has not been reported from this formation to date.

C.2 Background geological information – Southland

Morley Coal Measures
Morley Coal Measures (Figure C2) are part of the Ohai Group and occur mostly within the Ohai Coalfield (Turnbull and Allibone 2003). The Morley Coal Measures vary between 0 and 210 m thick, are the main coal-bearing unit in the Ohai Coalfield and have an extensive mining history by both underground and opencast methods. They are Late Cretaceous in age (c. 80 million years old) (Warnes 1988, 1990; Raine 1989). The coal deposits and associated sediments were deposited in a river valley setting removed from the sea, similar to the Paparoa Coal Measures (above) (Sykes 1985, 1988; Bowman et al. 1987; Shearer 1995). Drainages from mines in the Morley Coal Measures typically have low acidity, circum-neutral pH and elevated suspended solids (Craw et al. 2008).

Beaumont Coal Measures
Beaumont Coal Measures (Figure C2) are part of the Nightcaps Group and occur within the Ohai Coalfield and surrounding basins. Beaumont Coal Measures are between 20 and 25 m thick within the Ohai Coalfield but are thicker outside the coalfield. Beaumont Coal Measures have been mined at a small scale in the Ohai Coalfield and are Eocene in age (c. 40 million years old) (Couper 1960; Pocknall 1990; Raine 1989). Beaumont Coal Measures were deposited in river environments near to the sea, and are overlain by lake and marginal marine sediments of the Orauea Mudstone (Bowen 1964).
Figure C2: Map showing extent of coal measures in Southland.
There are no mine drainage chemistry data or ABA from the Beaumont Coal Measures. However, the marginal marine environment of formation of Beaumont Coal Measures and overlying Draurua Mudstone is similar to that of the Brunner Coal Measures and overlying Kaiafa Formation (described above), and similar ABA issues are to be expected.

**Gore Lignite Measures**

Gore Lignite Measures (Figure C2) are Late Oligocene to Miocene in age (c. 25-15 million years old) (Pocknall 1990) and were deposited in a river delta and flood plain environment near to the sea (Isaac and Lindqvist 1990). The Gore Lignite Measures locally rest on marine sediments, including limestones, and the lower parts of the lignite measures are interlayered with marine sediments (Chatton Formation) (Isaac and Lindqvist 1990). Sediment for the Gore Lignite Measures was derived from the Otago Schist, local basement greywackes, and some erosion of pre-existing coal measures (Isaac and Lindqvist 1990; Youngson et al. 2006). Groundwater alteration of Gore Lignite Measures includes alteration of some components of the sediment to clays (Craw et al. 2008), occasional silification of plant remains (Isaac and Lindqvist 1990), rare carbonate cementation and irregularly distributed sulphide (pyrite or marcasite) cementation of sandstone and conglomerates (Youngson 1995; Falconer et al. 2006).

Mine drainage from the Gore Lignite Measures has been investigated at mine pits (Dune 2001; Mulliner 2006; Mulliner et al. 2006). In general, mine drainage chemistry is circum-neutral. Rarely pH below 5 occurs and rarely this can contain elevated trace element concentrations. Datasets of ABA information for the Gore Lignite Measures are rapidly improving and indicate that most rocks are not acid forming but occasional PAF rocks are present (Pope et al. 2010).

**C.3 Minerals with implications for mine drainage**

**Sulphide minerals**

Sulphide minerals control the acid production potential and strongly influence the trace element geochemistry of mine drainage. Pyrite and marcasite are the most abundant acid-producing sulphide minerals in rocks likely to be disturbed by mining on the West Coast and in Southland. Pyrite oxidises to produce acid:

\[
\text{FeS}_2 + 7/2 \, \text{H}_2\text{O} + 13/4 \text{O}_2 \rightarrow \text{Fe(OH)}_3 + 2 \text{SO}_4^{2-} + 4\text{H}^+.
\]

Pyrite oxidation is a complex and stepwise process that is often catalysed by microbial processes (Evangelou 1998). Breakdown of other sulphide minerals can release trace elements with or without acid:

- \( \text{PbS} + 2\text{O}_2 \rightarrow \text{PbSO}_4 \) \hspace{1cm} (2)
- \( \text{ZnS} + 2\text{O}_2 \rightarrow \text{ZnO}_2^+ + \text{SO}_4^{2-} \). \hspace{1cm} (3)

Sulphide minerals are commonly impure and can contain several % of other transition metals that substitute for Fe in the mineral structure. Therefore breakdown of pyrite can also substantially increase trace element compositions.

**Carbonate minerals**

Acid-producing potential decreases with increasing abundance of common carbonate minerals such as calcite, dolomite, ankerite, and magnesite. Calcite and dolomite are present in some rocks disturbed by coal mining in the West Coast and Southland. Dolomite, ankerite and siderite carbonate minerals are common in rocks disturbed by hard-rock gold mining on the West Coast and in Southland (Christie and Brathwaite 2003):

- \( \text{CaCO}_3(s) + 2\text{H}^+ (\text{aq}) = \text{Ca}^{2+} + \text{H}_2\text{O} + \text{CO}_2(g) \) \hspace{1cm} (4)

Carbonate minerals can also be the source of trace elements, because trace element impurities are common in carbonate minerals and trace-element-specific species can form such as \( \text{ZnCO}_3 \), \( \text{MnCO}_3 \), \( \text{CaCO}_3 \) and many others.

Most carbonate minerals have acid-neutralising capacity except siderite:

- \( \text{FeCO}_3(s) + 2\text{H}^+ = \text{Fe}^{2+} + \text{CO}_2(g) + \text{H}_2\text{O} \) \hspace{1cm} (5)
- \( \text{Fe}^{2+} + 1/2 \text{O}_2 + \text{H}^+ = \text{Fe}^{3+} + 1/2\text{H}_2\text{O} \) \hspace{1cm} (6)
- \( \text{Fe}^{3+} + 3\text{H}_2\text{O} = \text{Fe(OH)}_3(s) + 3\text{H}^+ \). \hspace{1cm} (7)
C.4 Geological description

The following information should be recorded when describing rock samples:

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<tr>
<td>• Sample depth interval</td>
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</tbody>
</table>

<table>
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</tr>
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</tr>
<tr>
<td>• Informal or local name</td>
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</tbody>
</table>

<table>
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<table>
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</tr>
<tr>
<td>• Igneous/metamorphic information, mineralogy, textures</td>
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<tr>
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</tr>
<tr>
<td>• Weathering</td>
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</tbody>
</table>

<table>
<thead>
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<th>Rationale for sample collection</th>
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<tr>
<td>• Selected because it contains minerals that have implications for acid production or neutralisation:</td>
</tr>
<tr>
<td>• Sulphide minerals (pyrite, pyrrhotite, arsenopyrite, etc.)</td>
</tr>
<tr>
<td>• Secondary minerals after sulphide oxidation (gypsum, Fe oxyhydroxides)</td>
</tr>
<tr>
<td>• Carbonate minerals (calcite, siderite, magnesite, etc.)</td>
</tr>
</tbody>
</table>

C.5 Field and core samples

The following section provides a brief description and pictures of key minerals that have mine drainage implications (acid forming, acid neutralising, arsenic forming). Their presence during hand or core sampling indicates that there are definite implications for the quality of the mine drainage likely to be produced, and this should be taken into consideration during further geological sampling (e.g. sample selection for ABA analyses). However, these minerals are not always easy to see in hand specimens and cores, and formal chemical analyses are needed to confirm their presence or absence.

Pyrite

Pyrite ($\text{FeS}_2$), the primary acid-forming mineral in most AMD areas, can be identified in rock samples. It commonly has a gold colour (bright or dull) (Figures C3 and C4), although it can be greenish grey when fine grained or crushed and ground by drilling (Figure C5).
**Figure C3**: Pyrite in coal, Taratu Formation, South Otago.

**Figure C4**: Pyrite (elongate grey patches) in core from Brunner Coal Measures.
Figure C5: Pyrite cement (greenish grey) filling in between quartz white grains in quartz gravel, Taratu Formation. Acid runoff has killed grass around this specimen and stained the gravel brown with Fe oxyhydroxide.

Arsenopyrite

Arsenopyrite (FeAsS) is the most common source of As in mine drainage and is almost always present in hard-rock gold deposits. It has a silver-grey colour when coarse grained (Figures C6 and C7) and a black colour when fine grained or crushed by drilling. Grey stibnite (Sb2S3) superficially resembles arsenopyrite when it is fine grained in quartz veins (Figure C8).
Figure C6: Silvery grey arsenopyrite (FeAsS) crystals in a gold-bearing vein. The arsenopyrite crystals have a characteristic diamond shape and distinctive striations on crystal faces.

Figure C7: Grey streaks and patches in an auriferous (gold-rich) quartz vein are rich in fine-grained arsenopyrite.
Figure C8: Grey stibnite (Sb2S3) superficially resembles arsenopyrite when it is fine grained in quartz veins (as seen in core, top), but is much softer and has long thin crystals (bottom).
Calcite

Calcite (CaCO3) is the most common mineral that contributes acid-neutralising capacity to rocks. It can be formed during deposition of sedimentary rocks as cement, or as fossil shells. Calcite also forms during hydrothermal alteration of rocks after burial, and occurs in most hard-rock gold deposits as veins and scattered grains in rocks adjacent to veins (Figure C9). Calcite is commonly white and can be distinguished from other white minerals because it dissolves effervescently with addition of 10% HCl.

Figure C9: Small white calcite veins in Greenland Group rocks that host gold deposits.

Gypsum

Gypsum (CaSO4) is a sulphate mineral that can be present in hydrothermally altered or weathered rocks and, when present, is generally associated with pyrite. Gypsum can also occur as a salty encrustation on rocks that have been recently drilled or as a coating on rock surfaces (Figure C10). For this reason, sampling of fresh unweathered core is preferred for ABA measurements (Figure C11). Occurrence of gypsum in a core indicates that the core will contain elevated sulphur (S). However, S associated with gypsum is NAF and pyrite-specific S analyses are required to determine the acid-forming potential of rocks that contain gypsum.
Secondary iron oxyhydroxides

Secondary Fe oxyhydroxides are a suite of minerals (including ferrihydrite, schwertmannite, jarosite and others) that form when Fe oxidises or when water that contains dissolved Fe is neutralised. These minerals are commonly rusty coloured and occur as a surface coating on rocks (or drill core) (Figure C12). Jarosite (Fe oxyhydroxysulphate) can have a distinctive butter-yellow colour (Figures C13–C15). The Fe in these secondary precipitates can come from pyrite (Figures C16 and C17) and therefore can be an indirect indicator of acid-forming minerals in a rock. However, Fe oxyhydroxides are an unreliable indicator of acid-forming conditions because Fe can also come from other minerals (including carbonates) or fluids associated with rocks (Figures C18 and C19).

Figure C10: Gypsum encrustation on core from Gore Lignite Measures.

Figure C11: Gypsum on gravel.
Figure C12: Iron oxide staining (deep brown precipitate) on rocks in a creek draining an acid-producing coal mine. When the creek is low, the acid drainage dominates, and Fe is readily dissolved in the creek water. During rain events, the creek pH rises because of rain dilution, and Fe oxyhydroxide is deposited on the pebbles.

Figure C13: Mixed jarosite and iron oxide on coal.
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**Figure C14**: Jarosite: butter-yellow staining on surface of rock.

**Figure C15**: Jarosite: butter-yellow staining on surface of quartz gravels.
Figure C16: Fe oxyhydroxides associated with pyrite in Brunner Coal Measures.

Figure C17: Iron oxide staining (top half of picture) indicating an oxidised zone in alluvial gravels; the lower half is unoxidised and contains pyrite.
**Figure C18:** Fe oxyhydroxides (brown patches) that are not related to pyrite in conglomerate from the Gore Lignite Measures. Fe oxyhydroxides here are localised onto specific fragments in the rock that contain other Fe minerals such as chlorite (an iron silicate).

**Figure C19:** Iron-magnesium-calcium carbonates. These are less common than calcite, but can contribute to acid neutralisation. Even small amounts of Fe in these minerals cause a distinctive orange weathering appearance, as in the veins from a gold mine shown above.
Scorodite

Scorodite is a green arsenic oxide mineral that forms on arsenic sulphide minerals such as arsenopyrite (Figures C20 and C21). Fine-grained grey arsenopyrite can be difficult to distinguish in rocks and cores, but the green scorodite coating is a useful indicator of As-bearing rock in a mine setting.

**Figure C20:** Grey arsenopyrite has a coating of green scorodite (FeAsO$_4$.2H$_2$O) that forms from oxidation of the arsenopyrite.

**Figure C21:** Green scorodite coats a sample of dark sheared schist containing abundant arsenopyrite. Specimen is 20 cm across.
C.6 Notes on ABA analyses

Commonly used ABA analyses, and their limitations, are described below.

Maximum potential acidity (MPA)

Typically MPA analysis uses total sulphur to calculate potential acid generation assuming all sulphur in the sample is present as pyrite:

\[
\text{FeS}_2 + 3.25\text{O}_2 + 3.5\text{H}_2\text{O} \rightarrow \text{Fe(OH)}_3 + 2\text{H}_2\text{SO}_4. \tag{8}
\]

Weight % sulphur is converted into kg H_2SO_4/t through molar mass ratios:

\[
\text{MPA kg H}_2\text{SO}_4/t \rightarrow \text{S(%) x 10 x 98.08 / 32.06.} \tag{9}
\]

the assumption that all S is present as pyrite is often valid (for rocks – not coal) however, there are many other ways that sulphur might be incorporated into rocks (Smart et al. 2002) including:

- Sulphate minerals, (jarosite, alunite, gypsum)
- Native sulphur
- Other sulphide minerals, (arsenopyrite, chalcopyrite, sphalerite)
- Organically bound sulphur (common in carbonaceous rocks or coal)

When sulphur is present in these other forms, using total sulphur to calculate acidity is overly-conservative and might lead to false-positive analyses because most other forms of sulphur are non-acid forming or have lower acid-forming potential. Pyrite-specific sulphur analyses can be completed through a chromium reduction method (Sullivan et al. 1999, 2000). This method is not commonly performed in laboratories in New Zealand but could be completed at a research laboratory or sent to an Australian laboratory where it is routinely tested in samples from coal mines. Commercial instruments are available to perform chromium reducible sulphur analysis. Once the amount of reduced inorganic sulphur has been analysed this weight % value can be used to calculate MPA in the same way as a total sulphur value.

Acid-neutralising capacity (ANC)

Most ANC comes from carbonate minerals within rocks.

\[
\text{ANC kg H}_2\text{SO}_4/t = \text{ANC kg CaCO}_3/t \times \frac{100.09}{98.08}. \tag{10}
\]

Net acid generation (NAG)

NAG analysis is only suitable for rocks that do not contain substantial organic material and includes oxidation of sulphides material with hydrogen peroxide; however, this reactant is not selective for sulphides and it also oxidises organic matter in the sample. As the organic matter breaks down during NAG analysis it releases organic acids that titrate during acidity analysis and lower the pH, but do not represent acids that will form under environmental conditions. Therefore application of NAG analysis to carbonaceous sediments and rocks immediately adjacent to coal seams should be conducted with care (Miller et al. 1997; Stewart 2003; ACARP, 2008).

Additional data from NAG tests can be obtained by repeating the procedure on the same sample (Sequential NAG) or monitoring the evolution of the NAG liquor (Kinetic Nag) (Smart et al. 2002; ACARP, 2008), although completion of these tests and interpretation of results should only be undertaken by an experience geochemist.

Paste pH

Paste pH is a non-quantitative field-based indication of stored soluble H+ acidity (Lin et al. 2000). It has minimal application to calculating acid production or identification of total acidity that will be released by weathering of a rock because it does not include sulphide-bound potential acidity or acidity that is contributed by acidic metals such as Fe and Al (Equation 8) (Lin et al. 2000):

\[
\text{Al}^{3+} + \text{H}_2\text{O} \rightarrow \text{Al(OH)}_3 + 3\text{H}^+. \tag{11}
\]

This method can be successfully applied as a management tool and qualitative relationships are present between paste pH and quantitative ABA methods (Weber et al. 2006).

Carbonate bomb

The carbonate bomb method is a qualitative field method to determine the acid-neutralising potential of rocks. Case studies indicate good relationships between carbonate bomb methods
and acid-neutralising capacity (Hughes et al. 2004, 2007). However, there is potential for the carbonate bomb method to overestimate the ANC of rocks because this method does not discriminate between carbonate minerals that have high ANC such as calcite or those with no ANC such as siderite.

C.7 Notes on kinetic tests

Commonly used kinetic tests methods and their limitations are described below.

Kinetic NAG tests

This test extends data collected from a standard NAG test. Chemical information such as pH and temperature is monitored throughout the NAG reaction to gain information about relative reaction rates of acid-producing and acid-neutralising components within the rock. It can be used to identify lag periods between exposure and the onset of acid production in acid-forming rocks (Weber et al. 2006; ACARP 2008).

Leach columns

These tests typically use at least 1–2 kg of rock material and simulate an accelerated weathering process. Rocks are watered on a periodic basis to simulate a rainfall regime and samples of leachate are collected for analysis. Many variations are possible depending on the experimental requirements, for example, heat lamps can be used to accelerate weathering, slightly acidic leach water can be used or samples can be flooded and dried repeatedly.

The data collected indicate weathering rates or rates at which acid or neutralising capacity is released and this analysis highlights any differences between MPA or ANC and actual acidity or neutralising capacity. In addition, proposed management or treatment methods can be trialled such as addition of a neutralising agent to a PAF rock.

Lysimeter

Lysimeter tests complete leaching of larger samples (typically at least 1-2 m$^3$) under field conditions. Lysimeters are drainage collection devices emplaced into waste-rock dumps to collect leachate under natural field conditions. They can provide information about cover performance and mine drainage chemistry without laboratory-induced artefacts, although sample materials and surrounding rocks must have similar physical characteristics to ensure accurate results. Lysimeters can be used to measure real field rates for mine drainage reactions and test treatment methods including the effect of vegetation.

Leach pad

Leach pad tests are similar to lysimeter tests but on a larger scale (typically 5-50 m$^3$). All drainage from the sample of rock is collected so that field rates for mine drainage reactions are indicated. The main advantage of leach pad tests over lysimeter tests is that variations within a waste-rock dump can be estimated. For example, variations in oxygen and water ingress into a waste-rock dump might cause weathering to proceed slowly in some parts of a dump and rapidly in others. Leach pad tests can highlight potential variations and can be used to test treatment methods that limit water or oxygen ingress.

C.8 Arsenic chemistry

Arsenic is a metalloid, not a metal, and therefore it behaves differently from metals in the environment. One of the most significant differences between As and metals is the response to pH change in the environment. In particular:

Metals: these form positively charged ions (cations), and the cations become more soluble with decreasing pH. Hence, in an AMD setting, metals such as Fe and aluminium (Al) can be a significant environmental issue as dissolved Fe$^{3+}$, Al$^{3+}$, and other metals such as zinc (Zn), nickel (Ni), and copper (Cu) may also be present in elevated concentrations.

Arsenic (and antimony): these metalloids form complexes with oxygen that take on a positive charge. For As, the most important complex in the environment is HASO$_4^{2-}$. This complex, and other similar As complexes, becomes more soluble with increasing pH, from a low near pH 3-4 (Figure C22). Arsenic also becomes more soluble with decreasing pH below pH 3 (Figure C22). The maximum amount of As than can become dissolved in mine waters is generally controlled by the mineral scorodite (FeAsO$_4$.2H$_2$O). Arsenic concentrations can rise until they reach the scorodite solubility curve at the prevailing pH, and then scorodite will precipitate from the water (Figure C22).

The solubility behaviour of As has major implications for environmental issues around hard-rock gold mines, because these gold mines typically
have no AMD, and waters have pH between 7 and 8. Under these neutral to slightly alkaline pH conditions, As is potentially highly soluble (Figure C22). The dissolved As concentrations will not necessarily reach the maximum levels indicated in Figure C22, as other effects such as dilution and adsorption of As to Fe oxyhydroxides can lower dissolved As (Figure C23). Consequently, mine site management practices that limit As dissolution are important (see Section 6.4).

If an As mineral other than scorodite is present at a site, the above generalisations no longer apply. Such minerals are rare, but an example below provides some information on one mine site that contains the mineral arsenolite (As$_2$O$_3$), as well as scorodite.

![Figure C22: Dissolved As concentrations measured at a range of water pH at South Island mine sites. The maximum As concentrations are controlled mainly by the solubility curve for scorodite (Krause and Ettel 1988).](image)

**C.9 Load or flux**

Measurement of contaminant concentrations in water has many practical difficulties, because of factors such as dilution by rain events and adsorption to solid particles. In addition, it is difficult to quantify regional background concentrations in the context of locally-high concentrations at individual sites, such as mining discharges. Calculation of the load or flux can help with these issues.

The load or flux of a chemical is the concentration multiplied by the stream flow rate:

$$\text{Flux (mg/s)} = \text{Concentration (mg/L)} \times \text{Flow rate (L/s)}.$$  

This has the advantage of correcting for variable stream flow rates in high-rainfall areas. It also enables better comparison between large rivers with low concentrations and small streams with high concentrations. The following example, using real data, illustrates the use of load or flux.

**Case study: Arsenic discharge from an historical mine site, Waiuta, Westland**

Prohibition mill site at Waiuta is probably the most acutely toxic historical mine site in New Zealand (Haffert and Craw 2008a, b). Residues from the processing plant are lying exposed on the ground surface and in an old tailings dam. Most of these residues contain about 20% As, which are more than 1000 times higher than other As-contaminated sites in New Zealand such as orchards and sheep-dips.
Some of the As is present as arsenolite, which was a by-product of the gold-processing system. Arsenolite is highly soluble, and surface water typically contains up to 50 mg/L dissolved As, which is 5000 times higher than the New Zealand drinking water limit. The high dissolved As at the site decreases by over an order of magnitude in a dam structure below the site (Figure C23). However, water leaving the dam and entering the natural environment downstream still has 2 mg/L dissolved As, which is 200 times higher than the drinking water limit. The dissolved As concentrations decrease steadily downstream because of dilution. The Blackwater River, which is the receiving stream for the Prohibition Mine site discharge, has 0.003 mg/L dissolved As 14 km downstream from the Prohibition mill site (Figure C23).

**Figure C23**: Concentrations of dissolved arsenic in stream waters downstream of the Prohibition mill site and into the Blackwater River. Annotations on the graph indicate processes that affect the dissolved As concentration at each stage.

**Local scale**

The site has a unique geochemistry, which is characterised by:

- The presence of the highly soluble near-pure As mineral arsenolite. Arsenic at mine sites is usually present as significantly less soluble arsenopyrite, which is derived from arsenopyrite-rich ore. At this site the ore was roasted as part of the processing, which converted arsenopyrite to arsenolite and other by-products.
- Low pH (3-4) due to the oxidation of arsenolite. Despite the high neutralising capacity of the surrounding host rock, acidity at this site prevails because it does not receive buffered runoff from surrounding terrain.

Effective natural attenuation takes place in the man-made dam by adsorption to Fe oxyhydroxide that is being generated mainly by the dissolution of Fe-bearing carbonates. Effective carbonate dissolution, as is presently the case, only takes place if the water entering the dam is:

- Undersaturated with respect to the carbonate mineral, i.e. it has not previously interacted with carbonates
- Acidic and can carry dissolved As and therefore contribute sorption sites when Fe precipitates upon carbonate dissolution.

It is vital for the downstream environment to maintain effective As removal within the dam. However, some treatment options could negatively alter site chemistry in a way that the dam would lose efficiency. These include:

- Counteracting site acidity by adding neutralising agents, such as lime, to the tailings dam
- Increasing site safety by covering up the tailings dam with locally derived rocks that contain abundant carbonates.

In both cases, wetland water would become neutralised and saturated with respect to carbonates, which would make the dam ineffective.
Instead, either complete removal of the As minerals or covering with carbonate-free rock would be more desirable.

**Regional scale**

Because the Blackwater River has detectable As concentrations downstream from the Prohibition Mine site, it is reasonable to question whether or not the highly contaminated Prohibition site is adversely affecting water quality in the Blackwater River. It is not possible to answer this question using water chemistry alone. Instead, it is necessary to determine the load or flux of As in the relevant streams as follows:

- Prohibition dam discharge water As concentration = 2 mg/L
- Prohibition Stream flow rate = 0.3 L/s
- Arsenic load down the Prohibition Stream = (1)x(2) = 0.6 mg/s
- Blackwater River water As concentration = 0.008 mg/L
- Blackwater River flow rate = 900 L/s
- Arsenic load down the Blackwater River = (4)x(5) = 7 mg/s

Hence, around 90% of the dissolved As in the Blackwater River is from sources other than the Prohibition mill site. This As load constitutes a background value, and is largely derived by groundwater moving through As-rich rock in the Blackwater catchment. Dissolved As in the Blackwater River is mainly naturally-derived. Controlling the Prohibition mill site As discharge (highly desirable) will have only a minor effect on the regional As load down the Blackwater River.

**C.10 References**


Appendix D: BIOLOGICAL IMPACTS

D.1 Effects of mining on freshwater communities

Fresh waters receiving mine discharges are often impacted from high suspended solid loads, acidity, and high metal or metalloid concentrations. While impacts arising from high suspended solid loads are common for all mining types, the impacts arising from acidity and high metal concentrations are more specific to different mine types.

Freshwater ecosystems support many different types of living organisms, ranging from microscopic bacteria to large fish. These organisms form part of a complex, interacting food web. All food webs depend on the transfer of energy. In freshwater food webs the primary energy comes from two main sources: (1) photosynthesis by algae and plants within the system and (2) terrestrial material such as fine organic matter, leaves and wood that fall or are blown into the water; this dead organic material is usually quickly colonised by microbes (e.g. bacteria and fungi). Algae, bacteria, fungi and decomposing organic material are eaten by small insects and other invertebrates, which may be consumed in turn by predatory invertebrates, crayfish, fish, and other animals such as birds (see Figure 2 in main text).

Algae (sometimes called periphyton) and other primary producers use photosynthesis to convert inorganic carbon into glucose and cellular material. Algae include a wide range of photosynthetic organisms, many of them microscopic. Other photosynthetic organisms in fresh waters include mosses (or bryophytes), liverworts, and vascular plants (or macrophytes). Algae are usually eaten by invertebrates known as grazers, which commonly include snails, mayfly nymphs, and some caddisfly larvae.

Dead plant matter, including leaves and wood, enters streams, rivers and lakes from vegetation in the riparian zone. Once in the water, microbes such as bacteria and fungi colonise this material and begin to decompose it. Certain invertebrates (shredders) readily consume the dead material and the attached microbes as their food source. Invertebrate shredders are usually not very abundant in New Zealand streams and lakes.

In addition to grazers and shredders, some invertebrates can act as collectors and others as predators. Collectors find food by brushing algae and fine organic matter from the substrate or filtering stream water. Predators catch and consume other animals. All freshwater invertebrate species serve as food for fish and other vertebrates, including amphibians and birds.

Overall, healthy freshwater food webs support multiple trophic levels (producers, consumers, predators and omnivores). Most people commonly see and value the higher trophic levels, such as fish and birds, but these cannot survive without plentiful organisms lower in the food web. Natural ecosystems depend on healthy food webs. Disruptions in any of the pathways and trophic levels, caused by mining activities, can lead to reductions and extinctions of the higher animals.

The following section provides an overview of the biological effects associated with specific components of mine drainage relevant for the West Coast or Southland. As there are limited New Zealand data, this overview also draws on international literature where appropriate.

D.2 Effects of acidity, metals and sediment

Biological impacts arising from mine drainage may be due to lowered pH, dissolved metal concentrations (acid and neutral mining), increased sedimentation or a combination of these factors (AMD). In many cases, the concentrations of metals in AMD streams are closely related to pH; low pH streams have high metal concentrations. In such cases, the biota in the stream may be responding to a combination of negative effects. It can be difficult to separate the effects of each of these stressors on different types of aquatic life. Further, metal solubilities depend on pH, dissolved oxygen, water hardness and other factors, so these variables also influence and confound our ability to predict toxicity. Similarly, the deposition of metal precipitates, especially oxyhydroxides of Fe and Al, commonly occurs when acidic waters with high metal content mix with neutral streams, and their effect on stream biota can be difficult to distinguish from effects of pH or dissolved metals.

Acidity

The pH of natural waters is typically considered to be between pH 6.5 and 8.5 (e.g. ANZECC and ARMCANZ 2000). ‘Trigger values’ for pH for various
stream types in New Zealand, based on the 20th and 80th percentiles of measured values from selected sites from the National Rivers Water Quality Network, are 7.2–8.0 (Davies-Colley 2000). The Southland Regional Water Plan (Environment Southland 2010) indicates that pH 6.5–9.0 is appropriate for lowland and hill streams, although a narrower range of 7.2–8.0 is appropriate for other streams in the Southland Region.

In these naturally circum-neutral systems, lowering of pH may result in reduced biodiversity and changed community structures (Herrmann et al. 1993) as a result of mortality, or more subtle effects such as decreased growth and reproduction (Ikuta et al. 2003). Low pH may affect freshwater organisms by disturbing cellular ionic balance and hindering osmoregulatory processes, particularly through losses of sodium, calcium, potassium, and chloride ions (Poléo et al. 1994).

However, naturally acidic waters are relatively common on the West Coast (Winterbourn and Collier 1987). These are caused by natural humic and fulvic acids leaching from the podocarp rainforest and pahiki wetlands. Stream water in these systems may get as low as pH 4, and in very extreme circumstances 3.5. Despite these very low pHs, bacteria, fungi, algae, moss and benthic invertebrates, crayfish and some native fish can survive and maintain populations (Winterbourn and Collier 1987; Harding 2005; Grieg et al. 2010). The presence of plant and animal species that are naturally adapted to low pH means that some of our New Zealand species probably have some resistance to human acidification of waters (Collier et al. 1990).

This natural adaptation to low pH has been shown to occur even within the same species, with populations from differing streams showing different tolerance to pH. For example, experiments with the ubiquitous New Zealand mayfly, Deleatidium, indicated populations from naturally low pH streams had greater survival under low pH conditions than mayflies from circum-neutral streams (O’Halloran et al. 2008). In particular, there is also considerable evidence of intraspecific variation in pH tolerance from international studies across a range of taxa, including stream insects, crustaceans, amphibians and fish (reviewed in Collier et al. 1990; O’Halloran et al. 2008).

O’Halloran et al. (2008) found that no mortality of Deleatidium sourced from naturally low pH streams occurred at pH 4.0, and that survival at pH 3.5 was markedly improved compared with mayflies sourced from circum-neutral streams. Additional toxicity testing with Deleatidium and the stonefly Zelandobious also found no mortality at pH 4.0 (unpublished data). Although these laboratory tests are based on an acute response (mortality), this same value of effect matches observations from studies on mine-impacted streams on the West Coast. Specifically, our analysis of meta-data from numerous studies (Winterbourn and Collier 1987; Collier et al. 1990; Harding 2005; Harding et al. 2006) on the West Coast indicates that sites with pH < 4 usually have markedly reduced diversity of species and therefore food webs (Figure D1).

In the case of AMD, low pH co-occurs with high metal concentrations, and lowered pH potentially increases the toxicity of metals. However, O’Halloran et al. (2008) observed that metals only appeared to contribute additional toxicity to Deleatidium sourced from naturally acidic streams at pH ≤ 3.6, and that neutralising AMD to background pH (4.6) largely removed acute toxic effects. Further testing is required to validate these observations.

### Metal precipitates on the streambed

Metals typically stay dissolved in solution until the pH increases to a point where they precipitate out. This value varies with the metal (Table D1). Precipitation of metals occurs when acidic metal-rich water mixes with receiving water of a higher pH. Iron oxyhydroxide is the predominant precipitate of AMDs. Precipitation of metals can typically result in a smothering of the substrate and algae, which alters food and habitat for freshwater invertebrates and fish. Accumulation of precipitates on the gills of fish or invertebrates may also result in direct effects.

<table>
<thead>
<tr>
<th>Metal</th>
<th>Minimum pH</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fe 3+</td>
<td>3.5–4.3</td>
</tr>
<tr>
<td>Al</td>
<td>4.9–5.4</td>
</tr>
<tr>
<td>Cu 2+</td>
<td>7.2</td>
</tr>
<tr>
<td>Zn</td>
<td>8.4</td>
</tr>
<tr>
<td>Ni</td>
<td>9.3</td>
</tr>
<tr>
<td>Fe 2+</td>
<td>9.5</td>
</tr>
<tr>
<td>Cd</td>
<td>9.7</td>
</tr>
<tr>
<td>Mn 2+</td>
<td>10.6</td>
</tr>
</tbody>
</table>

Table D1: Minimum pH values for precipitation of metal ions as hydroxides (modified from Kelly (1988) and Niyogi et al. (1999)).
**Dissolved metals in the water column**

**Aluminium (Al)**

Aluminium is a key constituent of acid mine drainage. The effects of Al on aquatic biota varies with both pH and the Al species (e.g. Al\(^{III}\)) present (Gensemer and Playle 1999). The latter is related to pH and other components present in streams such as dissolved organic matter, calcium and fluoride that form complexes with Al and reduces its toxicity (Gensemer and Playle 1999).

At lower pH (<5) Al affects ion regulation, and electrolyte loss, especially of sodium (Na\(^+\) and chloride (Cl\(^-\)) occurs (Poléo 1995; Monett et al. 2009). This effect is similar to that arising from low pH alone and authors have variously suggested that, at lower pH, pH effects dominate (e.g. Havas and Likens 1985; Havas et al. 1984), Al enhances the effects of pH (e.g. Soucek et al. 2001), or Al can ameliorate the effects of pH (e.g. Vuori 1996; Herrmann 2001). This variability in response may be in part due to species differences. For example Stewart (1993) found that the addition of Al up to 0.5 mg/L ameliorated the effect of pH on the common New Zealand mayfly species (*Deleatidium*) at pH 4, and two amphipod species at pH 5, although the Al addition reduced survival of two caddisfly species at pH 4. Similarly, Baker and Schofield (1982) found that Al had different effects on different life stages of fish, with the presence of Al being beneficial to the survival of fish embryos prior to hatching at low pH. In contrast, the presence of Al was detrimental for survival of larvae and post-larvae (Baker and Schofield 1982).

At higher although still acidic pH, respiratory effects predominate (Gensemer and Playle 1999). Respiratory effects arise from the polymerisation and precipitation of Al hydrolysis products onto gill surfaces, resulting in suffocation (Poléo 1995; Gensemer and Playle 1999). Fish are more sensitive to respiratory effects due to the release of ammonia from the gill surface, giving rise to an alkaline environment in which Al can precipitate (Poléo 1995; Gensemer and Playle 1999). Invertebrates are less sensitive to respiratory effects as their gills do not release ammonia, and respiratory effects are more likely to result from active polymerisation and precipitation in the surrounding waters as a result of rapid neutralisation of acidic drainages, i.e. initial stages of mixing zones (Soucek et al. 2001). These stages also result in greater toxicity to fish (McMahon and Pascoe 1989; Poléo et al. 1994). Decreased toxicity along the mixing zone is observed and is suggested to be due to stabilisation of the Al polymers formed (Poléo et al. 1994). Respiratory effects in both fish and invertebrates may also arise from increased mucus production as a result of Al exposure (Hall et al. 1987; Gensemer and Playle 1999).

As indicated above, toxicity of Al depends in part on the Al species present, and the availability of complexing substances. In addition, different analytical methods for determining Al concentrations, and in particular whether the toxic species (typically considered to be reactive or labile monomeric Al) are determined directly or not, further confounds determination of critical concentrations of effect for Al (Herrmann 2001; Gardner et al. 2002). Based on a literature review Herrmann (2001) found that adverse effects (ranging from insect drift rate and respiration rate to mortality) on benthic invertebrates where observed at concentrations ranging from 0.1 mg/L to 0.5 mg/L or higher, with mayfly nymphs and crustaceans the most sensitive.

In New Zealand, and particularly on the West Coast, invertebrates and some fish are reasonably tolerant to low pH (~pH 4.5) (e.g. Winterbourn and McDifffett 1996; Greig et al. 2010). In particular, a surprisingly high diversity of invertebrates frequently occurs in low pH streams, which may also contain dissolved metals. Tolerance to low pH may give rise to tolerance to moderately high concentrations of reactive Al for these invertebrates (Winterbourn and McDifffett 1996). Laboratory testing appears to confirm this higher tolerance to Al. For example, O’Halloran et al. (2008) found that mortality of the mayfly, *Deleatidium*, was lowered (although non-significantly) at only the highest concentration of Al (5.2 mg/L) at pH 4.5. Toxicity testing using *Deleatidium* spp. and other invertebrate species, a cased caddisfly (*Heleicapsyche*) and a stonefly (*Zelandobius*) at pH 4 indicated that *Deleatidium* was most sensitive, and that its sensitivity appears to vary with time of year (unpublished data). The latter may reflect differing sensitivities of different life-stages. Up to 50% mortality of early instars was observed at 21 mg/L after 96 h, whereas a similar mortality rate for larger instars was observed at >130 mg/L. In comparison, Soucek et al. (2001) observed 50% mortality of a cladoceran species at 2.9 mg/L. Corresponding with the literature, West Coast fish appear more sensitive than invertebrates to dissolved metal concentrations with no fish found in streams with Fe and Al concentrations >2.7 mg/L (Greig et al. 2010). Al concentrations <1 mg/L and total dissolved metal concentrations < 2.5 mg/L were suggested as maximal metal limits to ensure fish species were not affected by mine drainage (Greig et al. 2010).
Iron (Fe)

Iron is an essential element and is required by plants and animals for their growth, thus iron deficiencies may lead to toxic effects. However, in acid mine drainages, it is excessive Fe concentrations that may cause toxic effects, with dissolved Fe suggested to cause disruption of cell membranes and DNA damage in aquatic plants and animals (Payne et al. 1998; Sinha et al. 1997). Additionally, precipitation of ferric hydroxide may cause direct effects, such as respiratory effects in fish due to accumulation of precipitate on fish gills and indirect effects through alteration of the benthic substrate (e.g. Soucek et al. 2000a; see also section D.2.2).

Laboratory toxicity tests have typically been conducted using Fe⁰ as this remains dissolved in waters with pH > 7. Acute toxicity testing indicates that fish are most sensitive, with the average concentration at which 50% mortality is observed (LC50) being >3 mg/L (Table D2). Toxicity testing using New Zealand insects (Deleatidium, Helicopsyche, Zelandobius) indicates a greater tolerance to Fe⁰ compared with most other invertebrate species tested (Table D2). The response at different pHs may vary and in an overseas study Gerhardt (1994) found that Fe⁰ was more toxic to a mayfly, Leptophlebia marginalis, at lower pH (4.5 cf. 7), although the 50% effect concentrations for behavioural effects were still >63 mg/L, and for mortality were >89 mg/L.

Table D2: Summary of acute mortality (LC50) for different taxonomic groups of freshwater organisms exposed to Fe2+. Data collated from US EPA Aquire database, except NZ insects (unpublished data).

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Ave LC50 (mg/L)</th>
<th>Standard deviation</th>
<th>No. observations</th>
<th>No. species</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fish</td>
<td>3.0</td>
<td>0.68</td>
<td>13</td>
<td></td>
</tr>
<tr>
<td>Crustaceans</td>
<td>10.9</td>
<td>0.16</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Insects</td>
<td>14.5</td>
<td>1.4</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Amphibians</td>
<td>17.6</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>NZ Insects</td>
<td>185</td>
<td>0.5</td>
<td>6</td>
<td></td>
</tr>
<tr>
<td>Worms</td>
<td>230</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

While Fe is a primary constituent of AMD, few studies have reported that dissolved Fe contributes to observed biological responses. An exception is Schmidt et al. (2002), who found that both dissolved Fe and Al were primary contributors to observed impairment of benthic macroinvertebrate communities impacted by AMD. Similarly, Soucek et al. (2000b) found the dissolved Fe concentrations were significantly negatively correlated with macroinvertebrate species richness in a watershed impacted by AMD. The mean Fe concentrations at the sites with the highest Fe concentrations were 17.1 mg/L (pH 3.71) to 4.3 mg/L (pH 6.6). Other studies have suggested that the presence of Fe in AMD can ameliorate the toxicity of other metals, in particular Al, Cu and Zn (Soucek et al. 2001). The latter may be due to co-precipitation of the toxic metals (Soucek et al. 2001). However, this needs further research.

Anthony (1999) suggested that Fe additions (up to 3.9 mg/L) increased survival of the New Zealand mayfly Deleatidium and stonefly Zealando bus at low pH (i.e. pH 3) while the addition of Fe up to 11 mg/L did not alter survival of Deleatidium at pH 5 and 6.7.

Arsenic (As) and Antimony (Sb)

Arsenic and antimony co-occur in mine drainages and are more frequently associated with hard-rock gold mine drainages. They are chemically similar in many respects. In aquatic systems, inorganic As and Sb can occur in +3 and +5 oxidation states. Both forms generally exist together, although the +5 species is more common under oxidising conditions (upper surface of water) and the +3 state predominates under reducing conditions (lower water column and near sediments).

There are limited studies on the toxicity of Sb, although Sb⁵⁺ is probably more toxic than Sb³⁺ (Oorts and Smolders 2009). Similarly As³⁺ is considered to be more toxic to aquatic organisms, with the exception of microalgae for which As³⁺ was demonstrated to be more toxic (Levy et al. 2005).

Regardless, the following discussion on As focuses on the toxicity associated with As⁵⁺ as this is the commonest form in surface waters, including those impacted by mine drainages. It is also important to note that the behaviour of As changes with pH, in particular above pH 4 As solubility increases with increasing pH (see also section C.8).

From the limited data that exist on the response of New Zealand invertebrates and fish to As, the mayfly Deleatidium appears more sensitive than several other taxonomic groups, based on acute toxicity testing (Table D3). A single study (Jeyasingham and Ling 2000) investigated the sensitivity of different instars of New Zealand chironomids (non-biting midges) to As toxicity and found that early instars were more sensitive. The reported LC50s seem remarkably high – up to 4190 mg/L (i.e. ~4 g/L), although some chironomids can...
be found in very polluted waters. In contrast, other studies on insect species indicate 96-h LC50s in the range 6–7 mg/L for As\(^{5+}\) (Valenti et al. 2005) and 1.6–15 mg/L for As\(^{3+}\) (Canivet et al. 2001).

**Table D3**: Summary of acute mortality (LC50) for different taxonomic groups of freshwater organisms exposed to As\(^{5+}\). Data collated from US EPA Aquire database, except for New Zealand insects (unpublished data).

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Geomean LC50 (mg/L)</th>
<th>Standard deviation</th>
<th>No. observations</th>
<th>No. species</th>
</tr>
</thead>
<tbody>
<tr>
<td>NZ Insects</td>
<td>2.4</td>
<td></td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Worms</td>
<td>23</td>
<td>0.5</td>
<td>7</td>
<td>1</td>
</tr>
<tr>
<td>Fish</td>
<td>44</td>
<td>0.4</td>
<td>38</td>
<td>1</td>
</tr>
<tr>
<td>Crustaceans</td>
<td>51</td>
<td></td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>Insects(^1)</td>
<td>460</td>
<td>0.5</td>
<td>21</td>
<td>3</td>
</tr>
</tbody>
</table>

\(^1\)These results are from a single paper (Jeyasingh and Ling 2000), testing different life-stages of New Zealand chironomids. The concentrations are reported as mg/L, and seem to be excessively high.

Data on the toxicity of Sb are also limited with most studies being conducted using Sb\(^{3+}\). We are not aware of any New Zealand studies. Chronic toxicity data indicate that a fish species was more sensitive than an invertebrate (*Daphnia magna*) and an algal species, with a no observed effect concentration of 1.13 mg/L (Oorts and Smolders 2009). Limited acute toxicity data indicate that *Daphnia magna* has an LC50 of 14 mg/L whereas fish LC50s range from 9 to 30 mg/L (US EPA Aquire database).

There are no published studies on the effect on streams communities of As and Sb arising from gold mining in New Zealand. However, overseas studies of streams associated with As mines indicate elevated As and Sb can substantially impact invertebrates. For example, Mori et al. (1999) observed a 37% reduction in species diversity in mine-impacted sites (dissolved As > 2.8 mg/L and Sb > 0.3 mg/L) compared with upstream reference sites, resulting largely from the loss of oligochaetes (worms), leeches and numerous mayfly species. Over 80% of the taxa absent from reaches below mines were those exposed to high As concentrations in the sediment (9.5 mg/kg). This exposure probably occurs either through the regular ingestion of sediment during feeding (e.g. shredders and collector/browsers) or their habitat preferences for living in depositional zones. Valenti et al. (2005) also observed a sharp decrease in benthic invertebrate density, including the absence of almost all EPT (Ephemeroptera [mayfly], Plecoptera [stonefly], and Trichoptera [caddisfly]) taxa in sites downstream of a mine drainage (As 0.3–8 mg/L) compared with unimpacted upstream sites.

**Mercury (Hg)**

Mercury is more commonly an issue in alluvial gold mines.

Mercury in the aquatic environment exists mainly as complexes of mercury (II) and as organomercurials. In waterways inorganic forms of Hg may be converted by bacteria in situ to organomercury complexes (particularly methylmercury), which are more toxic and tend to bioaccumulate. Only a small proportion of total Hg is found in the dissolved form, and dissolved Hg concentrations rarely exceed 12 ng/L in fresh waters (Gill and Bruland 1990).

The limited data on the toxicity of inorganic Hg indicate that the acute toxicity of mercury (II) to freshwater invertebrate species ranges from 2.2 μg/L for the plankton *Daphnia pulex* (Canton and Adema 1978) to 2000 μg/L for a mayfly (Warnick and Bell 1969 in ANZECC and ARMCANZ 2000). Acute values for fish range from 30 μg/L for a guppy to 1000 μg/L for *Tilapia* (Deshmukh and Marathe 1980; Quereshi and Saksena 1980). Due to the high bioaccumulation of Hg, the primary effect of Hg on fish populations is most likely to be reduced reproductive success resulting from maternally derived Hg to embryonic and larval stages (Wiener and Spry 1996).

**Suspended sediment**

High sedimentation and turbidity can have both direct impacts such as smothering of benthic organisms and eggs or deterring the migratory stages of native fish, and indirect impacts such as reduced algal growth due to reduced light penetration. The direct effects of increased suspended sediment also include smothering of the gills of fish or invertebrates, and abrasion of the sensitive gill tissue. The impacts of turbidity on the food resources of consumers, such as reduction in algal productivity (Davies-Colley et al. 1992) or reduced food quality (Ryder 1989), may indirectly influence consumer abundance. Similarly, increased turbidity may alter predator–prey interactions (e.g. through reduced feeding rates of predatory fish; Rowe and Dean 1998), resulting in potentially complex indirect effects on prey species.

Surveys and large-scale experiments have shown substantial impacts of sedimentation on stream macroinvertebrate diversity and density. For
example, Quinn et al. (1992) observed a 9-45% decrease in benthic invertebrate density in West Coast sites impacted by sediment from alluvial gold mines compared with unimpacted upstream sites. Fish communities can also be influenced by sedimentation and turbidity. Richardson and Jowett (2002) observed a four-fold decrease in fish diversity in streams with high sediment loads on the East Cape of the North Island. Similarly, the occurrence and abundance of banded kokopu was reduced in North Island streams with high sediment loads (Rowe et al. 2000). This reduction may have been caused by interruption of larval migration and adult feeding (Rowe and Dean 1998).

The effect of sedimentation varies among species for both invertebrates and fish. For example, Matthaei et al. (2006) found that half of the common species of benthic invertebrates in their study streams were not influenced by experimental additions of sediment. In contrast, the densities of snails (Potamopyrgus), mayflies (Deleatidium), sandfly larvae (Austrasimilium) and the amphipod Paraleptomphophus were strongly affected, and some taxa (e.g. microcrustaceans) actually increased in abundance after sediment addition. Laboratory testing found that five invertebrate species (two mayflies (Deleatidium and Zephlebia), two caddisflies (Polyplectopus and Triplectides), a damselfly (Xanthocnemis zealandica) and a crayfish (Paraneophas planifrons)) were directly affected by suspended solids concentrations ranging from 4000 to 21 300 NTU (Suren et al. 2005). Absence of these animals from catchments with high suspended solids concentrations may be due to adverse effects on in-stream habitat (such as filling of interstitial spaces or ‘colmation’ and contamination of food sources). The most vulnerable invertebrates to sedimentation appear to be those dependent on stone surfaces for food resources (e.g. algal grazers) or habitat (e.g. filter feeders). Fish species also show differential responses to suspended sediment. For example, choice experiments with migratory juveniles of six common New Zealand species indicated banded kokopu were the most sensitive to suspended sediment, koaro and inanga showed avoidance to only very high turbidity levels, and redfin bullies and long-finned and short-finned eel species were unaffected by sediment (Boubee et al. 1997). This may partially explain why redfin bullies and eels were often the only fish present in North Island streams with high sediment loads (Richardson and Jowett 2002).

D.3 Determining outcome groups for ecological effects

pH

As discussed in Section D.2.1 the commonly adopted New Zealand water quality guidelines (i.e. ANZECC and ARMCANZ 2000) do not cover the naturally low pH streams frequently found on the West Coast. Thus to develop criteria for this Framework we have conducted both field surveys and laboratory ecotoxicological trials to determine pH tolerances.

An ecological survey of 66 streams on the West Coast indicated that species diversity is greatly reduced as pH decreases (Harding unpublished data). Naturally highly acidic streams (i.e. pH < 4) are rare, while severely impacted AMD streams can frequently be less than pH 4. Laboratory toxicity testing at different pHs consistently indicate little or no mortality for benthic invertebrates from naturally acidic streams greater than pH 4, and for some species, including the common mayfly Deleatidium, no or minimal mortality was observed at pHs 3.5-4.0 in animals sourced from naturally acidic streams (O’Halloran et al. 2008, unpublished data). Analysis of the biological survey data in combination with toxicity testing indicates that below pH 4 significant impacts on the aquatic community are likely to occur (Figure D1). Particularly as these sites are also typically accompanied by high metal concentrations (see also below). For pH between 4 and 6, impacts on the aquatic community will be more variable. For stream communities pre-adapted to naturally acidic streams the impacts are likely to be more dependent on metal concentrations. By contrast, above pH 6, the impact of mine drainages will be dependent on metal concentrations. In our experience, circum-neutral streams with high metal concentrations are generally very rare in New Zealand.
Figure D1: Diversity of stream invertebrate species decrease with low pH in 66 streams sampled on the West Coast. The dashed line indicates the pH 4 value recommended in the ecological outcomes (triangles indicate streams associated with mines, circles streams without mines).

Studies on the distribution of native fish on the West Coast indicate the presence of fish in streams with a wide range of pH, including as low as pH 4.4 (Figure D2).

Figure D2: pH tolerance range of 12 common fish species from natural streams on the West Coast, South Island (after Main et al. 1985; Taylor and Main 1987; Collier et al. 1990; Harding et al. 2006; Olsson et al. 2006).
Metals

Determining the effect of metal contaminants is typically based on water quality criteria derived from laboratory toxicity data (e.g. ANZECC and ARMCANZ 2001). Statistical distribution methods or a range of different assessment factors are applied to the results depending on the type and extent of data available. For example, ANZECC and ARMCANZ (2000) preferentially use no observed effect concentration (NOEC) data from chronic toxicity testing in a statistical distribution analysis. This is considered to give a highly reliable estimate of the concentration that is protective of a given percentage of species in the system (the 95% protection level is the most widely used, although other levels of protection are also provided). In the absence of sufficient data then a factor of 10 may be applied to the lowest chronic NOEC, or, if only EC/LC50 data are available, a factor of 100 (or 20 for essential elements such as Al and Fe) is typically applied. A summary of ANZECC water quality guidelines for selected metals are shown in Table D4.

The ANZECC water quality guidelines are typically for streams with a pH range of 6.5-8.5. While this is applicable to streams in Southland, these values have less relevance for many streams on the West Coast where the presence of naturally acidic streams may provide indigenous species with an increased tolerance to low pH (e.g. O’Halloran et al. 2008). This naturally high pH may provide some protection to these species to elevated metal concentrations (Winterbourn and McDiffet 1996).

An alternative approach to developing metal tolerance criteria is through the use of field data based on the actual distribution of aquatic organisms such as benthic invertebrates and fish. In the case of Fe and Al, this approach takes into account the inherently toxicity of these metals in natural systems, and the direct effects associated with dissolved species, and indirect effects associated with precipitation and any chemical interactions. This approach has been widely used overseas (Linton et al. 2007). For this Framework, benthic invertebrate field data from 66 sites on the West Coast formed the basis of values used in our ecological outcomes. In particular, these data show that when low pH occurs (<4) with Fe and Al concentrations above ≈2 mg/L, the diversity of stream communities is markedly reduced (Figure D3). This forms the basis for the proposed 2 mg/L value for outcomes 1 and 2 described for Coal-PAF impacts (see also following section). Our field data from 66 sites indicate that, in general, AMD-impacted sites generally have combined dissolved Fe and Al values above 2 mg/L.

Table D4: ANZECC water quality guidelines for selected contaminants (ANZECC and ARMCANZ 2000).

<table>
<thead>
<tr>
<th>Toxicant</th>
<th>95% protection µg/L</th>
<th>Reliability1</th>
<th>99% protection µg/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aluminium, pH &lt; 6.5</td>
<td>0.8</td>
<td>Low (AF 20 applied to fish LC50)</td>
<td>NA</td>
</tr>
<tr>
<td>Aluminium, pH &gt; 6.5</td>
<td>55</td>
<td>High</td>
<td>NA</td>
</tr>
<tr>
<td>Antimony (III)</td>
<td>9</td>
<td>Low (AF 1000 applied to fish LC50)</td>
<td>NA</td>
</tr>
<tr>
<td>Arsenic (V)</td>
<td>13</td>
<td>High</td>
<td>42</td>
</tr>
<tr>
<td>Iron</td>
<td>NA</td>
<td>-</td>
<td>NA</td>
</tr>
<tr>
<td>Mercury (inorganic)</td>
<td>0.6</td>
<td>High</td>
<td>0.06</td>
</tr>
</tbody>
</table>

1Reliability is specified in ANZECC and ARMCANZ (2000) and is based on data availability. NA, not available; AF, assessment factor
Figure D3: Total species richness at 66 sites on the West Coast compared with total Fe and Al values from those sites (authors’ unpublished data). The dashed line indicates the 0.2 and 2 mg/L values recommended in the ecological outcomes (triangles indicate streams associated with mines, circles streams without mines).

Greig et al. (2010) surveyed fish communities in 42 streams on the West Coast and found that no fish species occurred in streams with dissolved metal concentration > 2.7 mg/L.

However, if high concentrations of other metals occur (e.g. As, Ni, Zn, Cu, Pb) then a significant toxicity may occur below 2 mg/L.

Multivariate analysis of the water quality data provided further evidence to support the 2 mg/L and 0.2 mg/L described for outcomes 3 and 4, and 5 and 6 respectively (Figure D4). This multivariate analysis shows that streams unimpacted by AMD on the West Coast rarely have pHs < 4.2 and combined Fe and Al values > 2.3 mg/L (see Groups 5 and 6 in Figure D4). In contrast, mildly AMD impacted streams (Group 4) had pHs ranging from 3.4 to 4.3 and combined dissolved metals as high as 9 mg/L. Severely AMD impacted streams (Groups 1, 2 and 3) could have pHs as low as 2.7 and extremely high dissolved metal concentrations (Figure D4).
Linton et al. (2007) investigated the toxicity of Fe using data from North America and suggested similar values. The first of these was 0.21 mg/L for no or minimal changes in community structure (equivalent to our outcomes 5 and 6), based on the concentration that resulted in a 20% decline in the 90th %ile numbers of the most sensitive taxonomic group. The second value (slight to moderate change, equivalent to our outcomes 3 and 4) was 1.74 mg/L, based on 50% decline in the fourth most sensitive taxonomic group, the beetle family Psephenidae (Linton et al. 2007). This value was compared with field-based effect concentrations (FEC) based on classification of streams according to a Stream Conditioning Index (SCI). This comparison indicated that an FEC of 1.25 mg/L for the upper value of sites may be considered ‘comparable to below-average reference sites’ based on the SCI. These values are also consistent with outcomes 3 and 4, although this value is for the combined concentrations of Fe and Al.

**Figure D4:** Multivariate cluster analysis showing water chemistry data for 54 sites and indicating that cluster groups 1, 2 and 3 are impacted by AMD, while groups 5 and 6 have pH > 4.2, and dissolved Fe and Al < 2.3 mg/L.

D.4 Detailed ecological outcomes: Coal - PAF

**Outcome 1**

Streams with very low pH (<4) and high metal concentrations (>2 mg/L of dissolved Fe, Al) do not seem to occur naturally in New Zealand. Any water with these water chemistries will almost certainly be impacted by acid-forming coal mine drainage, and these chemistries dramatically effect algae and are acutely toxic to many invertebrates and all New Zealand fish species.

Benthic invertebrate communities will be severely impacted. As a result the number of species present will be very low and may range from 0 to 5 (Winterbourn and McDifffett 1996). The species present are liable to be very tolerant of dissolved metals and may include chironomids (e.g. Orthocladiinae), scirtid beetles, hydrobiosid caddisflies (e.g. *Psilochorema*) and notonemourid stoneflies (e.g. *Spaniocercoides*) (Winterbourn 1998). Mayflies will almost certainly be absent. Internationally, benthic invertebrates are often used as indicators of pollution and typically the most sensitive orders of insects are the EPT orders (mayflies, stoneflies and caddisflies). However, these groups are not necessarily good indicators of acid or metal inputs (Hickey and Clements 1998). On the West Coast several stonefly and caddisfly species can occur in mine-impacted water. Crayfish have been collected in naturally acidic streams as low as pH 4.1 (Ball 1987); however, they have not been found in mine-impacted waters with pH < 5.

No native or introduced fish species seem to be able to survive in pH < 4 (Figure D4). Harding et al. (2006) investigated 52 sites in the Ngakawau, Waimangaro and Mokihinui rivers with pH ranging from 2.9 to 7.9, and a total of 12 fish species were collected. No fish were found in any streams with pH < 5.4. More
recently Greig et al. (2010) found two native fish species in waters as low as pH 4.4. While animals seldom occur in such waters, algae and microbes can still exist in the extreme conditions. Some taxa of algae, including *Euglena mutabilis* and *Klebsormidium acidophilum*, are abundant in highly acidic streams provided that other factors (stable substrate, lack of scouring flows) are suitable (Bray 2007). Additionally, aquatic fungi, which decompose organic matter in streams, can also be common and active in highly acidic, metal-rich streams (Niyogi et al. 2008). Although high rates of primary production and decomposition would support animals in most pristine streams, that does not occur in highly acidic streams because of the limited tolerance of animals discussed above.

We have suggested that a dissolved metal value of >2 mg/L (Fe and Al) should indicate likely degraded stream communities (see previous section). If high concentrations of other metals occur (e.g. As, Ni, Zn, Cu, Pb) then a significant toxicity may occur below 2 mg/L.

**Outcome 2**

Waterways with pH < 4 but very low dissolved metals (<2 mg/L) are very uncommon. Some natural pahiki wetlands have pH 3.5 (Winterbourn and Collier 1987). A survey of 66 sites on the West Coast recorded two sites with pH 3.4–3.9 and dissolved Fe < 2 and dissolved Al < 2 mg/L (Harding unpublished data). These sites have 4-5 benthic invertebrate taxa, and were dominated by the stonefly *Spaniocercoides*, the caddisfly *Oxyethira*, chironomids and oligochaete worms. A 1985 survey of fish communities in south Westland observed inanga (*Galaxias maculatus*) in a stream with pH 3.8 (Main et al. 1985) but subsequent surveys indicate this was an exceptional observation.

Algal and microbial communities that can survive in acidic conditions can also often tolerate high metals. Thus, similar communities and activities are found between Outcomes 1 and 2.

**Outcome 3**

Waterways with pH ranging from 4 to 6 may have impacted benthic communities, depending on the concentrations of dissolved metals. Field data from 66 streams from the West Coast indicated that at this pH, dissolved metals have frequently precipitated but might reach concentrations of 3 mg/L (Harding unpublished data). Benthic invertebrate communities will still be impacted with 2–10 species present (Winterbourn and McDiffett 1996). Communities include species found in more impacted waters, but now might also include the stonefly *Zelandoperla*, amphipods *Paraleptomphophos* and other diptera (e.g. *Muscidae*).

Although fish occur in streams within this pH range (Figure D1), their tolerance of dissolved metals under these conditions is unknown. Collier et al. (1990) recorded inanga (*Galaxias maculatus*) in pH 4 and giant kokopu (*G. argenteus*), longfin (*Anguilla diefenbachii*) and shortfin eels (*A. australis*), banded kokopu (*Galaxias fasciatus*), common bully (*Gobiomorphus cotidianus*), and brown mudfish (*Neochanna apoda*) in waters with pH < 4.5. Koaro (*Galaxias brevipinnis*) have also been observed in water as low as pH 4.1 (Olsson et al. 2006).

Algal and microbial communities also become more diverse as pH increases into this range (Bray et al. 2008; Niyogi et al. 2008). The effects of dissolved metals on these communities are not well understood, but in general, algae and fungi are more tolerant of dissolved metals than animals are. In this range, metal hydroxide precipitates can often influence algae on rocks and microbes on organic matter. At sites with high rates of deposition, primary production by algae and decomposition by microbes can be greatly impaired (Niyogi et al. 2002, 2008; Bray et al. 2008). In these cases, animals may not have adequate food supplies because of the indirect effect of mine drainage on lower trophic levels.

**Outcome 4**

On the West Coast many natural streams have this water chemistry (Collier et al. 1990). The average number of benthic invertebrate species can be >15 (Collier and Winterbourn 1987; Winterbourn and McDiffett 1996). Communities will frequently be dominated by the common New Zealand mayfly *Deleatidium*. Data from 66 West Coast streams indicate that *Deleatidium* occurs at pH > 4.2; however, a wide range of mayfly, stonefly, caddisfly and other groups should occur in these streams (Harding unpublished data). Crayfish will occur in these water chemistries.

Many native fish species should be tolerant of these water chemistries (Figure D4). For example, Harding et al. (2006) recorded koaro (*Galaxias brevipinnis*) and upland bullies (*Gobiomorphus breviceps*) at pH 5.4 and 5.8, respectively.

Algal and microbial communities in streams in this category should be relatively diverse and functional. Sites with deposition of metal oxide precipitates, however, may limit biomass and functioning of these communities, which in turn would influence animals that feed on them.
Outcome 5
Streams with neutral pH and elevated metals are generally rare for PAF coal types. Depending on the metal, benthic invertebrate communities may or may not be noticeably impacted. The presence of metal hydroxides will affect communities, however. Without metal hydroxides present, we would generally expect a relatively healthy community (e.g. > 15 taxa), dominated by mayflies.

Fish communities might not be strongly impacted, again depending on the metals. For example, Cu, Zn and As may be acutely toxic, while Al and Fe may present chronic effects. The diversity of fish communities will vary with distance from the sea, migratory barriers and other factors.

Algal communities in streams of neutral pH are generally very diverse. Additionally, microbial communities should be diverse and functioning. As discussed for other outcomes, algae and microbes are usually more tolerant of metals than animals are (Bray et al. 2008; Niyogi et al. 2008).

Outcome 6
Under these water chemistry conditions normal diverse invertebrate communities should be present (e.g. 15–30 taxa) (Winterbourn and Collier 1987).

Fish communities will probably be unimpacted by mining and may be affected by other conditions. Harding et al. (2006) recorded 12 fish species in sites on the West Coast < 15 km from the sea.

As discussed for Outcome 5, algal and microbial communities should be very diverse and active in streams of neutral pH.

D.5 Detailed ecological outcomes: hard-rock gold
New Zealand field data on the effect of metals associated with hard-rock mining on stream communities is rare. As a result the following outcomes have been based on a review of international literature. It is possible that New Zealand species may be more or less tolerant to some impacts.

Outcome 1
The most severe impact on stream ecosystems will occur when there are high concentrations of arsenic (As) and/or antimony (Sb). Fish are unlikely to be present and few macroinvertebrates of very limited diversity will be found in these streams. Algae and microbes, however, will be present, and even in high biomass in some cases. These communities tend to be dominated by a few taxa that are able to tolerate the stressful conditions.

Outcome 2
At higher concentrations of As and Sb (> 1 mg/L) sub-lethal and lethal effects on algae, invertebrates and fish will be expected. A lower diversity and abundance of these organisms would be likely.

Outcome 3
At moderate dissolution of As and Sb, some effects arise, although at concentrations < 1 mg/L, Sb does not exert any visible toxicity (Duran et al. 2007). Reproductive effects in juvenile fish and mortality in amphibians and invertebrates have been observed at As concentrations as low as 0.2 mg/L. In addition, some impacts may be observed on algal and invertebrate species, although diversity is expected to be moderately high.

Outcome 4
Refer to Outcome 6 in section D.3.

D.6 Biological monitoring
What should be measured in a monitoring programme?
A number of different characteristics can be used to determine the ‘health’ of a freshwater ecosystem. Most consent and monitoring programmes require collection of a suite of water chemistry variables, e.g. pH, conductivity, DO, turbidity and metals. However, it’s important to remember that water quality is not usually what is trying to be protected. Water quality gives an indication of the likely impacts on the ‘health’ of the waterbody. The most direct and efficient ways to measure ecosystem health are to measure the plants and animals (the biota) and how they are functioning (i.e. in RMA terms, ‘the life supporting capacity’ of the waterway).

Many monitoring programmes collect organisms (e.g. macroinvertebrates) and identify the species that are present. Freshwater ecosystems have several different communities of organisms, and there are advantages and disadvantages to monitoring these different groups of organisms (Table D5).

Additionally, one can measure the rates of ecosystem ‘processes’ or ‘functions’ as a measure of health. Some ecosystem processes, such as primary production or decomposition, reflect the flow of...
energy in food webs in fresh waters and are thus good measures of health (Gessner and Chauvet 2002). For example, iron hydroxide precipitating onto leaves may reduce bacterial and fungal decomposition resulting in leaves not being broken down as they should be and less food for invertebrates; this may result in less or poor-condition invertebrates as food for fish, crayfish and birds.

Table D5: Advantages and disadvantages of using different organisms for the biological monitoring of mining impacts (after Boothroyd 1999; Biggs and Kilroy 2000; Boothroyd and Stark 2000; Bray 2007).

<table>
<thead>
<tr>
<th>Organism</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>Algae</td>
<td>• Standard sampling protocols available</td>
<td>• Identification difficult</td>
</tr>
<tr>
<td></td>
<td>• Strong community response to AMD impacts</td>
<td>• Few taxonomists commercially available</td>
</tr>
<tr>
<td></td>
<td>• Moderate diversity of species with high AMD tolerance</td>
<td>• Rapid generation time reduces integration of long-term water quality trends</td>
</tr>
<tr>
<td></td>
<td>• Rapid generation time leads to rapid response to change</td>
<td>• Ecological knowledge still developing</td>
</tr>
<tr>
<td></td>
<td>• Individuals are sedentary</td>
<td></td>
</tr>
<tr>
<td>Macroinvertebrates</td>
<td>• Ease of sampling</td>
<td>• Patchy distributions within streams</td>
</tr>
<tr>
<td></td>
<td>• Ease of identification</td>
<td>• Taxonomic knowledge varies between groups</td>
</tr>
<tr>
<td></td>
<td>• Established sampling protocols</td>
<td>• Some species relatively mobile (e.g. drift downstream)</td>
</tr>
<tr>
<td></td>
<td>• Commercially available sampling and taxonomic skills</td>
<td>• Very few species tolerate high AMD impacts</td>
</tr>
<tr>
<td></td>
<td>• Strong community response to AMD</td>
<td>• Respond to multiple pollutants</td>
</tr>
<tr>
<td></td>
<td>• 1-2 year life cycle integrates water quality over time</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Relatively sedentary</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• High diversity of species</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Background ecological knowledge good</td>
<td></td>
</tr>
<tr>
<td>Fish</td>
<td>• High public awareness of aesthetic, recreational and economic value</td>
<td>• Highly mobile and may not reflect local conditions</td>
</tr>
<tr>
<td></td>
<td>Field identification possible</td>
<td>• Potential distribution limited by in-stream migratory passage</td>
</tr>
<tr>
<td></td>
<td>• Taxonomy well developed</td>
<td>• Strong inland diversity decline</td>
</tr>
<tr>
<td></td>
<td>• Sensitive to AMD and turbidity</td>
<td>• Relatively few species</td>
</tr>
</tbody>
</table>
| Freshwater communities

**Microbes**

Microbial (bacteria and fungi) communities are not often used in monitoring because of the challenges in measuring and identifying microbes. Most freshwater bacteria have similar morphology (rod-shaped) and other characteristics as well (e.g. Gram-negative staining). Recent molecular advances (e.g. Nikolcheva et al. 2003) now allow characterisation of microbial communities, but costs and the fact that they have not been well studied in natural systems are still obstacles to their use in monitoring fresh waters.

**Algae and plants**

Algal communities are somewhat similar to microbial communities in that identifying species is a challenge. On the other hand, certain algal species can be common and even characteristic of acidic waters, including those affected by mining. In this Framework document, we include data from New Zealand (Bray 2007; Bray et al. 2008) and elsewhere (Kelly 1988; Niyogi et al. 2002) on algal communities in streams affected by mine drainage. We have not included data on ponds and lakes as these have not been well studied in New Zealand. Compelling arguments have been made for the use of algae in the biological monitoring of human impacts, including AMD (Biggs 2000; Biggs and Kilroy 2000; Bray 2007), but the current limitations...
in both ecological knowledge and commercially available sampling and identification skills (Table D2) preclude their inclusion in this manual. A detailed description of algal sampling methods for biological monitoring can be found in Biggs and Kilroy (2000).

**Invertebrates**

Invertebrates (sometimes called macroinvertebrates or benthic invertebrates) have several key advantages over other communities for biological monitoring including:

1. **Ease of sampling and identification**
2. **Their long larval (juvenile) stage within fresh waters (months to several years) indicates water quality conditions over time, and**
3. **Species vary in their tolerance of stressors including pH and metals.**

Consequently, communities of macroinvertebrates are the most commonly used biological indicator of freshwater condition or health worldwide. In New Zealand, the use of macroinvertebrates in biomonitoring is especially well developed (reviewed by Boothroyd and Stark 2000). Every regional council conducts annual macroinvertebrate monitoring as part of State of the Environment (SOE) reporting, and there are well-developed indices for the tolerance of invertebrates to common pollutants (largely agricultural). Ecological knowledge of stream invertebrates is extensive, and sampling and identification skills are commercially available. Comprehensive field and laboratory protocols have been developed for stream macroinvertebrate sampling (Stark et al. 2001) and these are referred to in this framework. Sampling macroinvertebrates in ponds and lakes can be more difficult, although a number of standardised sampling devices exist (e.g. ponar grab sampler). The use of macroinvertebrates as indicators of lake health is less common.

**Fish**

Fish communities are an attractive choice for monitoring, as people generally consider their presence and abundance an important indicator of a waterway’s health or condition. However, using fish for biological monitoring poses several significant challenges (Table D5). First, fish are much more difficult to collect than other biota, and require specialised equipment. Second, fish are generally quite mobile, so the presence of fish at a given site does not mean that they can survive well there (they could just be passing through). Finally, the distributions of New Zealand native fish are strongly influenced by migratory passage, including distance from the sea (McDowall 1996). However, models based on GIS analysis of catchment characteristics and local habitat features (e.g. Joy and Death 2002, 2003; Eikaas et al. 2005) can generate predictions of fish communities occurring in stream reaches (not for lakes). As these models do not incorporate mining activities, the predicted communities they generate can be compared with those actually occurring in a stream in order to assess levels of mining impact (e.g. Harding et al. 2006). Fine-resolution models and the associated software are available for some North Island regions (e.g. Joy 2005, 2006), and nationwide predictive models have recently been developed (Leathwick et al. 2005). These models are likely to increase the usefulness of fish as biological monitoring agents in the future.

**Ecosystem processes**

In addition to biological communities, increasingly freshwater scientists are proposing the use of ecosystem processes as a good way to assess the health of freshwaters (Young et al. 2004, 2008). The two main processes that could be monitored are (1) ecosystem metabolism and (2) the breakdown of leaf litter and wood. These two often account for the major energy sources within freshwater food webs. The advantages and disadvantages of these techniques are outlined in Table D3.

Ecosystem metabolism is a combination of primary production by algae and plants and respiration by consumers; i.e. a measure of the production and consumption of organic carbon in a freshwater ecosystem. Because primary production involves the use of CO₂ and production of O₂, and respiration is the reverse, ecosystem metabolism can be quantified by measuring the change in CO₂ and O₂ concentrations in the water column. In practice, O₂ is significantly easier to measure than CO₂ and is the most commonly used method (Young et al. 2008). Numerous methods exist to quantify O₂ flux in streams (as compared to lakes), which involve either measuring changes in O₂ within a river, or enclosing part of the streambed in an airtight chamber and measuring O₂ flux within the chamber. The methods and relative merits of these approaches are beyond the scope of this document, and readers are referred to Young et al. (2008) for a thorough review of methods in streams and rivers. The utility of using ecosystem metabolism measurements for biological monitoring is primarily limited by the difficulty in
assessing $O_2$ changes in small, turbulent streams with low productivity – which are common on the West Coast (Table D6).

Litter and wood breakdown in streams occurs through biological processing by invertebrates, bacteria and fungi, and physico-chemical processes such as leaching and abrasion; all of which are potentially influenced by mining activities. Litter breakdown is most often measured using a simple litter bag technique, where a known weight of leaves is placed within coarse mesh (0.5 – 1 cm) bags and secured to the streambed for a period of time (e.g. 2 weeks). Bags are then removed, leaves are dried in an oven for 48 h, and weighed in order to calculate the percentage of leaf mass lost over a given period of time. Replicate leaf bags (e.g. 3-5) are used in each stream to estimate average breakdown rates. Many studies remove groups of leaf bags after various lengths of exposure to enable curves of mass lost over time to be calculated (see Benfield 2007) for a detailed description of methods). However, Young et al. (2008) suggest a single retrieval period of around one month may be adequate for biomonitoring purposes.

Other methods of assessing litter breakdown for biological monitoring include measuring leaf toughness, the mass lost of wooden sticks (e.g. tongue depressors) and the tensile stress of cotton strips after a period of submersion (Young 2006). The breakdown rate of cotton strips and wood sticks have low variability among replicates compared with litter bags, which makes them a potentially useful assay of detritus breakdown rates (Young 2006). However, both cotton strips and wood mass loss primarily indicate microbial processing rates, rather than breakdown by invertebrates. In addition, measuring the tensile strength of cotton requires the use of specialised equipment.

Leaf and wood breakdown trials have been conducted in streams impacted by AMD in New Zealand (Barnden and Harding 2005) and widely overseas (e.g. Maltby and Booth 1991; Niyogi et al. 2001; Simmons et al. 2005), and generally slower breakdown rates have been observed in mining-impacted streams than in reference streams. However, Barnden and Harding (2005) observed significant deposition of metal precipitates within leaf packs. Consequently, in streams affected by these precipitates they found leaf toughness was a better indicator of leaf breakdown than mass lost (Barnden and Harding 2005). The use of litter bags as a biomonitoring tool in West Coast streams may, however, be limited by unpredictable and severe flood events that frequently destroy deployed leaf bags.

**Table D6: Advantages and disadvantages of using litter breakdown and ecosystem metabolism as functional indicators of stream health (after Young et al. 2008).**

<table>
<thead>
<tr>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>Litter breakdown</td>
<td>Breakdown rate influenced by numerous factors</td>
</tr>
<tr>
<td>• Simple, inexpensive methods</td>
<td>• Leaf packs prone to loss in floods</td>
</tr>
<tr>
<td>• Strong background scientific knowledge</td>
<td>• Only representative of location of leaf packs within a stream reach</td>
</tr>
<tr>
<td>• Can be used in any aquatic habitat</td>
<td>• Difficult to convey results/implications to general public</td>
</tr>
<tr>
<td>• Established criteria exist to link breakdown to ecosystem health</td>
<td></td>
</tr>
<tr>
<td>Ecosystem metabolism</td>
<td>Specialised equipment required</td>
</tr>
<tr>
<td>• Measurements representative of entire reach</td>
<td>• Expensive equipment must be left in streams for at least 24 h</td>
</tr>
<tr>
<td>• Directly assesses energy flow in food webs</td>
<td>• Difficult to measure in small, turbulent streams with low productivity (i.e. many South Island streams)</td>
</tr>
<tr>
<td>• Short-duration (2-day) studies</td>
<td></td>
</tr>
<tr>
<td>• Importance of $O_2$ in streams easy to convey to public</td>
<td></td>
</tr>
</tbody>
</table>

**Stream monitoring - macroinvertebrate sampling protocols**

Numerous sampling devices and methods are used to sample macroinvertebrates in streams, each with their advantages and disadvantages. Fortunately, standardised field and laboratory protocols have been developed for stream macroinvertebrate sampling in New Zealand (Stark et al. 2001), and these form the basis of sampling and laboratory procedures suggested in this framework.

**Qualitative and semi-quantitative sampling**

This sampling is designed to assess the diversity or relative abundance macroinvertebrates within a
stream reach. It does not provide estimates of the density or absolute abundance of species (see Quantitative sampling below). Qualitative sampling generally uses a ‘kick net’ which consists of a 0.5-mm-mesh net attached to a ‘D’ or triangular shaped frame ~30-40 cm wide. Stream substrate is disturbed < 0.5 m above the net, and any dislodged invertebrates drift downstream and are caught in the mesh (Figure D5).

**Figure D5**: Qualitative sampling of macroinvertebrates on the Dennistion Plateau using a triangular kick net.

We recommend following protocol C1 of Stark et al. (2001), which involves taking a composite sample of five kicks at each site, with each kick disturbing 0.1-0.2 m² of streambed. Conducting each of these kicks in a different microhabitat (e.g. pool, backwater, macrophyte bed, accumulation of leaves) increases the likelihood of capturing the majority of species present at a site. Protocol C1 is most relevant to West Coast and Southland streams, as they contain ‘hard bottomed’ bed substrates of gravels, cobbles and boulders. However, if soft-bottomed streams are encountered (those dominated by sand, silt, mud, macrophytes or wood), sampling should follow protocol C2 (Stark et al. 2001). Samples are preserved in 70% ethanol until processing in the laboratory.

Laboratory protocol P1 yields semi-quantitative data, based on the coded abundance of each species. In practice, this involves estimating the ranked abundance of each different species within the sample (Table D7). This procedure can be simplified by not estimating coded abundance if purely qualitative data (i.e. a list of species present at the site) are required.

<table>
<thead>
<tr>
<th>Coded abundance category</th>
<th>Coded abundance score</th>
<th>Animals per sample</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rare</td>
<td>1</td>
<td>1-4</td>
</tr>
<tr>
<td>Common</td>
<td>5</td>
<td>5-19</td>
</tr>
<tr>
<td>Abundant</td>
<td>20</td>
<td>20-99</td>
</tr>
<tr>
<td>Very abundant</td>
<td>100</td>
<td>100-499</td>
</tr>
<tr>
<td>Very very abundant</td>
<td>&gt;500</td>
<td>&gt;500</td>
</tr>
</tbody>
</table>

Table D7: Coded abundance categories for SQMCI and other semi-quantitative metrics. Animal counts are based on combined kick-net samples of ~0.5 m² of streambed (see above) (after Stark 1998).

1 For use in SQMCI calculations (see quantitative metrics).

**Quantitative sampling**

Quantitative sampling estimates densities of species in a stream reach, and is achieved through the sampling and enumeration of macroinvertebrates within a known, defined area. Numerous types of quantitative benthic samplers exist, each similar in that they define and constrain an area of streambed within a frame or quadrat. Surber samplers, which consist of a frame of a known area affixed perpendicular to a net, are the
most commonly used in New Zealand (Stark et al. 2001). Benthos is disturbed within the frame, and solid side panels funnel dislodged invertebrates downstream into the net (Figure D6).

Figure D6: Surber sampler being used to assess the density of macroinvertebrates in an urban stream.

We recommend following the methods of field protocol C3 to conduct sampling, and laboratory protocol P3 to process samples (Stark et al. 2001). Because macroinvertebrates often exhibit patchy distributions (both in presence and abundance), replicate quantitative samples are required in order to calculate the average density of invertebrates. The degree of replication depends on the study question and sampling logistics, and is discussed in the sampling designs below.

Stream habitat assessment

It is often useful to measure stream habitat parameters during biological sampling to quantify potentially confounding variables and assess changes in physical habitat between sampling dates. Stream habitats consist of catchment, valley, reach and local-scale variables, each of which is important to assess. Harding et al. (2009) provide a detailed account of which physical parameters to include in habitat assessment, and a choice of three standard protocols that differ in the intensity of data collection. Protocol one (P1) is a rapid, subjective assessment ideal for site selection/characterisation purposes, but does not generate any quantitative data. Protocol two (P2) generates categorical data which can be analysed further, and is relatively quick to complete (< 1 hour). Protocol three (P3) generates replicated quantitative data but is time-intensive and may only be necessary for the most detailed assessments of environmental effects.

Stream monitoring - sampling design

Sampling designs for biological monitoring should be driven by the data required to answer the question being addressed. In reality, however, sampling designs are also influenced by budgetary logistical constraints, access to sampling sites and the expertise of available researchers. In general, sampling designs need to consider (Biggs and Kilroy 2000):

- What are the objectives/questions of the study
- Where to sample
- When and how often to sample
- Variables to consider
- Sampling protocol, e.g. quantitative or qualitative
- Laboratory processing and analysis
- Approaches to data analysis
- Reporting requirements.

Assessment of environmental effects (AEE) of a potential pollutant

Often biological monitoring aims to assess the impact of a stress that occurs at a well-defined point in the landscape, such as mine drainage. In
such cases, sampling should be concentrated on a few key stream reaches within the zone of potential impact, with appropriate non-impacted reference (control) sites. In this section we outline the sampling considerations for monitoring the effect of these point-source impacts. These sampling protocols may also be relevant for assessing the success of treatment activities.

Spatial and temporal sampling design

AEE questions are best suited to a comparison of impacted reaches (or streams), with controls in both space and time (Green 1979). This sampling design is commonly referred to as a BACI design (Before, After, Control, Impact; Figure 6 main text). However, when sampling prior to the impact is not possible, study designs should focus on spatial comparisons between unimpacted (control) and impacted reaches (i.e. a CI design).

Spatial comparisons can be made through sampling upstream of an impact, or in an adjacent unimpacted tributary or catchment. The most rigorous BACI designs for streams involve the use of both these spatial controls, as this enables the detection of effects that can be transmitted upstream of an impact, such as interrupted fish migration. Selecting several control sites in different streams creates a more rigorous assessment of baseline reference conditions (Underwood 1994), and including downstream impact sites provides information on the downstream extent of impacts. Including more spatial control sites is essential if sampling before an impact is not possible.

In the very minimum, BACI designs should include multiple upstream and downstream sites, both before and after the impact (see Figure 6 main text). Some current consent conditions require sampling of a single upstream site and a single downstream site. Unfortunately, this information is virtually useless and tells you nothing about any impact.

If a single upstream site does differ from a single downstream site all that you can conclude is that they are different. To confidently attribute the cause of any difference to a particular impact is very difficult. By contrast, if multiple upstream sites and controls can be compared with multiple downstream sites then differences can be much more confidently attributed to impacts.

If time and resources allow, multiple post-manipulation sampling periods are advisable, to detect long-term changes in stream biota. This is especially important in assessing the effectiveness of treatment activities, where it may take some time (e.g. multiple generations of stream insects) for new species to disperse into the remediated habitat.

Regardless of the BACI design, care must be taken to select control sites with similar physico-chemical conditions to enable valid comparisons. In particular, flow regime (floodling and drying), land use, stream size (width, depth and discharge) and substrate type can each have strong influences on stream communities that may hinder the assessment of environmental impacts. The selection of appropriate control reaches can be aided by desktop analyses of stream databases (e.g. FWENZ and REC), using existing data (e.g. from regional council SOE monitoring) and by ground-based assessments prior to sampling. Once selected, the physical similarity of control and impacted reaches can be evaluated by conducting assessment of stream habitat using standard protocols (Harding et al. 2009).

Spatially extensive surveys

Sometimes biological monitoring programmes aim to establish the background health of stream communities across a broad region or to place mining impacts in the context of other environmental influences on stream communities. For example, a spatially extensive fish survey was recently conducted in North Buller, Greymouth and Reefton to assess the relative effects of natural and AMD-generated acidity on the distribution of freshwater fish (Harding et al. 2006).

Site selection in spatially extensive surveys is determined by the sampling objectives, but in general, will involve a larger number of sites (e.g. 10-20) that differ in level of impact across a region. The selection of these sites should consider:

- Incorporating an even spread of sites across the impact gradient
- Minimising other physical and chemical differences between sites (such as land-use, stream size, flow regimes, barriers to fish migration, etc.)
- Selecting sites across the landscape such that impacted sites are not clustered together.

Measuring background stream habitat properties is especially important in spatially extensive surveys, as many confounding variables may need to be quantified in order to produce robust conclusions.

The logistical constraints of spatially extensive surveys mean sites are often only sampled once or
at best seasonally. Fortunately as seasonal variation in New Zealand stream communities is not as extreme as in Europe and North America, sampling 1-4 times per year (e.g. summer and winter) is sufficient to produce representative data (Boothroyd and Stark 2000). If detecting long-term trends across a region is important, then sampling should be conducted in the same season (or month) each year so temporal trends are not confounded by seasonal differences.

For example in the survey of fish distributions of Harding et al. (2006), sampling included multiple sites within two catchments: (1) potentially affected by mining, (2) a pristine catchment without mining. This sampling generated 52 sites along a gradient of both natural and mining-affected streams. Each site was sampled once, and the effect of natural and anthropogenic acidity was assessed by comparing (a) fish communities in pristine sites with those in mining-affected sites, and (b) fish distributions to those predicted by a GIS model based on species preferred habitat.

How many samples to take?

BACI and CI sampling designs rely on statistical comparison of impacted and control reaches, which requires data generated by replicated quantitative samples. Because of the small spatial scale of sampling (i.e. a small number of streams and sites within streams), a high level of sample replication should be attainable for a given time and money investment. Benthic invertebrates have notoriously uneven distributions across a streambed, so we suggest at least five semi-quantitative or quantitative samples be taken at each sampling site, with each sample randomly located among similar habitat type (usually riffles).

Broad-scale surveys

Often researchers wish to assess the relative influence of environmental parameters on stream ecosystems, or provide long-term trends of ecosystem health. For example, how are freshwater fish communities in West Coast streams influenced by pH? Such questions are best answered with broad-scale surveys, where multiple streams are sampled; each being a replicate of how local environmental conditions influence stream communities. Although selecting sites to sample in a large-scale survey is daunting, and conclusions can be influenced by a suite of confounding variables, several considerations can focus site selection:

The broader the spatial scale, the broader the inference

For example, conclusions from sampling within the Buller District alone are only relevant for the Buller District, whereas conclusions drawn from sampling in Hokitika, Grey and Buller districts are likely to be relevant to the entire West Coast.

The spatial arrangement of streams

Stream communities often differ across geographic localities due to differences in the underlying environment, and historical colonisation, extinction and evolutionary processes. As these variations can confound the detection of impacts of interest, researchers should make sure site characteristics are spread across the landscape (Figure D7).

Figure D7: Hypothetical example of poor and good spread of sites in space. In (a), site properties (indicated by degree of shading) are strongly confounded by latitude. Example (b) shows good stratification of site properties across the geographic range.

Fitting loose site categories can aid in site selection

The ideal sampling design covers similar numbers of streams across the range of the variable in consideration. One way to achieve this is to divide a gradient into broad categories, and select a similar number of streams to sample in each category. For example, selecting 30 streams to sample gradient of pH may benefit by selecting 10 streams impacted by AMD, 10 naturally acidic streams and 10 circum-neutral streams. This is likely to generate a gradient of pH with an even spread of sites.

Measuring potentially confounding variables

In addition to sampling biological parameters, such as macroinvertebrate communities, large-scale surveys should quantify physical habitat parameters at the reach, valley and catchment scales. This enables the inclusion of environmental
variables in analyses, and the subsequent assessment of potentially confounding variables.

**Summary metrics**

Complex data collected during surveys of macroinvertebrates (or other organisms) need to be summarised into simple metrics in order to generate conclusions. Here we present a selection of the more commonly used metrics (both qualitative and quantitative) that are derived from data on biological communities. Many are transferable across different types of taxa, but EPT and the three MCI metrics are specific to macroinvertebrate data only.

**Qualitative metrics**

**Species richness**

The total number of taxa present in a stream reach. Often, anthropogenic stressors reduce the number of taxa able to survive in a community, resulting in a decrease in species richness. However, species richness does not detect changes in the relative abundance or dominance of particular taxa in a community. Thus species-richness statistics are often combined with data on the relative abundance of taxa to create diversity indices (see below).

**% EPT**

The %age of species in a community that are Ephemeroptera (mayflies), Trichoptera (caddisflies) and Plecoptera (stoneflies) species. EPT taxa generally are less tolerant of pollutants than other aquatic orders. Thus, a decrease in % EPT may indicate a decrease in stream health arising from pollutants.

**MCI**

The Macroinvertebrate Community Index is a qualitative index that uses predetermined pollution tolerance scores of the species present in a stream to assess the health of waterways. Boothroyd and Stark (2000) contains a detailed description of the MCI and associated quantitative metrics, and Stark et al. (2001) provides revised MCI scores for New Zealand stream invertebrate taxa. MCI is calculated as:

\[
MCI = 20 \sum a_i / S
\]

where \(a_i\) = MCI tolerance score for the \(i\)th taxon, and \(S\) = total number of taxa.

Streams with high MCI scores are considered healthy, with lower scores indicating progressively polluted systems (Table D8).

**Table D8: Interpretation of MCI, SQMCI and QMCI scores for invertebrate communities in riffles (after Boothroyd and Stark 2000).**

<table>
<thead>
<tr>
<th>Interpretation</th>
<th>MCI score</th>
<th>SQMCI and QMCI score</th>
</tr>
</thead>
<tbody>
<tr>
<td>Good</td>
<td>&gt; 120</td>
<td>&gt; 6.00</td>
</tr>
<tr>
<td>Moderate</td>
<td>100-119</td>
<td>5.00 - 5.99</td>
</tr>
<tr>
<td>Fair</td>
<td>80-99</td>
<td>4.00 - 4.99</td>
</tr>
<tr>
<td>Poor</td>
<td>&lt; 80</td>
<td>&lt; 4.00</td>
</tr>
</tbody>
</table>

The index was originally developed to assess organic pollution in agricultural landscapes but has proved useful in the assessment of additional stressors on stream communities. However, the MCI performed poorly in assessing Coromandel streams with metal contaminants (Hickey and Clements 1998), as tolerance scores were not appropriate to assess vulnerability of taxa to heavy metals. Thus, the MCI and its quantitative derivatives (see below) should be used with caution for the biological monitoring of mining impacts.

**Quantitative metrics**

**Density**

The number of animals per unit area of streambed, calculated as the total number of animals in a sample divided by the area of streambed sampled.

**Biomass**

Biomass is the total mass of living organic matter per unit area of streambed. This can give a more accurate picture of the function of invertebrate communities as grazing, litter breakdown and other ecosystem functions often depend on both the size and abundance of organisms. As biomass is the amount of organic matter, the inorganic component of animal tissue (especially those with cases or shells) is usually subtracted in a process called ashing. First, the sample is dried in an oven at 60°C for 48 h and then weighed (the dry weight). This ‘dry’ sample is then combusted in a furnace 550°C for 4 h to leave only non-combustible inorganic material (the ‘ash’). Intuitively, the ash-free dry mass (AFDM) is calculated by subtracting the dry weight from the ash weight.
**Diversity indices**

These provide a more inclusive estimate of diversity than species richness as they incorporate species richness, abundance and evenness (the similarity of abundance of individuals within each taxon) into a single metric. This number can then be compared across sites or over time. Numerous diversity indices exist, but the more commonly used ones include:

- **Margalef’s index** is a simple and intuitive diversity index, and is calculated by dividing the total number of taxa by the total number of individuals using the equation:

  \[ d = \frac{(S-1)}{\ln N} \]

  where \( S \) is the total number of taxa and \( N \) is the total number of individuals. However, Margalef’s \( d \) is sensitive to sample size, and therefore unsuitable if comparisons incorporate different sampling efficiencies or have markedly different benthic densities.

- **Often highly impacted systems are dominated by a few key taxa that tolerate high levels of pollution. Thus, diversity indices that quantify species dominance are useful for biomonitoring activities. The Berger–Parker index, which is the proportional importance of the most abundant taxa, is an easy-to-calculate, intuitive index of the relative dominance of taxa:**

  \[ \text{Berger-Parker index } d = \frac{N_{\text{max}}}{N}, \]

  where \( N_{\text{max}} \) is the number of individuals of the most abundant species, and \( N \) is the total number of individuals in the sample. Often the reciprocal is used (i.e. \( 1/d \)) so that an increase in the index equates to an increase in diversity and a decrease in dominance (Margurran 1988).

**Quantitative derivatives of MCI**

Two versions of the MCI have been developed that use data on species abundance as well as species presence/absence in calculating metrics. These are:

- **Semi-quantitative MCI (SQMCI)**

  The SQMCI uses a five-point coded abundance scoring system to infer the relative abundance of taxa (Table D7):

  \[ \text{SQMCI} = \sum (c_i a_i / N), \]

  where \( c_i \) = coded abundance score for in the \( i^{th} \) taxon, \( a_i \) = MCI tolerance score for the \( i^{th} \) taxon.

- **Quantitative MCI (QMCI)**

  Uses invertebrate abundance data derived from quantitative sampling, and \( d \) is calculated by:

  \[ \text{QMCI} = \sum (n_i a_i / N) \]

  where \( n_i \) = number of individuals in the \( i^{th} \) taxon, \( a_i \) = MCI tolerance score for the \( i^{th} \) taxon, \( N \) = total number of individuals.

- **Both the SQMCI and QMCI are likely to be more sensitive biological indicators than the MCI, as they incorporate changes in the relative abundance of taxa as well as changes in species presence/absence. The QMCI has been used extensively in biomonitoring and other research in New Zealand (Boothroyd and Stark 2000), and the SQMCI provides a cost-effective alternative to quantitative sampling.**

**D.7 References and further reading**


Biggs BJ 2000. New Zealand periphyton guideline: detecting, monitoring and managing the


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Gerhardt A 1994. Short term toxicity of iron (Fe) and lead (Pb) to the mayfly Leptophlebia marginata (L.) (Insecta) in relation to freshwater acidification. Hydrobiologia 284: 157-168.


Gill GA and Bruland KW 1990. Mercury speciation in surface freshwater systems in California and other
Green RH 1979. Sampling design and statistical methods for environmental biologists. Wiley Interscience, Chichester, UK.


Rowe D, Hicks M, Richardson J 2000. Reduced abundance of banded kokopu (Galaxias fasciatus) and other native fish in turbid rivers of the North Island of New Zealand. New Zealand Journal of Marine and Freshwater Research 34: 545-556.


Soucek DJ, Cherry DS, Trent GC 2000a. Relative acute toxicity of acid mine drainage water column and sediments to Daphnia magna in the Puckets Creek watershed, Virginia, USA. Archives of Environmental Contamination and Toxicology 38: 305-310.


Appendix E: OPERATIONAL MANAGEMENT

Operational management is a critical factor in minimising the impacts of mining on streams. Operational management to prevent AMD is primarily accomplished in the assessment and design phases and more options and more effective options are available earlier in the mine life. The following sections provide more detail on the options specified in section 4.4.1. The information is largely drawn from INAP (2009).

E.1 Water management methods

Diversion
One of the easiest and cheapest methods for minimising the volume of acidic leachate is to control water volume and direction by diverting surface water above a mined site (Skousen 2000) and draining water away from pyritic materials as rapidly as possible to keep water from reacting and forming acidic products. Removing water before it contacts pyritic material may also be achieved by pumping.

Diversion of surface runoff to limit its uptake of suspended solids or acidified materials involves construction of drainage ditches to move non-impacted surface water quickly off the site. The diversion is accomplished either by ditching on the uphill side of surface mines or by providing new channels or impervious channels for existing surface streams to convey water across the disturbed area. There are very few sites where water diversion is not appropriate and it can provide savings in both cost and space.

In some cases, diversion of streams running through the proposed mining area may be necessary, although, it is preferable to move mining waste rather than move the stream, since the relocated stream can require considerable maintenance, particularly following high flow events.

Dewatering
In order to reduce the amount of groundwater that comes in contact with in situ PAF materials a drainage system can be designed to lower the water table and therefore change the hydraulic gradient; this is termed dewatering. It can also include diversion of groundwater away from mine workings. Dewatering is often a normal part of mining practice to provide safe working access to minerals. Although dewatering can reduce acid transport during mining it can also promote oxidation of pyritic surfaces if these surfaces have previously been inundated, and careful planning is required to prevent these oxidation products from entering waterways, particularly after mine closure. Examples include pit dewatering to reduce seepage through pit walls, and shallow groundwater collection ditches above tailing ponds and waste-rock dumps. The dewatering method used will be determined by the site, in particular its hydrogeological conditions (see Table E1). Any contaminated groundwater will require treatment after collection and pumping.

Flooding
The basis for this approach is outlined in MEND (2001). Factors to be considered for flooding include the status of the mine plan and schedule of waste production, potential for mobilisation of stored oxidation products, availability of open workings or pits, capacity to store waste products (mined material will swell, increasing in volume by approximately 25–30%), and climate – water levels need to be maintained so rainfall should exceed evaporation.

Hydrogeological controls
Hydrogeological controls are primarily used for controlling groundwater flow. For example, placement of low-permeability materials such as tailings in an open pit with a highly permeable surrounding material creates a large permeability contrast causing groundwater to flow around, rather than through the low-permeability material. The use of such control options is site-specific and depends on factors such as topography, hydrology and hydrogeology.

Seals
Seals are primarily used for decommissioning underground mines, and effective design relies on knowledge of the most geochemically reactive areas within the mine, and water ingress and discharge locations. Seals are most often used to promote flooded conditions. Bricks and mortar or concrete ‘plugs’ are used to seal a mine tunnel to prevent discharge of water, which then results in the mine tunnel being flooded.
Table E1: Dewatering techniques suitability or different mine hydrogeological conditions

<table>
<thead>
<tr>
<th>Dewatering method</th>
<th>Outline of method</th>
<th>Suitable mine conditions</th>
</tr>
</thead>
</table>
| Sump              | Roads and ditches within the mine are graded and routed to deliver all water to one or more centralised sumps, whence it is pumped from the mine. | Mine above water table  
Mine below water table but low-permeability strata.  
Open pit in chemically inert rocks  
Small or isolated mine |
| Adit              | A drainage adit with gradient >1:500 is driven from a portal in a valley beneath the mine area, and all mine drainage is directed to the adit via roads, shafts, pipes, etc. | Maximum mine depth above elevation to which an adit can be economically constructed.  
Requires evaluation of long-term effects of permanent water table drawdown. |
| External          | Boreholes or shafts in aquifers or old workings outside of the mine are used to pump water to prevent water ingress or to lower the water table below active mining. | Underground mine surrounded by highly permeable aquifers or flooded old workings.  
Mines containing pyritic or other reactive strata that could impact on water quality.  
Deep mines where external wells can intercept the water table at shallower depth. |

E.2 Special handling methods

Special handling of PAF rocks must be incorporated into mine planning, and segregation of PAF and NAF materials is often a critical component for special handling.

Encapsulation and layering

The effectiveness of encapsulation and layering is governed by availability of materials, the general balance between PAF and NAF materials, the type and reactivity of acid-consuming material, deposit geometry, the nature and flow of water through the deposit, and chemical armouring of alkaline materials (MEND 1998, 2001).

Blending

Homogeneous and thorough mixing of acid-generating and acid-neutralising materials is generally required to achieve maximum benefit (MEND 1998, 2001). Operational experience indicates that, for effective blending of PAF rock with limestone, it is essential that all size fractions within the blend be at least acid-base neutral (i.e. ANC/MPA ratio of at least 1) (INAP 2009). Since the particle size of limestone is generally coarser than that of PAF rock, the acid–base balance of the bulk waste rock needs to be greater than balanced.

E.3 Amendments and additions

Alkaline addition

The addition of alkaline materials to PAF rocks is the most common form of amendment. This addition can be achieved in a number of ways and Table E2 provides a summary of the benefits and limitations of various methods.

Table E2: Benefits and limitations of alkaline amendments (from INAP 2009)

<table>
<thead>
<tr>
<th>Configuration</th>
<th>Benefits</th>
<th>Limitations</th>
</tr>
</thead>
</table>
| Liquid amendment | Excellent initial control of solution pH  
Versatile – allows localised treatment  
Proven to work | Time - alkaline materials are consumed by even pH neutral water  
Cost and availability of reagents  
Particle size and release of alkalinity  
Effort for mixing or blending |
| Layering       | Easy to implement and manage                                             | Difficult to obtain mixing of alkaline and acid leachate due to preferential flow |
| Encapsulation and | Easy to implement and manage                                             | Cost and availability of material                                           |
### Configuration and Benefits

<table>
<thead>
<tr>
<th>Configuration</th>
<th>Benefits</th>
<th>Limitations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alkaline cover</td>
<td>Versatile and allows localised treatment</td>
<td>Time and release rate of alkalinity</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Alkaline materials are consumed by pH neutral water</td>
</tr>
<tr>
<td>Blending</td>
<td>Excellent pH control</td>
<td>Cost</td>
</tr>
<tr>
<td></td>
<td>Proven to work</td>
<td>Effort for mixing and blending</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Availability of materials</td>
</tr>
<tr>
<td>Injection and mine filling</td>
<td>Proven to work</td>
<td>Cost and availability of material</td>
</tr>
</tbody>
</table>

### Determination of the amount of acid-neutralising material required

ABA data can assist in the determination of the amount of acid-neutralising material required to neutralise PAF rocks using the following two-step process:

- Convert net acid-producing potential (NAPP) result of \( x \, \text{kg/t(H}_2\text{SO}_4) \), for the overburden rocks that will be removed during mining, to calcium carbonate equivalents: \( \text{NAPP} \, \text{kg/t(CaCO}_3) = \text{NAPP} \times \text{kg/t(H}_2\text{SO}_4) \times \frac{98.08}{100.09} \).
- Multiply this value by the tonne of overburden material to be removed.

### Other amendments

Less common amendments include passivation and the addition of organic materials.

Passivation is the treatment of reactive rock surfaces to limit release of contaminants by installing a chemically inert and protective surface layer. This technique appears to be most cost-effective on freshly mined surfaces as weathered surfaces often have high acidity and contain an oxidation rind that limits the effectiveness of passivation. The long-term stability and full-scale applicability of this treatment is yet to be determined, but pilot-scale experiments have shown that it can substantially reduce contaminant release for more than 5 years.

Mixing waste rock with organic materials such as sewage sludge and paper waste will consume oxygen and promote metal reduction in an anoxic environment by naturally occurring bacteria. The bacteria reduce available sulphate and create insoluble metal sulphide precipitates. The method is limited by exhaustion of organic materials. An additional concern is that the reducing conditions generated may dissolve precipitated iron and manganese hydroxides.

### E.4 Waste-rock-pile management

The management of excavated waste rock, and particularly waste-rock-pile design, and the use of covers are integral components of effective operational management. Covers are often constructed as part of closure or rehabilitation works, and thus the final landform design of the waste-rock pile is important and should blend with the surrounding environment (MEND 2007). Cover performance may be monitored on a micro-scale (e.g. individual rock piles) or macro-scale. Micro-scale monitoring entails monitoring of the actual cover itself, while macro-scale monitoring entails measurement of water quality and flow from seepage or overland flow discharged from the mine site (MEND 2007).

The use of covers over reactive waste materials is intended to minimise the formation and transportation of AMD. The choice of suitable covers is dependent on a number of variables including climate, hydrology, rainfall and temperature. Figure E1 provides a guide to suitable cover systems for different climatic regions. Based on this, [dry] covers that have low permeability for oxygen, water covers, and organic covers are likely to be the most suitable for mines in New Zealand, particularly those on the West Coast.
Dry covers

Dry covers should be resistant to erosion and provide a suitable rooting zone for vegetation. The presence of vegetation improves the performance of all dry cover systems and in particular, moisture store-and-release covers. In most situations some monitoring and maintenance of a cover system will be required. The long-term performance of dry covers has been evaluated by INAP (2003); Wilson; 2011 – SMI AMD Workshop, Darwin).

Soil covers

Soil covers are the most widely used cover. Single-layer soil covers were not found to be effective in reducing AMD formation in North America, and multi-layer designs are more widely used (INAP 2009). A number of factors will influence soil cover design (Table E3). Another key factor is that performance of a soil cover on a sloping surface can be much different to that on a horizontal surface, and slope needs to be considered in cover design. Finally, the longevity of a soil cover should be evaluated in relation to site-specific physical, biological, and chemical processes that will alter as-built performance and determine long-term performance.

Table E3: Considerations and limitations of soil covers

<table>
<thead>
<tr>
<th>Considerations</th>
<th>Limitations</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Climate – wetting, drying, freezing, thawing</td>
<td>• Soil covers do not stop infiltration and may not stop acid drainage.</td>
</tr>
<tr>
<td>• Type and performance of waste (such as surface settlement), reactivity of wastes</td>
<td>• Permeability of water infiltration barriers may increase with time when subjected to climate and vegetation.</td>
</tr>
<tr>
<td>• Surface water flow and erosion</td>
<td>• Oxygen barrier covers are especially vulnerable to relatively small imperfections in the cover – such as differential settlement, holes caused by animal burrows, desiccation cracking, – that effectively render permeable an otherwise sealed area.</td>
</tr>
<tr>
<td>• Topography</td>
<td>• Soil covers may be prone to erosion and long-term maintenance requirements.</td>
</tr>
<tr>
<td>• Hydrogeological setting and basal flow</td>
<td>• Soil covers may be vulnerable to vegetation, animal, and human activity including vehicle traffic.</td>
</tr>
<tr>
<td>• Availability of cover materials</td>
<td></td>
</tr>
<tr>
<td>• Construction quality control and maintenance</td>
<td></td>
</tr>
<tr>
<td>• Design of landforms - final land use</td>
<td></td>
</tr>
</tbody>
</table>

Vegetation

Establishing vegetative cover on soil covers can be used for interim, long-term, or permanent erosion control. Non-native species may be used initially so that vegetation can be established quickly, but over time the objective may be to establish a complete

Figure E1: Covers and climate types (from INAP 2009).
and dense vegetative cover comprised of native species. The vegetation assists in maximising evapotranspiration of water retained in the soil cover.

Proper reclamation and revegetation of a site can help reduce the potential for AMD by preventing excess infiltration. However, infiltration cannot be completely eliminated and even exceptional quality reclamation will not prevent AMD in an otherwise poor setting. Effects of vegetation must be considered in engineered soil cover design with respect to AMD. Vegetation may physically alter cover systems by way of holes caused by roots, tree throw, or blow down, and may take up and transport contaminants from below the soil cover. Root penetration of cover systems may effectively bypass capillary break layers and provide a pathway to the surface ecosystem.

**Wet covers**

Disposal of acid-generating materials below a water cover is one of the most effective methods for limiting AMD generation in a suitable climate. In water, the maximum concentration of dissolved oxygen is approximately 30 times less than in the atmosphere. More importantly, the transport of oxygen through water by advection and diffusion is severely limited (approximately 10,000 times slower) relative to transport in air.

Water covers also promote sulphide reduction by bacteria, metal hydroxide precipitation, and development of sediment layers, which inhibit interaction between tailings and overlying waters.

However, the ability to maintain a water cover over the long term, the integrity of the containment structures, locality and site-specific potential risks due to seismic events, severe storm events, etc., can negate the use of this technology.

**Inundation or sub-aqueous disposal**

An overview of the factors that should be considered for the use of water covers is given in Table E4.

### Table E4: Some considerations for the use of water covers (from INAP 2009)

<table>
<thead>
<tr>
<th>Considerations for use of natural water bodies (lakes or oceans)</th>
<th>Considerations for engineered water-retaining structures (dikes and dams, pits, underground workings)</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Proximity to mine</td>
<td>• Dams and dikes</td>
</tr>
<tr>
<td>• Water level</td>
<td>• Detailed design of embankment and spillway required</td>
</tr>
<tr>
<td>• Tides and currents</td>
<td>• Measures required to control seepage</td>
</tr>
<tr>
<td>• Other lake uses</td>
<td>• Geotechnical, maintenance, and inspection requirements of dikes, dams, and hydraulic structures required to maintain a water cover</td>
</tr>
<tr>
<td>• Regulatory environment</td>
<td>• Hydrogeology and hydrology, - a reliable water source is required to maintain flooded conditions, even during drought</td>
</tr>
<tr>
<td>• Potential toxicity of wastes including metal leaching, reagents and metals from the milling process</td>
<td>• Potential risks to downstream receptors of catastrophic failure and release of retained water and/or wastes</td>
</tr>
<tr>
<td>• Residue from blasting processes as a potential nutrient source</td>
<td>• Pits and underground workings</td>
</tr>
<tr>
<td>• Potential increase in turbidity</td>
<td>• Potential mobilisation of stored oxidation products upon flooding</td>
</tr>
<tr>
<td>• Potential effects on local flora and fauna including commercial and recreational fisheries</td>
<td>• Potential leaching of metals from flooded materials</td>
</tr>
<tr>
<td>• Potential loss of habitat</td>
<td>• Resulting pit lake chemistry</td>
</tr>
<tr>
<td>• Natural mixing processes</td>
<td>• Potential oxidation of materials above the water line</td>
</tr>
<tr>
<td>• Lake water chemistry</td>
<td>• Continuity with other underground workings</td>
</tr>
</tbody>
</table>

**Partial water cover**

This method is useful for operations where two types of tailings are being generated: high sulphur concentrate that is stored at depth and NAF tailings that are used as cover above the level of the pond. It may also be used where the underlying non-oxidised tailings have sufficient neutralising capacity to assimilate the entire acid load that is produced from the overlying rind of unsaturated tailings.

A key design consideration is to raise the water table above the acid-generating material by placing non-acid-generating material as a cover, controlling the water level by the pond spillway elevation, or both placing non-acid-generating material and
controlling the water level. Use of the partial water-cover method must consider climate, topography, hydrology and hydrogeology, the residual neutralising capacity of unoxidised tailings, and the water characteristic retention curve of the tailings material.

E.5 State of knowledge

Table E5: Summary of prevention and mitigative measures and climate considerations (adapted from INAP 2009).

<table>
<thead>
<tr>
<th>Widely demonstrated</th>
<th>Limited demonstration</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Oxygen limiting</strong></td>
<td></td>
</tr>
<tr>
<td>• Submergence (all climates except B)</td>
<td>• Oxygen-consuming cover (all climates)</td>
</tr>
<tr>
<td>• Water covers (A, C, D)</td>
<td>• Saturated soil cover (A, C, D)</td>
</tr>
<tr>
<td>• In-pit disposal (all climates)</td>
<td></td>
</tr>
<tr>
<td>• Elevated water table (A, C, D)</td>
<td></td>
</tr>
<tr>
<td>• Mine backfilling (all climates)</td>
<td></td>
</tr>
<tr>
<td>• Membrane covers (all climates)</td>
<td></td>
</tr>
<tr>
<td><strong>Water limiting</strong></td>
<td></td>
</tr>
<tr>
<td>• Diversion (all climates)</td>
<td>• Low-permeability covers in wet climates (A, C, D)</td>
</tr>
<tr>
<td>• Store-and-release covers (B)</td>
<td>• Paste tailings (C, D)</td>
</tr>
<tr>
<td>• Membrane covers (all climates)</td>
<td></td>
</tr>
<tr>
<td><strong>Geochemical</strong></td>
<td></td>
</tr>
<tr>
<td>• Segregation (all climates)</td>
<td>• Alkaline covers (all climates)</td>
</tr>
<tr>
<td>• Avoidance (all climates)</td>
<td>• Passivation (all climates)</td>
</tr>
<tr>
<td>• Tailings desulphurisation (C)</td>
<td>• Blending (all climates)</td>
</tr>
<tr>
<td>• Encapsulation (coal) (C)</td>
<td>• Encapsulation (metal) (A, C)</td>
</tr>
<tr>
<td><strong>Climate</strong></td>
<td></td>
</tr>
<tr>
<td>(A) Tropical humid</td>
<td>(D) Continental severe mid-latitude</td>
</tr>
<tr>
<td>(B) Dry</td>
<td>(E) Polar</td>
</tr>
<tr>
<td>(C) Temperate mild mid-latitude</td>
<td>(H) Highland</td>
</tr>
</tbody>
</table>

E.6 References


MEND (Mine Environment Neutralising Drainage) 2007. Macro-cover scale cover design and performance monitoring. Natural Resources Canada.

Appendix F: REDUCING IMPACTS – TREATMENT TECHNIQUES

F.1 AMD treatment (coal PAF regions)

Selection between active and passive treatment
A number of factors will influence the decision as to whether to use active or passive treatment. For example, where mine drainage exceeds specific thresholds and a large amount of neutralising material is required to ensure appropriate treatment, then a large passive system would be required, which can be prone to failure (see below), so active treatment is likely to be a better choice. Particularly relevant in New Zealand are variables such as flow rate and acid load. On the West Coast, for example, where the majority of AMD sites are located, annual rainfall is high (up to 6 m on the plateaus north of Westport), and AMD sites are often located in very isolated areas with steep topography. This can result in AMD with very high flow rates and acid loads that can be markedly influenced by rainfall events. This is very different to the south-eastern USA (West Virginia) where much of the early development of AMD treatment occurred. There, topography plays a minor role in the selection of treatment systems and the dominant variables are climatic – the very cold winters and hot summers. Therefore, for New Zealand AMD sites, base and peak flow rates and acid loads need to be carefully considered along with the degree of isolation and access to power when making a choice between active and passive treatment. Further details of how each parameter influences the choice of active or passive treatment is provided below.

Acidity and pH
Acidity is comprised of proton acidity (pH) and mineral acidity, which is from dissolved metal species which produce more proton acidity upon hydrolysis (Rose and Crayotta 1998). To treat a low pH and/or high acidity AMD with passive remediation, a very large system is necessary to achieve a long enough residence time for neutralisation and a very large amount of neutralising material is necessary to maintain system longevity. Very large systems can be prone to short-circuiting and failure as preferential pathways may develop over time. Rather than constructing one large system, it may be better to split the flow among several parallel systems. Waters et al. (2003) document that the most successful passive treatment systems treat AMD with pH > 2 and acidity < 800 mg/L (as CaCO3). An acidity of 800 mg/L equates to a pH of 2, an Fe concentration of 50 mg/L and Al concentration of 30 mg/L. The relationship between acidity and pH is exponential, with much greater increases in acidity below pH 2. As such, AMD with extremely low pH (<2) and/or high acidity (>800 mg/L) is better treated with active systems than passive systems.

Flow rate
High flow rates are difficult to treat with passive systems because the systems need to be very large to achieve the necessary residence time and very large systems can be prone to short circuiting and failure. However, flow rates should be carefully considered along with the practicabilities of treatment. Significant flow rate variations during storm events are common at AMD sites in New Zealand, and treatment systems (active or passive) may not be able to treat the AMD 100% of the time, but rather might be designed for base flow conditions and allow high flow events to bypass the treatment system. Before this decision can be made, quantification of acid loads during base flow versus total flow (including storm events) should be completed and compared with the treatment requirements for the site. In general, active treatment systems can accomodate some variability in flow rates by changing chemical dosing rates to match flow rates, whereas, passive treatment systems can only manage variability if an equilisation header pond is constructed to dampen peak flows.

Acid load (acidity x flow rate)
A high acid load (generated by high acidity and/or high flow) consumes the neutralising material faster, and in a passive system this can limit life expectancy. In addition, high flow rates require construction of very large systems that can be prone to short circuiting and failure. Therefore, for a situation with very high acid load, it is recommended that consideration be given to using an active treatment system, although a passive system can perhaps be used if a shorter life expectancy (< 25 years) is acceptable. The choice will often be based on a cost–benefit assessment of active versus passive treatment. Alternatively, if sufficient space is available at the site, the AMD can be split into several smaller flows with lower acid loads, and multiple passive treatment systems can...
be constructed. This is essentially using a process principle to potentially enable use of passive treatment in a high acid load situation. High acid loads also can result in the generation of much greater volumes of precipitates during neutralisation and the longevity of passive systems can be compromised if permeability drops as precipitates accumulate in system pore spaces (Waters et al. 2003).

As discussed above, significant flow-rate fluctuations are common at AMD sites in New Zealand, and systems (active or passive) must be designed with this in mind. But research is lacking on the effect of flow rate fluctuations on acid load in New Zealand; it is possible that correlations may be site specific and even variable for a given site. Research is also lacking on ecosystem effects resulting from a sudden pulse of high-acid-load water if treatment systems are not designed to accommodate storm events (where acid loads are increased). It is recommended that these issues be addressed prior to deciding on the level of treatment required.

Operational mine site with power

In general, active treatment systems are more commonly used at operational mine sites, whereas passive systems are typically used at closed and abandoned mines. Operational mine sites typically have limited space for treatment systems and a drainage chemistry and flow rate that can change as mining proceeds. These factors are addressed more easily with active treatment systems than with passive systems. However, if sufficient space is available, and chemistry and flow rates are not expected to change significantly with time, passive treatment can be a suitable solution at active mine sites. Further, passive treatment may be used to complement active treatment.

Power is a critical factor for active treatment systems. Pumps are often used to convey the water to the treatment system and between various components of the system; power is usually needed to meter additives to the water such as neutralising chemicals, flocculants, and coagulants, and power is necessary for mixing and oxidation of the water. If no power is available at the site, a semi-passive system, such as the Aquafix system, can be used, which relies on a paddlewheel in the AMD stream to operate a hopper that dispenses neutralising chemicals into the AMD (Skousen and Jenkins 1993). Otherwise, lack of power limits AMD treatment to passive systems unless it is warranted to bring power to the site.

F.2 AMD active treatment systems

Selection of neutralising chemicals – Dosing-with-alkali stage

The most commonly used chemicals for raising the pH of acidic mine drainages are soda ash (sodium carbonate; Na₂CO₃), hydrated lime (calcium hydroxide; Ca(OH)₂), quicklime (calcium oxide; CaO), caustic soda (sodium hydroxide; NaOH), and ammonia (NH₃). Magnesium oxide or hydroxide (MgO or Mg(OH)₂) and lime stone (CaCO₃) are occasionally used. A brief description of these chemicals and their use is provided below.

Selection of an appropriate chemical is dependent on the flow rate of the AMD, and the concentration of dissolved Fe (see main text section 5.4.2). Other common metal ions in AMD, such as Al, and to a lesser extent Zn, and Ni, are removed along with Fe, and are not a factor in deciding among the neutralising chemicals. Other factors will also influence chemical selection. These include: chemical cost, neutralising efficiency, maximum pH attainable, dispensing mechanism required, mixing mechanism required, health and safety issues, sludge settling rates and therefore requirement for flocculants or coagulants, and resulting sludge volume and density (Skousen et al. 2000; Waters et al. 2003; Means 2006). Chemical cost in particular can be significant, as over the long term the largest single cost component in most systems is the neutralising chemical (Waters et al. 2003). It is recommended that bench scale tests are conducted on various chemicals before final selection (Younger et al. 2002) and also a sequential titration acidity analysis as described in Hilton (2004) and in section F.3.1. A comparison of the characteristics of the common chemicals is provided in Table F1.

If Mn is present at high concentrations in mine drainage, there needs to be additional consideration of what chemicals to use. Manganese is a difficult metal to remove from solution as it exhibits high solubility over a broad pH range (4.5-8) and the chemical oxidation of Mn is kinetically slow (Bamforth et al. 2006). The most effective way to remove Mn from water is to raise the pH above 9 and allow Mn²⁺ to oxidise to Mn³⁺ or Mn⁴⁺ and form insoluble Mn oxides or Mn carbonates (Evangelou 1998). Where it is necessary to lower the concentrations of Mn, although Ca-based chemicals can be used, the most commonly-used chemicals for Mn removal are sodium hydroxide (NaOH) and ammonia (NH₃) (Skousen 1988; Skousen et al. 1990, 2000).
**Table F1:** Characteristic of chemicals used to neutralise AMD in active treatment systems (Skousen et al. 2000; Waters et al. 2003; Means 2006).

<table>
<thead>
<tr>
<th>Chemical</th>
<th>Conversion factor</th>
<th>Neutralisation efficiency (%)</th>
<th>Cost of chemical per tonne of acid neutralised</th>
<th>Dispensing mechanism</th>
<th>Key benefits</th>
<th>Key limitations</th>
<th>Risk of failure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soda ash or sodium carbonate (Na₂CO₃)</td>
<td>1.06</td>
<td>95–100 (powder) 60 (briquettes)</td>
<td>$830 – $870 (powder)</td>
<td>Briquettes placed in wooden box or large drum/reactor in AMD stream.</td>
<td>High efficiency in powder form, most metals precipitate, low sludge volumes.</td>
<td>Health and safety issues, poor sludge settling rates, potential sodium toxicity.</td>
<td>Potential for reduced treatment effectiveness when using briquettes if acidity loading rates increase significantly (best as an interim treatment or only for low flow, low acidity AMD).</td>
</tr>
<tr>
<td>Hydrated lime or calcium hydroxide (Ca(OH)₂)</td>
<td>0.74</td>
<td>90–95</td>
<td>$330 – $350</td>
<td>Silo or hopper with mechanical feed screw to dispense powder. Batching tank to mix powder with water. Can use aqueous slurry. Mixing suggested.</td>
<td>High efficiency, most metals precipitate, low cost, widely available.</td>
<td>Health and safety issues, reagent saturation can lower efficiency.</td>
<td>If acidity loading rates increase beyond system capacity to neutralise and settle hydroxides, treatment effectiveness will drop. Poor maintenance can result in plugged dispensing mechanism and complete failure.</td>
</tr>
<tr>
<td>Quicklime or calcium oxide (CaO)</td>
<td>0.56</td>
<td>90</td>
<td>$233</td>
<td>Silo or hopper with mechanical feed screw to dispense powder or water wheel feeder with one-tonne storage bin (no power). Batching tank to mix powder with water. Mixing suggested.</td>
<td>High efficiency, most metals precipitate, very low cost, widely available.</td>
<td>Health and safety issues, reagent saturation can lower efficiency, possible armouring of pebbles.</td>
<td>If acidity loading rates increase beyond system capacity to neutralise and settle hydroxides, treatment effectiveness will drop. Poor maintenance can result in plugged dispensing mechanism and complete failure. Must be watertight or will hydrate and form calcium hydroxide and plug dispensing mechanism.</td>
</tr>
<tr>
<td>Ammonia (NH₃ or NH₄OH)</td>
<td>0.34</td>
<td>100</td>
<td>$2,200</td>
<td>Compressed and stored as liquid in tank, gas injected near bottom of pond or water inlet. No mixing required.</td>
<td>Very high efficiency, most metals precipitate, low sludge volumes.</td>
<td>Health and safety issues, poor sludge settling rates, can be toxic to aquatic life, high cost.</td>
<td>If acidity loading rates increase beyond system capacity to neutralise and settle hydroxides, treatment effectiveness will drop.</td>
</tr>
<tr>
<td>Chemical</td>
<td>Conversion factor⁴</td>
<td>Neutralisation efficiency (%)⁵</td>
<td>Cost of chemical per tonne of acid neutralised⁶</td>
<td>Dispensing mechanism</td>
<td>Key benefits</td>
<td>Key limitations</td>
<td>Risk of failure</td>
</tr>
<tr>
<td>----------------------------------------------</td>
<td>--------------------</td>
<td>--------------------------------</td>
<td>-----------------------------------------------</td>
<td>--------------------------------------------------------------------------------------</td>
<td>-------------------------------------------------------------------------------</td>
<td>--------------------------------------------------------------------------------</td>
<td>--------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Caustic soda or sodium hydroxide (NaOH)</td>
<td>1063 (50% liquid)</td>
<td>100</td>
<td>$1,050</td>
<td>Stored as a liquid in tank, dispense through metering pump or valve and feeder hose near top of pond or water inlet. No mixing required.</td>
<td>Very high efficiency, most metals precipitate, low sludge volumes.</td>
<td>Health and safety issues, poor sludge settling rates, potential sodium toxicity, highest cost of all chemicals, low freezing point.</td>
<td>If acidity loading rates increase beyond system capacity to neutralise and settle hydroxides, treatment effectiveness will drop. If insufficient antifreeze added, can freeze in winter resulting in complete failure.</td>
</tr>
</tbody>
</table>

1 Conversion factor is the mass of chemical needed to neutralise the acidity relative to limestone. The conversion factor is used along with the neutralisation efficiency to calculate the tonnes of chemical needed to neutralise each tonne of acidity per year. For sodium hydroxide, the conversion factor gives litres of chemical needed per tonne of acid.

2 Neutralisation efficiency estimates the effectiveness of the chemical in neutralising acidity relative to sodium hydroxide and is used along with the conversion factor to calculate the tonnes of chemical needed to neutralise each tonne of acidity per year. For example, if 46 tonnes of acid needs neutralisation per year, 38 tonnes of hydrated lime would be needed \( \frac{46(0.74)}{0.90} \).

3 Cost of chemical is per tonne of acid neutralised in 2010 NZD. This is an effective way to compare costs of treatment with different chemicals. Cost is calculated using purchase price, conversion factor and neutralisation efficiency. For example, if calcium oxide costs $375 per tonne, to neutralise one tonne of acidity would cost $233 \( \frac{(375)(0.56)}{0.90} \).
Factors to consider in selecting a calcium or sodium compound for an AMD treatment system are shown in Table F2.

**Table F2:** Factors that may influence the selection of a calcium or sodium compound for an AMD treatment system (from Skousen 1988).

<table>
<thead>
<tr>
<th>Factor</th>
<th>Calcium</th>
<th>Sodium</th>
</tr>
</thead>
<tbody>
<tr>
<td>Solubility</td>
<td>slow</td>
<td>fast</td>
</tr>
<tr>
<td>Application</td>
<td>requires mixing</td>
<td>diffuses well</td>
</tr>
<tr>
<td>Hardness</td>
<td>high</td>
<td>low</td>
</tr>
<tr>
<td>Gypsum formation</td>
<td>yes</td>
<td>no</td>
</tr>
<tr>
<td>High TSS or clay particles</td>
<td>helps settle clay</td>
<td>disperses clay particles and keeps clay in suspension</td>
</tr>
<tr>
<td>Chemical cost</td>
<td>lower</td>
<td>higher</td>
</tr>
<tr>
<td>Installation and maintenance costs</td>
<td>high</td>
<td>low</td>
</tr>
</tbody>
</table>

**Soda ash or sodium carbonate (Na2CO3)**

Soda ash is generally only used in small-flow cases with low amounts of Fe and where Mn is not a problem (Skousen et al. 1990). It can also be used as a good interim solution until a more permanent system is constructed (Means 2006). Selection of this chemical is sometimes based more on convenience than cost-effectiveness. The chemical is delivered in briquette form and placed in a box, barrel or drum. Water flows through this dispensing mechanism dissolving a small amount of the chemical. Gravity keeps the briquettes feeding into the water for constant treatment.

**Figure F1:** Treatment of AMD using sodium carbonate.

**Hydrated lime or calcium hydroxide (Ca(OH)2)**

Hydrated lime is the most common chemical used for neutralising AMD. Although the capital cost for a system using hydrated lime is high, low chemical cost makes this a very economical treatment method over the long term. It is generally delivered in powdery form, which is hydrophobic, so extensive aeration and mixing is recommended for high flow rates to make it soluble in water (Skousen et al. 1990). It is usually contained in large hoppers and dispensed into the AMD using a mechanical screw feeder (Figures F2 and F3) (Means 2006).
Figure F2: Treatment of AMD using calcium hydroxide. Hoppers dispense chemical directly into AMD.

Figure F3: Treatment of AMD using calcium hydroxide in large water treatment plants.

Quicklime or calcium oxide (CaO)

Quicklime is the non-hydrated form of hydrated lime. Quicklime is the least expensive chemical used in treatment of AMD, but it must be stored in a water-tight container to avoid premature hydration and formation of hydrated lime (Means 2006). If it does form hydrated lime, it can bridge and eventually plug the dispensing mechanism. It is commonly dispensed from a hopper through a waterwheel-powered auger into the AMD (Figure F4), requiring less operation and maintenance than a conventional system, and useful for areas without power (Skousen and Jenkins 1993). At high flow rates, use of a water-driven mixer is recommended (Means 2006).
Caustic soda or sodium hydroxide (NaOH)

Caustic soda is the most expensive chemical used in AMD treatment, is hazardous to handle, and freezes more readily than lime; however, it is 100% efficient, can raise the pH very high and therefore is able to remove Mn, and a treatment system using caustic soda has low capital costs (Means 2006). It can be delivered to the site in pellet form or more commonly as an aqueous solution (20–50%) (Figure F5). It is often stored in large tanks and dispensed either through a valve and feeder hose or with a metering pump into the top of the AMD (it is denser than water). Along with ammonia, no mixing is required; however, the resulting sludge is loose and gelatinous (Skousen et al. 2000). Sodium hydroxide shows a nearly linear relationship with pH (to 12), unlike ammonia which shows a logarithmic curve, with only small changes in pH occurring above 9.2 with the addition of more chemical (Skousen et al. 1990). It is often not economical to use Ca-based chemicals for Mn removal due to the very long residence times needed to attain a high pH (slower dissolution rates than NaOH and NH₃) and the large amounts of unreacted chemical that are produced when raising the pH above 7 (Skousen 1988; Skousen et al. 2000; Means 2006).

Ammonia (NH₃)

After caustic soda, ammonia is the next most expensive chemical used in AMD treatment and is very hazardous to handle. However, like caustic soda it is 100% efficient, can raise the pH to 9.2 and therefore is able to remove Mn, and a treatment system using ammonia has low capital costs (Waters et al. 2003). It is delivered as a...
compressed gas (anhydrous ammonia), stored as a liquid in large tanks (Figure F6, left) and bubbled into the water near the bottom of the water column where it returns to the gaseous state (Figure F6, right) (Faulkner and Skousen 1991). As it is a compressed gas, it does not have to be pumped or gravity fed. No mixing is required, but the resulting sludge is loose and gelatinous (Skousen et al. 2000). If ammonia is used, a pH-driven monitoring system is recommended, because over application can lead to toxicity of the treated water. Ammonia reacts with sulphuric acid to form ammonium sulphate, consuming hydrogen ions (raising the pH), and forming hydroxyl ions which react with Fe and Al to form hydroxides (Faulkner and Skousen 1991). It is one of the fastest ways to raise the pH.

**Figure F6:** Treatment of AMD using ammonia. Chemical is dispensed from tank (left) under pressure through piping into the AMD and precipitates form immediately (right).

*Magnesium oxide or hydroxide (MgO or Mg(OH)₂)*

Although magnesium oxide has a relatively low cost (but slightly more expensive than calcium hydroxide) and is highly efficient, it needs mixing, has a low reaction rate, and most importantly, is not generally widely available (Waters et al. 2003). It is dispensed in a similar manner to and behaves like calcium hydroxide (Skousen et al. 1996).

*Limestone (CaCO₃)*

Limestone has the lowest material cost and is the safest and easiest to handle of the chemicals commonly used for AMD treatment. Unfortunately, its successful application is limited due to its low solubility and tendency to develop an external coating, or armour, of ferric hydroxide when added to AMD (Phipps et al. 1996). In addition, limestone has the lowest treatment efficiency of the common chemicals used for AMD treatment. Limestone is generally only applicable when Fe concentrations are below 5 mg/L and total acidity is less than 50 mg/L (Skousen et al. 1990).

If limestone is ground to a powder it can be dispensed into an AMD stream by a waterwheel-powered dry powder limestone doser raised the pH to neutral and treated water with Fe concentrations of about 7 mg/L and Al concentrations of about 8 mg/L. Zurbuch (1996) describes water-powered rotary drums which autogenously grind limestone aggregate inside them into a slurry which is released into a river. The limestone drops into a small hopper where it enters the drum via an inverted screw feed which is part of the drum axle. Small holes in each drum vane permit water to enter the drum and CaCO₃ slurry to exit. A back-up limestone powder doser was constructed adjacent to the drum station. In New Zealand powdered limestone (< 0.1 mm in size) is currently being used to treat AMD at the Stockton Mine (Weber et al. 2007). Powdered limestone is released from a hopper via an electrically-driven screw feed into the Mangatini Stream (Figure F7). The rate of the screw feed is regulated by a pH probe placed downstream of the hopper. The main disadvantage of simply dispensing a neutralising chemical into an AMD stream is that metals precipitate and settle out of solution along the watercourse resulting in an increase in TSS in the stream and large amounts of Fe and Al floc (Weber et al. 2007).
Figure F7: Treatment of AMD using powdered limestone. Chemical is dispensed from hopper by screw feeder directly into AMD. Stockton Mine, New Zealand.

Rarely used chemicals
Other chemicals occasionally used to neutralise AMD include kiln dust, fly ash, fluidised bed combustion ash, calcium peroxide, potassium hydroxide, barium carbonate, hydroxyapatite and seawater-neutralised red mud (Waters et al. 2003).

Oxidation
Examples of different oxidation systems are shown in Figures F8 and F9.

Figure F8: Oxidation using rotating blades (left) and trickle filter (right).
Sedimentation

The chemical composition of AMD sludge is generally hydrated Fe$^{2+}$ or Fe$^{3+}$ oxyhydroxides, CaSO$_4$, Al(OH)$_3$·6H$_2$O, CaCO$_3$, and Ca(HCO$_3$)$_2$ with trace amounts of Si, PO$_4$, Mn, Cu, and Zn (Ackman, 1982). Leaching of trace elements from the sludge may influence disposal in a landfill or on-site and leach tests should be performed.

Standard leaching tests, both of which should be used, include the toxicity characteristic leaching procedure (TCLP; US Environmental Protection Agency test method 1311; EPA 1995) and the synthetic precipitation leaching procedure (SPLP; US Environmental Protection Agency test method 1312; EPA 1995). Each procedure produces a leachate from the sample which is then analysed for the chemicals of concern in the AMD. The TCLP assumes worse case scenario in which the test material is subjected to an acidic extraction buffer, whereas the SPLP uses an extraction buffer more similar to rainfall.

If significant leaching of trace elements occurs, stabilisation of the sludge may be required. Stabilisation of sludge involves incorporating an additive (often cement) to prevent leachate forming. See Brown et al. (1994a, b, c) for information on volume, stability, and composition of sludge generated from active treatment of AMD.

Examples of different sedimentation techniques are shown in Figures F10-F12, while photos of sludge dewatering techniques are shown in Figures F13-F14.
**Figure F10:** Sedimentation pond used to collect precipitates formed during treatment of AMD.

**Figure F11:** Side (left) and top (right) view of a clarifier used to collect precipitates formed during treatment of AMD.

**Figure F12:** Equipment used to add coagulants during AMD treatment to assist sedimentation of the precipitates formed.
F.3 AMD passive treatment systems

Ten passive remediation systems are described in this section. The first six utilise the oxidising strategy (open limestone channel, diversion well, limestone leaching bed, slag leaching bed, aerobic wetland, and dosing with limestone sand) and the next four utilise the reducing strategy (anoxic limestone drain, anaerobic wetland, vertical flow wetland, and sulphate reducing bioreactor). Each system is listed in Table F3, providing a general description, design factors, indicative cost, key benefits, limitations, risk of failure, and references. Full-scale systems can be designed using the computer program AMDTreat (Means et al. 2003). Figures F15 and F16 provide information on potential construction costs, lifetime treatment costs and overall effectiveness of passive treatment systems based on literature studies.
### Table F3: Passive treatment systems

<table>
<thead>
<tr>
<th>System type</th>
<th>General description</th>
<th>Design factors</th>
<th>Key benefits</th>
<th>Limitations</th>
<th>Construction cost</th>
<th>Risk of failure</th>
<th>References</th>
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<tr>
<td><strong>Oxidising strategies</strong></td>
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<td>Open Limestone Channels (OLC)</td>
<td>Channel, ditch, or streambed lined with limestone cobbles. Dissolution of limestone and neutralisation occurs as AMD flows down the channel.</td>
<td>Rock sizes should be &gt; 10 cm (15-30 cm), Slope should be &gt;20%, if possible. Base residence time on acid load, limestone volume on alkalinity generation rate.</td>
<td>Low cost. Simple. High level of reliability.</td>
<td>Armouring with hydroxides. Can construct at &gt;20% gradient to minimise but then must be long channel to achieve residence time.</td>
<td>$135,000</td>
<td>If Fe loading rates too high, armouring of limestone will reduce effectiveness and system may fail.</td>
<td>Ziemkiewicz et al. 1994, 1997; Ziemkiewicz and Brant 1996; Cravotta and Trahan 1999; Trumm et al. 2005, 2008</td>
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<td>Open Limestone Drains (OLD)</td>
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<td>Diversion Wells (DW)</td>
<td>Round chamber filled with crushed limestone aggregate. AMD flows into chamber outpipe near bottom of well creating turbulence which abrades particles preventing armouring. Water flows upward and out of chamber.</td>
<td>Design based on trial and error. Well dimensions typically 1.5 m in diameter, 2 m deep, half-filled with 1-2 cm diameter limestone gravel (low hardness). Residence time about 15 min.</td>
<td>Low cost. High limestone efficiency.</td>
<td>Requires refilling with limestone chips about every 2-4 weeks. Requires elevation change and constant flow rate. Precipitates not captured. Pockets of air in piping can reduce flow rate.</td>
<td>NA</td>
<td>If limestone gravel too coarse it may not be churned enough to prevent armouring by Fe oxides and system may fail.</td>
<td>Sverdrup 1983; Arnold 1991; Faulkner and Skousen 1995; Skousen et al. 1998</td>
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<td>Limestone Leaching Beds (LLB)</td>
<td>Rectangular chamber filled with limestone cobbles. AMD flows horizontally or vertically through cobbles dissolving limestone which neutralises acidity.</td>
<td>Limestone mostly between 38 and 90 mm size. 15 h residence time in unit.</td>
<td>Simple. High level of reliability.</td>
<td>Armouring with hydroxides if Fe concentrations too high.</td>
<td>$152,000</td>
<td>If Fe loading rates too high, armouring of limestone will reduce effectiveness and system may fail.</td>
<td>Black et al. 1999; Danehy et al. 2002; Hilton et al. 2003; Watzlaf et al. 2003</td>
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<tr>
<td>Slag Leaching Beds (SLB)</td>
<td>Rectangular chamber filled with steel slag fines. AMD flows horizontally or vertically through slag dissolving CaO which neutralises acidity.</td>
<td>Steel slag mostly less than 3 mm. Residence time 1-10 h depending on acidity.</td>
<td>High pH. Generates more alkalinity than limestone (up to 2,000 mg/L). Stable silicate sludge produced.</td>
<td>Armouring with hydroxides. Slow dissolution. Leaching of metals from slag.</td>
<td>NA</td>
<td>If Fe loading rates too high, armouring of slag may reduce effectiveness and system may fail.</td>
<td>Ziemkiewicz and Skousen 1998; Simmons et al. 2002; Trumm et al. 2009</td>
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<tr>
<td>System type</td>
<td>General description</td>
<td>Design factors</td>
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<td>Limitations</td>
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<td>Aerobic Wetland</td>
<td>Large surface area pond with emergent wetland species plants. With or without limestone. AMD flows horizontally through pond and over substrate. Oxidation reactions precipitate oxides and hydroxides.</td>
<td>Design based on removal rates: 10-20 g Fe/m²/d, 0.5-1 g Mn/m²/d, 3.33 g H₂SO₄/m²/d.</td>
<td>General precipitate storage. Low cost.</td>
<td>Best for pH &gt; 5.5. Removes selected metals.</td>
<td>$25,349</td>
<td>If pH is too low, plants may not survive, oxidation rates may lower, and system may fail.</td>
<td>Skousen et al. 1992; Hedin et al. 1994; MEND 1999; Skousen et al. 2000; Batty and Younger 2002</td>
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<td>Dosing With Limestone Sand</td>
<td>Large stockpile of crushed limestone placed on edge of AMD. Water washes limestone into stream and acidity neutralised.</td>
<td>Limestone very small grain size. Stockpile placed partly in stream.</td>
<td>Very simple, armouring prevented due to small grain size.</td>
<td>Requires restocking pile.</td>
<td>NA</td>
<td>If limestone sand is too coarse, it may armour with Fe oxides before it can dissolve and treat the water.</td>
<td>Mills 1996; Zurbuch 1996; Skousen et al. 1998; Watzlaf et al. 2003</td>
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<td>Reducing Strategies</td>
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<td>Anoxic Limestone Drains (ALD)</td>
<td>Buried limestone-filled drain. AMD flows horizontally through limestone in an anoxic environment. Alkalinity added by limestone dissolution.</td>
<td>Limestone 6-15 cm in diameter. 14 h residence time. Need to exclude oxygen.</td>
<td>Low cost. Simple.</td>
<td>Best for low Al, low DO, low Fe³⁺ or armouring occurs. Maintenance difficult.</td>
<td>$128,000</td>
<td>If DO levels are too high and/or Fe³⁺ concentrations too high, armouring with Fe oxides can cause system to fail. Too high Al concentrations can plug pore spaces.</td>
<td>Turner and McCoy 1990; Skousen 1991; Hedin and Watzlaf 1994</td>
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<tr>
<td>Anaerobic Wetlands</td>
<td>Large pond with a layer of organic substrate - typically spent mushroom compost with about 10% CaCO₃. AMD flows horizontally within the substrate. Emergent vegetation helps stabilise substrate and provide organic material. Sulphate reduction removes sulphate and metals.</td>
<td>Hydraulic conductivity of substrate 103-104 cm/s. Design based on removal rates: 3.5 g acidity/m²/d; 2.9 g H₂SO₄/m²/d; sulphate reduction rate 300 mmoles/m³/d.</td>
<td>Relatively stable sulphide sludge storage.</td>
<td>Requires long residence time.</td>
<td>$57,938</td>
<td>Without sufficient organic matter, anaerobic conditions may not occur and Fe oxides can plug pore spaces causing system failure.</td>
<td>Hedin et al. 1994; Skousen et al. 1992, 2000</td>
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<tr>
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<td>Vertical Flow Wetlands (VFW); Successive Alkalinity Producing System (SAPS); Reducing and Alkalinity Producing System (RAPS)</td>
<td>A rectangular unit with limestone at the base covered by spent mushroom compost and free water. AMD flows vertically down through the unit. Sulphate reduction removes sulphate and metals in the compost, alkalinity generated in limestone.</td>
<td>15-30 cm organic matter; 6-15 cm diameter limestone. Design based on 15 h residence time in limestone layer; 35-40 g acidity/m2/d removal rate.</td>
<td>Small area required. Good for high Fe, low Al.</td>
<td>High capital costs. Armouring and plugging with hydroxides.</td>
<td>$171,000</td>
<td>If compost thickness or composition inadequate to establish reducing conditions, metals will oxido, limestone will become armoured and system can fail. Inadequate flushing may not remove accumulated metals and lead to plugging of system.</td>
<td>Kepler and McCleary 1994; Zipper and Jage 2001; Watzlaf et al. 2004; Rose 2006</td>
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<td>Sulphate Reducing Bioreactor (SRB)</td>
<td>A rectangular unit filled with a mixture of organic substrates such as hay, lucerne, sawdust, paper, or woodchips, crushed limestone, and compost or manure. AMD flows vertically through unit. Sulphate reduction removes sulphate and metals.</td>
<td>Design based on removal rates: 0.3 mol metals/m3 of substrate/d or 0.3 mol sulphate/m3/d.</td>
<td>Small area required. Good for high Fe, low Al.</td>
<td>High capital costs. Reduced permeability with time. Potential for armouring and plugging with hydroxides.</td>
<td>$274,000</td>
<td>Without sufficient organic matter, anaerobic conditions may not occur and Fe oxides can plug pore spaces causing system failure.</td>
<td>Gusek 2002, 2004; Neculita et al. 2007</td>
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1Cost to treat hypothetical AMD (acidity 150 mg/L, Fe° 10 mg/L, Fe³+ 1 mg/L, Al 12 mg/L, pH 2.9, DO 1 mg/L, flow rate 10 L/s) determined using AMDTreat computer program (Table F4).
**Figure F15**: Comparison of potential construction costs for AMD passive treatment systems. A, Average cost in evaluation of 137 treatment units in the Eastern USA (Ziemkiewicz et al. 2003a, 2003b). B, Range in costs for treatment systems in Australia (Waters et al. 2003). Aew, aerobic wetland; Anw, anaerobic wetland; OLC, open limestone channel; ALD, anoxic limestone drain; RAPS, reducing and alkalinity producing system; SLB, slag leaching bed; LLB, limestone leaching bed; DW, diversion well; Bio, bioreactor.

**Figure F16**: Comparison of the cost-effectiveness of AMD passive treatment systems (Ziemkiewicz et al. 2003a, 2003b). A, Cost to treat each kg of acidity per year over the system lifespan (usually 20 years) using the following equation: Acid removal cost ($/kg/yr) = [Construction Cost/(kg acid treated per year)](system lifespan)]. B, %age of installed systems which had positive treatment measured as net acid reduction in the AMD. Aew, aerobic wetland; Anw, anaerobic wetland; OLC, open limestone channel; ALD, anoxic limestone drain; RAPS, reducing and alkalinity producing system; SLB, slag leaching bed; LLB, limestone leaching bed.
Oxidising systems

Open limestone channel and open limestone drain (OLC, OLD)

Open limestone channels (OLCs) and open limestone drains (OLDs) are systems in which limestone is placed along the sides and bottom of culverts, ditches or stream channels (Ziemkiewicz et al. 1994; Cravotta and Trahan 1999) (Figure F17). As AMD flows down the channel, dissolution of the limestone neutralises the acidity and raises the pH.

Iron and Al are removed in OLCs by formation of metal hydroxide, oxyhydroxide or hydroxysulphate precipitates as the pH is raised (Nordstrom and Alpers 1999; Skousen et al. 2000). Iron precipitates above a pH of about 2.5 and Al precipitates above a pH of about 5 (Nordstrom and McCleskey 2006). As the AMD flows down the channel and is neutralised, the precipitates typically accumulate in the channel; however, due to the gradient in pH, the Al and Fe precipitates are spatially separated and little Fe-Al coprecipitation occurs (Bigham and Nordstrom 2000). Zinc and Mn are removed in these systems by adsorption onto reactive Fe hydroxide surfaces rather than formation of oxides (Stumm and Morgan 1996; Bostick et al. 2001).

One of the key benefits of an OLC is its simplicity and low cost. In an analysis of long-term performance and cost-benefit of passive treatment systems, Ziemkiewicz et al. (2003a, b) found that OLCs were one of three systems that provided a high level of reliability (success versus failure), high acid load removal, and low treatment cost.

A key limitation, however, is the tendency for the limestone to become armoured with precipitates which can reduce the dissolution rate and therefore the effectiveness of the system (Ziemkiewicz et al. 1997; Trumm et al. 2008). Selfridge et al. (2003) found that only a relatively thin coating of less than 1 µm is needed to reduce dissolution rates for iron-coated limestone by 20%. Eventually dissolution rates in OLCs can drop by 50-80% (Skousen et al. 2000). In addition, if the gradient is relatively low, precipitates can accumulate in pore spaces and this limits the reactive surface area of the limestone. Cravotta and Trahan (1999) recommend that velocities through the system are kept high enough (>0.1 m/min) so that the solids are kept in suspension for subsequent settlement in a lagoon, wetland, or settling pond. To reduce the amount of armouring of limestone and build-up of precipitates within pore spaces, Ziemkiewicz et al. (1997) recommend that OLCs be constructed at gradients of greater than 20%.

Even if OLCs are constructed at low gradients or used to treat AMD with very high Fe concentrations, recent work suggests that armouring on limestone can be removed through agitation. Santomartino and Webb (2007) found in a laboratory experiment that Fe hydroxide coatings on limestone consist of three distinct layers, with a 60-µm-wide void between the limestone substrate and the first layer. They suggest that mechanical agitation can dislodge the coatings and restore effectiveness of the treatment system.

Operation and maintenance, therefore, consists of periodically moving and breaking the rocks with a dozer to loosen and remove precipitates if they have accumulated in the channel.

Open limestone channels are sized according to residence time necessary to neutralise acidity. A model was developed by Ziemkiewicz et al. (1994) to estimate limestone volumes and channel dimensions for achieving neutralisation of AMD based on first-order kinetics. The required residence time to achieve target acidity from the initial acidity is first solved. Then the amount of limestone required is based on flow rate and measured acidity of the AMD. Wetted channel dimensions are computed to obtain sufficient residence time of the water in the limestone channel at specified water velocities and channel slopes (using the Manning equation).

In their initial publication (1994), Ziemkiewicz et al. recommended that channels be built five times larger than the model suggests to account for reduced dissolution rates as the limestone became armoured. However, in a later publication (Ziemkiewicz et al. 1997), they found that OLCs did not need to be built this big. Santomartino and Webb (2007) provide a formula to predict when an OLC (or open limestone drain) may need agitation to dislodge precipitates armouring limestone rocks. The formula incorporates reactive surface area, Fe concentration, fraction of Fe retained in the system, and flow rate.
Diversion wells (DWs) are partially-buried large-diameter vertical chambers constructed of concrete or metal and half-filled with limestone chips (Arnold 1991). Acid mine drainage flows through a PVC pipe that enters vertically down the centre of the well and ends shortly above the bottom (Figure F18). The hydraulic force of the pipe flow causes the limestone to turbulently mix and abrade into fine particles. The water flows upward and overflows the well through a notch or a hole, where it is diverted back into the stream. The limestone is replenished on a regular basis (typically 2-4 weeks); therefore, this is considered a semi-passive treatment system. Diversion wells were initially developed for treatment of acidity caused by acid rain in Norway and Sweden (Cementa 1983).

To ensure a velocity high enough to fluidise the limestone bed, there must be sufficient head drop above the DW from a dam or other AMD collection device (typically 2.5 m or more). Work by Arnold (1991) shows that a DW with a diameter of 1.5-1.8 m and height of 2-2.4 m, filled with limestone 1-2 cm in diameter, can treat a flow rate of 1500 L/s. The PVC piping should be 20-30 cm in diameter. Waters et al. (2003) suggest DWs can be constructed with diameters between 1 and 3 m and heights below 5 m.

Acid mine drainage is treated in a DW through dissolution of the limestone, which raises the pH and causes Fe and Al hydroxides to precipitate. Turbulence within the well abrades any armouring that forms on the limestone chips, thereby continually exposing fresh limestone surfaces for dissolution. Water flowing out the well carries Fe and Al floc as well as some limestone fines. Settling ponds can be constructed following the DW to collect the precipitates (Skousen et al. 1998). Sometimes only a portion of the AMD needs to flow through the DW, and the treated AMD can be mixed with untreated AMD in the settling ponds to complete the treatment.

The key benefits of a DW are its simplicity and the high efficiency of limestone dissolution due to the lack of armouring with Fe and Al hydroxides.

The major limitation is that the limestone must be replenished on a regular basis. In addition, because the residence time within the well is relatively short (about 15 min), alkalinity generation is limited and a pH rise of only 1 to 2 units can be expected (Arnold 1991). To compensate, several DWs can be constructed in series to treat a low-pH AMD. Other design considerations include increasing efficiency by installing a metal collar at the point of AMD discharge into the well with lateral holes so that the water is directed sideways rather than down against the bottom, and flaring the top of the well to cause reduced water velocity, which allows undissolved particles to sink back into the fluidised bed (Arnold 1991).

Wratten and Schwartz (1996) developed a pulsed-bed limestone-water contactor that accelerates limestone dissolution rates through use of a carbon dioxide pretreatment step. Mineral acidities in excess of 1000 mg/L were neutralised and unusually high levels of alkalinity were achieved during treatment. However, their design adds additional complexity to the system and requires power.

Operation and maintenance consists of regularly adding limestone chips to the DW and clearing piping of accumulated floating debris.

Diversion wells are sized according to flow rate and available head drop. Although some workers have gone to considerable lengths to calculate the exact consumption rate and particle size distribution under various conditions (Sverdrup 1983; Fraser et al. 1985), Arnold (1991) suggests that local variation will in most cases negate the accuracy of such calculations and that a trial-and-error process is necessary to achieve optimum results.
Figure F18: Diversion well passive treatment system, side (left) and top (right) views.

**Limestone leaching bed**

Limestone leaching beds (LLBs) are rectangular chambers or beds filled with limestone cobbles ranging in size from 5 to 10 cm in diameter (Black et al. 1999; Watzlaf et al. 2003) (Figure F19). Acid mine drainage typically flows through the bed horizontally and is neutralised through dissolution of the limestone. To ensure adequate residence time in the LLB, the water level is kept below the surface of the limestone.

Aluminium and Fe are removed in LLBs during neutralisation of AMD by formation of metal hydroxide, oxyhydroxide or hydroxysulphate precipitates as the pH is raised (Nordstrom and Alpers 1999; Skousen et al. 2000). Zinc and Mn are likely removed in these systems by adsorption onto reactive Fe hydroxide surfaces (Stumm and Morgan 1981; Bostick et al. 2001). Watzlaf et al. (2003) recommend using LLBs only if Fe is not present, and suggest that they primarily be used to remove Mn through biological Mn oxidation. Rose et al. (2003) used an LLB inoculated with Mn-oxidising bacteria to remove Mn from an AMD with a pH of 6.5. Hilton et al. (2003) used an LLB to successfully treat AMD containing low Fe (1.65 mg/L) but high Al (56.0 mg/L).

The key benefits of an LLB are its simplicity and its high treatment efficiency. In an analysis of long-term performance and cost–benefit of passive treatment systems, Ziemkiewicz et al. (2003a, b) found that LLBs were one of three systems that provided a high level of reliability (success versus failure), high acid load removal, and low treatment cost.

A key limitation, however, is the tendency for the limestone to become armoured with Fe precipitates, which can reduce the dissolution rate and therefore the effectiveness of the system (Ziemkiewicz et al. 1997; Trumm et al. 2008). Selfridge et al. (2003) found that only a relatively thin coating of less than 1 μm is needed to reduce dissolution rates for iron-coated limestone by 20%. Therefore LLBs are best reserved for AMD with low Fe concentrations.

A common problem with leaching beds used for treating AMD is the clogging of passages with precipitates that decrease permeability and, hence, residence time (Faulkner and Skousen 1994; Watzlaf and Hyman 1995). Often flow rates through the open passages in the bed increase, resulting in enlargement of these preferential passageways and complete plugging elsewhere. To help overcome this problem, researchers in the USA have modified an existing system (an LLB) to have a horizontal flow with vertical flush capability (Hilton et al. 2003). During normal operation, preferential horizontal pathways start to become established as precipitates build up. During rapid vertical flushing, the precipitates are dislodged and removed from the system, thereby restoring permeability throughout the bed and eliminating these preferential pathways. We recommended that this horizontal flow - vertical flush concept is used in the construction of LLBs.

Operation and maintenance consists of regularly flushing the system to remove built-up precipitates and capture them in a holding pond. Piping and valves may need cleaning or replacement with time.

Limestone leaching beds are sized according to residence time necessary to neutralise acidity. Residence times are typically 12-17 h (Black et al. 1999; Ziemkiewicz et al. 2003b).
Slag leaching bed

Slag leaching beds (SLBs) are rectangular chambers or beds filled with steel slag generated from the arc furnace steel-making process (Figure 20). AMD typically flows through the bed horizontally and is neutralised through dissolution of the slag. The slag from New Zealand Steel Limited in the North Island contains approximately 15% CaO and ranges in size from 20 to 60 mm.

As with CaO active treatment systems, neutralisation by steel slag is relatively fast and the pH can be raised substantially, possibly in excess of 10 (Ziemkiewicz and Skousen 1998). Iron and Al are removed in SLBs by formation of metal hydroxide, oxyhydroxide or hydroxy sulphate precipitates as the pH is raised. Manganese is typically removed in these systems through precipitation of oxides, hydroxides, or carbonates at high pH, and is the primary target metal of SLBs.

The key benefits of an SLB are its simplicity and its ability to raise the pH relatively high to remove metals such as Mn, due to a high neutralisation potential (Simmons et al. 2002). If the source of the slag is local it can prove to be as economical as limestone, otherwise it is typically more expensive than limestone.

A common problem with leaching beds used for treating AMD is the clogging of passages with precipitates that decrease permeability and, hence, residence time (Faulkner and Skousen 1994; Watzlaf and Hyman 1995). Often flow rates through the open passages in the bed increase, resulting in enlargement of these preferential passageways and complete plugging elsewhere. To help overcome this problem, researchers in the USA have modified an existing system (an LLB) to have a horizontal flow with vertical flush capability (Hilton et al. 2003). During normal operation, preferential horizontal pathways start to become established as precipitates build up. During rapid vertical flushing, the precipitates are dislodged and removed from the system, thereby restoring permeability throughout the bed and eliminating these preferential pathways. We recommended that this horizontal flow - vertical flush concept is used in the construction of slag leaching beds.

Limitations of slag include cost, the potential to leach other elements, and the potential of armouring if Fe concentrations are significant. If the source of the slag is far from the site, it can be cost prohibitive. Slag is a by-product of the steel-making process, and as such, contains impurities from iron ore in the production of steel. There is the potential for these impurities to be leached from the slag as the CaO dissolves (Simmons et al. 2002). It is prudent, therefore, to conduct trials of the slag to determine the stability of these impurities in the slag prior to use in full-scale construction. As with limestone-based systems, armouring can be a significant problem which can reduce dissolution rates, and therefore, neutralising capability. Selfridge et al. (2003) found that only a
relatively thin coating of less than 1 μm is needed to reduce dissolution rates for Fe-coated limestone by 20%. The upper limits for Fe concentration in SLBs are unknown.

Operation and maintenance consists of regularly flushing the system to remove built-up precipitates and capture them in a holding pond. Piping and valves may need cleaning or replacement with time.

Slag leaching beds are sized according to the residence time necessary to neutralise acidity. Residence times are typically between 1 and 10 h and are determined largely through trial and error (Trumm et al. 2009).

**Aerobic wetland**

Aerobic wetlands are the earliest passive treatment system used to treat AMD. Huntsman et al. (1978) and Wieder and Lang (1982) first noted treatment of AMD flowing through naturally occurring *Sphagnum* bogs in Ohio and West Virginia, USA. Since then, over a thousand wetlands have been constructed to treat AMD (Skousen et al. 2000) (Figure F21). Aerobic wetlands consist of *Typha, Juncus* and *Scirpus* sp. and other wetland vegetation planted in shallow (<30 cm), relatively impermeable sediments comprised of soil, clay or mine spoil (Skousen et al. 2000; Ziemkiewicz et al. 2003b). The plants encourage more uniform flow, help stabilise the substrate, help maintain microbial populations, add organic matter, and provide aesthetic qualities to the wetlands.

Water flows horizontally through the wetland with depths varying between 15 and 45 cm. Because of their extensive water surface area and slow flow, aerobic wetlands promote metal oxidation and hydrolysis, thereby causing precipitation and physical retention of Fe, Al and Mn hydroxides. Variations in water depth within the wetland cell may be beneficial for performance and longevity. Shallow water zones enhance oxygenation and oxidising reactions and precipitation, while deeper water zones provide storage areas for precipitates. Aerobic wetlands perform best when water is net alkaline (Skousen et al. 2000; Watzlaf et al. 2003; Ziemkiewicz et al. 2003a); however, they can be used for low pH water but need to be a larger size due to lower removal rates, and plants are seldom very vigorous at low pH (PIRAMID 2003).

Metals are removed in aerobic wetlands by sedimentation of suspended flocs, filtration of flocs by stems of plants, adsorption of aqueous metal species, precipitation of hydroxides on plant stems and the wetland sediment surface, and direct plant uptake of iron and other metals (which are retained primarily in the plant roots; PIRAMID 2003). Metal oxidation, hydrolysis, and precipitation are considered the most important metal removal mechanisms and it is best to have net alkaline water to neutralise acidity from hydrolysis (Skousen et al. 2000). Aeration prior to the wetland, via riffles and falls, increases the efficiency of the oxidation process.
and the microbial mass in the wetland is necessary for microbial Fe oxidation. Although plant uptake is generally considered less important, Batty and Younger (2002) have shown that it is crucial in removing the last few mg/L of Fe.

The key benefits of an aerobic wetland are its simplicity, low cost, aesthetic qualities, and ability to serve as general precipitate storage.

The most significant limitation of aerobic wetlands is their poor performance with low pH water. As such, they have not been included in the flow charts for selecting AMD passive treatment systems, but they are commonly used as polishing systems following other passive treatment systems. Ziemkiewicz et al. (2003a) found that wetlands treating water with pH > 6 showed optimal performance.

There is very little required operation and maintenance for aerobic wetlands. Inflow and outflow devices/channels may need to be regularly cleared of debris.

Aerobic wetlands are typically sized based on metal loading rates. For example, Skousen et al. (1992) recommend a sizing criteria of 20 g Fe/m²/d, Brodie et al. (1993) recommend 21 g Fe/m²/d and Hedin et al. (1994) recommend 10-20 g Fe/m²/d. PIRAMID (2003) provide examples of successful treatment of drainage containing over 50 mg/L Fe. A review of aerobic wetland performance and design criteria is provided in MEND (1999). When constructing aerobic wetlands, Watzlaf et al. (2003) suggest (1) precede the wetland with a pond 1.5-2 m deep with 8-24 h retention time or have a deep section at the start of the wetland, (2) plant reed rhizomes (or other wetland species) in soil or spread seeds, (3) design with an average depth of 10-50 cm with some shallow and deep marsh areas and a few 1-2 m deep areas, and (4) design with 1-m freeboard and size to hold 20-25 years’ accumulation of Fe. Construction details are also provided in Hammer (1992) and PIRAMID (2003). Plant selection and establishment details are provided in Skousen et al. (1992).

**Figure F21:** Aerobic wetland passive treatment system.

**Dosing with limestone sand**

Dosing with limestone sand is perhaps the simplest of passive treatment systems. With this technique, sand-sized limestone is dumped into and adjacent to AMD streams (Figure 22), the sand is picked up by the stream flow, dissolves, and thereby neutralises the acidity (Mills 1996; Zurbuch 1996; Skousen et al. 1998; Watzlaf et al. 2003).

There is no system as such through which water flows. Instead the neutralising material (limestone) is directly added to the AMD, similar to how an active treatment system works. During base flow conditions, only a small amount of the limestone is swept into the AMD stream adding some alkalinity; during storm flow conditions, much greater amounts of limestone are carried by the stream and transported throughout the watershed where the limestone is incorporated in
the stream sediments and dissolved to add alkalinity over time (Watzlaf et al. 2003).

Aluminium and Fe are removed with this technique by formation of metal hydroxide, oxyhydroxide or hydroxysulphate precipitates as limestone dissolves and the pH is raised. Zinc and Mn are likely removed by adsorption onto reactive Fe hydroxide surfaces. Coating with Fe-hydroxides can occur but agitation and scouring of the limestone keeps fresh surfaces available (Skousen et al. 1998).

The key benefits of dosing with limestone sand are its simplicity, low cost, and the high efficiency of limestone dissolution due to minimal armouring with Fe and Al hydroxides.

There are two main limitations to this technique. The limestone pile needs replenishment on a regular basis, effectively making this a semi-active system. At 41 sites in the Middle Fork River in West Virginia treated with limestone sand dosing, workers found that limestone needed to be replenished three times per year to maintain a pH above 6 (Zurbuch 1996). Another limitation to this technique is that the length of stream in which the limestone is carried before dissolving is sacrificed for downstream improvement in water quality (Watzlaf et al. 2003).

Operation and maintenance revolves around ensuring that the correct amount of limestone is being carried away by the AMD stream. This requires having the pile in a location where the stream can continuously erode the limestone at an appropriate rate. It requires regular replenishment.

The only design criterion for this system is to choose an appropriate grain size for the limestone. If too large a grain size is chosen, Fe-hydroxides can armour the limestone before it is dissolved into the AMD. Too small a grain size may result in much of the limestone being blown away from the site as dust. Higher gradient streams, and hence, those with higher turbulence, can be treated with larger-grain-sized limestone. Optimal grain size can be chosen through trial and error. Zurbuch (1996) found that grain sizes between 0.075 and 2.36 mm work well for the Middle Fork River in West Virginia.

Reducing systems

Anoxic limestone drain

Anoxic limestone drains (ALDs) are buried trenches filled with limestone gravel (Hedin and Watzlaf 1994) (Figure 23). Water flows through the system under anoxic (low DO) conditions, dissolving the limestone thereby neutralising the acidity. ALDs are only effective when the majority of Fe is in the Fe$^{2+}$ form and very little DO is present. The common problem that plagues most oxidising passive treatment systems, armouring of limestone with Fe oxides, can be avoided with ALDs. Ferrous iron requires oxygen to oxidise to Fe$^{3+}$ and Fe$^{3+}$ readily precipitates without the need for oxygen at pH around 3 (resulting in armouring). Therefore, if the majority of Fe is Fe$^{2+}$ and no DO is present, the pH can be raised substantially without Fe oxides precipitating since Fe$^{2+}$ will not oxidise to Fe$^{3+}$ without oxygen. It is possible to form Fe$^{2+}$ precipitates, but only at higher pHs than can be achieved through limestone dissolution (Stumm and Morgan 1996).

In an ALD, the limestone is buried because conditions within the trench must be anoxic, so that all dissolved iron remains as Fe$^{2+}$, rather than converting to Fe$^{3+}$, which would quickly lead to hydrolysis and armouring of the limestone with Fe precipitates, leading to a reduction in limestone dissolution rate and, ultimately, clogging of the pore spaces of the ALD. ALDs were developed by the Tennessee Valley Authority who found that AMD seeping through a coal refuse dam was being treated passively by high
calcium carbonate limestone in an old haul road buried under a dam (Turner and McCoy 1990; Brodie et al. 1993).

Once water exits the ALD, sufficient area in the form of a ditch, settling pond, or wetland must be provided for metal oxidation, hydrolysis and precipitation to occur (Skousen 1991). Dissolved oxygen levels will increase, Fe$^{2+}$ will oxidise to Fe$^{3+}$ and Fe oxides will precipitate. Zinc and Mn are likely removed by adsorption onto reactive Fe hydroxide surfaces.

Aluminium, however, will precipitate in the ALD and, if the concentration is too high, will tend to reduce the permeability of the drain with time, potentially leading to eventual failure. Skousen et al. (2000) recommend a maximum Al concentration of 25 mg/L, but Watzlaf et al. (2000) found ALDs could fail with Al concentrations above 21 mg/L, and Watzlaf et al. (2003) recommend that Al concentrations be below 1 mg/L as an extra precaution against failure. We recommend ALDs can be used with Al concentrations up to 25 mg/L but extra permeability should be built into the ALD at the higher values.

The key benefit of an ALD is its high efficiency of limestone dissolution due to minimal armouring with Fe hydroxides. Other benefits of an ALD are its simplicity and low cost. In an analysis of long-term performance and cost–benefit of passive treatment systems, Ziemkiewicz et al. (2003a, b) found that ALDs were one of three systems that provided a high level of reliability (success versus failure), high acid load removal, and low treatment cost and they were the most consistently efficient passive treatment system in terms of cost per tonne of acid removed (Ziemkiewicz et al. 2003b).

The main limitations to this technique are its strict water chemistry limits. Dissolved oxygen levels should be below 2 mg/L and Eh at 0 or less (Skousen 1991). Some argue for DO to be less than 1 mg/L (PIRAMID 2003). Ferric iron should comprise at most 10% of the total Fe, but some workers recommend a concentration limit of 1 mg/L for Fe$^{3+}$ (PIRAMID 2003; Watzlaf et al. 2003). And Al levels should be at most no greater than 25 mg/L (Skousen et al. 2000). In addition to these, Nairn et al. (1991) and Robbins et al. (1999) show that high levels of sulphate (> 1500 mg/L) can result in precipitation of gypsum in the drain.

There is very little required operation and maintenance for ALDs. The most important is to ensure that air is continuously excluded from the system. Inflow and outflow devices/channels may need to be regularly cleared of debris.

Design criteria are largely based on residence time. The theoretical maximum level of alkalinity that can be generated is 300 mg/L, mostly in the form of bicarbonate (HCO$_3^-$; Skousen 1991). Hedin et al. (1994) recommend a residence time of 14 h, and in a review of long-term performance of ALDs, Watzlaf et al. (2000) found maximum levels of alkalinity were reached after 15 h retention. Limestone particle size is another important design criterion. Too small a size and the drain may plug and too large reduces limestone reactive surface area substantially. Skousen (1991) recommends sizes between 4 and 10 cm, and PIRAMID (2003) recommend sizes between 5 and 7.5 cm. Some 8-25 cm rocks should be used as well, especially when Al is present (Faulkner and Skousen 1996). Construction details (such as recommended width and depth) and design equations are provided in Skousen (1991), Anonymous (2001) and Watzlaf et al. (2003).

**Figure F23:** Constructing an anoxic limestone drain passive treatment system.

**Anaerobic wetland**

Anaerobic wetlands (sometimes referred to as compost wetlands) are similar in design to aerobic wetlands, except the organic substrate thickness is much greater (> 30 cm) and the thickness of free-standing water is much less (0-8 cm; Hedin et al. 1994; Skousen et al. 2000; PIRAMID 2003; Waters et al. 2003; O'Sullivan 2005). The result of this design is that water flows horizontally through the substrate material in a reducing environment rather than over
the substrate as it does in aerobic wetlands. Some wetlands have limestone mixed into the lower layers of the substrate. Emergent vegetation helps stabilise the substrate and helps maintain microbial populations, adds organic matter, and provides aesthetic qualities to the wetlands.

Anaerobic wetlands perform better with AMD containing low DO and pH above 3.5 (Waters et al. 2003), but they can tolerate high levels of DO and lower pH (Skousen et al. 2000).

Metals are removed in anaerobic wetlands by a combination of mechanisms (Skousen et al. 1992, 2000). In the aerobic surface layers, some metal oxidation and hydrolysis occurs, especially if the concentration of Fe$^{2+}$ is significant. Aluminium will form Al hydroxides on the surface of the substrate material. In the upper layers of the substrate, metal exchange and complexation reactions can occur (but only initially until sorption sites filled).

However, the dominant mechanism is chemical and microbial reduction, which precipitates metals and neutralises acidity. Microbial sulphate reduction occurs in the presence of an oxidisable carbon source (the organic substrate), which results in the formation of bicarbonate and hydrogen sulphide (H$_2$S). The H$_2$S then combines with metals to form mono- and di-sulphides (such as FeS and FeS$_2$). Compared with diffuse amorphous Fe oxyhydroxides, Fe mono- and di-sulphides are compact and dense precipitates, which would therefore hinder permeability to a lesser extent (Skousen et al. 2000). The removal of metals from solution results in a lowering of the mineral acidity and excess bicarbonate produced through sulphate reduction will consume proton acidity, thereby raising the pH (Hedin and Nairn 1992).

Dissolution of carbonates within the substrate material will also generate alkalinity (Skousen et al. 1992; Waters et al. 2003). Under the reducing environment within the substrate, the limestone will not become armoured with Fe oxyhydroxides.

The key benefits of an anaerobic wetland are its simplicity, low cost, aesthetic qualities, and ability to serve as general precipitate storage. Another main benefit is the high efficiency of limestone dissolution due to minimal armouring in the anaerobic environment. In addition to this, anaerobic wetlands are useful in locations where there is a minimal elevation drop between the AMD and the treatment system location.

Limitations of anaerobic wetlands include potential metal overloading and excessive acidity in the AMD. Metal overloading can result in exhaustion of sorption sites on the organic material and a lowering of the permeability of the organic material due to sulphide precipitation (Waters et al. 2003). High acid loads (> 300 mg/L) can overwhelm pH-sensitive microbial activities (Skousen et al. 2000). Mono-sulphides within the substrate material can oxidise and release acidity if exposed to air (Trumm 2007).

There is very little required operation and maintenance for anaerobic wetlands. Organic matter may need to be added to the wetland if performance drops (Waters et al. 2003). Inflow and outflow devices/channels may need to be regularly cleared of debris.

Anaerobic wetlands are typically sized based on acidity loading rates. For example, Hedin et al. (1994) recommend a sizing criterion of 3.5-7 g acidity/m$^2$/d and MEND (1999) recommend a loading rate of 1 kg acidity per 200-500 m$^2$/d. A review of anaerobic wetland performance and design criteria is provided in MEND (1999). Material selected for the substrate must provide sufficient dissolved organic carbon to support sulphate-reducing bacteria activity. Typical material includes woodchips, sawdust, peat moss, compost, manure, leaf litter or straw/hay (Skousen et al. 2000; Waters et al. 2003). Skousen et al. (2000) suggest that selection and enrichment of naturally occurring microbial populations help long-term performance. Plant selection and establishment details are provided in Skousen et al. (1992) and Skousen et al. (2000). PIRAMID (2003) recommend following an anaerobic wetland with an aerobic wetland to increase the DO and remove any remaining Fe and Al.

Reducing and alkalinity producing system / Bioreactor

Reducing and alkalinity producing systems (RAPS), also known as vertical flow wetlands (VFWs) (Figure 24) and successive alkalinity producing systems (SAPS), are a combination of an ALD and an organic substrate (Kepler and McCleary 1994; Zipper and Jage 2001). Sulphate reduction and metal sulphide precipitation can occur in the compost material while the underlying limestone adds alkalinity to the AMD.

Water flows vertically downward through the system. Typically, there is 1 m of standing water over 13-30 cm of organic compost, underlain by 0.5-1 m of limestone (Kepler and McCleary 1994). The water first
encounters the top of the compost layer where aerobic bacteria consume the available oxygen, then within the compost layer, where DO levels are very low, sulphate-reducing and Fe-reducing bacteria operate, then when the water reaches the limestone layer, dissolution of limestone can occur without significant armouring of Fe oxides since Fe\(^{3+}\) will have been reduced to Fe\(^{2+}\) (Skousen et al. 2000). The water leaves the system via a network of drainage pipes at the base of the limestone layer, and the outlet consists of a riser to maintain head within the system (Danehy et al. 2002).

In RAPS, reducing conditions in the compost layer reduce Fe\(^{3+}\) to Fe\(^{2+}\) and reduce sulphate to hydrogen sulphide, generating bicarbonate alkalinity (Hedin et al. 1994; Jage et al. 2001). The Fe\(^{2+}\), hydrogen sulphide and bicarbonate react to form Fe mono-sulphides. Laboratory studies demonstrate that under reducing conditions the pathway to pyrite involves the formation of mackinawite (FeS), followed by reaction with sulphide to form greigite (Fe\(_2\)S\(_4\)), which then reacts with sulphide to form pyrite (FeS\(_2\); Hunger and Benning 2007). Zinc removal in RAPS is likely through formation of Zn sulphide or as impurities in FeS (likely to be sphalerite). Manganese does not precipitate as a sulphide at circum-neutral pH, therefore it is not removed in these systems (Kepler and McCleary 1994; Trumm 2001). The Fe\(^{2+}\), hydrogen sulphide and bicarbonate are removed from the system. Solutions can include installing settling ponds before the RAPS to remove Fe, or Fe hydroxides can be removed from the top of the compost periodically.

In a review of the long-term performance of 40 RAPS, Rose (2006) found four main problems can occur.

1. AMD with high Fe concentrations can result in precipitation of Fe hydroxides on top of the compost layer which can result in a reduction in the permeability of the compost, reducing flow rates through the compost and plugging the system. Solutions can include installing settling ponds before the RAPS to remove Fe, or Fe hydroxides can be removed from the top of the compost periodically.

2. AMD with high concentrations of Al (>20 mg/L) can result in plugging of the limestone pore spaces with Al precipitate and/or decreased limestone dissolution rates due to coating of the limestone with Al precipitates. Aluminium does not form sulphides, but rather precipitates as a hydroxide at a pH of about 5 (Nordstrom and McCleskey 2006). Although not to the same extent as Fe hydroxides, Al hydroxides can armour limestone to some extent and potentially reduce dissolution rates (Hammarstrom et al. 2003). To reduce problems with Al, RAPS can be constructed with automatic flushing systems or be flushed manually more frequently (Kepler and McCleary 1997; Watzlaf et al. 2002; Weaver et al. 2004).

3. Preferential pathways can form resulting in short circuiting of the water through the compost and limestone and into the underdrain without the required residence time for treatment. This can be prevented through proper design and construction.

4. If RAPS are under-designed for the flow rate and acidity levels, inadequate treatment can result. Systems are typically designed for at least 15 h residence time in the limestone layer; however, there is also an acidity treatment limit of 35-40 g/m\(^2\)d (Rose 2004).

Operation and maintenance for RAPS consist of regularly flushing the systems to remove built-up precipitates and capture them in a holding pond. Piping and valves may need cleaning or replacement with time.

Sizing for RAPS is based both on residence time and acidity removal rates. Residence time in the limestone layer should be at least 15 h (Zipper and Jage 2001; Watzlaf et al. 2004). Jage et al. (2001) found that alkalinity generation in a RAPS is at first rapid but decreases over time, with the response of alkalinity generation to residence time logarithmic. Design recommendations for acidity removal rates range from 25-30 g/m\(^2\)/day (Watzlaf et al. 2004) to 35-40 g/m\(^2\)/day (Rose 2004).

Typical systems are designed with 13-30 cm compost underlain by 0.5-1 m limestone (Kepler and McCleary 1994), although Watzlaf et al. (2003) suggest compost thicknesses between 15 and 60 cm and Demchak et al. (2001) suggest at least 50 cm of compost. Thicker layers are important for highly oxidised AMD. Limestone grain size should be between 6 and 15 cm. Details and guidelines on designing systems can be
found in Kepler and Mccleary (1994), Skousen et al. (1998), and Skovran and Clouser (1998).

**Figure F24:** Vertical-flow wetland passive treatment system.

*Sulphate-reducing bioreactor*

Bioreactors are similar to VFWs except that the neutralising material (usually limestone) is homogeneously mixed with the organic material and there are no plants growing in the substrate (Gusek 2002, 2004; Neculita et al. 2007). Bioreactors are relatively new to the arsenal of passive treatment for AMD.

As with VFWs, water flows vertically through the bioreactor (usually down, but sometimes up) and AMD is treated through sulphate reduction reactions and dissolution of limestone (Figure F25). The water leaves the system via a network of drainage pipes at the base of the system, and the outlet consists of a riser to maintain head within the system.

Metal removal is dominantly through reduction as metal sulphides through the activity of sulphate-reducing bacteria (Nordwick et al. 2006). Microbial sulphate reduction occurs in the presence of an oxidisable carbon source (organic material in the bioreactor), which results in the formation of bicarbonate and hydrogen sulphide (H$_2$S). The H$_2$S then combines with metals to form mono- and di-sulphides (such as FeS and FeS$_2$). Compared with diffuse amorphous Fe oxyhydroxides, Fe mono- and di-sulphides are compact and dense precipitates, which would therefore hinder permeability to a lesser extent (Skousen et al. 2000). The removal of metals from solution results in a lowering of the mineral acidity, and excess bicarbonate produced through sulphate reduction will consume proton acidity, thereby raising the pH (Hedin and Nairn 1992). Dissolution of carbonates within the substrate material will also generate alkalinity. Bioreactors, therefore, are good in situations where multiple metals (such as Cu, Cd, Zn) are present. Aluminium does not precipitate as a sulphide but rather as an Al hydroxysulphate (Thomas and Romanek 2002).

Other metal removal processes that can operate include ion exchange of metals by an organic-rich substrate, precipitation of metal hydroxides, adsorption of metals by precipitated Fe hydroxides, precipitation of carbonates, and precipitation of silicates (Gusek 2002, 2004; Nordwick et al. 2006).

Gusek (2002) provides a list of the benefits of bioreactors over other systems. They can treat low pH water containing multiple metals, they are resilient to metal loading, and they are less prone to plugging by Al since Al hydroxysulphate is more compact that Al hydroxide.

The main limitation of bioreactors is the potential for short circuiting through the unit (Gusek 2004). Careful selection of material and complete mixing can help to prevent this problem. Also, constructing multiple smaller systems is better than one large system.
Very little operation and maintenance is required for bioreactors. Piping and valves may need cleaning or replacement with time. Inflow and outflow devices/channels may need to be regularly cleared of debris.

Wildeman et al. (2006) recommend a design criterion of 0.3 moles of metal removal/m$^3$ of substrate/day for bioreactors with a mixture of organic materials and crushed limestone. Gusek (2002, 2004) provides details on design and materials for bioreactors. Organic material can include wood chips, hay and straw (spoiled), sawdust, cardboard, rice hulls, soy bean hulls, yard waste, waste alcohols including antifreeze, mushroom compost, waste dairy products, animal manure, and sugar cane processing residue (bagasse). McCauley et al. (2009) used mussel shells for the neutralising material in laboratory trials of bioreactors treating low pH AMD containing multiple metals.

Figure F25: Sulphate-reducing bioreactor passive treatment system.

F.4 AMDTreat

AMDTreat is a free Windows-based computer program designed to estimate the capital and annual costs to treat mine discharges (Means et al. 2003). It was developed cooperatively by the Pennsylvania Department of Environmental Protection (http://www.dep.state.pa.us/), the West Virginia Department of Environmental Protection (http://www.dep.state.wv.us/), and the U.S. Office of Surface Mining Reclamation and Enforcement (http://www.osmre.gov/). It can be downloaded from the AMDTreat Home Page (http://amdtreat.osmre.gov/). Version 4.1C was released on 4 February 2008.

AMDTreat uses a three-step approach to estimate treatment costs:
- Users enter water quality and quantity data.
- Users ‘build’ an active and/or passive treatment system by selecting the applicable treatment components from the software menu.
- Users customise each treatment system to site-specific conditions by controlling the size, quantity, and unit cost of treatment components.

The water quality and quantity data to be entered include: acidity, alkalinity, total Fe, ferric Fe, ferrous Fe, filtered Fe, total Al, filtered Al, total Mn, filtered Mn, sulphate, pH, specific conductivity, total dissolved solids, DO, typical flow, design flow. Not all of the parameters are used in the calculations (the program only requires acidity, alkalinity, total Fe, ferrous Fe, total Al, total Mn, sulphate, pH, typical flow, and design flow). If acidity is not known, an acidity calculator can be used to estimate the value.

The passive treatment systems that can be constructed using the program include vertical flow ponds, anoxic limestone drains, anaerobic wetlands, aerobic wetlands, manganese removal beds, oxic limestone channels, limestone beds, and bioreactors. Active treatment systems that can be constructed include caustic soda, hydrated lime, pebble quicklime, ammonia, oxidation chemicals, and soda ash. The program does not help the user to choose the most appropriate solution for a given water chemistry and flow rate. It is up to the user to identify which solutions may or may not work for the given site. The program then can be used to compare relative costs between different types of systems (Table F4).

To help select among appropriate active treatment systems, Means (2006) provides specific details for
each chemical, including: system description, typical set-up, chemical facts, benefits, limitations, maintenance requirements, and economics.

The model combines costs from these treatment methods with costs of ancillary treatment components, such as settling ponds, roads, land access, ditching, and engineering costs to calculate a site-specific capital cost. Similarly, AMDTreat calculates annual costs by taking into account user-provided information regarding sampling, labour, maintenance, pumping, chemical consumption, and sludge removal. Capital and annual costs can be used in conjunction with AMDTreat’s financial forecasting utility to evaluate the economics of long-term treatment. Over 400 user-modifiable variables are provided for modelling costs.

AMDTreat also contains several scientific tools to help select and plan treatment systems. These tools include an acidity calculator, a sulphate reduction calculator, a mass balance calculator, and a passive treatment alkalinity calculator.

Table 1: Hypothetical AMD chemistry used for comparison of passive treatment system construction costs using the program AMDTreat. Selected chemistry can be treated by any passive treatment system. For construction costs see Table 3 and Figure 21 in main text.

<table>
<thead>
<tr>
<th>Acidity</th>
<th>150 mg/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow rate</td>
<td>10 L/s</td>
</tr>
<tr>
<td>Al</td>
<td>12 mg/L</td>
</tr>
<tr>
<td>Fe total</td>
<td>11 mg/L</td>
</tr>
<tr>
<td>Ferrous</td>
<td>10 mg/L</td>
</tr>
<tr>
<td>Ferric</td>
<td>1 mg/L</td>
</tr>
<tr>
<td>pH</td>
<td>2.9</td>
</tr>
<tr>
<td>DO</td>
<td>1 mg/L</td>
</tr>
<tr>
<td>Acid load</td>
<td>52.1 tonnes/year</td>
</tr>
</tbody>
</table>

F.5 Acidity and treatment

The acidity of a solution, not the pH, is probably the best single indicator of the severity of AMD (Rose and Cravotta 1998). The hydrolysis of metals in AMD, as the pH is raised, produces hydrogen ion acidity as shown in the following reactions (Stumm and Morgan 1996):

\[ Fe^{2+} + 0.25 O_2 + H^+ = Fe^{3+} + 0.5 H_2O \]

Therefore, acidity in AMD is comprised of mineral acidity (the hydroxide ion demand by cations of Fe, Al, Mn and others) and hydrogen ion acidity (measured as pH units). Acidity can either be measured in a laboratory through the hot acidity procedure in which hydrogen peroxide and heating are used to oxidise and hydrolyse metals, followed by titration with base to a pH of 8.2 or 8.3 (USEPA 1979; APHA 1980; ASTM 1994), or it can be calculated using the concentrations of dissolved metals and pH according to the following formula (Hedin et al. 1994):

\[
Acidity = 50 \left( \frac{2Fe^{2+}/55.85 + 3Fe^{3+}/55.85 + 3Al^{3+}/26.98 + 2Mn^{2+}/54.94 + 1000(10^{-pH})}{26.98} \right)
\]

(Fe\textsuperscript{2+}, Fe\textsuperscript{3+}, Al\textsuperscript{3+}, Mn\textsuperscript{2+} are dissolved concentrations in mg/L)

Although Fe\textsuperscript{2+}, Fe\textsuperscript{3+}, Al\textsuperscript{3+}, Mn\textsuperscript{2+} and H\textsuperscript{+} are the major components of acidity in coal-mine drainage (Ott 1986), other dissolved species that precipitate as hydroxides or oxides or change form during the acidity titration, including Mg\textsuperscript{2+}, H\textsubscript{2}CO\textsubscript{3}, or H\textsubscript{2}S, can contribute to acidity (Payne and Yeates 1970). Therefore, if possible, the laboratory acidity measurement is preferable to the calculation method.

Acidity is commonly expressed as milligrams of CaCO\textsubscript{3} per litre of solution (mg/L as CaCO\textsubscript{3}) on the basis of the following stoichiometric relation:

\[ 2H^+ + CaCO_3 = Ca^{2+} + CO_2 + H_2O \]

Sequential titration methodology

Recent work has shown that often there is poor agreement between calculated acidity and laboratory-measured acidity and that sometimes neither method adequately predicts the amount of neutralising material necessary to neutralise the AMD and lower the concentrations of dissolved metals to acceptable levels (Means and Hilton 2004). Hilton (2004) suggests that multiple titrations are conducted in the field to determine the amount of neutralising material needed to treat the AMD to a specified pH level or specified concentrations of dissolved metals. This methodology is known as sequential titration. The procedure lends itself to field applications for instantaneous results, or for laboratory analysis where more detailed analyses are required.
In the methodology, raw AMD water is repeatedly titrated with NaOH. Each sequential titration is conducted with an increasing amount of NaOH and each titration is analysed for dissolved metals and pH. Once the goals for water quality treatment are determined, these titration data can be used to design an appropriately sized treatment system. See Trumm and Cavanagh (2006) for an example of the titration conducted at a New Zealand AMD site.

F.6 Small-scale trials for AMD passive treatment

For sites with existing AMD where passive treatment is being considered, data can be collected from the AMD over a period of time to add confidence to the selection of potential treatment techniques. Once potential treatment solutions have been identified through the use of the flow chart presented in the main text, Trumm (2007) recommends that small-scale trials be conducted on site to test the effectiveness of the various options before investing in full-scale system construction (see Trumm et al. 2006, 2008, 2009; McCauley et al. 2008 for examples of small-scale trials in New Zealand). Even if only one option is indicated through the use of the flow chart, field trials should still be conducted because unknown factors can influence the effectiveness of treatment systems. The choice of the full-scale system should be based on the results of the field trials and a review of the cost, effectiveness, limitations and risk of failure for each option. The following sections detail methodologies for small-scale trials.

Collect data and determine remedial action objectives

If possible, AMD water chemistry and flow rate should be measured monthly for at least 12 months. Water chemistry parameters should include the field measurement of pH, conductivity, temperature, dissolved oxygen, and ferrous Fe concentration, and laboratory analyses for total acidity, Fe and Al concentrations (dissolved and total), and sulphate. Any additional metals or metalloids should be added to the list of analytes that are at concentrations of concern. A sequential titration (Hilton 2004) should be conducted at least once (see section above).

Flow rate should be determined by either manual measurement on the same day as water chemistry is measured or preferably by data logger on a continuous basis. If possible, additional water quality and quantity data should be collected during high and low flow events to enable determination of flow variation on water chemistry.

Using the acceptable level of impact (from stakeholders), analyse the data to determine the levels by which metals should be reduced and the required increase in pH. Determine the acid load (kg/day) above acceptable limits and validate the calculations with the treatment level determined from the titration results (NaOH equivalent to lower concentration of metals and raise pH to acceptable levels).

Design and construct pilot trials

Using water chemistry data (Fe and Al concentrations, DO) and available land area, identify potential passive treatment solutions with the flow chart in the main text (there is often more than one). Further verify appropriate solutions by reviewing descriptions, examples, effectiveness, limitations, risk of failure, and relative cost of systems in the section AMD Passive Treatment Systems later in this appendix.

For each potential solution being considered for the site, design and construct a pilot trial to be operated at the site or to be operated in a controlled laboratory situation. If the trials are to be operated in a laboratory, AMD water from the site should be transported to the laboratory and sent through the systems. Three factors need to be considered when designing and constructing the systems:

- Size and dimensions of the trials
- Treatment media specifications for each system
- Amount of AMD to be treated for each system

The budget and availability of materials are typically used to determine the size and dimensions of the small-scale systems to be constructed. Typical sizes range from 0.5 m³ (Trumm et al. 2008) to 1 m³ (McCauley et al. 2008; Trumm et al. 2009), and containers and piping are usually plastic. For treatment media type, particle size, and volume, refer to system details in the section AMD Passive Treatment Systems later in this appendix. If particle sizes in the trials are much smaller than those recommended for full-scale systems, grain size surface area should be considered when later analysing the data for system performance and scaling up to full size. To determine the amount of AMD that can be treated by each system, the
computer program AMDTreat can be used (see previous section above) and formulas for designing each system should be consulted (see section AMD Passive Treatment Systems below). Piping and valves are used to convey only the amount of AMD needed to pass through each system.

**Operate pilot trials and collect data**

During operation of the pilot trials, data should be collected on a regular basis according to the duration of the trials to monitor system effectiveness with time and variations in inlet water chemistry (weekly, fortnightly, or monthly). Typical trial durations are 3–6 months. Data should include the field parameters: pH, conductivity, temperature, DO, ferrous Fe concentration and flow rate, and samples should be collected for laboratory analysis of Fe and Al (dissolved and total concentrations), sulphate, sulphides (where applicable), acidity, and any other metals or metalloids of concern.

Ecotoxicity experiments should be conducted using treated water to verify that treatment will enable restoration of the aquatic ecosystem (for examples of ecotoxicity experiments see Trumm et al. 2003; Trumm and Gordon 2004).

To determine system limits and effectiveness, flow rates (and therefore, residence times) should be varied during the duration of the trial.

For systems with flushing mechanisms, such as vertical flow wetlands and leaching beds, the pilot trials should be flushed at least once to test flushing effectiveness at removing accumulated contaminants (for piping design and sampling during flushing see Danehy et al. 2002; Watzlaf et al. 2002; Weaver et al. 2004; Trumm et al. 2008).

**Analyse data and select full-scale system**

Once operation of the systems has ceased, the system should be deconstructed (autopsy) and additional data should be collected. This includes photographing treatment media, analysing armouring on treatment media using SEM or other applicable methods (see Hammarstrom et al. 2003), and analysing treatment media for contaminants of concern removed during the trial.

Data analysis from the trials should include the following:

- Calculate residence times in the systems
- Calculate contaminant removal rates and compare to residence times and duration of system operation
- Determine system lifespan, if possible, by comparing treatment media before and after the pilot trial and analyse applicable analytical results to determine consumption of neutralising material in kg/day
- Analyse data collected during flushing, and from system deconstruction to determine effectiveness of flushing mechanisms in removing accumulated contaminants
- Analyse system deconstruction data to determine where contaminants accumulated
- Determine if treatment was effective in reducing toxicity of untreated water

For examples of data analysis and interpretation of pilot trials see Trumm et al. (2006, 2008, 2009) and McCauley et al. (2008).

The results of pilot trials or laboratory experiments are used indicate the effectiveness of a given system in meeting the treatment goals. Selection of a full-scale treatment system is based on the effectiveness of that system, in addition to ease of implementation and cost. Ease of implementation considers the practicability of implementing the different treatment systems and can include consideration of the site-specific constraints (such as space availability, terrain, etc.), as well as the experience of personnel involved in the construction of different systems. Cost should be an order-of-magnitude cost estimate to allow comparison of the alternatives from a cost standpoint. These costs should approximate the total implementation cost, which would include design, construction, operation, and engineering services during construction. These estimates are developed using quotations by vendors and estimating values.
F.7 Arsenic treatment - hard rock gold mines

Active treatment systems
There are many active treatment technologies that can be used to remove As from water. The following discussion is largely based on USEPA (2002). A brief description is provided below, with further details, including benefits and limitations of the technology provided in Table F5.

Oxidation
Oxidation of As$^{3+}$ to As$^{5+}$ improves the performance of As removal technologies. It often forms a pretreatment step or may be an intrinsic part of another technology, although it is not typically used alone as an As treatment. Technologies that are typically considered as As removal processes are:

Coagulation/precipitation
This process uses chemicals to transform dissolved contaminants into an insoluble solid, either through direct precipitation or adsorption onto another species that is precipitated (co-precipitation). The precipitated solid is then removed from the liquid phase by clarification or filtration. Conventional chemicals used include Al and Fe hydroxides, ferric salts (e.g., ferric sulphate, ferric chloride) ferrihydrite, lime and limestone. The effectiveness of this technology is less likely to be reduced by characteristics and contaminants other than As, compared with other water treatment technologies. It is also capable of treating water characteristics or contaminants other than As, such as hardness or heavy metals. Systems using this technology generally require skilled operators; therefore, precipitation/coprecipitation is more cost effective at a large scale where labour costs can be spread over a larger amount of treated water produced. Chemical costs may also influence costs.

This technique is the most widely used active treatment technology in the US for As removal from a variety of As-contaminated waters (USEPA 2002). This process is also used in New Zealand, for the treatment of As-contaminated water at OceanaGold’s Globe Progress Mine at Reefton (below).

Case study: Active treatment at Globe Progress Mine, Reefton
OceanaGold’s Globe Progress Mine at Reefton uses a coagulation/precipitation process and treats water with As concentrations up to 2.8 mg/L. A schematic diagram of the treatment plant is shown in Figure F26. In the plant all water potentially containing As (mine water, tailings return water, waste-rock seepage) and plant sump return water is initially fed into an oxidation tank to which Fe chloride is added in small amounts. Pilot trials were used to determine the amount of Fe chloride to be added. This tank is aerated by an agitator and air injection. The water then enters the second tank (precipitation tank) where lime is added with mixing, to adjust the pH to the target of 7.0. Flocculent overflow from the clarifier and raw water are added to the tank. Flocculent is added to the water overflow from the precipitation tank, which then enters the clarifier. The solids settle out and are pumped from the bottom into a tailings dam. The clean water is pumped into a discharge lagoon that overflows to the nearby stream. The stream is monitored regularly for As and is required to remain below a trigger level set by the regional council (maximum As 3.3 mg/L, 90th percentile As 0.25 mg/L).
Adsorption

Adsorption techniques are typically based on contaminated water passing through a fixed bed of media. Dissolved contaminants concentrate at the surface of the media (sorbent), thereby reducing their concentration in the water. As contaminated water is passed through the column, contaminants are adsorbed. When adsorption sites become filled, the column must be regenerated or disposed of and replaced with new media. Competition for adsorption sites could reduce the effectiveness of adsorption because other constituents may be preferentially adsorbed, resulting in a need for more frequent bed regeneration or replacement. Phosphates and silicates are reported to compete with As\(^{5+}\), while sulphate and chloride will compete more favourably with As\(^{3+}\).

Common adsorbents are activated alumina, activated carbon, Fe-based adsorption media (including granular ferric hydroxide, Fe-oxide-coated sand and Fe ferricydrite). Of these, Fe-rich absorbents may be particularly useful in New Zealand as they are reasonably cheap and abundant, and often available naturally (Fe-Mn ore and hydrated Fe oxides from Fe-rich waters) or as a by-product from mining and processing such as AMD sludge (amorphous Fe oxide).

Ion exchange

Ion exchange is a physical/chemical process in which ions held electrostatically on the surface of a solid are exchanged for ions of similar charge in a solution. This process typically uses a resin made from synthetic organic materials, inorganic materials, or natural polymeric materials that contain ionic functional groups to which exchangeable ions are attached. Specifically, ions are removed from the aqueous phase by the exchange of cations or anions between the contaminants and the exchange medium. Ion exchange resins must be periodically regenerated to remove the adsorbed contaminants and replenish the exchanged ions.

Membrane filtration

In this technique, As is removed by passing contaminated water through a semi-permeable membrane or barrier. There are four types of membrane processes, categorised by the size of the particles that pass through the membranes: reverse osmosis (RO), nanofiltration (NF), ultrafiltration (UF), and microfiltration (MF). All four are pressure-driven
and the force required to drive fluids across the membranes depends on the pore size; NF and RO require a relatively high pressure (50–150 pounds per square inch [psi]), while MF and UF require a relatively low pressure (5–100 psi). The low-pressure processes primarily remove contaminants through physical sieving and the high-pressure processes primarily remove contaminants through chemical diffusion across the permeable membrane. These processes are often expensive and membranes are prone to fouling due to the small pore sizes, and it is more likely to be used as a finishing step for As removal.

**Biological**

Biological treatment for As removal is an emerging technique that uses biological activity to promote precipitation of As from water. Arsenic removal may occur either as a result of biological activity, typically using microbes, creating ambient conditions that result in As precipitating or by transforming As species into those that are more amenable to precipitation. The microbes may be suspended in the water or attached to a submerged solid substrate. Iron or hydrogen sulphide may also be added.

**Passive treatment systems**

There are few examples of passive treatment of mining water contaminated with As in the international literature. Treatment systems typically use Fe-rich adsorbants, and Fe oxyhydroxides as these are naturally formed. For example, The Wheal-Jane constructed wetland can remove As (initial As <2 mg/L) by precipitation of amorphous Fe oxyhydroxides in acidic conditions (pH 3–4) (Swash and Monhemius 2005). This system consists of a pH-conditioning cell, 5 aerobic cells; 1 anaerobic cell and 9 rock filtration cells in which As is removed in the aerobic cells and rock filter (Swash and Monhemius 2005). Wildeman et al. (2006) describe benchtop passive treatment of As (initial concentration 129.8 mg/L) and cyanide-rich water at a gold mine site in Brazil by zerovalent Fe.

More recently, passive treatment of As using AMD sludge (which contains Fe oxyhydroxides) has been investigated in New Zealand and has provided promising results. This research is described below.

**Case study: Passive treatment of As-contaminated water at Waiuta, West Coast, using AMD sludge**

The efficiency of AMD sludge at removing As from surface water was investigated in laboratory and field experiments. AMD sludge was obtained from active treatment of AMD at a working coal mine (Stockton, ST) and natural precipitation from untreated AMD at the abandoned Blackball Coal Mine (BB) on the West Coast, New Zealand. The sludge contained Fe oxide at concentrations of 13 wt% (Stockton sludge) and 74 wt% (Blackball sludge) respectively. Water used for laboratory trials was obtained from a contaminated gold mine site at Waiuta, West Coast, New Zealand (Haffert and Craw 2008a, b). Laboratory batch and column experiments were initially performed to determine the ability of AMD precipitate to remove As from mine drainage. Subsequently, a field trial was established at the Waiuta site. Further details are available in Rait et al. (2010).

**Laboratory trials**

Batch and column experiments were performed in the laboratory to determine the ability of AMD precipitate to remove As from mine drainage. Water used in the laboratory experiments contained up to 99 mg/L As. In the initial batch experiments rapid reduction in As concentrations occurred within 15 h, with BB sludge reducing As levels below that for ST sludge. After 48 h at a ratio of 10 g sludge to 1 L water As concentrations were lowered from 99 to 0.016 mg/L (BB) and 0.55 mg/L (ST). For a higher ratio of 50 g sludge to 1 L water As concentrations after 48 h were lowered to 0.0017 mg/L (BB) and 0.008 mg/L (ST). A column leaching experiment was then conducted to determine long-term adsorption potential. Water (at average concentrations of 40 mg/L As) was passed through columns, at an average flow rate of 1.8 L/day, with sand coated in the powdered AMD sludge for 10 days at a constant contact residence time of 1 day. Two columns were packed at a ratio of 50 g sludge (ST and BB) to 1 L water and one at a ratio of 10 g sludge (BB) to 1 L water. All columns lowered As concentrations in the effluent to <0.01 mg/L for the first 2 days. However, after 2 days the columns with ST sludge and BB sludge at a ratio of 10:1 showed a steady increase in effluent As concentrations, suggesting that adsorption sites were being exhausted. The column with the BB sludge at a ratio of 50:1 continued to have effluent As concentrations of <0.01 mg/L for 9 days, only increasing to 0.04 mg/L on day 10, suggesting many more available adsorption sites on the naturally-
precipitated AMD sludge. A toxicity characteristic leaching procedure test on the BB sludge/sand mixture packed at the 50:1 ratio showed <0.021 mg/kg As, indicating relatively good stability of the adsorbed As. These results suggest that As can be treated with AMD sludge if the ratio of sludge to water and contact residence time are optimised.

**Field trials**

Based on the above laboratory trials, a small-scale field trial was established at the Waiuta site. At this location, water from a wetland flows through a manmade dam and emerges on the other side at a flow rate of about 2 L/s and As concentrations about 2.4 mg/L. The dam acts as a natural As removal system reducing concentrations from 52 (wetland) to 2.4 mg/L (Haffert and Craw 2008a). The field trial was constructed using 1000-L plastic tubs (filled with sand/AMD mixtures), PVC piping and valves. Three tubs were set up: one with ST sludge and two with BB sludge (BB-1 and BB-2) at loading ratios of 139, 99 and 99 g sludge/L water, respectively. A flow rate of 0.006 L/s was used as the primary flow rate, which provided the optimal residence time of 20 h. Water inflow and outflow were sampled on a daily basis for the first 3 days, weekly thereafter for 11 weeks, and fortnightly for 9 months. The influence of residence time on As removal was examined by varying the inflow rate, which is inversely proportional to residence time, for BB-2 to assess As removal at 1x, 2x, 3x, and 4x the optimal residence time. Longer residence times (i.e. closer to 20 h) were more effective in lowering As concentrations. The results of the trials showed that removal rates closely matched the adsorption isotherms determined in the laboratory study and that percent removal is predictable. These results have been used to design a full-scale system for the site (Trumm et al. 2012).
Table F4: Summary of arsenic removal technologies

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<tr>
<th>Technology</th>
<th>Techniques</th>
<th>Chemicals and benefits</th>
<th>Limitations</th>
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<tr>
<td>Oxidation</td>
<td>Air oxidation, chemical oxidation, photo catalytic, electrochemical preoxidation</td>
<td>Mainly used in conjunction with other technologies as oxidises As(III) to As(V) for removal. By stirring or cascade in ponds. Relatively simple, low-cost but slow process. Also oxidises other inorganic and organic constituents in water. Simple, rapid process. Minimum residual mass. Chemicals include calcium hypochlorite, ozone, bleach, Mn oxide, permanganate, chlorine. Fe(VI) electrochemical oxidation of As(III). Can be used on a wide range of water contaminants.</td>
<td>Mainly removes As(V) and accelerates the oxidation process. Efficient control of the pH and oxidation step is needed. Can be costly. Interfering reductants can inhibit. As sludge needs to be disposed.</td>
<td>Buisson et al. (1979), Hug et al. (2001), Arienzo et al. (2002), Lee and Choi (2002), Ghurye and Clifford (2004), Dutta et al. (2005), Light and Yu (2005), Moore (2005), Pokhrel et al. (2005), Swash and Monhemius (2005), Ferguson and Hering (2006), Yuan et al. (2006), Zhang and Itoh (2006), Habuda-Stanic et al. (2007)</td>
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<tr>
<td>Coagulation/precipitation</td>
<td>Al coagulation, Fe coagulation, ferric ions plus calcite, lime and limestone</td>
<td>Low capital cost. Simple process, Al and Fe hydroxide. Effective over a wider range of pH. Common chemicals are available, ferric sulphate, ferric chloride, ferrihydrate. Medium to high removal of As(III). Fe2(SO4)3·5H2O with calcite to enhance removal. Most common chemicals are available commercially, Granulated lime, calcium carbonate, limestone.</td>
<td>Produces sludge. Disposal of sludge required. Anaerobic bacteria can reduce ferrihydrate to mobilise As. Competing species. Sedimentation and filtration needed</td>
<td>Buisson et al. (1979), Meng et al. (2001), Chen et al. (2002), Brown et al. (2003), Wickramasinghe et al. (2004), Dousova et al. (2005), Tadanier et al. (2005), Davis et al. (2006), Parks and Edwards (2006), Song et al. (2006), Violante et al. (2006), Yuan et al. (2006), Lee et al. (2007), Masue et al. (2007)</td>
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<tr>
<td>Technology</td>
<td>Techniques</td>
<td>Chemicals and benefits</td>
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<tr>
<td>Membranes, reverse osmosis, exchange resin</td>
<td>Nanofiltration, ultrafiltration, reverse osmosis, electrodialysis, compost-based permeable reactive barrier, on exchange membrane bioreactor</td>
<td>Nanofiltration is well-defined and high-removal efficiency. Pre-oxidation required. Micellar enhanced ultrafiltration with high molecular weight cut-off membranes.</td>
<td>High-cost medium. Not very economic if high TDS. Disposal of residuals. Toxic wastewater produced. Requires high-tech operation and maintenance. May be retarded by other trace metals</td>
<td>Mondal et al. (2007), Zhang et al. (2007), Ning (2002), Wang et al. (2002), Clifford et al. (2003), Moore (2005), Köber et al. (2005), Nguyen et al. (2006), Oehmen et al. (2006), Ramakrishna et al. (2006), Beolchini et al. (2007), Habuda-Stanic et al. (2007)</td>
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Appendix G: EXTREME EVENTS

This section provides an overview of extreme events that may impact on mining operations, and subsequently on nearby aquatic ecosystems. There is limited research on the impact of extreme events on aquatic ecosystems, and these impacts will be highly variable for the same type of event. Similarly, it is often difficult to gauge the likelihood of the various events occurring. For these reasons, this section is based on the current state of knowledge and is presented in a qualitative rather than quantitative format.

Extreme events that may result in additional impacts on aquatic ecosystems from gold and coal mining can be divided into events directly associated with mining activities (e.g. treatment system failure, tailings dam failure, cyanide spill) and natural events (e.g. extreme high rainfall events, landslide, drought). This appendix provides a discussion of the likely impacts arising from various extreme events, prevention and treatment activities, as well as consideration of how extreme events can be taken into account during mine planning.

G.1 Mining-related extreme events

Treatment system failure

Treatment system failure may occur as a result of poor fit of treatment system type to site, under-design of treatment system, mechanical failure, or inadequate operation and maintenance.

Selection between active and passive treatment and design of system within each category are critical to achieve required water quality. Active treatment systems are much more forgiving to inadequate design, as modules can be added to the system to correct most shortcomings. Passive systems, however, are much more difficult to rectify if performing inadequately since the system type might not be appropriate for the site and an entirely different system may need to be constructed. For example if an anoxic limestone drain is constructed for an AMD with high concentrations of Fe^{3+}, armouring of limestone and plugging of pore spaces with precipitates can result in complete failure of the system and a new system, such as a vertical flow wetland (VFW), may need to be constructed. Poor fit of passive treatment system type to site chemistry, flow rate, and topography is the most common form of treatment system failure and is an area of active research.

Mechanical failure is more likely in active treatment systems rather than passive systems since passive systems generally do not have any moving parts. Active treatment involves the operation of many pumps, valves, metering devices, mixing paddles, etc., each of which has a definite lifespan and can fail prematurely. Passive systems usually have piping and valves to convey water to and through the systems. Examples of mechanical failure in passive treatment systems include build-up of rust or precipitates on valves limiting operation, build-up of precipitates in piping clogging passageway, and breaking of piping due to excessive overburden pressure.

Inadequate operation and maintenance can occur in both active and passive treatment systems. Active systems are much more labour intensive than passive systems and typically have personnel on site on a daily basis. As such, treatment system failure due to inadequate operation and maintenance can usually be rectified quickly. Passive systems, however, may not be visited for up to a month or more, so a failure may have a much greater impact. Typical operation and maintenance requirements for active and passive systems are included in Chapter 8 (Decision making and monitoring).

Treatment system failure for active treatment systems can result in discharge of water with chemistry similar to untreated water or can have worse water chemistry. If excessive neutralising agent is added, the pH can be elevated much higher than 7 and unreacted neutralising chemicals can be released in the discharge water. Depending on the chemicals used, this can result in excessive sodium, ammonia, or chloride in the water. Excessive flocculant or coagulant can occur in the discharge water as well.

Failure of passive systems usually results in water discharge with chemistry similar to untreated mine discharge, but in some cases can have worse water chemistry. Some systems, such as slag leaching beds (SLBs), can raise the pH higher than 7, and some systems, such as SLBs and VFWs, can release other elements into the water. SLBs can contain numerous elements that were impurities in the iron ore and VFWs can contain various pesticides and metalloids (such as As) in compost.

Impacts on stream communities

The impacts on stream communities will depend on what the treatment system was treating, for example, suspended solids, pH, or trace elements. The release of sediment from suspended solid treatment-systems may result in physical choking of the stream and its biota when deposited. Sediments with the finest particles, when dried on stream banks, can form a hard deposit.
that radically changes the nature of the streambed. Fine particles also infill spaces between large substrate within the wetted streambed, reducing the quantity and quality of habitat available for macroinvertebrates.

Prevention

The best prevention is to select and design the most appropriate treatment system for the site and to conduct adequate operation and maintenance. Treatment systems should have alarms to notify operators of a failure and, if possible, excess capacity should be built into the system to contain untreated water on site.

Treatment

Treatment after treatment-system failure depends on what water chemistry was discharged from the site. If metals or metalloids are discharged only in dissolved form, then there are no remedial activities that can be conducted in the receiving stream. However, if particulates are discharged into the stream, these may settle on the streambed and alter the aquatic ecology. Artificial or natural treatment may be required.

Artificial treatment involves collection and removal of the precipitates to a repository or back into the treatment system. This process can be disruptive to the stream, as access roads are needed and the streambed must be disturbed with earth-moving equipment.

Natural treatment consists of allowing rain events such as regular floods to remove the precipitates from the stream. Remnants of precipitates are progressively colonised by the stream biota.

Tailings dam failure

Tailings dam failures are surprisingly common around the world, so possible failure of a tailings dam on a New Zealand mine site has to be taken seriously. Examples of tailings dam failures, the reasons for the failures, and consequences, around the world are documented on the following website: http://www.wise-uranium.org/mdaf.html.

The tailings dam failure event that has attracted the most scientific study is that of the Los Frailes mine, Aznalcollar, Spain, in 1998. An extensive scientific literature is available on this event (starting with The Science of the Total Environment, volume 242, 1999, Elsevier), and new research is still being published.

Failure of a tailings dam will typically involve dispersal of large volumes of tailings as solid-rich slurry into the nearest stream. The tailings settle progressively on the bed and banks of the stream, with coarsest material (sand and larger particles) near the dam, and finer material (silt and mud) further downstream. Sediment transport can be for tens of kilometres.

Impacts on stream communities

The sediment, when deposited, may result in physical choking of the stream and its biota. Sediments with the finest particles, when dried on stream banks, can form a hard deposit that radically changes the nature of the streambed. Fine particles also infill spaces between large substrate within the wetted streambed, reducing the quantity and quality of habitat available for macroinvertebrates. Alternately, chemical dissolution of elements such as As may occur, or if the tailings contain pyrite, oxidation of that pyrite may result in AMD issues.

Prevention

Prevention of tailings dam failure should be a key aim in original design. The following points are significant in this regard:

- **Construction material.** Many overseas tailings dams are built with dried tailings, and these are most prone to failure. Construction of dams with a majority of core as waste rock is more likely to produce a long-term stable structure.

- **Site selection.** New Zealand has steep topography, and many hillsides are underlain by active or inactive landslides. Loading of these hillsides with dams and their contained tailings, and earthworks associated with site construction, can activate or enhance landslide activity. The Golden Cross gold mine (Coromandel, NZ) was found to be sited on an active landslide, and major treatment work was required to stabilise the structure to prevent dam failure in the 1990s. This work is ongoing, after the mine has closed.

- **Water management.** Many tailings dam failures result from water erosion and related phenomena during extreme rain events. The Opuha irrigation water dam (South Canterbury, NZ) failed in a major rain event during construction in 1997, suddenly releasing large volumes of water downstream (Lees and Thompson 2003). Diversion channels should be designed to cope with such extreme rain events.

- **Height additions.** Most tailings dams, unlike dams for irrigation and hydroelectricity generation, are built progressively, with additional height added as the tailings levels rise. Problems with construction of these additions have caused many tailings dam failures.

The International Commission on Large Dams (ICOLD) has a large number of technical publications related to all aspects of dam design, construction, operation, monitoring and maintenance, including UNEP (2001) and

Treatment

Treatment activity after a tailings dam failure is dependent on the scale of the failure, the thickness of sediment resulting, and the potential chemical effects of that sediment.
- **Artificial treatment.** This typically involves collection and removal of the tailings to a repository, normally the repaired dam from which they came. The treatment process can be disruptive to the stream, as access roads are needed and the streambed must be disturbed with earth-moving equipment.
- **Natural treatment.** This mainly depends on natural rain events such as regular floods to remove the tailings from the stream. Remnants of tailings are progressively colonised by the stream biota. The process can take decades (Black et al. 2004).

Monitoring of the success of treatment of streams after a tailings dam failure is greatly facilitated if the geological, chemical and biological state of the affected stream was known before the tailings arrived. This background information provides a baseline against which the treatment activity can be measured. Many mines are developed in areas where mining has occurred historically, and there are natural geochemical anomalies around such sites, so these streams are typically different from surrounding streams. Hence, clear definition of background data for streams that could be impacted by tailings dam failures constitutes important insurance for mining companies and regulatory bodies.

Cyanide spill – hard-rock gold mining

Routine use of cyanide in a hard-rock gold processing plant has little downstream environmental significance because cyanide residues in tailings and supernatant water are typically at low levels (see section 6.2). However, concentrated cyanide compounds are transported to the mine site and regularly handled on-site. Hence, there is potential for accidental spillage of this material because of:
- Transporting vehicle accident
- Cyanide container leakage
- Cyanidation-plant pipeline failure (see section G.1.1)
- Failure of ‘cyanide kill’ system, leading to high-cyanide tailings (see section G.1.1)
- Tailings dam failure (see section G.1.2)

If this spillage involves discharge to the environment, an extreme ecological event is inevitable. Cyanide spills occur regularly around the world, with more than 30 significant incidents since 2000, including one in New Zealand (http://www.rainforestinfo.org.au/gold/spills.htm).

The most famous and most thoroughly documented recent example is the Baia Mare event (Romania) of 2000 (UNEP 2000).

Impacts on stream communities

Cyanide is highly toxic to most animals, so a cyanide spill into a stream is likely to kill all or most species present almost instantly. This will occur for several kilometres downstream until sidestream dilution lowers the cyanide concentration sufficiently for animal survival. The cyanide effect will persist in the stream until:
- clean water from upstream or rainfall dilutes the cyanide, and/or
- the cyanide decomposes in ultraviolet light (sunshine).

These mitigating effects can take days to weeks to occur. Ecosystem recovery after a cyanide spill can take weeks, months, or years, depending on the severity of the spill.

Mitigation options

The effects of a cyanide spill can be decreased by use of ‘neutralising’ chemicals that decompose the cyanide faster than UV light. Oxidising agents such as sodium hypochlorite are most commonly used for this purpose. These sorts of chemicals should already be on-site as part of routine cyanide management. Use of these chemicals is likely to be most relevant to an on-site spill or a local stream spill. There is potential for this ‘cure’ for a cyanide spill to be as bad or worse for an ecosystem than the cyanide itself.

It is important that a site-specific cyanide spill contingency plan be developed for the whole cyanide pathway, wherever cyanide is to be transported to a mine site and used on-site (ICME 1999; ICMI 2002, 2008). This plan should involve maintenance of adequate stockpiles of cyanide decomposition chemicals.

G.2 Natural events

For the most part no prevention or mitigation options are suggested for natural events, except for extreme high rainfall events. It is expected that obvious precautions, such as not locating tailings dams or treatment systems in earthquake-prone or landslide-prone areas, would be taken.
Extreme high rainfall events

Extreme high rainfall events are a natural phenomenon, and depending on their frequency, stream systems may be more or less adapted to them.

Impacts on stream communities

Extreme rain events causing flooding may have effects independent to mining, but can also interact with mining impacts to alter the severity of outcomes.

Natural effects on organisms

Stream communities are strongly influenced by natural flood disturbances. Floods have direct impacts on the diversity and density of organisms through removal of individuals, or indirect effects through influencing their habitat or food supply (e.g. removing algae and leaf litter). Consequently, stream communities often undergo a process of recolonisation after flood events. The severity of these events relates to the intensity of the flood, the frequency of floods, and the recovery time of fauna (driven by colonisation rates and the extent of changes to physical habitat). Thus, in frequently disturbed streams, communities may be in a constant state of recovery since the last disturbance event. High-flow events are relatively common in the headwaters of West Coast streams and often have a strong impact on invertebrate density. This physical template acts on top of mining activities, and can potentially confound the detection and quantification of AMD impacts.

Mining-related effects

Flood events may generate a pulse of low pH water with high metal concentrations through an increase in mine discharge. Acute toxicity during flood events may cause mortality that results in more severe impact outcomes than predicted by water quality under normal flow levels. These discrete events are often difficult to detect as water quality sampling is usually taken as spot measurements.

Alternatively, high-flow events may actually decrease mining impacts if increased stream volume dilutes an existing source of AMD. This should result in a temporary increase in pH and decreased concentrations of dissolved metals. However, the benefits arising from pulses of less impacted water is likely to be overwhelmed by the more severe, chronic toxicity under baseflow conditions.

Prevention of additional mine-related impacts

Tailings dams or treatment systems should be appropriately designed so that they can cope with significant rainfall events (World Meteorological Organization 1986).

Landslides

Landslides are most likely to occur on the West Coast, due to the mountainous terrain and steep bedrock underlying the topsoil. Landslides may occur as a result of an earthquake, a high rainfall event, or as a result of destabilisation by mining and/or related roadworks. Depending on where the landslide occurs, it may result in either the deposition of rocks and sediment directly into a stream, or high amounts of sediment being transported to the stream and resulting in highly turbid waters.

Impacts on stream communities

Where rocks and sediment are deposited directly into a stream there is an obvious catastrophic effect on the benthic stream community in that location through smothering. Downstream effects as a resulted of increased turbidity and sediment deposition will also be observed. This sedimentation can persist for some time if the exposed bank or debris continues to be eroded by stream flow. The effect of increased turbidity of the water will depend on its duration. The impacts of turbidity on stream communities occurs through the direct effects of physical abrasion, smothering and irritation (e.g. fish gills), and indirect effects of reduced food quality and quantity, and altered species interactions such as predation (see section 4.3 and Appendix D for more detail).

Prevention of additional mine-related impacts

Ensure that tailings dam or treatment systems are not located in landslide-prone areas. Ensure that appropriate stabilisation works are undertaken during mining operations.

Droughts

Droughts result in the reduction of stream flow, with extreme cases leading to the cessation of flow and the loss of surface water. Droughts cause effects independent from mining, as well as effects that interact with mining to increase the severity of outcomes.

Impacts on stream communities

Natural effects

The severity of drought impacts on stream communities depends on the extent of the reduction in stream flow. Reduced flow causes stress as a result of increased water temperature and reduced dissolved oxygen (DO), potentially impacting less tolerant taxa. In addition, the contraction of wetted habitats intensifies interactions between species, which may reduce the densities of prey species or poor competitors.
When surface flow ceases, isolated stagnant pools form refuges for fish and invertebrates. However, in the absence of flow, animals in refuge pools will experience considerable stress from reduced DO and increased temperature. Salmonid fish and EPT taxa are likely to be most vulnerable to these conditions.

In extreme cases, surface water may dry completely. The resulting effect on stream communities depends on the duration and harshness of drying. Many stream invertebrates and some native fish (non-migratory galaxiids) can burrow into the substrate to reach permanent water underground, and some can even survive for days to weeks in moist conditions under stones. Streams with permeable gravels are likely to have more of these refuges than streams with lots of bedrock or those that are choked with fine silt. Many West Coast headwater streams have large, armoured substrate as frequent floods remove smaller particles. Therefore, West Coast streams are likely to be limited in drought refugia. In contrast, many Southland streams contain loose gravels that provide good access to groundwater zones.

**Mining-related effects**

Droughts may reduce the dilution of AMD, especially if unimpacted tributaries normally dilute a constant mine discharge. Thus, during low-flow conditions, impacts may increase in severity, and be transmitted further downstream. In addition, stress from increased temperature and low DO may increase the vulnerability of organisms to low pH or elevated metals from AMD.

**Earthquake**

Earthquakes may result in landslides, and destruction of tailings dams and treatment systems (see Sections G.1.1 and G.1.2). Earthquake damage mitigation is a key part of design of all engineering works in earthquake-prone parts of New Zealand.

**Impacts on stream communities**

Where rocks and sediment are deposited directly into a stream there is an obvious catastrophic effect on stream community in that location. Downstream effects as a result of increased turbidity and sediment deposition will also be observed. The effect of increased turbidity of the water will depend on its duration.

**G.3 Consideration of extreme events during mine planning**

The occurrence and potential impact of extreme events should be assessed during the planning of mining operations to ensure appropriate design and operation to minimise potential negative impacts. Further, most of these issues are likely to be required to be considered during the bonding process. As part of the bonding process, a risk assessment of events leading to negative environmental outcomes during mining operations and after mine closure is generally required. This risk assessment can be completed through an appropriate forum of experts, with appropriate background data. A risk matrix can include information on the type of risk, consequences resulting from the realisation of that risk, likelihood of occurrence, and options for prevention and/or treatment.

Extreme events are by their nature unpredictable. Thus, only qualitative assessments, mostly based on professional opinion, of the likely consequences can be made. Similarly, often there are only limited data with which to estimate the likelihood of occurrence and this too will mostly be based on professional opinion. For some events, such as extreme high rainfall events or droughts, a reasonable estimate of the likelihood of occurrence could be obtained from historical rainfall data.

Examples of a risk assessment matrix outlining the likely extreme events relevant for coal mine on the West Coast, and alluvial and hard-rock gold mines in Southland are shown in Tables G1-G3. Additional less ‘extreme’ events that may result in environmental impacts, such as failure of revegetation or spillage of hazardous substances, may also be included in a full risk assessment used for bonding purposes. Bonds will also need to be sufficient to ensure a mine site is appropriately restored and rehabilitated should the agreed remedial measures not be undertaken.
**Table G1:** Risk matrix of ‘typical’ extreme events associated with a coal (NAF or PAF) mine on the West Coast.

<table>
<thead>
<tr>
<th>Risk</th>
<th>Likely consequences</th>
<th>Annual frequency (1 in x years)</th>
<th>Annual probability</th>
<th>Prevention, treatment, or mitigation measures</th>
<th>Most likely treatment measure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Localised Instability</td>
<td>Slumping of material into stream</td>
<td>5</td>
<td>0.20</td>
<td>Move access road</td>
<td>Excavators and dump trucks to remove sediment</td>
</tr>
<tr>
<td>Global Instability – mine wall, overburden stack</td>
<td>Slumping of material into stream, mobilisation of sediment</td>
<td>100</td>
<td>0.01</td>
<td>Re-establish access road and stabilise rocky bluffs. Recontouring of wall/stack.</td>
<td>Excavators and dump trucks to remove sediment</td>
</tr>
<tr>
<td>Instability (from earthquake or storm event)</td>
<td>Slumping of material into stream, mobilisation of sediment</td>
<td>1000</td>
<td>0.001</td>
<td>Move access road.</td>
<td>Excavators and dump trucks to remove sediment</td>
</tr>
<tr>
<td>Reduction in water quality of mine drainage (AMD) – treatment system failure</td>
<td>Unacceptable offsite sediment and chemistry discharge</td>
<td>100</td>
<td>0.01</td>
<td>Repair or reinstate treatment system</td>
<td>Repair or reinstate treatment system.</td>
</tr>
<tr>
<td>Reduction in water quality (chemistry and TSS) - failure of sediment retention works during storm or instability event (ponds)</td>
<td>Unacceptable offsite sediment and chemistry discharge</td>
<td>100</td>
<td>0.01</td>
<td>Reinstall ponds and drainage.</td>
<td>Reinstall ponds and drainage.</td>
</tr>
</tbody>
</table>

1. Actual items will be specific for individual operations, but examples are included to provide an indication of how different events may impact and the relative likelihood of occurrence.

**Table G2:** Risk matrix of ‘typical’ extreme events associated with an alluvial gold mine in Southland.

<table>
<thead>
<tr>
<th>Risk</th>
<th>Likely consequences</th>
<th>Annual frequency (1 in x years)</th>
<th>Annual probability</th>
<th>Prevention, treatment, or mitigation measures</th>
<th>Most likely treatment measure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tailings dam failure during a 100-year flood event. Water overtops dam and erodes the structure causing a breach which then discharges tailings</td>
<td>Discharge of arsenic-bearing water and sediment into downstream catchments.</td>
<td>100</td>
<td>0.01</td>
<td>Remove sediment from accessible reaches downstream, and redeposit it in the repaired tailings dam. Water and remaining sediment will be flushed over 10-50 years.</td>
<td>No action for minor discharge of tailings.</td>
</tr>
<tr>
<td>Large cyanide spill from processing plant during a major rainstorm is washed off site</td>
<td>Cyanide kill of most aquatic species several kilometres downstream until dilution reduces the toxicity.</td>
<td>50</td>
<td>0.02</td>
<td>Immediate addition of cyanide-neutralising agent, stored on-site and accessible. Warn nearby population of risks of toxicity.</td>
<td>1000 litres of neutralising agent. Regular monitoring of downstream water quality between mine site and the sea until cyanide is no longer detected.</td>
</tr>
</tbody>
</table>
### Table G3: Risk matrix of ‘typical’ extreme events associated with a hard-rock gold mine in Southland.

<table>
<thead>
<tr>
<th>Risk</th>
<th>Likely consequences</th>
<th>Annual frequency (1 in x years)</th>
<th>Annual probability</th>
<th>Prevention, treatment, or mitigation measures</th>
<th>Most likely treatment measure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waste rock pile collapse during a major rain event</td>
<td>Debris enters nearest creek, and sediments mobilised downstream catastrophically.</td>
<td>50</td>
<td>0.02</td>
<td>Remove sediment build-up from accessible reaches onto nearby banks or other disposal site.</td>
<td>Excavators and dump trucks to remove sediment</td>
</tr>
<tr>
<td>Earthquake greater than Magnitude 7</td>
<td>Instability and possible breaching of tailings dam; landslides from valley walls into tailings dam; debris enters nearest creek, and sediments mobilised downstream catastrophically.</td>
<td>200</td>
<td>0.005</td>
<td>Local and regional infrastructure damage will preclude immediate action on environmental issues such as arsenic discharge. Human life and access will be top priority.</td>
<td>Excavators and dump trucks to remove sediment</td>
</tr>
<tr>
<td>Major rainfall event causes overtopping of tailings dam (silt pond)</td>
<td>Discharge of suspended sediment into downstream catchments</td>
<td>100</td>
<td>0.01</td>
<td>Remove deposited sediment from accessible reaches downstream to clear waterway.</td>
<td>No action for minor discharge of tailings</td>
</tr>
<tr>
<td>Failure of mine water containment structure during drought</td>
<td>Discharge of acid water with elevated Al, Fe, Ni, As</td>
<td>50</td>
<td>0.02</td>
<td>Flush immediate streams with water from a non-acid source.</td>
<td>Water tanker loads</td>
</tr>
</tbody>
</table>
G.4 References


