NEW ZEALAND MINERALS SECTOR ENVIRONMENTAL FRAMEWORK

A User's Guide

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January 2015

ACKNOWLEDGEMENTS

The Minerals Sector Environmental Framework extends *A framework for predicting and managing water quality impacts of mining on stream ecosystems: a user's guide* previously developed by Landcare Research [as Contract Report LC0910/114], CRL Energy [CRL Report 10-41100] and the Universities of Canterbury and Otago) in conjunction with end users including the Department of Conservation, the West Coast Regional Council, Environment Southland, Solid Energy, OceanaGold, Pike River Coal, Francis Mining and a number of consultants. Contributors to the previous framework also included Rachel Rait, Hamish Greig, Dev Niyogi, Rowan Buxton, Olivier Champeau, Tony Clemens and Amanda Black.

The Minerals Sector Environmental Framework [Landcare Research Contract Report LC2033] has been developed with input from end users including the Department of Conservation (DOC), the West Coast Regional Council (WCRC), Environment Southland, Solid Energy, OceanaGold, Francis Mining, Bathurst Resources, Newmont, Waikato Regional Council and the Tui Mine Iwi Advisory Group. Specific thanks go to Rob Harrison (DOC) and Jackie Adams (WCRC) for their input. Liz Law and Christine Bezar (Landcare Research are thanked for their editorial and formatting assistance.

Funding was provided by the Science and Innovation Group, Ministry of Business, Innovation and Employment.

EXECUTIVE SUMMARY

Coal and gold mining are important economic activities in New Zealand, and the West Coast of the South Island and Southland, for example, have long histories of mining. The process of mineral extraction inevitably affects the surrounding environment, but few tools exist to help mining companies and regulators assess and predict the environmental impacts of mining operations. This framework has been developed as part of a collaborative research programme with key mining partners to assist with planning of future mine developments on the West Coast and in Southland.

This framework focuses on water quality issues associated with coal and gold mining, specifically pH, acidity, metals and, suspended solids; and rehabilitation of mined areas. It draws together research on rock geochemistry, aquatic chemistry, freshwater ecology, aquatic toxicity, and management, treatment and rehabilitation techniques for mining to provide a process for data collection and decision making. The main body of the document outlines the process, including the data required and methods for collecting and interpreting those data. A series of appendices provide the more-technical and scientific results that underpin the processes and decision trees used in the document. Specifically the framework provides information on collection of the water, rock and biological information used to (a) predict water quality prior to mining, (b) monitor water discharges from mines, (c) identify mining-related impacts and d) undertake rehabilitation of mined areas. In addition, the framework includes information on state of the art techniques for *prevention* of poor water quality in mine drainages and optimal strategies for *management* of mine waste or overburden and *treatment* of mine drainages, if necessary. The appendices also include discussion on the impact of extreme events on mining operations.

The framework is written for a wide audience (e.g. regulators, mining companies, landholders, and the community) and should assist with regulatory processes, such as access arrangements with the Department of Conservation, assessment of environmental effects (AEE) for resource consenting, and the setting of resource consent conditions. The document should also aid internal decision-making by mining companies.

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1 INTRODUCTION

The New Zealand Minerals Sector currently contributes ~1.2% of GDP (Statistics New Zealand 2007) (~$2.0B), employs ~14,000 people (MBIE 2013) and generates ~$90M in royalties (Statistics New Zealand 2013) (excl. petroleum). New Zealand has a long history of mining, starting with the extraction of pounamu by Maori followed by European settlers extracting coal and gold in the 1860s. Most residential, commercial and industrial products use minerals during their production and thus mining will remain an important industry into the future. However, the removal of minerals from the ground may result in significant negative impacts on the environment. Increasingly, mining companies are being required to reduce and minimise these impacts.

The West Coast of the South Island and Southland are key mining areas, with potential for increased mining activity.

This document describes a framework to assist with planning of future mine developments in New Zealand. The framework results from a collaborative research programme between CRL Energy, the University of Canterbury, University of Otago and Landcare Research, and it has been developed in conjunction with end users including the Department of Conservation, the West Coast Regional Council, Environment Southland, Solid Energy, Oceana Gold, Pike River Coal, Francis Mining, Bathurst Resources, Newmont, Waikato Regional Council and consultants. It is intended to provide consistency and transparency in decision making for proposed mining operations, specifically around preventing or minimising impacts on streams. The framework is intended for use in internal decision-making by mining companies and to assist with certain regulatory requirements, such as access arrangements with the Department of Conservation, and in the resource consenting process, such as during consultation, assessment of environmental effects (AEE), and the setting of resource consent conditions. The framework may also be useful in developing future regional plans for water quality.

This framework focuses on water quality issues associated with coal and gold mining, specifically pH, acidity, metals and turbidity, and rehabilitation of mined areas. It does not address other water quality measures such as salinity, temperature, or environmental issues that also may need to be considered during mine planning and consenting such as stream diversions, water quantity, noise, traffic, visual, dust, and subsidence issues.

Commencing with a general introduction to mining in New Zealand and existing water quality and terrestrial impacts associated with mining, this document outlines a process for predicting and minimising environmental impacts of mining. The framework draws together research on rock geochemistry, aquatic chemistry, freshwater ecology, aquatic toxicity, and management, treatment and rehabilitation techniques for mining, and provides guidance on:

- Interpretation of regional trends in rock geochemistry and prediction of mine drainage chemistry
- Sampling strategies for rock and streams, and analysis methods for mine drainage prediction and management
- Effects-based ecological impact thresholds
- Sampling strategies and methods for water quality and biological impact assessment and monitoring
- Management strategies for waste rock
- Optimal selection of active and passive treatment systems
- Identification of resources for rehabilitation and rehabilitation techniques

1.1 Mining in New Zealand[1]

Coal and gold mining have a rich history in New Zealand, with commercial mining commencing around the mid-1860s. Today, the main coal mining areas are the West Coast and Southland in the South Island and the Waikato Region in the North Island. The three largest gold mines are in Otago, Coromandel and the West Coast Region, whereas alluvial gold mining and exploration occur mainly in Otago, West Coast and Southland. The scale and number of mining operations have expanded and declined several times in response to commodity price changes. Currently the global minerals sector is declining after an unprecedented run of increasing commodity prices throughout the early part of the 21st century. Mining of deposits that have marginal profitability can stop and restart depending upon commodity prices. Up until the 1970s mining was conducted with little regulation of the impact on downstream water quality or ecosystems. Many historical mine workings occur throughout New Zealand, particularly in the West Coast Region, with some historical mining workings causing significant impacts on downstream water quality.

1.1.1 Coal mining

On the West Coast mining occurs in 13 coalfields of various sizes mostly between Greymouth and Seddonville (c. 40 km north of Westport). The most productive of these are the Buller, Greymouth, Reefton and Garvey Creek coalfields. Of the 983 million tonnes (megatonnes; Mt) of in-ground coal in the region, over three-quarters of recoverable reserves are in the Greymouth (mostly underground) and Buller (mostly opencast) coalfields. Rocks mined for coal belong primarily to either the Brunner or Paparoa sedimentary sequences and these sequences are found in most of the coalfields. Both Paparoa and Brunner sedimentary sequences contain coal seams up to 20m thick.

This coal is primarily bituminous and as a result almost all of the coal mined on the West Coast is exported for steel production.

1 This information was primarily sourced from Te Ara - the Encyclopedia of New Zealand, updated 4 Dec 2008: Sherwood and Phillips, 'Coal and coal mining', http://www.TeAra.govt.nz/EarthSeaAndSky/MineralResources/CoalAndCoalMining/en; and Carl Walrond, 'Gold and gold mining', http://www.TeAra.govt.nz/EarthSeaAndSky/MineralResources/GoldAndGoldMining

Currently 1.5–2 Mt are mined on the West Coast each year, which accounts for about half of New Zealand's annual coal production.

In Southland, coal and lignite are currently mined from the Ohai coalfield and the Eastern Southland lignite deposits. The Ohai coalfield has relatively small recoverable resources of about 50 Mt. In contrast, the eastern Southland lignite fields are New Zealand's biggest fossil fuel energy resource and contain over six billion tonnes (gigatonnes, Gt) of lignite, much of which could be opencast mined.

Current annual production from Southland is about 300,000 t and this coal and lignite is mostly used for domestic industry.

However, there is potential for lignite mining in Southland to grow by more than an order of magnitude if economic conditions and demand are favourable.

In the Waikato Region Sub-bituminous coal is mined for use steel production (Glenbrook) power production (Huntly) and agriculture (e.g. dairy, meat works, cement manufacture). About 1.5Mt is mined currently from underground and open cast mines with production likely to decrease in future.

The predominant coal mining regions in New Zealand are shown in Figure 1.

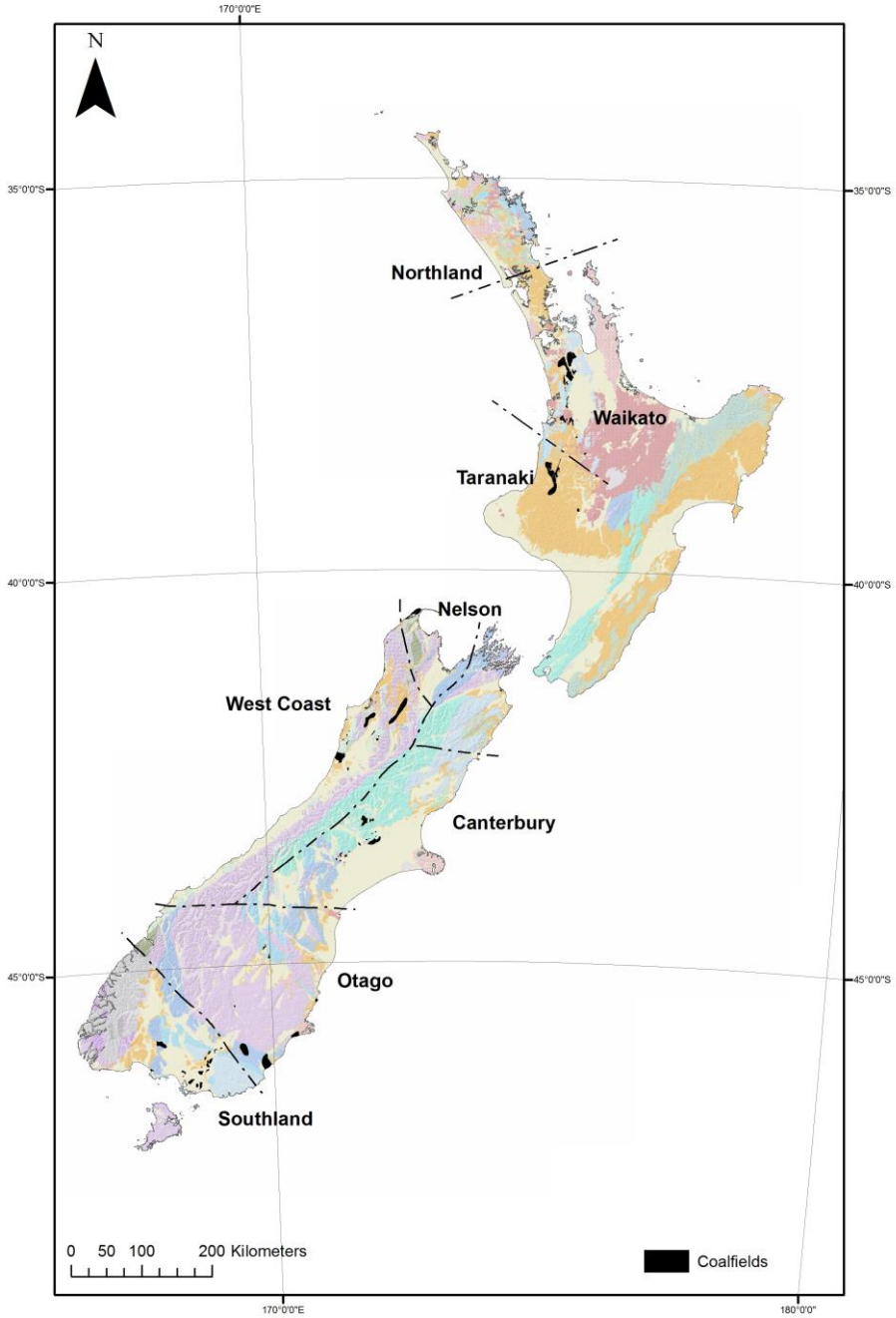


Figure 1: Map of coal resources in New Zealand

1.1.2 Gold mining

South Island

Mesothermal or orogenic gold deposits are confined to the South Island. They occur in two main belts of metamorphosed sedimentary rocks that have been offset by the Alpine Fault. The western belt includes a range of geologically old (400 million years) metamorphosed sedimentary rocks in NW Nelson, Buller, Westland, and SW Fiordland. The eastern belt includes the Otago Schist and its equivalent metamorphosed sedimentary rocks in Marlborough. Historical gold production from these hard-rock gold deposits was generally small, although some mines near Reefton were important producers at times. Most historical gold mining production was from placer gold deposits of varying ages that were derived by erosion and river transport of the gold from the mesothermal deposits. This river transport of gold was on a scale of tens to hundreds of kilometres, so some placer deposits are now far removed from their hard-rock sources. Placer mining has continued sporadically for the whole 150 years of gold mining history, but historical hard-rock mining largely ceased in the 1950s.

A resurgence of hard-rock mining occurred in the 1990s with the reopening of the Macraes mine in Otago Schist. Historical hard-rock production at that site was trivial, but the modern mine developed into the largest gold producer in New Zealand, and is a world-class deposit. The large capital investment in the processing plant at Macraes facilitated the reopening of the Globe-Progress mine at Reefton, in the western sedimentary belt, and ore concentrate was trucked and railed in containers 700 km from Reefton to Macraes for final gold extraction. Production from these mines has surpassed historical placer gold production. The recent history of Macraes and Reefton mines has helped to spark ongoing exploration in both the eastern and western sedimentary belts.

The Sams Creek hard-rock gold prospect in NW Nelson is notable because there was no historical production, and the deposit was only discovered in the 1980s, opening up an entirely new prospecting area. Sams Creek prospect is also notable because, although it occurs in the same broad western sedimentary belt as Reefton mines, it is found entirely within a small granite intrusion into those sediments. Hence, this is a new style of hard-rock deposit. Prospecting has occurred in several phases since first discovery, and is ongoing.

Mining at Reefton recommenced in 2007 with the opening of a major opencast gold mine, Globe-Progress, which is currently nearing the end of production.

North Island

North Island gold deposits are mostly hosted in Coromandel area within Miocene to Pliocene volcanic sequences of the Coromandel and Whitianga Groups. These rocks are typically felsic to intermediate volcanic rock sequences, commonly rhyolites or andesites. These deposits are epithermal veins and stock-work systems with classic zoned alteration and mineralised systems.

Mining of gold in the Coromandel area commenced in the late 1800s and continues presently at large opencast and underground mines. Important mineral deposits that have been mined in the last 50 years include the Tui Deposit, Golden Cross, Martha and related underground deposits. Tui is an andesite-hosted polymetallic epithermal style deposit mined mostly through underground methods near Te Aroha, south of the Coromandel Peninsula, until the 1970s. Golden Cross is an andesite-hosted deposit mined by both opencast and underground methods that ceased operations in 1998. Golden Cross is west of Waihi at the south of the Coromandel Peninsula. The Martha deposit is a large quartz vein system that occurs in andesitic rocks currently mined by opencast methods. The Martha deposit has several satellite vein systems (Favona, Corenzo etc.) that are currently mined by underground methods.

Other epithermal style mineralisation occurs in Northland and the Taupo area. However, no active mining currently takes place in these areas. The host rocks in Northland are the Pliocene Pura Beds and the deposit style is shallow epithermal or sinter deposits. In the Taupo area rare vein mineralisation is hosted by ignimbrites. Northland and Taupo epithermal deposits are currently being explored.

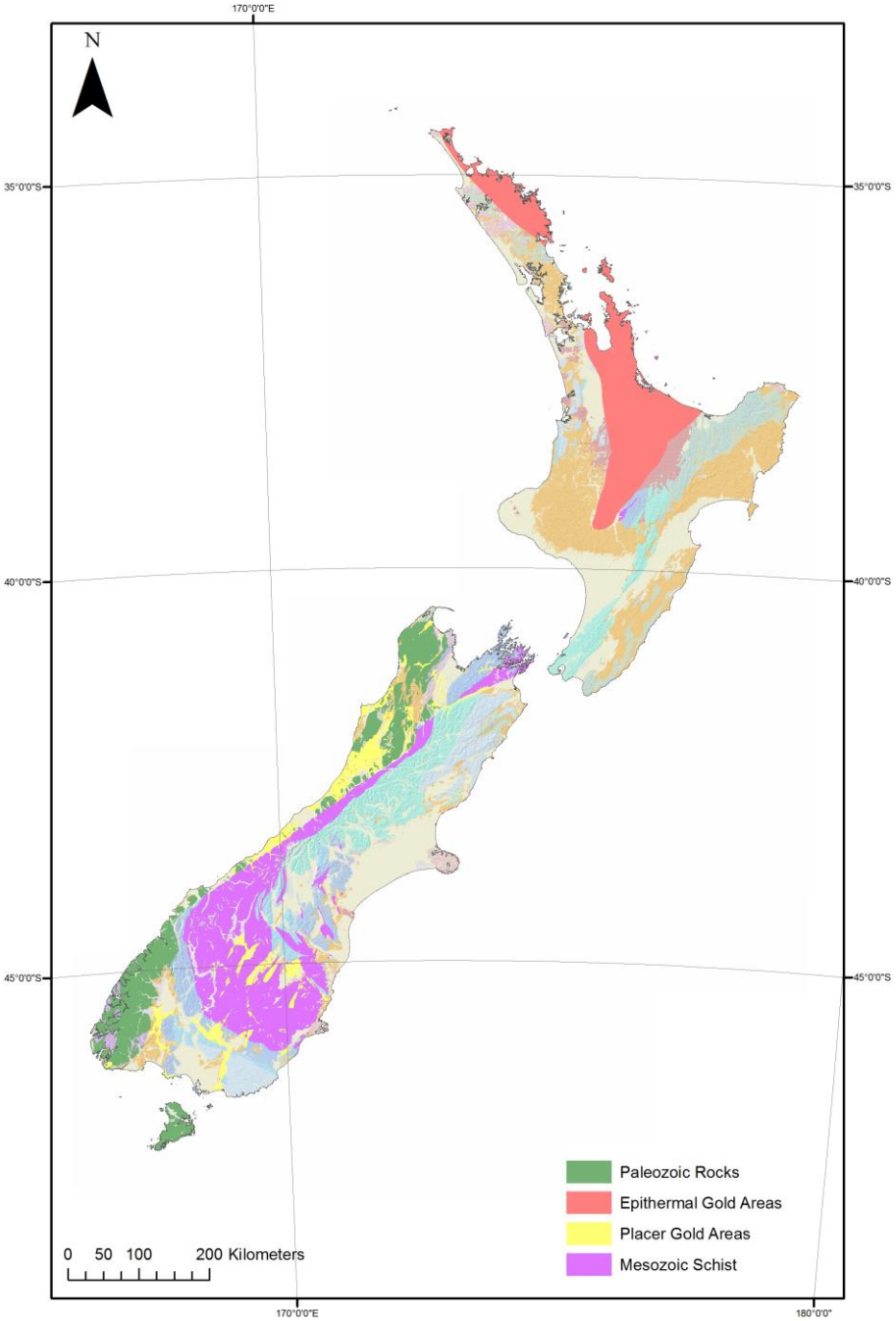


Figure 2: Location of gold provinces in New Zealand

1.1.3 Other mining

New Zealand produces Iron Sand from Port Waikato and Taharoa deposits in the Waikato Region on the West Coast of the North Island. From 2002 to 2012 production ranged between 1.7 and 2.4Mt of concentrate and current operators are Bluescope. There are several other companies actively exploring for similar resources both onshore and offshore on the West Coast of New Zealand.

Other active extractive industry in New Zealand include quarries for aggregates, building materials, industrial minerals and limestone.

New Zealand, has potential to host a variety of other mineral deposits, and by world standards is poorly explored for a politically stable first world country. Other commodities that have recognised potential in New Zealand mineral deposits include, platinum group elements, base metals (Cu, Pb, Zn, Ni), rare earth elements and coal seam gas. In addition, energy extraction trials by underground coal gasification have been conducted on Waikato coal.

1.2 Potential environmental effects of mining

Mining can have diverse environmental effects. These can be categorised according to whether the effect is restricted to the mine site or occurs at some distance from the mining activity. For example, environmental effects that are restricted to the mine site include those arising from clearing of vegetation and habitat modification, modification of soil profiles, changes to topography and slope, or subsidence. Environmental effects that occur away from the mine site include increased noise, dust deposition, and impacts on downstream water quality and ecosystems.

Current expectations of mining operations are that environmental effects, whether on or off site, will be mitigated or minimised (e.g. DITR 2007; Environment Canada 2009). With some exceptions, such as subsidence or mine fires, environmental effects that occur on the mine site are relatively predictable and more easily managed than those that affect areas away from the mine.

Effects on downstream aquatic systems arising from mine drainage are among the most difficult mining-related environmental impacts to predict, mitigate, manage or remediate. Often these effects can be severe. The presence or absence, severity and extent of chemical impacts on downstream ecosystems depend on a complex variety of local, regional, natural and anthropogenic factors. Timely and effective terrestrial rehabilitation can prevent post-mining downstream aquatic impacts. Opencast mining removes terrestrial ecosystems to access ore, and its effects are generally easier to understand and predict than aquatic impacts. However, the range of outcomes and rate of recovery that can be achieved are highly variable, depending on whether resources can be salvaged and reused and what techniques are used. This framework focuses on the prediction of water chemistry downstream of mine operations, the potential ecological impacts caused by mine drainage, the management and mitigation of mine drainages, and the optimal rehabilitation techniques at New Zealand mine sites.

1.2.1 Mine drainage chemistry

New Zealand's geology, climate and topography vary considerably so both natural and mine drainages have diverse water quality. Groundwater, surface water runoff, and mine process water at a mine site (collectively 'mine drainage') all have potential to chemically interact with mineralised rocks. Mining results in increases in reactive surface area of rocks, water ingress and oxygen availability to previously reduced mineral suites and so mine drainage waters can develop variable and distinctly different compositions to natural background waters.

Coal mines may produce mine drainage ranging from neutral mine drainage (NMD) with neutral pH and low dissolved metal concentrations to acid mine drainage (AMD) with acidic pH values and elevated concentrations of dissolved iron (Fe), aluminium (Al) and trace elements. Acidic drainages may also occur in non-mined areas; these are typically referred to as acid rock drainage (ARD) and have similar characteristics to AMD though often with concentrations of dissolved components. Acid mine drainage can be the most significant chemical water quality impact from mining. It commonly has low pH (2-3), elevated Lewis acids (Fe and Al up to ~1000 mg/L), extremely elevated trace element concentrations, low alkalinity, and increased chemical oxygen demand. Trace elements that can have elevated concentrations in coal mine drainage include zinc (Zn), nickel (Ni) and manganese (Mn). At low pH, these metals are very soluble in water and are transported downstream in dissolved form. These metals become less soluble with increasing pH, and can precipitate or adsorb to substrates at various pH thresholds. The precipitates are commonly Fe and Al oxyhydroxides, or occasionally more complex chemical compounds. The presence of these precipitates is often the most distinguishing characteristic of AMD and the bright yellow-orange colour of streambeds, due to Fe oxyhydroxides, historically led people to call it 'yellow-boy' (Figure 3).

Figure 3: Unimpacted tributary (left) mixes with AMD-contaminated water, Cascade Creek (Denniston)

Mesothermal gold mines and rocks that host gold mineralisation in New Zealand can produce neutral drainage enriched with trace elements, commonly arsenic (As) or, less frequently, antimony (Sb) or AMD with a suite of enriched trace elements. Rocks that host gold mines cause regional elevation of the trace elements that are naturally rich in those rocks, including As. There are processes that naturally attenuate dissolved As transport and the natural enrichment often masks impacts from historical mines. However, at active gold mines, discharge water is often treated to remove As or Sb.

Epithermal gold mines and rocks that host epithermal mineralisation produce acidic or neutral drainages that are variably enriched in a broad suite of trace elements, including Cu, Zn, Pb, Mn, Hg, and others. Further, mine drainage chemistry may change with time so that neutral mine drainages become acidic as reactions proceed within the rock mass that is impacted by mining. At historical mines, the impact of untreated drainages in downstream environments is variable depending on the chemistry and volume of discharge and amount of dilution. Active gold mines treat site water to prevent discharges that would cause unacceptable downstream impacts.

1.2.2 Biological effects

In this framework assessment of the response of biological communities to mining has focused on stream ecosystems. We have few data on how lake and standing water communities respond.

Stream ecosystems are home to large and often distinct communities of plants and animals. Most people probably think mainly of fish, but almost all streams also have large communities of invertebrates, algae, and microbes. These organisms interact to form food webs (Figure 4), which support higher trophic levels, including fish and birds. As each species is reliant on its food resource, disruptions to feeding pathways in food webs caused by mining activities can lead to reductions in numbers of the higher animals.

The effects of mine drainage on stream life can be direct or indirect. Direct effects include toxicity associated with low pH or high metal concentrations and may be acute (lethal) or sub-lethal (e.g. affect reproductive systems). Indirect effects can occur if the mine drainage affects the food supply (e.g. invertebrates for a fish) or habitat of stream organisms (Figure 4).

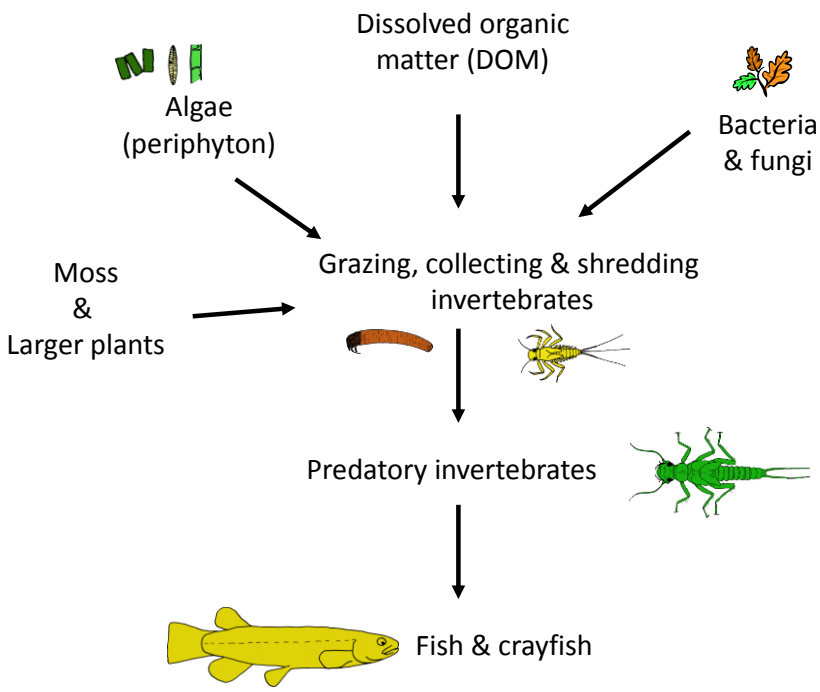


Figure 4: **Simplified food web of a stream ecosystem. Effects of mine drainage on fish can be direct, such as toxicity from low pH, or indirect, such as effects on algae, which affects invertebrates and fish.**

Biological effects from gold and coal mining may arise from high sediment loads or low pH and high metal concentrations. For example, sediment from active mining areas (including alluvial works), mine tailings, or mining roads can lead to high turbidity and fine sediment in streams. Coal fines and fine sediments may also make their way into streams from active mines and cause high turbidity. High turbidity can have both direct impacts, such as smothering of benthic organisms or deterring the migratory stages of native fish, and indirect impacts, such as reduced algal growth due to reduced light penetration.

However, of all these effects, AMD has the most potentially severe impacts on life in streams. Acidic water can be stressful or even lethal to some organisms, and high concentrations of dissolved metals can also be toxic to aquatic biota. In addition,

the metal precipitates associated with AMD (Fe and Al oxyhydroxides), while generally not toxic to aquatic biota, alter the streambed. These precipitates can coat the streambed and clog the areas around rocks where animals live, leading to poor habitat for all aquatic life.

Aquatic biota (algae, invertebrates and fish) are useful for monitoring the influence of mining and subsequent treatment on stream ecosystems because they reflect a history of water quality across an organism's life. Macroinvertebrates are the most commonly used biological monitoring tools in New Zealand, as they are:

- Easy to sample and identify
- Long lived, and thus reflect water quality changes over time
- Have variable tolerance to stressors such as low pH and metals.

Fish and algae are also promising biological monitoring agents, but each has difficulties. Algae are difficult to identify and ecological knowledge on their responses to AMD is just being developed. Fish are charismatic components of food webs, but they are highly mobile and many New Zealand species have migratory stages. Consequently, the presence or absence of a fish species in a reach may not be indicative of current or past water quality. Under certain circumstance riverine birds may also be useful indicators, although no studies have been done on these on New Zealand mine sites.

This document focuses on macroinvertebrates as biomonitoring tools, both in-stream and also for toxicity testing: for example to assess the toxicity of treated or untreated mine drainage. The predicted impacts of mine pollutants on fish and algae are also considered. In the foreseeable future, other useful biological indicators of stream health might become available, and these could be used to complement macroinvertebrate data.

As multiple mine inputs can occur along a river system, AMD impacts can be cumulative. These cumulative impacts may result in mild AMD impacts higher up a river system and severe impacts downstream; however, if significant point-source AMD inputs occur in the upper reaches of a catchment then significant impacts can remain along much of the river continuum. This is particularly common on the West Coast where many short streams and rivers are impacted along their entire length.

Lakes and standing water

Mining operations often create standing water bodies ranging from small ponds to large lakes. These form in a variety of structures such as remnant pits, washing stations and water treatment structures. The structure, age and chemistry of these systems can be quite variable and little is known about the biotic communities in them in New Zealand. A preliminary survey undertaken in 2014 surveyed water chemistry, habitat characteristics, and the littoral and planktonic macroinvertebrate communities in lakes spanning gradients of geographic location, mining type and age across New Zealand (Mulet et al 2014).

This study found that Al, Fe, Zn and Cd usually exceeded ANZECC water quality guideline values, especially in young lakes located in artificial (mining) sites located in indigenous forest areas, although the community structure of these and other newly created mining lakes are also affected by additional stress factors related to acid generation and metals dissolution, and poor physical habitat**.** Climate, geomorphology, local geology, vegetation type and surrounding land-cover are important factors influencing water chemistry and aquatic communities in mining lakes. Catchment land-cover (especially agriculture) can overlay the effect of mining in long-term water quality and community structure due to nutrient and some trace metal inputs, highlighting factors for consideration in the rehabilitation of mine sites.

1.2.3 Reducing impacts – management and treatment

Impacts on stream ecosystems from coal and gold mining can be mitigated through management of mining operations, particularly management of mine waste (tailings, mine water and waste rocks), through water treatment techniques, or a combination of both. In general, best management practices to prevent or reduce the formation of AMD and high total suspended solids (TSS) will be more cost-effective than ongoing treatment of AMD discharge. In particular, mine waste management techniques are critical to minimising AMD. However, in many situations mine waste management will be insufficient to mitigate the impacts of such drainage on receiving systems, and additional treatment may be required.

Operational management

Operational management can be a cost-effective means of minimising mining impacts on adjacent streams, and is the preferred first stage of any environmental management programme. Operational management to reduce mine drainage impacts focuses on preventing or reducing the amount of water entering the mined area, reducing the contact of water and/or oxygen with acid-forming materials, and neutralising or reducing the level of contaminants present in any mine drainage. Methods to achieve these goals involve evaluating the factors that influence mine drainage at each site and applying appropriate site-specific management options to reduce the amount of impacted mine drainage (Nieman & Merkin 2000; Osterkamp & Joseph 2000; Terrence & Black 2000). Several factors – including local topography, climate, mine waste composition and physical properties, and groundwater conditions –can influence the effectiveness of each mine waste management technique. Therefore the combination of mine waste strategies selected for a particular site may be unique to that site. It is also likely that in the life of a mine no single strategy for management will suffice, and ongoing monitoring of mine waste and water quality is required to ensure that appropriate management strategies are being adopted, and that monitoring and performance data enable strategies to be adapted if conditions change.

Key variables that need to be evaluated during selection and evaluation processes for mine waste and water management strategies at each site include background water quality; the volume, geochemical composition and potential toxicity of mine waste material; and the position of the mine waste relative to surface water and groundwater.

Treatment

Treatment of mine drainage may still be required even with good mine waste management practices. Treatment can be accomplished by either active or passive treatment systems, or a combination of both.

Active systems typically require continuous dosing with chemicals (such as lime); they consume power and require regular operation and maintenance, but are very reliable. Their main advantages are that they are very effective at removing acid and metals from mine drainage, particularly from AMD, and can be designed and operated to produce specific water chemistries. Further, they can be accommodated in locations where only a small land area is available. The main disadvantages of active treatment are the high capital cost and high ongoing operational and maintenance costs. Active systems are more suited to active mine sites, which may have limited land area available for treatment systems, a changing drainage chemistry and flow rate, a power supply, and personnel to manage the system.

Passive systems rely on natural physical, geochemical and biological processes, but can fail if not carefully selected and designed. Passive systems have limitations with respect to treating high flow and high acidity drainages. Mine drainage must have long enough residence times in these systems to allow these processes to occur, which means that these systems typically require large areas of land. For example, for AMD, most passive treatment systems rely on the dissolution of a neutralising material (usually limestone) to neutralise the acidity in AMD, and sufficient residence time in the system for this dissolution to occur (Skousen et al. 2000). In the long term however, treatment using passive systems is typically more economic than using active systems, especially after mine closure (Skousen & Ziemkiewicz 2005). At closed and abandoned mines AMD typically has a more stable chemistry and flow rate than AMD at active mine sites and land is usually more readily available for treatment systems – factors that fit well with passive treatment.

For drainage from coal and gold mines in non-acid-forming regions, suspended solids are the main focus for treatment. For coal and gold mines in acid-forming regions, treatment options are driven by pH, acidity and metal loads of the mine drainage. For drainage from alluvial gold mines, suspended solids are the main focus for treatment, although AMD may occasionally occur. In this case treatment for low pH and elevated soluble metal loads is required. Hard-rock gold mines in the current study typically have neutral discharge, which commonly has elevated As concentrations thus requiring treatment for arsenic removal.

1.2.4 Rehabilitation

Effective rehabilitation is required to minimise medium- and long-term adverse effects associated with mining disturbance. Rehabilitation begins with identifying the resources available for rehabilitation and understanding the ecosystems that are impacted by a mining project. The available resources (soils, plants, beneficial overburdens and rocks, fauna) influence and potentially dictate the range of outcomes that can be achieved. The range of possible outcomes forms the envelope within which specific post-mining rehabilitation landforms, ecosystems and success criteria can be agreed. However, mine rehabilitation plans need to be flexible. It is rare to have full knowledge of the geology of resource or overburden characteristics before mining, and the cut-off grade of the resource or the resource-to-overburden ratio depends on resource value and input costs (diesel, labour) that may rapidly change. Consequently, mine footprints may extend or contract. In larger, longer-term developments it is not unusual for personnel, contractors, and companies to change. Written records of rehabilitation principles, priorities, and success criteria are therefore important. Adaptive management gives miners the flexibility they need to achieve agreed outcomes in the most cost-effective manner.

Rehabilitation requires landforms that are safe and geotechnically stable. Landform design and development should ensure root zones that support the post-mining land use(s) are created and maintained. This includes the development of tarns, ponds and lakes, which, as noted earlier needs to consider restoration of the physical habitat as well as ensuring enhancement and maintenance of a good water quality. Finally, the establishment of vegetation and change in plant species and composition over time (succession) determine the post-mining landscape and use(s).
This framework outlines a process and information required for effective planning and implementation of mine rehabilitation to selected outcomes: farming (pasture), plantation forestry and native ecosystems.

1.3 Development of a framework to minimise environmental impacts from mining

This framework draws on fundamental research undertaken by the research team, and has been developed in conjunction with end users including the Department of Conservation (DOC), West Coast Regional Council, Environment Southland, Waikato Regional Council, Solid Energy, OceanaGold, Bathurst Resources, Newmont, Francis Mining and consultants. It is intended to be used for internal decision-making by mining companies and to assist with regulatory requirements.

This draft framework extends a previous framework developed to prevent and minimise aquatic impacts from mining (Cavanagh et al. 2010) by including a focus on mine rehabilitation and making the framework nationally relevant. However, not all geologies, and their associated environmental impacts, have been investigated to the same extent. Specifically, the most comprehensive research has been undertaken in acid-generating coal mining systems, and

reasonable amounts of research in non-acid coal mining areas, alluvial gold systems, and mesothermal gold regions. Limited research has been conducted in epithermal gold regions. Further, it is recognised that significant knowledge gaps remain for all geologies to enable effective management of mining operations to minimise environmental impacts. Future research will address some of these including:

- Geochemistry of mine tailings
- Long-term predictions and uncertainty analysis
- Trace element treatment processes
- Overburden management
- Extending of treatment systems, in particular adapting new techniques to New Zealand conditions
- Recovery of impacted stream systems
- Determining appropriate closure and post-closure rehabilitation endpoints
- Links between mine operations, economics and good environmental outcomes, including ecosystem services analysis

1.3.1 Regulation of the mining industry

Mining is a regulated industry primarily governed by requirements under the Crown Minerals Act 1991 and Resource Management Act 1991 (RMA). Three types of regulatory requirements need to be met prior to mining operations proceeding[2]:

- A permit or licence granted under the Crown Minerals Act[3]
- An access arrangement negotiated with all landowners and occupiers; this may include individuals or government departments such as DOC[4]
- Resource consents (e.g. use of land and water, discharges to water, air) (district and regional councils).

Permits or licences granted under the Crown Minerals Act are mostly related to the economic aspects of mining, whereas granting of access (in particular that granted by DOC) and resource consents are often concerned with the potential for environmental impacts from proposed operation. The framework is intended to provide a guide to information requirements and interpretation of this information for access arrangements and resource consents. This includes background information that is useful in undertaking assessment of environmental effects (AEEs) required for the development of resources under the RMA or for access arrangements with DOC. The framework is directly applicable to aquatic systems that are managed for aquatic ecosystem purposes (Class AE) according to Schedule 3 of the RMA, although other criteria may need to be taken into account for aquatic systems managed for other purposes. A more detailed overview of the regulatory requirements the framework aims to address is provided in Appendix A.

However, this document does not replace or supersede advice directly from consenting organisations such as regional councils or landholders such as DOC. All potential applicants are advised to liaise closely with regional councils and landholders to discuss any site-specific water quality issues and aquatic biological impacts. Further, the information provided in this document is correct as at September 2014, and users are advised to check with relevant councils and landholders for any updated documentation or requirements.

1.3.2 The framework

The New Zealand Minerals Sector Environmental Framework outlines a five-stage process to determine the potential extent of environmental impacts of mining on downstream aquatic systems and options to minimise these impacts. Included within the framework is consideration of the rehabilitation of mined areas, as effective rehabilitation will minimise downstream aquatic impacts, particularly post-mining, as well as being critical to the recovery of terrestrial ecosystems disturbed by mining.

A key aspect of the framework is that explicit 'acceptable' water quality criteria or rehabilitation outcomes are not established, because these are likely to be different at different sites and because social, economic and cultural factors may also influence decision making. Instead the framework provides a robust scientific basis for this decision to be made by end users during consultation processes.

More detailed technical and background information on all aspects of the framework is provided in Appendices B–F. In addition, a discussion on both natural events (e.g. high rainfall) and mining-related extreme events (e.g. tailings dam failure) that may impact streams downstream of a mine is provided (Appendix G) along with options for the minimisation of these impacts.

Each stage of the process is briefly discussed below. Figure 5 outlines the specific steps required for working through each of the five stages. Each step is described in further detail in the following chapters for the different mine types.

Stage 1: Potential for ecological impact

The potential for ecological impact downstream of a mine is determined by the water quality (pH, metals) that arises from mining operations, which can be predicted from the local and regional geology, geochemical analysis of rocks, and the water quality and quantity of the receiving system. The potential impact on terrestrial ecosystems depends on the complexity and age of the removed ecosystem(s). The degree to which these impacts can be mitigated depends on the resources available for rehabilitation.

[2] Further information about mining in general and regulatory requirements can be found at: http://www.crownminerals.govt.nz/cms and www.minerals.co.nz/html/index.html.
[3] If the minerals are privately owned a permit under the Crown Minerals Act is not needed, but all of the other permits and consents are still required, together with the consent of the mineral owner.
[4] In some cases, a concession from DOC may also be required if a track across DOC land is required to access the land in which the minerals exist; similar access would also be required to be granted by private landholders.

The framework is based on research undertaken by the research team and provides a guide and methodology for the prediction of water quality downstream of mines. Information to enable assessment of potential impacts and options for mitigation relevant for all mining operations is provided in Chapters 2–4 with specific information relevant for different mine types in Chapters 5-9. Further detail on geochemical requirements is presented in Appendix C.

Stage 2: Level of potential ecological impact on aquatic systems

A combination of biological survey data and toxicity testing using indigenous West Coast macroinvertebrate species has been used to establish the potential level of ecological impact. Our framework identifies several ecological impact thresholds associated with various water qualities. This information is presented in Chapters 5-9. Further detail on the biological effects arising from coal and gold mining and the development of the ecological impact thresholds is provided in Appendix D.

Stage 3: Determining the acceptability of the potential ecological impact

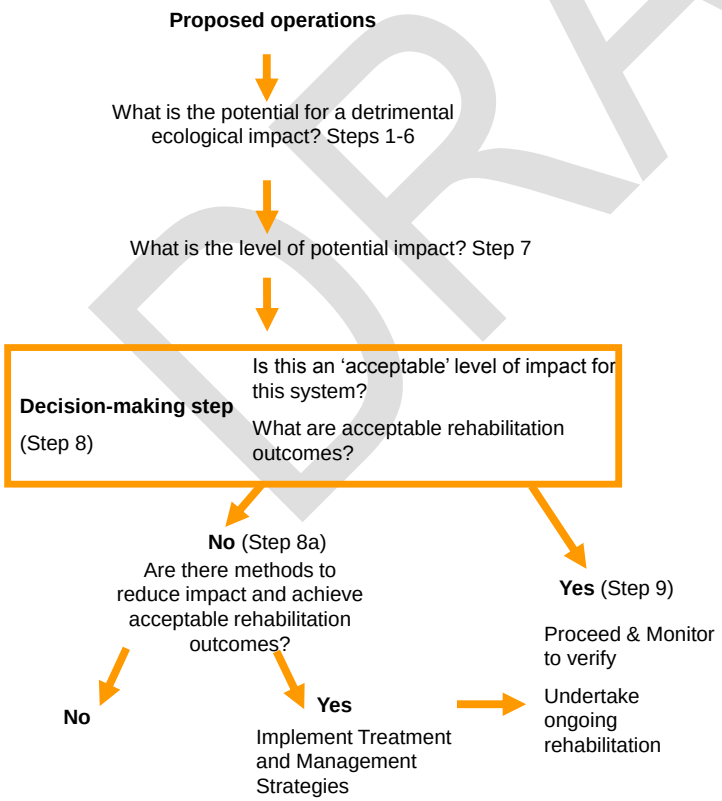
The predicted water quality (Step 1) can then be compared with these ecological impact thresholds to determine the likely ecological impact. This information can be used to determine whether the likely level of impact for the specific operation is acceptable or not. As stated above, the framework does not specify what 'acceptable' or 'unacceptable' impacts are, but rather provides a robust scientific basis for this decision to be made by end users during consultation processes.

Stage 4: Options to reduce the potential ecological impact

Where impacts are deemed unacceptable, options for the management of mine waste (termed operational management) and/or treatment of mine drainage through active or passive treatment systems to mitigate or prevent the formation and release of poor quality mine water are provided in the framework. This guidance is based on a literature review of operational management and treatment techniques, combined with fundamental research on treatment techniques. This information is presented in Chapters 4-7 with further details provided in Appendices E and F.

Stage 5: Developing an ongoing monitoring programme

Ongoing monitoring provides the information needed to adapt management systems if changes occur. Guidance is provided on monitoring (geological, treatment systems, water quality and biological), particularly that which should be undertaken on an ongoing basis after mining operations begin. This guidance has been developed from research experience and a literature review. This information is presented in Chapter 11, with further details on biological monitoring provided in Appendix D.



Step 1. Background and Baseline information
Step 2. Identify resources for rehabilitation and **collect rock samples** for geochemical testing
Step 3. Geochemical testing to determine acid-forming status and acid-neutralising capacity of rock samples
Step 4. Management of **suspended solids**
Step 5. Predict mine drainage chemistry using geochemical testing
Step 6. Predict stream water chemistry at downstream point of interest

Step 7. Determine the potential ecological impact on the stream

Step 8. Consider whether **impacts are acceptable, agree rehabilitation outcomes**
Step 8a. If unacceptable, determine whether **treatment and management strategies** to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes

Step 9. If acceptable, **design monitoring programmes** and **plan rehabilitation processes**

Figure 5 **General framework and detailed step-by-step guide for predicting and managing water quality (pH and metal) impacts from mining on streams.**

1.4 Document structure

The following chapters provide the details required to work through each step of the framework, and the structure of the document is outlined in Figure 6. Three general chapters outline the collection of historical (existing) and baseline (current) data, information required to predict water quality (pH, metals) downstream of mines, and impacts and management of sediment discharges to aquatic systems. Water quality (pH and metal) issues are discussed specific to different mine types in separate chapters relating to the five mine types: coal – potentially acid-forming (PAF), coal – non-acid-forming (NAF), gold – hard rock, and gold – alluvial. For each of these mine types a more detailed discussion is provided of the likely downstream water quality, potential aquatic ecological impact, and options for waste rock management and treatment to reduce aquatic impacts and preparation for rehabilitation. The processes for undertaking rehabilitation of mined areas are covered in a separate chapter (Chapter 10). Guidance on all monitoring that should be undertaken after mining operations commence, including rehabilitation, is provided in Chapter 11. At the conclusion of each chapter, the step-by-step guide is presented as a checklist expanding on the detailed information for the relevant steps provided in that chapter. To illustrate application of the framework, a worked example is provided in Chapter 12, along with a list of service providers.

The main text of this document gives an overview of information required for decision making; the appendices provide more detailed supporting information, including information on the potential impact of extreme events.

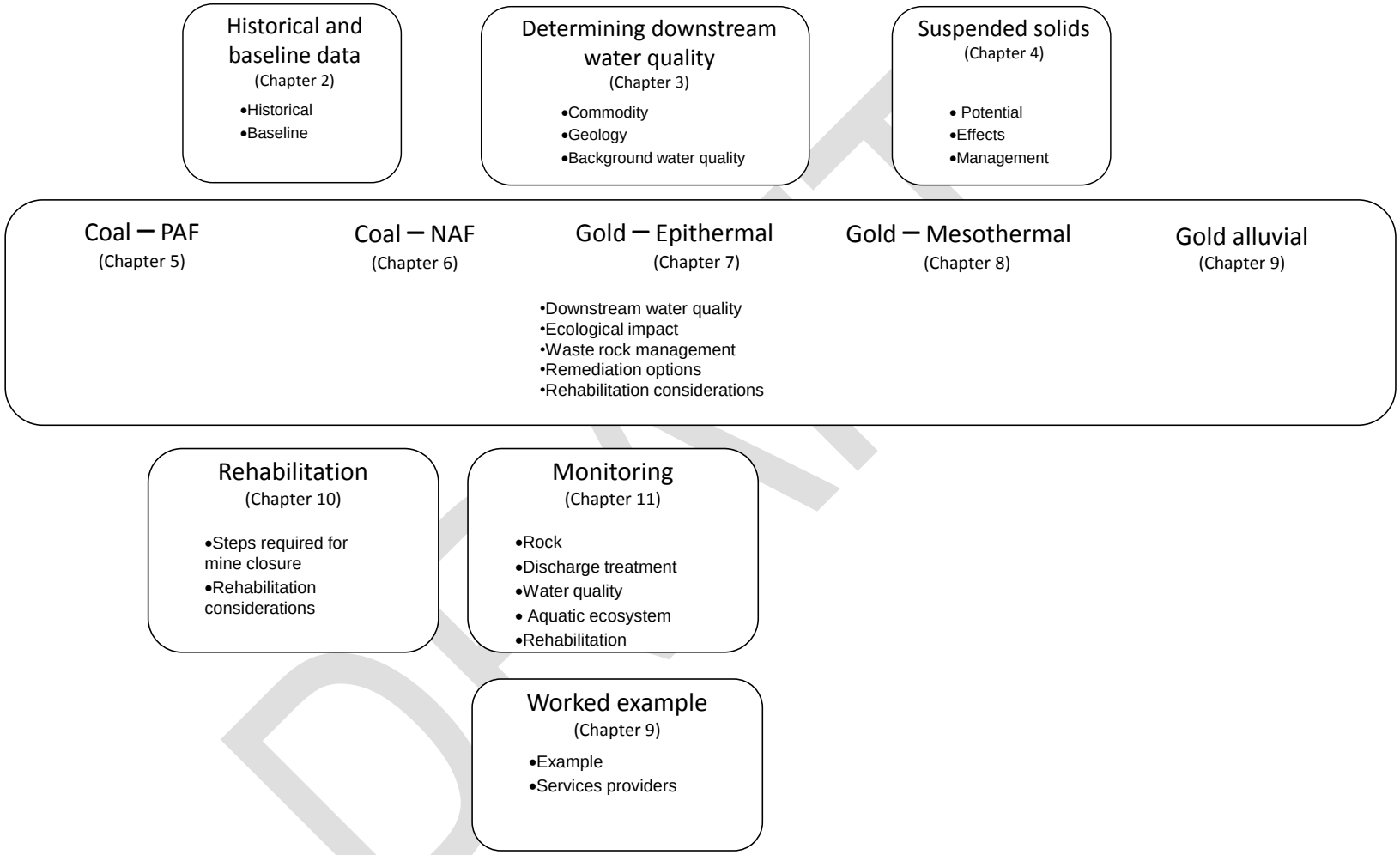


Figure 6: **Outline of document structure.**

2 HISTORICAL AND BASELINE DATA

2.1 Introduction

To predict the potential ecological impact of a mining activity on stream ecosystems, information is required on the site's water quality and current ecological status. This information will usually be sourced through a combination of previous reports or databases (historical data) and field surveys to establish current conditions (baseline data). Field surveys to collect baseline data are particularly important in identifying any unique features of waterways in the mine area, such as threatened ecosystems and rare species or impact from historical mining activity or other industries. These data may also influence subsequent management decisions.

This chapter provides an overview of the data required to determine the potential impact of mining on downstream water quality and to assist in determining achievable rehabilitation of on-site terrestrial ecosystems. The type and scope of field data collected are likely to be influenced by the historical data existing from previous work, in addition to any aspects that come to light in the early planning stages.

2.2 Historical data

Collation of existing information on the hydrogeology, water quality and ecological status of any waterways that may receive mine drainage discharge is a critical first step in being able to determine the potential impact of that discharge. In particular, where proposed new mine operations overlap or are adjacent to historical mines, information on drainage from historical mines gives an indication of mine drainage chemistry for the proposed operations. However, for these interpretations to be accurate, care must be taken to check that both the proposed and historical mines are in the same geological formation. Different geological formations with different mine drainage characteristics can be adjacent to each other and might have different mine drainage chemistries. Discussion on the use of historical information to provide a qualitative desktop assessment of potential mine drainage chemistry is provided in section **3.4**.

Relevant historical information will include geological maps, soil maps and existing data on water quality, stream flow, terrestrial ecology, stream ecology and climate. The extent of historical information available will influence the amount and type of baseline data required to be collected.

Historical information and sources of this information include:

- Aerial photos – available from Land Information New Zealand (LINZ) at http://www.linz.govt.nz/topography/index.aspx
- Soil maps (S-map), Landcover Database 4, NZLRI Land Use Capability, and soils data from LRIS Land Resources Information Systems (www.landcareresearch.co.nz/reources/data/lris)
- Ecosystem attributes from Land Environments of NZ (LENZ, https:\\lris.scinfo.org.nz) includes climate, soil attributes and native vegetation from NVS (National Vegetation Survey Databank)
- Geological maps – available from GNS Science at http://www.gns.cri.nz/store/publications/maps.html
- Climate and river flow data – available from NIWA at http://cliflo.niwa.co.nz/ and http://edenz.niwa.co.nz/map/riverflow respectively. These are free databases, although registration is required to download data. Additional river-flow data may be obtained by contacting NIWA directly, and may incur some cost.
- Regional councils may hold relevant information in technical reports, reports supporting resource consent applications, or monitoring data including biological, flow and water quality and quantity monitoring; these can be obtained by contacting the relevant council directly.
- During the development of this framework raw data relating to mine drainage chemistry, stream water quality and biological monitoring were collated into the 'DAME' (Database for Assessment of Mine Environment) database. Please contact the West Coast Regional Council or Environment Southland to discuss accessing this database.
- Data collected during previous mining operations and exploration were reported to Crown Minerals (now New Zealand Petroleum and Minerals, Ministry of Business Innovation and Employment) and are available via the Internet (http://www.nzpam.govt.nz/cms , tools section) or from the NZ Petroleum and Minerals library. Registration is required to access the technical data from the website, although there is no associated cost.
- Currently no comprehensive and freely available databases exist for water chemistry, stream algae or invertebrates. Data on fish distribution are available from NIWA (http://www.niwa.co.nz/our-services/online-services/freshwater-fish-database). Registration is required to access data from this site, although there is no associated cost.
- The Department of Conservation uses a freshwater classification system; Freshwater environments of New Zealand classification (FENZ http://www.doc.govt.nz/conservation/land-and-freshwater/freshwater/freshwater-ecosystems-of-new-zealand/) and has developed a list of Waters of National importance (WONI)

2.3 Baseline information

Collection of data on the current hydrogeology, water quality and ecological status of any streams that may receive mine drainage discharge is a critical first step in being able to determine the potential impact of that discharge. This information may subsequently be used in models to predict downstream water chemistry at a specific site. In addition, baseline information might form the basis for consent application details where consent conditions are related to maintaining previous water quality.

2.3.1 Site hydrogeology

Surface water hydrogeological data, including detailed knowledge of stream channels and measurements of flow volume, are required for characterisation of catchment conditions. Ultimately, a hydrogeological model is required of the catchment to be disturbed by mining. In particular, hydrogeological models that include calculation of the amount of rainfall that contributes to surface flows, compared with groundwater, are required to calculate the flow volumes from mining disturbances. These models are best completed by a suitably qualified specialist.

Collection of flow data

A suite of average-stream-flow measurements is required to predict the effects of mixing mine drainage with other catchment water. Stream flow measurements are required for all tributaries that contribute more than 5% of surface flow volumes or 5% of any dissolved components to the main stream of interest.

The frequency and duration of sampling will be dependent on the variability of the background flow conditions. At a minimum, flow should be measured concurrently with the collection of water quality samples (see below), although continuous flow measurements from at least one site in each catchment will provide additional robustness.

Additional information on flow rates and their variability with season, rainfall, and drought is useful for detailed planning of water management strategies at mine sites. This can be coupled with water chemistry data to assist in the design, optimisation and implementation of water management or treatment strategies.

Subsurface hydrogeology

At some mines, e.g. underground mines or possibly deep opencast mines, groundwater flow will be more important than surface water flow. At these sites three-dimensional hydrogeological models will be required and the influence of groundwater flow into rivers should be compared with measured surface runoff. Groundwater data collection and modelling is a specialist field and is best completed by an experienced or suitably qualified hydrogeologist. Background data are used to characterise the environment prior to mining and some of the data collected can be used to predict the impact of mining downstream should consent applications proceed.

2.3.2 Baseline chemical water quality

The chemistry of the receiving environment can have a significant impact on water quality downstream. For example, if the background water has sufficient alkalinity, addition of small volumes of AMD might have minimal downstream effect. Or where streams contain naturally elevated concentrations of trace elements, additions of small volumes of NMD with higher trace elements might cause only a very small change to overall water quality. In contrast, streams with low alkalinity or low trace elements could be significantly impacted by small volumes of AMD or NMD.

On the West Coast, water quality from streams draining different geological formations can be highly variable. For example, water draining from Greenland Group rocks often has neutral pH, elevated alkalinity and naturally elevated As concentrations, whereas water draining from Brunner Coal Measures often has low pH (down to <4) as well as elevated Al and trace elements (Pope et al. 2006, 2010). Therefore it is essential that baseline water quality data are collected prior to mining so that the likely level of environmental impact can be estimated.

There are several key chemical factors to consider in an assessment of baseline water quality. These include:

- Natural sources and concentrations of alkalinity
- Natural sources of acid rock drainage (ARD)
- Background or baseline physiochemical properties (pH, electrical conductivity (EC), dissolved oxygen, etc.) and concentrations of sulphate, dissolved Fe and Al, and other dissolved trace elements such As and Zn
- Existing sources of mine drainage

Collection of baseline water quality data

A representative suite of samples collected from the entire catchment of a proposed mine site during typical flow conditions will generally be sufficient for characterisation of background water quality. If the rock geochemistry is characterised (Chapter 3), these water samples are also important for prediction of downstream water chemistry, should mining proceed). As a rule of thumb, this suite of water samples should include samples collected at a sufficient number of locations to capture all inputs that contain greater than 5% of flow volume or 5% of any dissolved component to the most downstream site. Samples should be collected concurrently with any biological monitoring being undertaken. The characterisation of background site chemistry requires dissolved concentrations of all relevant components and samples should be collected according to standard methods (e.g. Standards New Zealand 1998a, b). The minimum analysis should include:

- Physicochemical properties – pH, EC, TSS, and DO

- Major chemical components including bicarbonate (HCO^{3-}), sulphate (SO_4^{2-}), calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), Fe, Al
- Trace elements

Selection of analyses of trace elements such as Zn, As, Sb, Ni, or Mn may be relevant for certain systems and in some cases non-filtered samples might be required to identify the mode of transport for some trace elements. The importance of different trace elements at different types of proposed mines can be established by analysis of data from other similar operations. Where a new type of mine is proposed, a cautious approach to trace element analyses is recommended.

Other parameters such as oxidation/reduction potential (Eh), salinity, or Fe speciation can be used to refine water quality assessments and predictions. More detailed analytical procedures such as repeat sampling, data-logging and monitoring of chemistry throughout rainfall events can all be used to improve and strengthen water quality predictions.

Charge-balanced analyses, which sum up all major cations and anions, can provide a check that all major components have been analysed (Standards New Zealand 1998a, b). This analysis provides a useful tool for interpreting water chemistry but is not essential at all sites.

The sampling strategy and analyses required to characterise a site prior to mining are site specific and experienced water quality scientists should be consulted to determine the location and number of samples and types of analyses.

Drainage from historical mining activities

Sites that are impacted by historical mine drainage should have a baseline chemical and hydrological survey completed. This survey should include chemistry and quantity of the historical mine drainage in addition to inputs from unimpacted streams. Assessment of historical mine drainage chemistry has predictive value for future operations and provides a baseline from which change due to new operations can be measured.

2.3.3 Baseline biological monitoring of aquatic systems

Biological monitoring involves the assessment of stream communities in order to determine their current health or condition. Assessments may also be appropriate in lakes, wetlands and aquifers. However, these other systems are not covered in this framework. It is essential to conduct baseline monitoring prior to any mining as these data provide the background for any future comparisons of impacts or recovery. The same general techniques are used for monitoring after mining operations commence (Chapter 8).

Because stream organisms live most of their lives under the water, stream communities reflect an integrated record of water quality over an organism's lifetime. Fungi, bacteria, meiofauna (e.g. plankton and mites), algae, macroinvertebrates and fish are all important in natural stream communities and each is vulnerable to mining impacts. In this document we focus on using macroinvertebrates (Figure 77) as biomonitoring agents, largely because they are relatively easy to identify, sampling procedures are simple and well developed, and there is considerable knowledge of their ecology including how they respond to mine drainage (see Appendix D for further discussion). In addition, most regional councils in New Zealand conduct annual biomonitoring of macroinvertebrates as part of their state of the environment reporting, so there are numerous organisations experienced in using macroinvertebrates for biomonitoring.

Figure 7 **Examples of macroinvertebrates that commonly live in streams and rivers throughout New Zealand. Top left: the spiral-cased caddisfly, Helicopsyche. Top right: the stonefly, Zelandoperla. Bottom left: ubiquitous mayfly, Deleatidium. Bottom right: the common stonefly, Zelandobius.**

Macroinvertebrates can be sampled at differing levels of intensity:

- Qualitatively, where the presence or absence of a species is recorded at a site
- Semi-quantitatively, where the relative abundance of each species is determined
- Quantitatively, where the abundance of each species is determined in a sample of known streambed area. Abundance is usually expressed as number of animals per square metre of streambed.

New Zealand has standard protocols for sampling wadeable streams, which cover each of these sampling methods, and they are detailed in Stark et al. (2001) and paraphrased in Appendix D.6.

Of the three levels of data intensity, qualitative data are the fastest and cheapest to collect, and a number of measures (metrics) can be calculated using the presence or absence of species (Appendix D.6). However, qualitative data ***do not*** detect changes in the relative abundance of key species, and therefore overlook a potentially significant component of change in stream communities. Results can also be influenced by the sampling effort as greater sampling effort collecting the samples will capture more species. By contrast, semi-quantitative data can detect changes in relative abundance and require no additional sampling effort and only a minor increase in laboratory effort. Thus, the majority of stream monitoring should obtain at least semi-quantitative data. The third method, quantitative sampling, provides higher resolution data that can detect subtle changes in community structure. Quantitative sampling provides the strongest data for any assessment. However, the increased number of samples (or replicates) and greater laboratory processing time mean that this sampling will be more expensive.

In addition to biological sampling, assessing the physical and water chemistry properties of sites is strongly recommended. These assessments should occur at the same time as biological monitoring. Physical habitat conditions can also be incorporated into analyses to increase the ability to detect subtle changes in stream condition. Standard protocols for physical stream habitat assessment are outlined in Harding et al. (2009).

Collection of baseline biological data

A baseline biological survey should be completed prior to any mining and will usually be a requirement as part of any assessment of environment effects (AEE). Sampling should include both 'reference' and potentially-impacted sites to enable the subsequent detection and quantification of mining-induced change. A reference site is a site that represents a typical non-mine-impacted site in the region. Ideally, it should also be physically comparable (e.g. similar size, elevation and substrate) to impacted sites. It is absolutely essential that reference sites are included in any baseline survey as data from these sites provide the ability to detect any changes that might occur independently from mining activities; for example, changes caused by large floods, droughts, vegetation regeneration or other factors. A critical aspect of designing a biological monitoring programme is selecting the location and number of sampling sites, as they directly influence the ability to detect and monitor change. The most important considerations are:

- The location and comparability of potentially-impacted and reference sites
- Replication both between and within impact and reference sites

To enable both temporal and spatial reference comparisons, sites used for future monitoring and consents should be selected from those sampled during the initial baseline survey.

Analyses should involve before-after-control-impact comparisons (BACI designs), which are widely accepted as the standard method to detect and quantify ecological impacts (see Figure **8: Before-after-control-impact (BACI) sampling designs for assessing the environmental effects of human impacts on stream ecosystems. Each circle represents a sampling site. A rigorous BACI design includes multiple sampling sites in impacted reaches, and upstream and additional stream control (reference) sites, both before and after an impact.**

for an example of the sample design and Appendix D.6 for more detail).

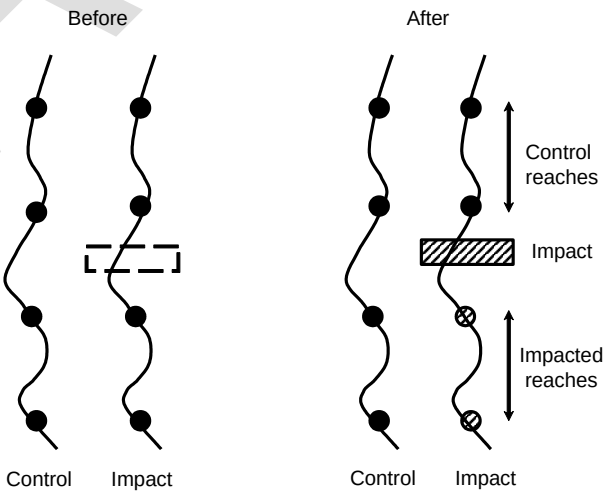


Figure 8: Before-after-control-impact (BACI) sampling designs for assessing the environmental effects of human impacts on stream ecosystems. Each circle represents a sampling site. A rigorous BACI design includes multiple sampling sites in impacted reaches, and upstream and additional stream control (reference) sites, both before and after an impact.

Baseline biological sampling for AEE occurs before access or resource consent proceedings and generally only needs to be done once (although multiple surveys will always provide stronger data). During a survey, all sites should be sampled within a few days of each other to minimise the likelihood of events such as floods or stream-drying influencing differences among sites. Furthermore, as invertebrate communities can be

in a state of recovery after flooding events, a general rule of thumb is to sample at least 5 days after any major flood that has moved large bed material. Smaller floods can probably be sampled after 2–3 days. In many regions around the country a longer period might be preferred. However, on the West Coast this is often impractical.

Although stream invertebrates can be sampled at any time of the year in New Zealand they are often larger (and easier to identify) during late winter to early summer (August-January). So it is preferable to conduct surveys during these months.

Baseline biological surveys often focus on biodiversity, thus the emphasis is often placed on collecting qualitative or semi-quantitative data across multiple sites, rather than quantitative data over fewer sites. Although this framework focuses on macroinvertebrates, baseline assessments would also normally qualitatively sample fish communities. Standard protocols for sampling freshwater fish in New Zealand have been released (David & Hamer 2010). These surveys should be conducted by an experienced freshwater scientist.

In summary, baseline sampling should include:

- An absolute minimum of three reference sites (ideally five or more) including sites both upstream of the impact zone and preferably also in unaffected catchments
- Sites in all potentially impacted tributaries and sites arranged longitudinally downstream on the mainstream river
- Extensive semi-quantitative or quantitative data.

2.3.4 Baseline monitoring of terrestrial systems

Pre-mining information will be required to identify the ecological and/or production values within areas impacted by a proposed mine. Identifying floral and faunal diversity and distribution is important, with special attention to locally or nationally threatened native species, ecosystems or habitats. The features that influence the significance of native ecosystems or species are usually defined in regional council plans. These can include the landscape pattern, for example the connectivity and function, for example wetlands. Production values can be measured by crop yields, monthly pasture dry matter, stocking rates, tree index.

Baseline sampling should inform rehabilitation planning by providing information on:

- Pre-mining condition of farmland and production forest, including the land use capability (and what limits productivity)
- Flora and fauna. Monitoring may need to be in a particular season, for a minimum duration (or intensity of searching), or under specific environmental conditions. Standard protocols are available for monitoring specific fauna and flora, for example, invertebrates are usually sampled during summer months, *Powelliphanta* land snails on nights above a specific temperature and moisture.
- Age and complexity of the landform and ecosystems. This indicates the time frame and difficulty of rehabilitation
- Potential rehabilitation materials and their salvageability
- Natural revegetation and plant succession processes (land slips etc.) for native ecosystems
- Associated infrastructure such as water supplies, access (vehicle, walking, cycling) and buildings.

2.4 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide.

Step 1. Background and Baseline information

Background data

- Mineral exploration reports ☐
- Geological maps and reports ☐
- River flow and climate data ☐
- DAME database ☐
- Regional council database and reports ☐
- Soil and land capability maps and reports ☐
- Land Environments of New Zealand (ecosystems) and Land Cover Data Base (land use, LCDB4) ☐

Baseline data

- Site hydrogeology ☐
- Water quality ☐
- Tributaries and main receiving system ☐
- Existing mine drainage ☐
- Biological (aquatic and terrestrial) ☐
- Reference sites (aquatic) ☐
- Potentially impacted sites (aquatic) ☐

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☐

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☐

Step 4. Management of suspended solids ☐

Step 5. Predict mine drainage chemistry using geochemical testing ☐

Step 6. Predict stream water chemistry at downstream point of interest ☐

Step 7. Determine the potential ecological impact on the stream ☐

Step 8. Consider whether impacts are acceptable, agree rehabilitation outcomes ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

3 PREDICTION OF MINE DRAINAGE AND DOWNSTREAM WATER CHEMISTRY

3.1 Introduction

Prediction of mine drainage chemistry is economically and environmentally significant for all parts of the mine environment life cycle. Mine drainage chemistry influences mine planning, mine operations, and mine closure and post-closure activities and costs. During mineral exploration a preliminary prediction of mine drainage chemistry can be used to prioritise exploration targets. Further, pre-mining activities provide the opportunity to identify the resources available for rehabilitation, which is critical in determining feasible outcomes. During mine planning and operations, detailed predictions of mine drainage chemistry and evolution with time will assist with water management and treatment. Prediction of mine drainage chemistry can help to determine potential treatment requirements during mine closure and post-closure. The following section covers the types of information that should be collected to predict mine drainage and downstream water chemistry.

3.1.1 Commodity and region

At the broadest scale regional geological information can be used to predict mine drainage chemistry through comparison with other deposits that have been mined in similar rocks. The key information to be determined is the geological formation(s) that will be disturbed by a proposed mine, which can be determined from knowledge of the commodity being mined and the location. Assessments based on regional geological and historical information are indicative. Detailed analyses of the various rock types that will be disturbed at a proposed mine site are required to provide quantitative assessment.

A geological formation is a group of rocks that are recognisable over large areas (typically >tens of km^2) or stratigraphic thicknesses (>100 m). These groups of rocks have similar characteristics such as age, composition, geological history and depositional process. Rocks of a single geological formation have a limited range of mineral compositions, and the mineral composition of rocks disturbed by mining will determine the characteristics of mine drainage. Therefore identification of the different geological formations that will be disturbed by a proposed mine can provide qualitative information on likely mine drainage chemistry (Pope et al. 2006, 2010; Craw et al. 2008).

The way in which mining disturbs the rocks will influence the resultant drainage. Mining disturbances may range from groundwater chemistry changes in the rocks surrounding mines to removal, crushing, pulverising and chemical alteration of rocks during mineral/ore processing. Of particular interest for the prediction of mine drainage chemistry is the acid-forming or neutralising potential of rocks and the trace element concentration and mobility. Rocks that are unlikely to form acid during mining are called non-acid forming (NAF); rocks that are likely to form acid during mining are called potentially acid forming (PAF). Both NAF and PAF rocks can release trace elements at elevated concentrations.

Relatively few geological formations host gold or coal deposits in New Zealand and often the host rocks are different in each region. Typically gold deposits occur within different geological formations to coal deposits. Figure 9 shows the geological formations that are likely to be disturbed by coal and gold mining in New Zealand, along with comments on likely mine drainage characteristics. Detailed information relating to the location and extent of geological formations can be obtained by analysis of geological maps. The most up to date compilation of geological maps for New Zealand is the 1:250,000 Q Map series (GNS Website - http://www.gns.cri.nz/research/qmap/aboutqmap.html). More detailed geological maps for specific areas may be available from GNS, university theses, published reports and scientific papers, or from mineral exploration reports (Crown Minerals Website - http://www.crownminerals.govt.nz/cms). Generalised maps and descriptions for examples of the geological formations in Figure 9 drawn from these sources are shown in Appendices C.1 and C.2.

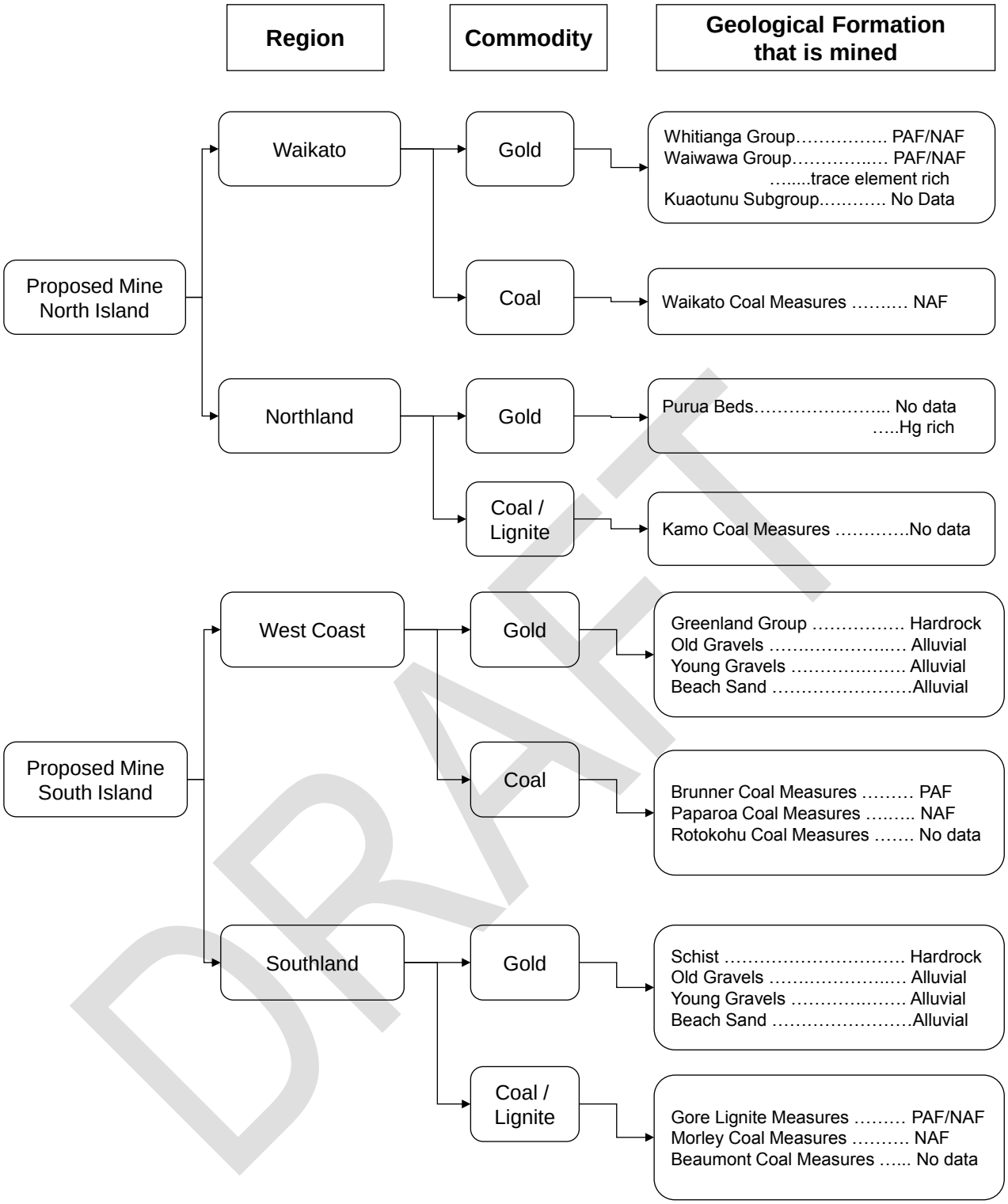


Figure 9: **Formations that could be disturbed by coal and gold mining on the predominant mining regions on the north and south islands of New Zealand.**

Where proposed new mine operations overlap or are adjacent to historical mines, information on drainage from historical mines may give an indication of mine drainage chemistry for the proposed operations. However, to ensure these interpretations are valid, care must be taken to check that both the proposed and historical mines are in a similar geological formation. Different geological formations with different mine drainage characteristics can occur adjacently and thus will have different mine drainage chemistry. In addition, mine chemistry evolves with time and some direct analogy of historical mine drainages to future mine drainage chemistry could be difficult.

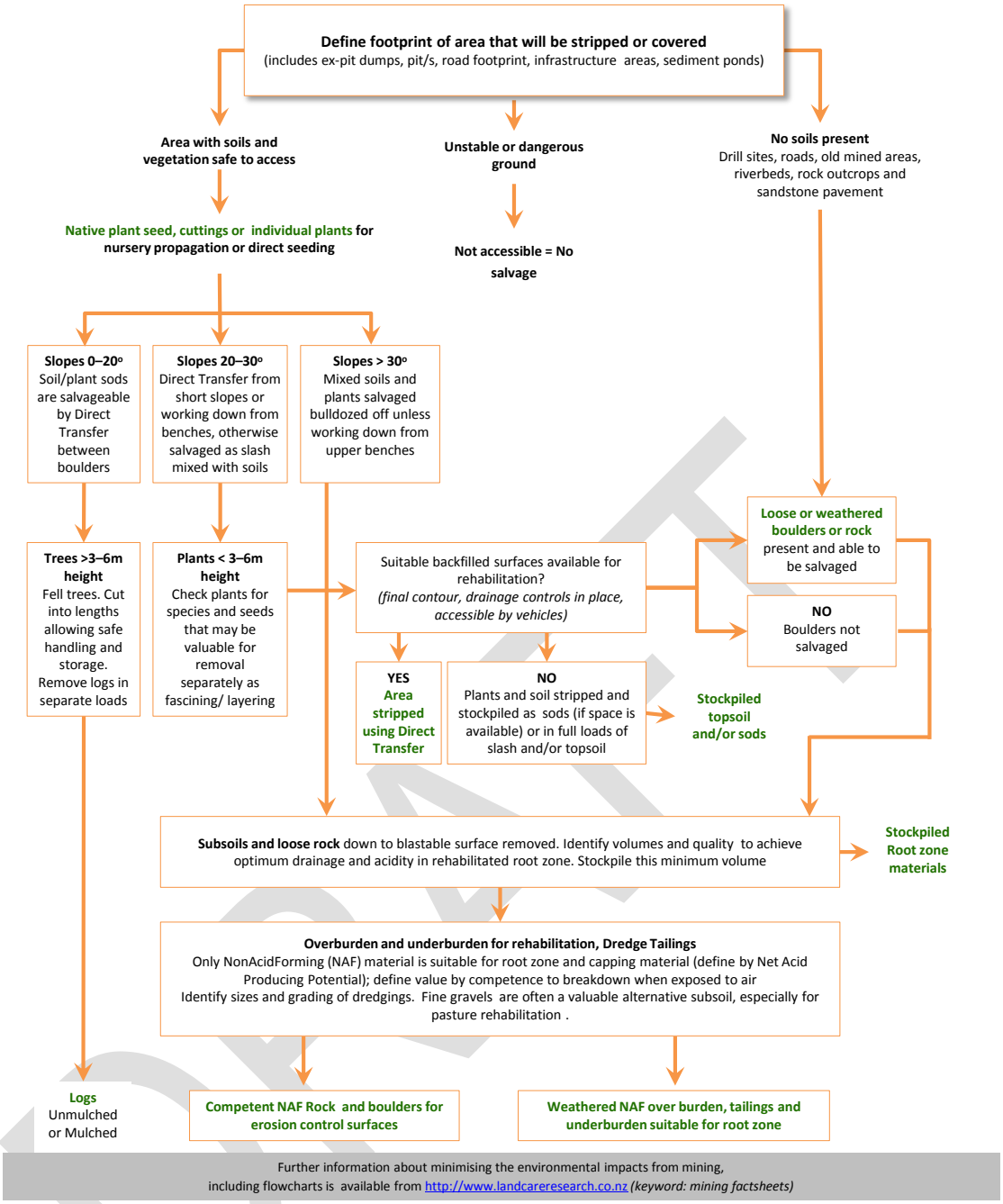
3.2 Identifying resources for rehabilitation

Successful rehabilitation starts with the identification, salvage, and conservation of rehabilitation resources as this determines what rehabilitation outcomes are possible. Resources include:

- Plant growth materials in the overburden and soils that will be used to create new root zones
- Materials for controlling and limiting erosion. These include competent rock suitable for rip-rap used to create erosion-resistant drainage paths
- Plant and fungi propagules, including seeds and cuttings. This includes plants suitable for using as intact sods, as direct transfer
- Fauna, for relocation and potential reintroduction to rehabilitated areas, including invertebrates within direct-transfer sods (earthworms)
- Infrastructure such as posts, water troughs and buildings.

Figure 10 provides an overview of the general process to identify and salvage materials for rehabilitation. The location of where resources will be stored if not used immediately during progressive mine rehabilitation is also required to be identified. However, miners can maximise the value of rehabilitation resources by using staged stripping which enables no-stockpiling techniques, such as direct transfer ('DT'). Mine scheduling needs to allow time and access to the vegetation ahead of bulk stripping, and mine planning needs to allow designated stockpile space where materials are kept separate form overburden.

The identification and salvage of resources should be undertaken before mining starts and repeated during mine operations as more detailed information on overburden materials becomes available, machinery changes, and mine plans are updated. Together the information helps refine practical and cost-effective rehabilitation options.



(Green text highlights the resources for rehabilitation, NAF = Not Acid Forming.)

Figure 10: Overview of the typical steps to identify and salvage materials for rehabilitation.

3.3 Analysis of rocks from the proposed mine site

Field observations and laboratory analysis of rocks to be disturbed by mining are used to assess the likely quality of mine drainage. Field observations are useful for the selection of rocks for further analyses and interpretation of data once analytical results are obtained. Laboratory analysis of rock composition and reactivity can enable qualitative and quantitative assessment of the mine drainage chemistry that can be expected from a prospective mine area. The interpretive value of these analyses is improved by having a good collection of geological data, a robust geological model, and thorough observations of field characteristics or rock samples. Analytical procedures vary from rapid and standardised laboratory analyses to specialised testing procedures that can be designed to investigate site-specific geochemical issues. This section provides a guide to the appropriate sampling and analyses that could be undertaken, and the interpretation of results.

3.3.1 Important minerals and field observations

The type, abundance, distribution and reactivity of minerals within rocks can strongly influence the chemistry of mine drainages, associated potential environmental impact, and management or treatment strategies required. The most important groups of minerals that influence mine drainage

chemistry are sulphides and carbonates (Plumlee & Logsdon 1999; Appendix C.3). Oxidation of sulphide minerals causes acid production and strongly influences the trace-element geochemistry of mine drainage. In general, carbonate minerals can neutralise acid produced by sulphide minerals and also contribute trace elements. There are several other groups of minerals, including secondary minerals that form after sulphide oxidation, that influence the acid-producing or neutralising characteristics of rocks.

The distribution of sulphide, carbonate and secondary minerals varies within and between rock types within a coal or gold deposit. Therefore a general geological/mineralogical description of any samples collected is essential for interpretation of analytical data. Detail of what should be included in a general geological description is provided in Appendix C.4.

Field or hand-specimen observations of minerals in rocks to be disturbed by mining are a qualitative tool for assessment of mine drainage potential. These observations are useful for the selection of rocks for further analyses (see section 3.3.2) and interpretation of data once analytical results are obtained. Important observations include:

- General geological description of a rock type or sample (Appendix C.4)
- The presence of primary sulphide minerals, particularly pyrite and arsenopyrite
- The presence of carbonates, particularly calcite and siderite
- The presence of secondary minerals that indicate the reactivity of rocks when exposed to surface, such as Fe^{3+} (ferric) oxides, hydroxides and hydroxysulphates, Al hydroxides, sulphate minerals

Further details, including photos of different rock types, are provided in Appendix C.5. Field observations of minerals in hand specimens or outcrops are qualitative and should not replace laboratory analysis of rocks. Rather, these observations assist with the interpretation of laboratory data.

Field observations should also be completed on a regular basis throughout all mining and resource development phases to increase the level of confidence in the existing data and so that previously unidentified rocks that may have implications for mine drainage chemistry are identified, analysed and appropriately managed.

3.3.2 Sampling strategies for geochemical characterisation of rocks

Quantitative laboratory analysis of rock geochemistry is required to make more robust prediction of likely mine drainage chemistry from a proposed new mine site. Quantitative tests for acid production potential are collectively called acid–base accounting, while trace element concentrations in rocks can be determined by techniques such as X-ray fluorescence (XRF). Quantitative testing can be undertaken for several purposes and sample collection strategies will differ accordingly. However, fresh samples are always preferred as weathered rocks can give misleading results in some acid–base accounting tests. For the purposes of characterisation of rock at a new mine site or mine development, some general rules have been developed during the research programme that can be applied to ensure analysis density is sufficient.

Sampling density should suit the complexity and variability of the various rock types in the mine development area. For example, coal measure formations may require more samples than homogeneous rock formations such as sandstone, because coal measure formations may contain a wide variety of rock types. Samples collected during site characterisation for acid–base accounting or trace element analysis should be collected with the following aims:

- Characterisation of representative rock types within the sequence of rocks to be disturbed by mining including ore rocks or coal
- Identification and sampling of specific geological features that have anomalous acid-producing or neutralising characteristics. This includes:
    - For coal deposits, geological structures such as faults or beds that contain pyrite or calcite, and the roof and floor of coal seams, could have anomalous acid-producing or neutralising characteristics.
    - For hard-rock gold deposits, anomalous acid-producing or neutralising minerals or trace elements could be localised in mineral alteration zones and faults.
    - For alluvial gold deposits, the presence of oxide minerals or sulphide minerals should be noted and gravel coarser than 2 cm should not be included in the sample. [Grain size directly influences the rate of reaction with coarse-grained material reacting more slowly. In addition, the mass of sample collected can be biased toward a particular rock type if several large rocks are included and therefore, for a 1-kg sample, material larger than 2 cm could bias the sample]
- Sufficient data should be collected so that statistically meaningful acid–base accounting values can be identified or calculated for important rock types.

In general, sampling of rocks for environmental geochemical purposes can be completed at the same time as the collection of exploration data. The mine drainage implications of all rock types can be treated in a similar manner to exploration data so that three-dimensional models of rocks with different environmental implications can be compiled and these rock types may be selectively mined and managed appropriately (see also section **5.4.1**). Additional sampling of PAF and NAF rocks is required throughout resource development and mining so that rocks continue to be appropriately managed (see also Section 11.2).

How many drill holes should be sampled?

The density of data required for assessment of mine drainage chemistry is usually less than that required to identify and define a coal or mineral deposit. Therefore, data collection for mine drainage assessment can accompany exploration or

resource definition drilling and not all drill holes need to be sampled for environmental purposes.

Where geology is complex (>5 different rock types present in any 20 m of core) we recommend that at least one drill hole should be sampled for every block of 250 × 250 m. Where geology is simple (<5 different rock types present in any 20 m of core) at least one drill hole should be sampled for every block of 1000 × 1000 m. The decision to use a drill-hole density greater than 250 × 250 m for acid–base accounting analyses should be made by an experienced geologist.

The drill-hole densities recommended here provide minimum data for an initial assessment of acid–base accounting data or trace element data. Once operational, sample density may need to be increased to enable effective management of materials.

How many samples per drill hole?

Density of sampling within drill holes should ensure that the complete sequence of rocks to be disturbed by mining is sampled. At least 5–10 samples of each rock type encountered in the sequence that is to be disturbed by mining should be submitted for acid–base accounting analysis. At most exploration targets, it is likely there are 4-8 different rock types; therefore a minimum of 20-40 rock samples should be analysed for acid–base accounting for each block of 250 × 250 m.

If drill cores are not collected and/or drilling processes produce rock chips, then rock chips may also be collected for analysis. Prior to submission for analysis the rock chips should be sieved and rinsed because they may be contaminated with additives used during drilling. These additives can alter results of the analyses, although are unlikely to change results of acid-base accounting and net acid generation (NAG) tests.

Where alluvial gravels are to be sampled, samples should be taken from areas that are beneath the oxidised zone. The oxidised zone can be identified by the presence of Fe oxide staining (Appendix C.5).

In summary, the following samples should be collected over the series of drill holes sampled for environmental analysis:

- A minimum of five samples from each rock type to be disturbed by proposed mining
- Roof and floor rocks from coal seams in coal deposits
- Rocks from the alteration zone at hard-rock gold deposits
- Rocks from below the oxidised zone in alluvial sediments.

What kind of samples should be collected if there is no drilling?

Where drilling is not part of the exploration programme, or it is not practicable to undertake drilling, alternative sample types should be collected. This might include samples from:

- Rock outcrops and/or road cuttings where rocks have been exposed
- Trenches or test pits
- Old mine workings.

The strategy and density of the sampling should be similar to that for drill cores. Five to ten samples of all rock types that will be disturbed by mining should be sampled for acid–base accounting analysis for each 250 × 250 m area, and selective sampling of rocks with acid-forming or neutralising implications should be completed, where possible. Samples collected from such locations are typically of poorer quality than drill cores because of weathering processes. Weathering generally affects the top 10 m of outcropping rocks and this can influence the acid–base accounting properties of the rock. Effort should be made to collect the least weathered rock where possible and to provide detailed geological descriptions (Appendix C.4).

3.3.3 Acid–base accounting (ABA)

Acid–base accounting (ABA) tests can be used to identify the rocks that have the potential to change pH or increase the acidity or alkalinity of mine drainage chemistry. There are several reasons to undertake ABA analysis, including:

- To determine the presence or absence of potentially acid-forming (PAF) rocks
- To determine the presence or absence of acid-neutralising rocks
- To predict mine drainage chemistry
- To establish relationships between specific rock types and acid production or neutralisation
- To optimise management of waste rock or overburden/interburden with respect to mine drainage chemistry
- To select rock types for more detailed geochemical analyses

Acid–base-accounting analyses provide information on the geochemical characteristics of the rocks, but do not provide information on the rate (kinetics) at which different rocks react or on trace element concentrations and potential mobility. The relationships between ABA properties, rock reactivity and trace element concentrations can be determined if sufficient additional data such as kinetic test information or data from historical mine drainages are available.

Acid–base-accounting analyses are the most common tests carried out to determine whether mines will produce AMD. In general, ABA analyses identify the maximum amount of acid produced, and the maximum amount of acid that can be neutralised by a rock during weathering. Acid–base-accounting results can be combined with geological data relating to the distribution of different rock types to identify particular rock types or areas of concern, which can influence mine planning.

A brief description of the different tests that are commonly used, their limitations, and a brief guide on the interpretation of the results are listed below with further details provided in Appendix C.6.

Maximum potential acidity (MPA)

- Total sulphur (S) is determined and the maximum potential acidity (MPA) that could be generated is calculated assuming all S is sourced solely from the mineral pyrite (FeS_2) with the results expressed as units of kilograms of H_2SO_4 per tonne of rock (kgH_2SO_4/t).
- Usually MPA values are between 0 and 200 kgH_2SO_4/t.
- MPA analyses are commonly combined with ANC analyses for interpretation.
- There are some important limitations to MPA testing that should be understood when interpreting the results of MPA analyses (Appendix C.6).

Acid-neutralising capacity (ANC)

- ANC is the amount of acid that can be neutralised by a rock sample and mostly relates to the amount of carbonate minerals within that sample.
- ANC is commonly measured by the amount of acid consumed when a crushed rock sample is added to a known quantity of concentrated acid.
- ANC is best measured in units of kgH_2SO_4/t so that it can be directly compared with MPA.
- ANC values for rocks are commonly between 0 and 200 kgH_2SO_4/t, although highly carbonaceous materials such as limestone have higher ANC values (>200).
- ANC analyses are commonly combined with MPA analyses for interpretation.
- Rocks with less than 1 kgH_2SO_4/t should not be relied on for ANC.

Net acid production potential (NAPP)

- Net acid production potential (NAPP) is the calculated difference between MPA and ANC, i.e. NAPP = MPA − ANC, and is reported in units of kgH_2SO_4/t.
- Positive NAPP values more than 10 kgH_2SO_4/t are likely to be PAF, while negative NAPP values less than 10 kgH_2SO_4/t are likely to be NAF (see also Table 1).

Net acid generation (NAG)

- Net acid generation (NAG) testing assesses net acid generated during accelerated weathering. Specifically, a crushed rock sample is oxidised with hydrogen peroxide to release acid that can react with the neutralising minerals in the rock.
- Titrations (for acidity and pH measurements of the NAG solution are used to quantify the acid-producing potential).
- Net acid generation potential is measured in kgH_2SO_4/t and the end pH of the NAG solution is also measured.
- Samples with a NAG capacity of >10 kgH_2SO_4/t or NAGpH > 4.5 are defined as PAF.
- Samples with a NAG capacity of 1–10 kgH_2SO_4/t and NAGpH < 4.5 are defined as PAF – low capacity.
- Samples with NAGpH > 4.5 are defined as NAF.

There are some important limitations in the applicability of NAG analysis, and potential for false-positive interpretations with some samples from coal mines (Pope et al. 2010). These limitations are discussed in further detail in Appendix C.6).

Paste pH

- Paste pH is a field-based analysis that provides an indication of the readily soluble acidity in crushed rock and is commonly used as a qualitative tool to identify and manage areas that are already acidic.
- This analysis is carried out by mixing a crushed rock sample in a 1:5 volume ratio with deionised water and measuring the pH immediately.
- A sample with a pH less than 4.5 indicates the rock has elevated stored soluble acidity and may contribute acid rapidly during weathering.
- Paste pH is not an indicator of dynamic acid contributions from rock because it does not analyse those components that require long-term exposure to air and water to release acid.

Carbonate bomb

The carbonate-bomb field test is not widely used. It combines a powdered rock sample with concentrated acid to release carbon dioxide (CO_2) gas. The pressure of gas in the bomb is used to estimate the carbonate mineral content of the sample (Appendix C.6). The carbonate bomb can be used as a field method to assess the distribution of rocks that contain carbonate minerals and ANC of rocks.

Interpretation of acid–base accounting analyses

MPA, ANC and NAG analyses are the primary analyses that are useful in determining the acid-forming status of rock samples. Table **1** provides a generalised guide for the preliminary interpretation of ABA results in relation to the acidity of the collected rock samples.

Table 1 Summary to preliminary interpretation of ABA results

	MAPP (MPA-ANC)	NAG
>10 kgH2SO4/t	PAF (strongly acid producing)	PAF (strongly acid producing)
1-10 kgH2SO4/t	PAF (moderately acid producing)	PAF (moderately acid producing)
0-1 kgH2SO4/t	Low – no acid producing	Low – no acid producing
-1-0 kgH2SO4/t	Low – no acid neutralising	
-1-10 kgH2SO4/t	NAF	
<-10 kgH2SO4/t	NAF (potentially acid neutralising)	

Commonly, ABA data are plotted with NAG values on a *y*-axis and NAPP or the ratio of MPA/ANC on an *x*-axis (Figure **11** and **12**). Graphs of this type divide samples between the four quadrants of the diagram and into fields that are acid-producing, non-acid-producing, and uncertain. Samples that

plot as uncertain usually do so due to interference in the analytical method or a breakdown of the assumptions underlying the test method (see Appendix C.6). The geological description of the sample (Appendix C.4) is often the most important piece of information used to interpret samples that plot in the uncertain quadrants.

The results of the ABA are provided on a per-sample basis and need to be considered alongside geological data, particularly the volume and distribution of each different rock type, to determine the potential mine drainage chemistry. These data should be integrated with geological models, the mine plan, including mine scheduling information and kinetic test data (section **3.3.5**), to provide the most complete interpretation. In general, if all samples are NAF or have negative NAPP and have high NAG pH and low NAG acidity values then mining will not produce substantial AMD and the whole deposit would be considered to be NAF (see Section 5: Coal – non-acid forming). If any samples are PAF or have a positive NAPP value then interpretation of the geological data is required to determine the extent of the occurrence of the PAF rock and the overall likely impact – as few as 5% PAF samples or samples with a strongly positive NAPP value could result in the production of substantial AMD if not managed appropriately. If PAF rocks are distributed sporadically and are equalled in quantity by rocks with neutralising capacity (strongly negative NAPP values) then it is unlikely that the mine will produce AMD, although localised AMD issues may still arise as a result of preferential flow of AMD through the rocks and/or AMD seeps. If PAF rocks represent a small but predictably acidic suite of samples and are not balanced by rocks with negative NAPP values then mining of these areas is likely to produce AMD.

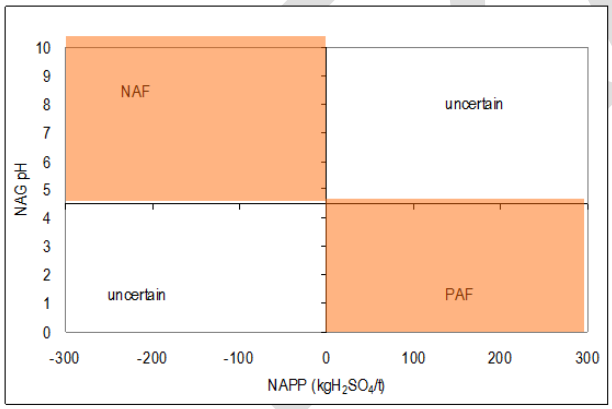


Figure 11: **NAG pH vs NAPP graph. Samples that are acidic according to both NAG and NAPP tests plot in the PAF field, while those that are non-acidic by both tests plot in the NAF field. Samples where one of the analyses indicates acidic and the other indicates non-acidic plot in the uncertain field and require further investigation.**

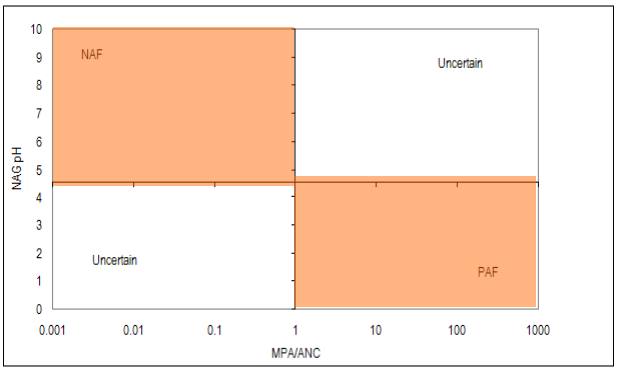


Figure 12: **NAG pH vs MPA/ANC. This plot is interpreted in the same manner as Fig. 11.**

Ongoing monitoring of the rocks disturbed during mining (section **11.2**) will be required on a regular basis to ensure that rocks with implications for mine drainage chemistry (i.e. PAF rocks, or rocks with neutralising capacity) are identified and appropriately handled.

If ABA analyses indicate PAF rocks, read section 5.2 for further description of predicting the chemistry of mine drainage

If ABA analyses indicate NAF rocks, read section 6.2 for further description of predicting the chemistry of mine drainage

3.3.4 Geochemical testing to assess trace element content of rocks

The trace element concentration of rock samples can be determined by a number of analytical methods, including XRF. These methods provide information about the total trace element concentration of the rock but do not provide any information about the form the trace elements are in (i.e. 'speciation'). Trace element speciation can assist in the determination of the mobility or reactivity of trace elements during weathering processes. Trace elements present in reactive minerals such as sulphides and carbonates may be reactive and can be released into solution. In contrast, trace elements in silicates or some oxides are less reactive and less likely to be released in to solution.

3.3.5 Assessing the reactivity of rocks – kinetic tests

Kinetic testing, or rock reactivity testing, provides additional information about rock drainage chemistry over time, in particular trace element chemistry of mine drainage, lag-period preceding acid generation, and rate of acid or alkalinity generation. Specifically, kinetic tests are designed to assist in the prediction of changes in mine drainage chemistry with time. These changes occur because the rates at which reactive minerals such as sulphides and carbonates weather is variable. In general, kinetic tests expose a rock sample to laboratory-simulated weathering or field-based weathering, and leachate chemistry is analysed frequently. Kinetic tests can be designed to provide information on:

- Sulphide oxidation rates, acid generation and mine drainage evolution trends
- Carbonate reactivity and alkalinity generation
- Lag periods prior to acid-producing or neutralising reactions
- Trace element concentrations likely to be present in mine drainage
- Effectiveness of mine drainage management methods
- Optimisation of management procedures

Kinetic tests should commence on selected samples of interest during initial geochemical assessment of a site as they are real-time tests and provide critical information to facilitate optimal mine site management, in particular waste-rock management and effective mine drainage treatment. These tests are required if initial ABA analyses or multi-element analyses indicate that the mine drainage requires treatment, or sometimes to demonstrate no treatment is required.

Kinetic test hierarchy

Kinetic test analysis is commonly phased throughout mine life where the scale and complexity of the test work that can be completed are determined by the availability of sample material (Figure **13**). Early in resource development, samples are only available from drill cores and laboratory-scale testing can be completed including kinetic NAG testing, standardised leach column testing/humidity cell testing. At some sites field leach column testing can be completed if enough drill core is available. Once mine operations commence, larger scale kinetic testing can be completed and this might include trial waste rock dumps with alternative capping materials or neutralising agents added. As mining progresses and waste rock dumps are constructed, kinetic test work could include lysimeters and instrumentation inside the waste rock dump, as well as full-scale capping and rehabilitation trials.

Each phase of kinetic test work reduces the chemical uncertainty related to mine drainage chemistry and volume at closure. At some mine sites a minimal approach to kinetic testing might be appropriate because predicted water quality is relatively benign or there is a high degree of certainty from early predictions. At other mine sites a comprehensive approach to kinetic testing is required either because uncertainty related to water quality is large or because the risks related to water quality are high if predictions are imprecise. Kinetic testing can become progressively more complex as larger amounts of waste rock become available; however, there might be cause to use simple kinetic tests at any time during mine life, depending on the information required.

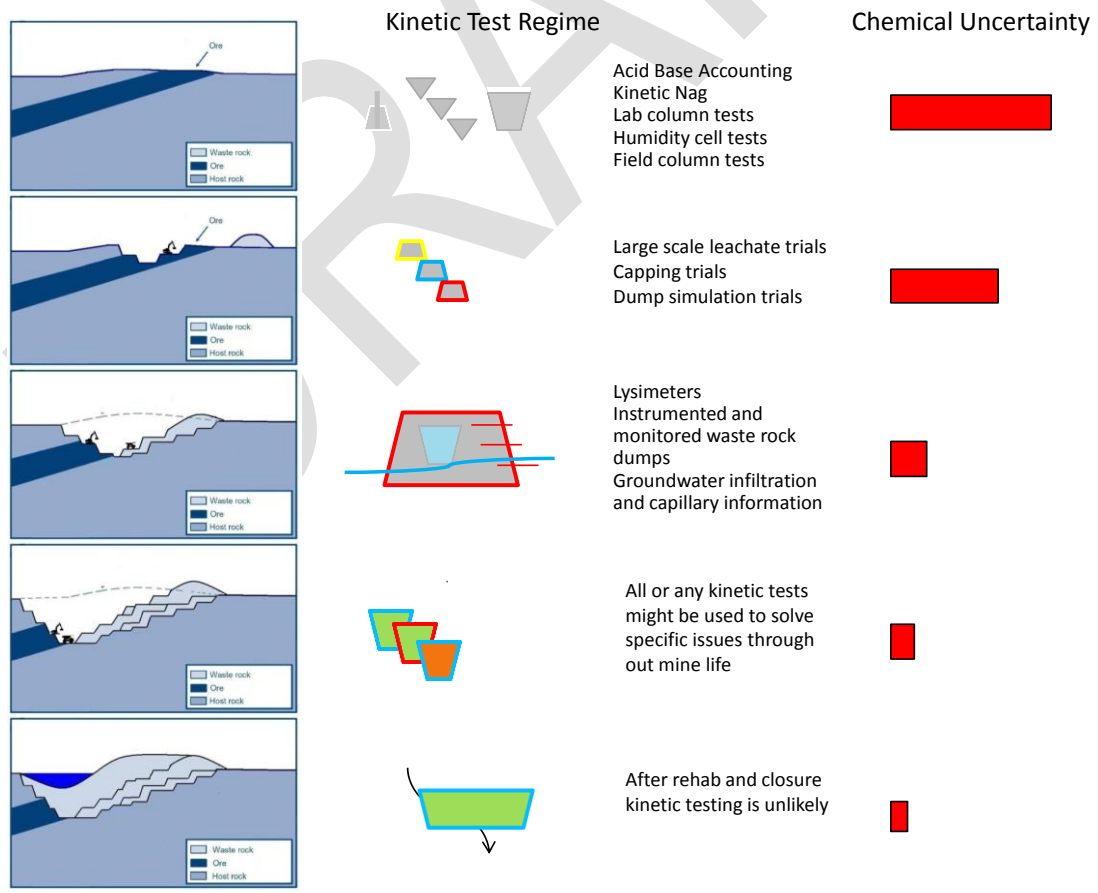


Figure 13: Schematic representation of the hierarchy of kinetic testing.

The value of kinetic testing is related to the certainty it provides for mine drainage chemistry at closure and after closure. There are several limitations to the application of small-scale kinetic testing (column tests, humidity cells, etc.) that are progressively removed by use of larger testing regimes. The limitations mostly relate to:

- Grain size (column tests are conducted on crushed samples)
- Reactive surface area (waste rock might contain many large clasts with low reactive surface area compared with a crushed rock sample)
- Oxygen availability (oxygen can be progressively consumed with depth into a waste rock dump)
- Water ingress (water ingress can be substantially reduced by compaction in a real waste rock dump compared with a column test)
- Secondary minerals (secondary minerals can precipitate and might vary in their distribution in a real waste rock dump compared with a column test).

At the point kinetic test work reflects monitoring the geochemical conditions in full-scale waste rock dumps then the results should accurately reflect mine conditions and the only source of uncertainty becomes forward projection of current results through time.

In, New Zealand, published kinetic test data from both field and laboratory column trials are available from coal and mesothermal gold-bearing rocks (e.g. Pope et al. 2011; Kerr et al. 2013; Pope & Weber 2013). In addition, large-scale dump monitoring data are available from some coal mines (Weber et al. 2013). These datasets and papers provide generic information on the rates of acid and trace element release from coal and gold mines. There are additional datasets in reports held at regional council for epithermal gold deposits. While none of these datasets are adequate for site-specific studies at future mine sites, they provide a guide on interpretation and limitations of kinetic test data. In addition, they provide a base for future studies and some background information that can be built upon by site-specific studies rather than repetition/duplication of previous tests.

Only general characteristics related to the acid and trace element release from waste rock disturbed by coal and gold mining in New Zealand are available to date (see below). This is an active research area.

PAF coal rocks

- Commonly Brunner Coal Measures.
- Acid release is rapid and under fully oxidising conditions could be 90% complete in 2-10 years.
- Lag periods can be present prior to acid release; however, these are generally short because acid-neutralising capacity in these rocks is generally low.
- Oxygen ingress into waste rock dumps can be limited (5-10 m) and therefore acid production from waste rock dumps is likely to be lower than predicted by assuming excess oxygen.

NAF coal rocks

- Commonly all other coal measures in New Zealand.
- Commonly there is some acid release, but this is neutralised by the relatively high ANC of these rocks.
- Some trace element release - sub ppm levels of Zn or As.

Mesothermal gold

- Limited acid release from ore rocks only; generally waste rock is strongly acid neutralising overall because of carbonate minerals in the metamorphic mineral assemblage.
- Release of As +/- Sb from ore rocks.

Epithermal gold

- Either strongly-acid-forming or acid-neutralising rocks are common.
- Lag periods to acid production are common because carbonate minerals are common in alteration system.
- Diverse suite of trace elements can be enriched including Mn, Fe, Cu, Zn, Pb, Hg and others.

3.4 Prediction of water quality downstream of a mine

Site-specific factors such as dilution, natural alkalinity, acidity, baseline or background concentrations of dissolved components, and historical mine drainages will impact on the water quality at a proposed mine. To predict water quality at a point of interest downstream of a mine (e.g. consent compliance points), information is required on the hydrogeology and water chemistry and how these parameters change with time at the proposed mine site. This information can then be integrated with predicted chemistry and volume of potential mine drainage to predict the water quality downstream of a mine.

Site hydrogeology and background water quality information is integrated with information on mine drainage to predict downstream water quality using reactive transport modelling (Figure **14**). Reactive transport modelling can be used to predict downstream water chemistry because reactive components (both acid and neutral) are present in stream waters and mine drainage. This means the prediction of downstream water chemistry based on only dilution ratios of different mine drainage components is inadequate. Required information may include:

- Alkalinity, acidity and pH
- Dissolved oxygen
- Dissolved Fe^{2+} and Fe^{3+}, Al
- Fine-grained (colloidal) particulate Fe^{3+} and Al minerals
- Major cations and anions (Ca, Mg, K, Na, SO_4, Cl, HCO_3)
- Trace elements
- Redox conditions

Reactive transport modelling requires specialist knowledge and should be completed by appropriately qualified and experienced personnel. A comprehensive overview of reactive-transport models has been completed by Mayer et al. (2003).

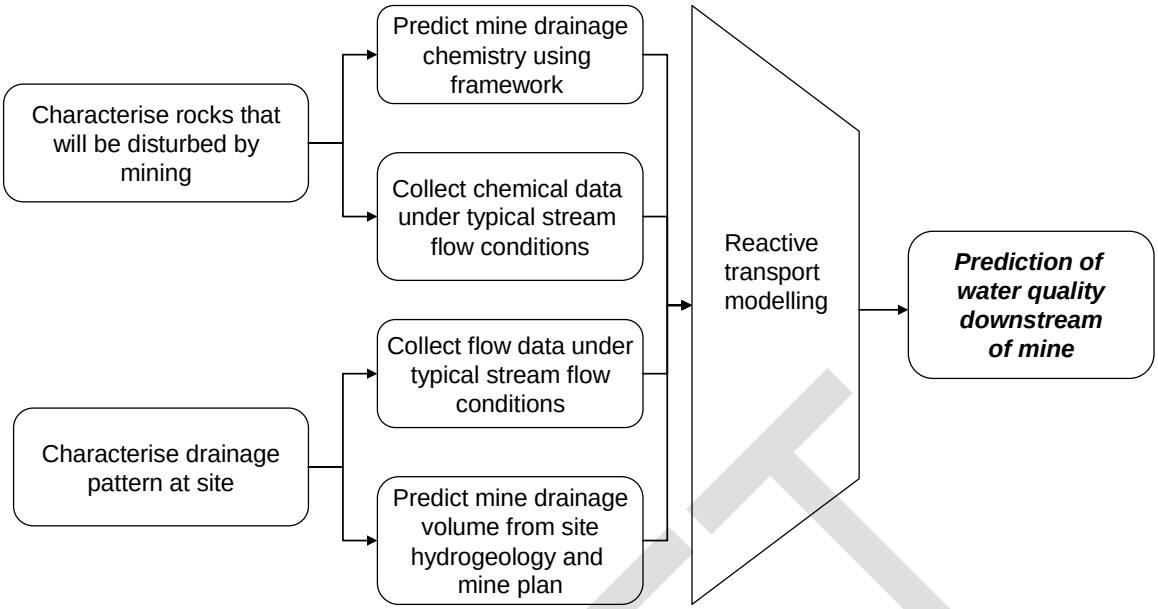


Figure 14: **Basic process for determining water quality downstream of a mine.**

The likely mine drainage chemistry can be determined using information inputs described in subsequent chapters (5–9), while collection of relevant site hydrogeological and background water quality is outlined below.

3.4.1 Site hydrogeology

Site hydrogeological data for predicting downstream water quality are similar to those used for determining baseline hydrogeology (section 2.3.1), but include a projected value of mine drainage volume. The volume of mine drainage relates to the type of mining (opencast vs underground), mine scheduling, the area of disturbance, as well as hydrogeology. Predictive models should aim to produce a site water balance model. These models include water use and storage, rainfall, evapotranspiration, surface flows, groundwater contributions, snowfall, evaporation/ablation, and how these inputs change with time and season. These models are best completed by a suitably qualified specialist and are often developed and refined during the life of a mine.

A very conservative mine drainage flow at an opencast mine could be calculated by assuming that 100% of rainfall or stream flow for the area of disturbance becomes mine drainage.

3.5 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide.

Step 1. Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing

- Identify resources for rehabilitation ☐
- Develop sampling strategy ☐
- Identify number of samples required ☐
- Field survey including drill-core samples ☐

Step 3. Geochemical testing to determine acid-forming or acid-neutralising capacity of rock samples

- Conduct acid–base accounting analysis ☐
- Determine potential neutralising capacity of rocks ☐

Step 4. Management of suspended solids ☐

Step 5. Predict mine drainage chemistry from geochemical tests (Chapters 6-10) ☐

Step 6. Predict stream water chemistry

- Identify downstream point of interest ☐
- Predict mine drainage chemistry ☐
- Analyse background water chemistry, site hydrogeology and historical mine drainages ☐
- Complete reactive transport modelling ☐

Step 7. Determine the potential ecological impact on the stream ☐

Step 8. Consider whether impacts are acceptable, agree rehabilitation outcomes ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

4 SUSPENDED SOLIDS

4.1 Introduction

In any mine operation, suspended solids will occur in a significant portion of mine discharge, due to the earthworks involved. For some mine types, suspended solids and associated turbidity issues will be the primary environmental issue. The amount of total suspended sediment (TSS) is site specific and varies with geology and climate-controlled factors such as rainfall and the presence of areas that can generate dust. Suspended solid concentrations can be qualitatively predicted with geological information (Craw et al. 2008; Druzbicka & Craw 2011, 2012), but these relationships require additional research for particular mine settings and rock types. Difficulties in accurately predicting suspended solid concentrations mean that TSS management must be proactive. This includes on-site management of surface water, and possibly treatment of mine discharges. Unsealed mining roads may also be a significant source of TSS to aquatic systems, and thus require management. When mining operations commence, suspended solid loads should be monitored to assess the adequacy of treatment systems.

4.2 Water quality

Suspended particulate material in a water sample can include many different entrained mineral or rock fragments, viable and non-viable organic material, and precipitates formed within the stream. Suspended particulate material in a stream can be quantified by several parameters using various analytical methods (Lewis & McConchie 1993).

- Total suspended solids (TSS) refers to the mass of non-filterable material in a water sample and is measured by mass differences on filters. TSS concentrations are reported in milligrams of suspended solids per litre of water (mg/L).
- Turbidity refers to the influence of suspended material on light transmission or scattering from a sample and can be measured by:
  - light transmissivity methods, either electronic or Secchi disk
  - light-scattering methods with a turbidity meter in NTU (nephlometric turbidity units).
- Clarity refers to how far you can see through the water and is measured optically by looking at a standardised object through a tube of sampled water.

Total suspended solids is the preferred method of measurement for suspended solids as optical methods may be confounded by other parameters such as dissolved organic carbon.

In general, suspended sediment loads are at a minimum during sustained periods of base flow and increase naturally during high rainfall events as particulates are swept into streams.

Clay minerals are the most common components of turbidity, and the finer the grain size of these mineral particles, the longer the settling of the particles will take. Clay-rich turbidity typically settles after about 4 days if the water remains still. This settling rate can be enhanced with flocculent chemicals that cause the fine particles to aggregate.

A simple field test aiming to assess the approximate turbidity level at a mine site without the need for lengthy laboratory analyses has been developed, as shown in Figure 15 (from Druzbicka & Craw 2011). The squares and lines were printed on white paper, the paper cut into manageable smaller pieces (5 lines/shades per piece) and mounted onto a 'handle' for easier manoeuvrability. The test strip is viewed through 450-ml plastic cups filled with the turbid water. Correlations between this field method and laboratory-measured NTU units are presented in Table **1**.

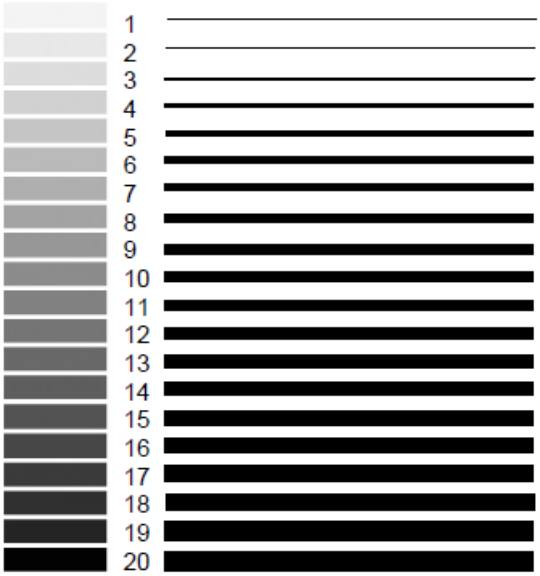


Figure 15 Field turbidity measurement strip for use in conjunction with turbid water in 450-ml clear plastic cups (see Table 2).

Table 2 General correlation between field turbidity estimation chart and NTU units of water in 450-ml clear plastic cups (after Druzbicka & Craw 2011).

Line/shade	Approx. corresponding turbidity, NTU
1	0-10
2-5	10-30
6-15	30-60
16-20	60-100
>20	>100

4.3 Biological effects

Impacts on aquatic ecosystems arising from high turbidity are largely physical in nature, such as smothering of benthic organisms and reduction in light penetration. Impacts arising from high turbidity and resulting deposited sediment can be considered as direct or indirect. Direct effects on an organism include smothering of food resources such as algae and organic matter, benthic organisms (Figure **16**) or eggs of some species, and clogging of the gills in fish. Excessive deposited sediment also reduces habitat for stream life. In contrast, indirect effects include reduction in primary production (algal growth) due to decreased light penetration, and changes in predator-prey relationships due to prey species being hidden to predators. A number of reviews on the effects of sediment in aquatic systems have been undertaken in New Zealand (Ryan 1991; Crowe & Hay 2004; Reid & Quinn 2011) and a more detailed overview of the effects on fish is also provided in Cavanagh et al. 2014).

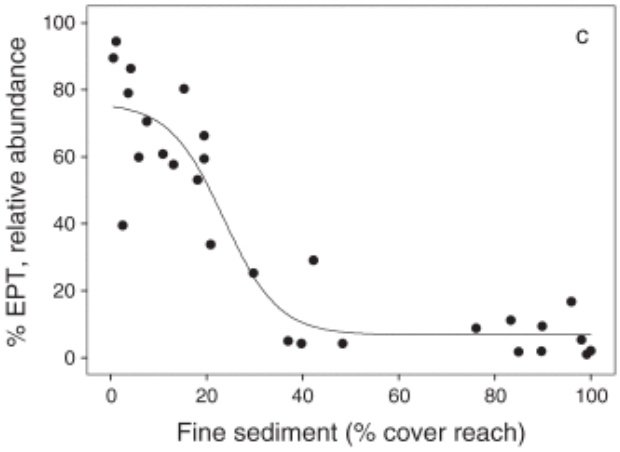


Figure 16 **Fine deposited sediment cover exceeding 20% cover on the bed has been shown to negatively affect stream invertebrates (i.e. mayfly, stonefly and caddisfly taxa) (from Burdon et al. 2013).**

Further, washed sediment can also degrade terrestrial areas. For example, sediment can smother short vegetation in both rehabilitated and natural sites, create sites where weeds establish, and degrade the soil resource. Sediment movement is associated with unstable sites, although site stability is a prerequisite for successful re-vegetation. For pasture sites sediment can seal or cap the soil, reducing infiltration (further exacerbating erosion) and inhibiting seedling establishment.

4.4 Operational management/treatment and prevention

Total suspended solids are expected to be elevated at all mine sites and management of TSS is routinely undertaken on mine sites, often using sediment traps, wetlands or active treatment using flocculants and thickeners (see below). Best management practices to prevent or reduce high TSS will be more cost effective than ongoing treatment of mine drainage. In particular, waste-rock- management techniques can help minimise the formation of TSS in mine drainage, although further treatment may also be required. The efficacy of TSS management options in place may vary, and therefore ongoing monitoring, particularly during the early stages of mining, is a critical part of management.

4.4.1 Waste-rock management

Stopping sediment generation and movement is the most effective form of management. This approach both conserves the soil resource and minimises impacts of sediment. Techniques for preventing sediment generation include the following.

- Keep clean water clean, by using cut-off drains to prevent water moving into and over areas of unstable soils.
- Minimise the area that may generate sediment. Delay stripping vegetation, particularly during winter.
- Create erosion-resistant landforms by manipulating slope length and slope steepness and catchment size. Focus on the erosion-prone components of the landscape where resources are limited.
- Creating rough surfaces with micro-relief by using tines or ripping across the slope. Rough surfaces slow water velocity and trap sediment. The microsites facilitate vegetation establishment.
- Spread organic material (logs, mulches, woody slash) from mining-related clearing across the surface. Forestry waste or agricultural materials (straw, hay) can be distributed over exposed areas to reduce water velocities and trap particulates.
- Surface with inorganic materials (gravel, rock, geotextiles).
- Rapid re-vegetation – establishment of plant cover on exposed materials provides protection from erosion. Direct transfer of intact sods is the most effective method of re-vegetation. Hydroseeding and seeding into mulches also encourage rapid vegetation.

Techniques for minimising elevated TSS at mine sites relate to management of surface water and include:

- Sediment traps – a variety of techniques, including the use of straw bales, check dams and sediment ponds, often used in conjunction with surface water diversion (see also Utah Oil Gas and Mining 2000)
- Effective drainage network to control the volume and velocity of surface water flows
- Sediment control fencing – temporary fine netting (shade cloth) fences or straw bales can be used in low-water-flow zones to slow water velocities and catch particulates
- Regrading - reducing the slope length and angle to reduce the velocity and volume of runoff
- Effective maintenance of sediment traps and ponds. Such sediment may be suitable for recycling and reuse for plant growth media.

Local topography and climate influence the effectiveness of each technique, and therefore the combination of strategies that may ultimately be used at any particular site.

4.4.2 Tailings management

Mine tailings are residues from any processing of the ore rocks or coal on the mine site. Processing might include crushing, washing, screening, separation and metallurgical processing on a gold mine site, or washing and screening at coal preparation plants. Mine tailings are typically discharged from a processing plant as water-rich slurry, and are accumulated in settling ponds or behind a dam so that the solids will settle. Commonly water forms a lake on the surface, and can either be recycled through the plant or discharged.

Water can pass through the tailings and seep to the surface below a dam. Treatment to remove the elevated dissolved solids may also result in the formation of suspended solids that also need to be removed. The most appropriate form of management will be specific for different mine types; for example in coal mines, a large portion of the TSS may be coal fines, which float and thus require different treatment to solids that may be removed through settling. Once the process is in place, suspended solids, including Fe oxyhydroxides, may develop, and a settling pond is necessary to allow removal. This is especially important at gold mines, where the Fe oxyhydroxides may have elevated As and Sb adsorbed to particle surfaces.

Fine tailings may also be dewatered and disposed of either in tailings piles or mixed with waste rock (co-disposal). Some of these materials may be useful for supplementing plant growth material. Laboratory testing and plant growth trials should be used to confirm the suitability of tailings or tailings mixtures as plant growth media with required amendments. This latter technique creates a more stable deposit than tailings or waste rock alone and can potentially remove the need for a tailings dam thereby decreasing the overall footprint of the mine waste facility.

Fine tailings can also be dewatered and used to line landfills or to make low-grade fuel such as briquettes (Williams 1990; National Research Council 2002), although these techniques have not been used in New Zealand to date.

4.4.3 Treatment

Waste-rock management options to prevent suspended sediment (section B.4.1) should be integrated as necessary into mine discharge planning prior to release into any receiving environments. However, it is likely that management will be insufficient to mitigate suspended sediment, and treatment will be required. In the first instance, settling ponds to capture TSS in mine discharge prior to release to any receiving environments should be considered as a bare minimum. However, in some instances, active treatment will be required. This could occur if very fine dispersive clays are present; geological factors that control characteristics of suspended sediment are currently being researched (Craw et al. 2008). As such, the effectiveness of settling ponds in removing turbidity should be closely monitored during the early stages of mining. If high turbidity levels are found, then testing can be undertaken to determine the characteristics of suspended particulates. This can include:

- Determining the settling velocities of particles entrained in a water column
- Grain-size analysis, through sieving or by laser and optical methods
- Flocculant efficacy testing

Power availability and land area are the key parameters determining whether active or passive systems are appropriate for use (Figure 17 Flow chart to select a treatment system for total suspended solids (TSS).
). Active systems are typically used where there is limited land area and power is available. If there is considerable land area available on a powered site, passive systems (settling ponds) could be used instead. If active treatment is considered, flow variability and predicted TSS specific gravity are used to determine which processes may be appropriate to use.

Details of active and passive treatment options are provided in Appendix B.

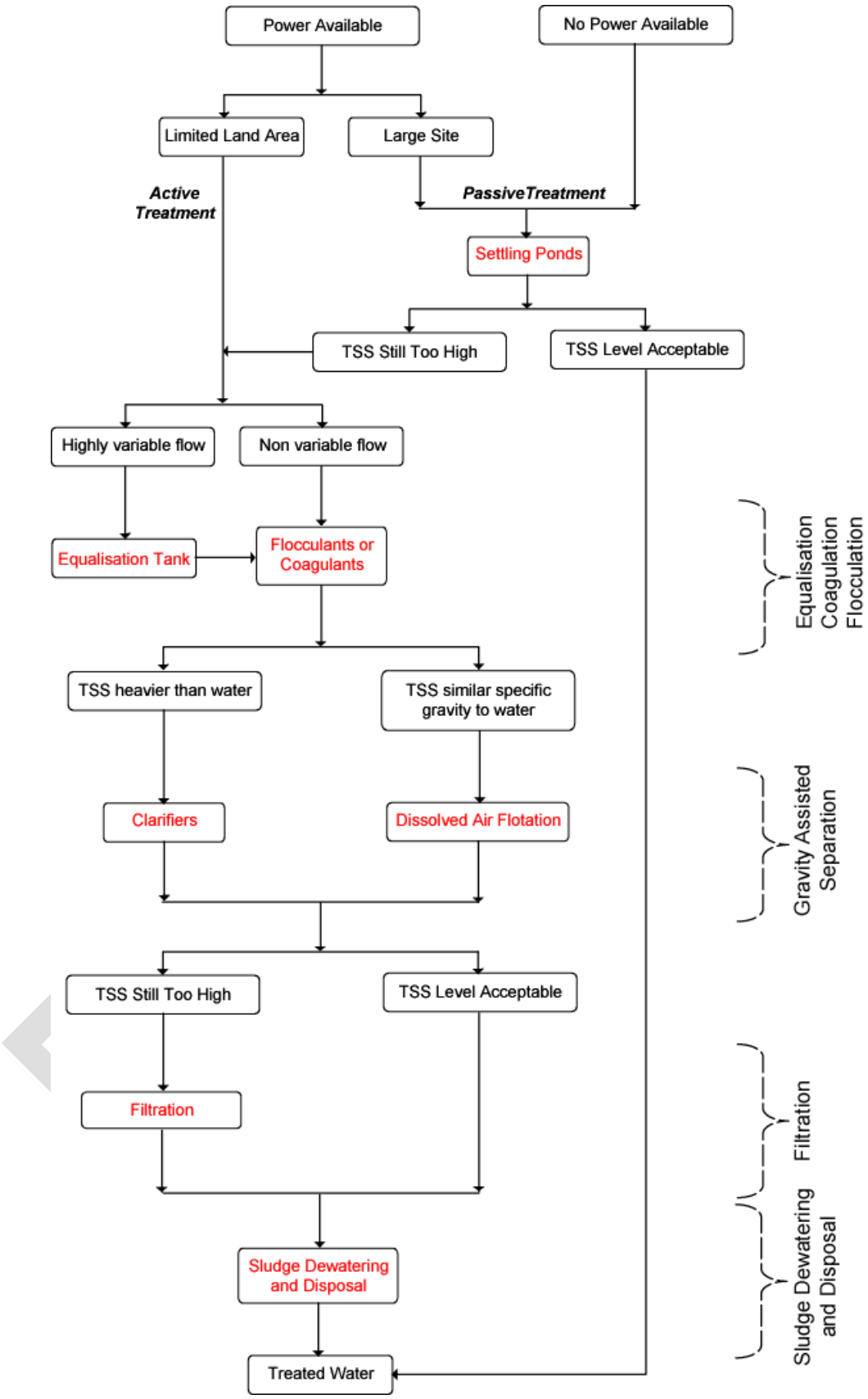


Figure 17 Flow chart to select a treatment system for total suspended solids (TSS).

4.5 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide.

Step 1. Background and Baseline information	☑
Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing	
Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples	☐
Step 4. Management of suspended solids	☐
Step 5. Predict mine drainage chemistry using geochemical testing	☐
Step 6. Predict stream water chemistry at downstream point of interest	☐
Step 7. Determine the potential ecological impact on the stream	☐
Step 8. Consider whether impacts are acceptable, agree rehabilitation outcomes	☐
Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes	☐
Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes	☐

5 COAL MINES WITH POTENTIALLY ACID-FORMING (PAF) MATERIAL

5.1 Introduction

Surface runoff and seepage from disturbed areas of mines in PAF regions can be a major source of AMD and adversely affect the receiving environment around these mines (Eary 1999; Nordstrom & Alpers 1999; Skousen et al. 2000; Ezpana et al. 2005). For opencast mines, the disturbed areas are usually large and include mine pits, exposed coal seams, mine facilities, disturbed waste-rock material, coal stockpiles, access roads and ditches. For underground mines, the disturbed areas largely correspond to the coal measures exposed in the mine walls, ceiling and floor as well as the area where oxygenated water penetrates due to modified groundwater profiles. Oxidation of sulphide minerals such as pyrite in these disturbed areas can produce drainage with low pH and elevated concentrations of dissolved metals such as Fe, Al, Mn, Ni, Zn and sulphate (Rose & Cravotta 1998; Cravotta 2008a, b).

Equations 1–4 demonstrate the sequences of reactions of pyrite (FeS_2) with water and oxygen to make sulphuric acid (H_2SO_4). This reaction can also release trace elements that are either associated with pyrite or within minerals that are dissolved by sulphuric acid.

$$FeS_2\,(s) + {}^{7}/_{2}O_2 + H_2O \rightarrow \quad Fe^{2+} + SO_4^{2-} + 2\,H^+$$

$$Fe^{2+} + {}^{1}/_{4}O_2 + H^+ \rightarrow \quad Fe^{3+} + {}^{1}/_{2}H_2O$$

$$Fe^{3+} + 3H_2O \rightarrow \quad Fe(OH)_3(s) + 3H^+$$

$$FeS_2 + 7/2\,H_2O + 15/4O_2 \rightarrow \quad Fe(OH)_3(s) + 2\,H_2SO_4$$

Opencast mines in PAF (and NAF) regions can also be a source of suspended solids that can adversely affect the receiving environment around these mines (Shapely & Bishop 1965). The high concentrations of TSS consist mostly of sand and clay particulates and coal fines (Osterkamp & Joseph 2000), but also include precipitates formed from reactions between AMD and surface waters. High TSS concentrations cause elevated turbidity in aquatic environments.

5.2 Predicted water quality

5.2.1 Mine drainage

The compositions of AMD from PAF rocks and other PAF mine waste materials at coal mines have some similar characteristics:

- Low pH (typically 2-4)
- Enriched dissolved Fe and Al concentrations (typically 1 to >200 mg/L)
- Enriched trace element concentrations (typically 1-3 orders of magnitude greater than background)

However, specific seeps have AMD of varying chemistries so the interaction of each with the wider catchment is also variable and subsequently will result in different levels of environmental impact. In addition, selection of optimal treatment or management techniques (section **5.4**) will depend on a detailed understanding of the mine drainage chemistry.

From an investigation of AMD from mines located in the PAF Brunner Coal Measures on the West Coast, several parameters that influence AMD chemistry have been identified (Pope et al. 2006, 2010). In particular, the following parameters were identified as being important:

- Mine type (opencast or underground)
- Hydrogeology (above water table or below water table)
- Local variations in rock type (mudstone rich or sandstone rich).

Using these parameters, the likely mine drainage chemistry associated with specific mine types within Brunner Coal Measures in different locations can be estimated (Figure 18). There are insufficient data to construct a hazard model of PAF rocks in other areas; however, similar relationships are likely and these water quality predictions can be regarded as broadly applicable to other PAF rocks. Therefore the chemical composition of AMD produced by the Brunner Coal Measures can be used as a guide for that of AMD arising from other sequences of PAF rocks (Figure **18**: Potential water chemistry from PAF coal measures.

). Equation 2

Equation 3

Equation 4

The type of proposed mine (opencast or underground) influences the ratio of dissolved Al to Fe in mine drainage, which in turn influences the likely downstream water chemistry. If rocks disturbed by mining are kept below the water table then AMD chemistry is much less acidic because reactions that involve oxygen proceed more slowly. This interpretation is speculative for opencast mines.

Trace elements in mine drainages from PAF Brunner Coal Measures typically include Fe + Al ± Mn > Zn > Ni > other trace elements (Cr ± Co ± Cu) (Pope & Trumm 2014). Concentrations of Mn up to about 10 ppm occur, of Zn up to about 5 ppm and Ni up to about 1 ppm. Concentrations of other trace elements seldom reach above hundreds of parts per billion.

Most trace elements, except Fe and Al, behave conservatively (remain dissolved) in untreated AMD. There are several processes that can attenuate metals in a downstream environment and the impact of these processes on trace element concentrations varies depending on site-specific conditions. Two of the main processes are adsorption onto iron oxyhydroxide and organic complexation. Generic modelling of these processes indicates that they are not important at pH<7 (Pope & Trumm 2014).

Where AMD is likely to form, the oxidation state (form of the chemical) and level of oxygenation (dissolved oxygen concentration) in mine drainage are important because these determine the types of treatment activities that can be undertaken (section **5.4.2**). These parameters can only be broadly predicted. In general, the concentration of dissolved

oxygen (DO) and ratio of Fe^{3+} to Fe^{2+} dissolved in mine drainage from opencast mines is higher than in underground mine drainage because the availability of atmospheric oxygen is greater at opencast mines. However, site-specific factors such as presence of small waterfalls, thickness of waste-rock sequence, and reaction time for AMD formation influence the oxidation and oxygenation state substantially. In general further analysis (e.g. kinetic testing) and consideration of site-specific factors will be required to quantitatively predict the DO concentration and the ratio of Fe^{3+} to Fe^{2+}.

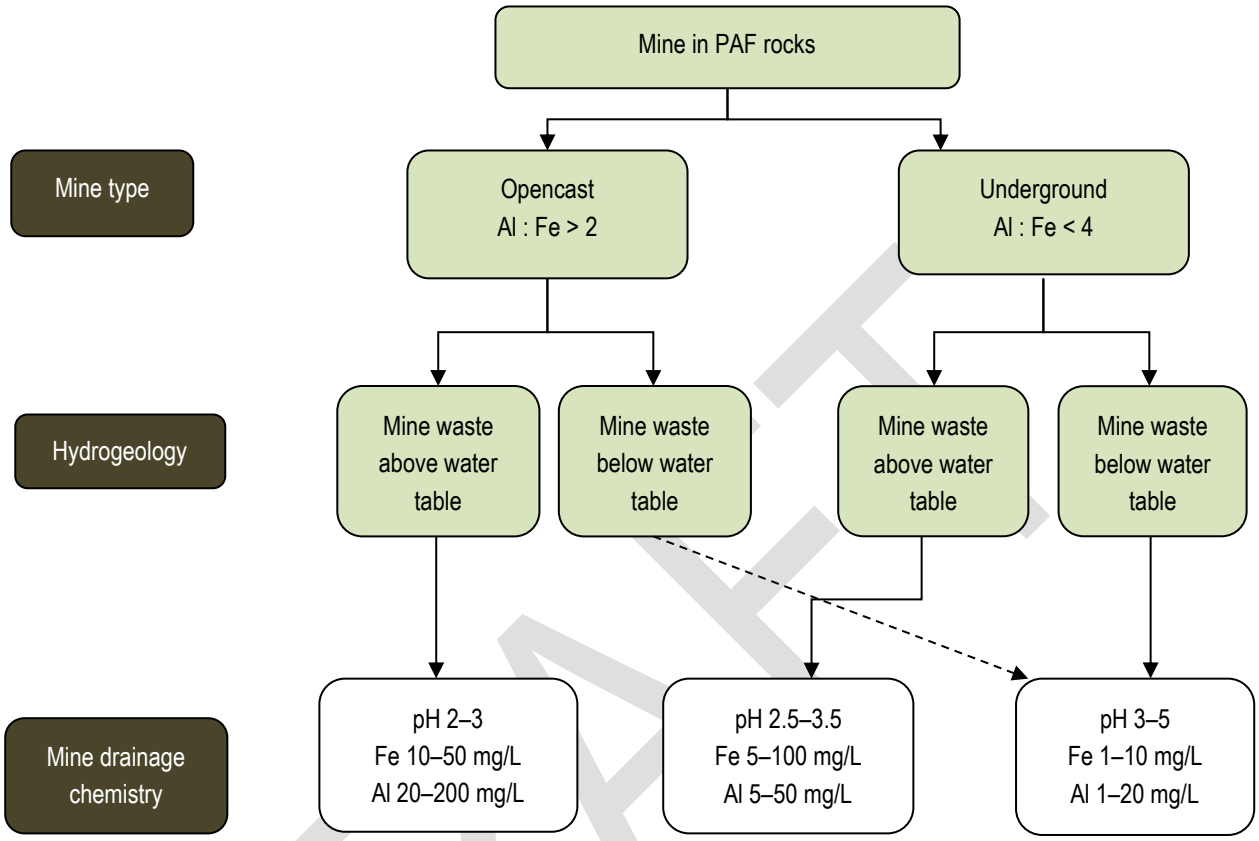


Figure 18: Potential water chemistry from PAF coal measures.

All predictions of AMD composition relate to the chemistry that can be found at mine drainage seeps. There are several limitations to the level of detail that can be predicted because mine drainage chemistry varies due to site-specific factors and evolves with time as reactive components within the rocks are consumed. Kinetic testing of rocks from specific mine sites can be useful to refine predictions of mine drainage chemistry and mine drainage evolution (section 3.3.5; Appendix C.7).

Total suspended sediment is generally expected to be elevated at mine sites, and management of TSS is routinely undertaken at all mine sites. The amount of TSS is site specific and varies with climate-controlled factors such as rainfall, the presence of areas that can generate dust and geotechnical properties of the rock, and the variable efficacy of the TSS management options in place. Coal fines may form a high proportion of TSS, and pose particular management challenges as they are less dense than sediment and can be difficult to remove when suspended. As there is limited predictive capability, TSS concentrations should be mitigated where possible and managed as mining operations commence. Further detail on the mitigation and management of TSS is provided in Appendix B.

5.2.2 Downstream water quality

Prediction of water quality downstream from a PAF mine site requires integration of data on the chemistry of the AMD, site hydrogeology and water quality of any potential receiving environment as outlined in Figure **14** (section 3.4). The likely mine drainage chemistry can be determined from Figure **18**: Potential water chemistry from PAF coal measures.
, while collection of relevant site hydrological and background water quality data is outlined in section 2.3.

5.3 Predicted ecological impact

Once water quality has been predicted, the ecological impact associated with that water quality can also be estimated. Coal mines with untreated AMD will have the most severe impact on streams. High acid production from mining areas leads to water characterised by low pH and elevated concentrations of metals and sulphate, while root zones affected by acid rock prevent plant establishment and growth.

The impact of AMD on stream biota may be influenced by the natural water chemistry. For example, on the West Coast there are numerous brown-water streams. These brown-water

systems are associated with podocarp rainforests and are widespread from Nelson to Fiordland. They are characterised by dissolved humic and fulvic acids that can reduce pH to less than 4.0. These naturally acidic streams contain diverse macroinvertebrate and fish communities, indicating some taxa pre-adapted to tolerate low pH. For example, toxicological experiments indicate that mayfly populations from naturally acidic streams are more tolerant of low pH than those from circum-neutral streams. However, pre-adaptation to low pH does not necessarily preclude impact from mine drainage as streams impacted by AMD typically contain high concentrations of metals (See Appendix D for further discussion). The abundance of naturally acidic streams on the West Coast and the consequent adaptation of endemic species mean that international water quality guidelines such as ANZECC & ARMCANZ (2000) are not appropriate when identifying and setting water quality targets for mining impacts for such regions. A combination of biological survey data (from the West Coast) and toxicity experiments using West Coast invertebrates has been used to establish the potential impacts outlined below (Figure **19**). Six general outcomes are illustrated and described briefly below. The pH limits described are reasonably well defined, while the metal limits are based on Fe and Al concentrations only and are qualitative. Further details about these outcomes, and the toxicological effects of selected metals, are provided in Appendix D.

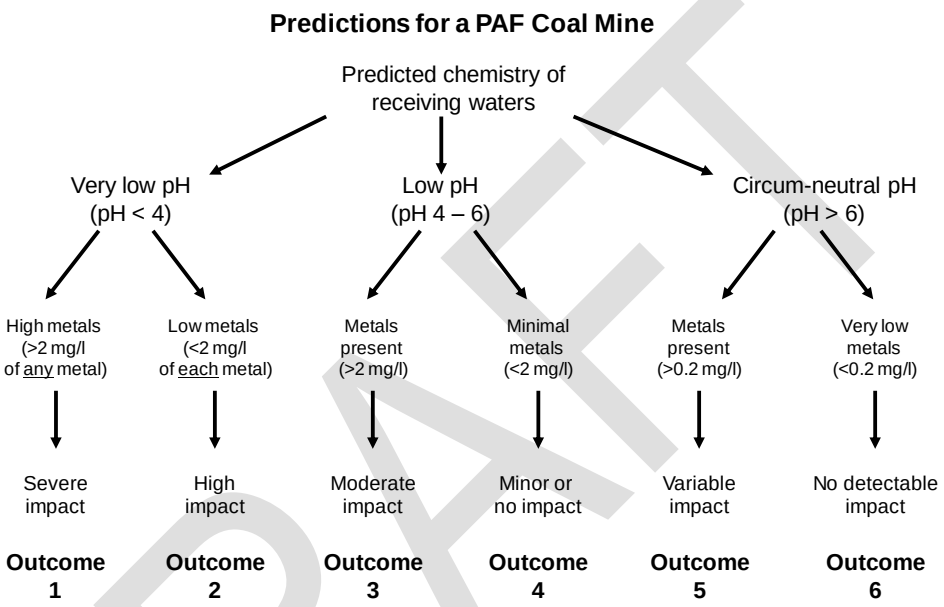


Figure 19 **Potential ecological outcomes arising from a PAF coal mine on the West Coast. Metal limits are dissolved metals and refer to the sum of Fe and Al concentrations.**

Relatively little is known about the responses of Southland stream invertebrates to acidification. In Southland, naturally occurring acidic streams are rare, so invertebrates are unlikely to exhibit the pre-adaptation to low pH observed in West Coast streams. Consequently, AMD impacts are likely to be more severe for a given water chemistry than those predicted for West Coast stream invertebrates. While the ANZECC water quality guidelines (ANZECC & ARMCANZ 2000) may be more relevant to Southland streams because of the lack of naturally acidic streams, guideline values for some relevant metals (e.g. Fe) are missing. More information on the toxicological effects of selected metals is provided in Appendix D.2.

Outcome 1

The most severe impact on stream ecosystems occurs when water is highly acidic (pH < 4) and has a high concentration of metals. No New Zealand fish can survive for long in such water. Few macroinvertebrates, of very limited diversity, will be found. Algae and microbes, however, may be present, and even in high abundance in some cases. These communities tend to be dominated by a few taxa that are able to tolerate the stressful conditions. Remedial and treatment strategies will be essential.

Outcome 2

Streams with very low pH usually have high metal concentrations as well, but there can be cases where acidic streams have low metal levels. These streams would also be highly impacted, although not as severe as Outcome 1. More algal and invertebrate taxa can be found, although fish would still be rare or not found. Some moss can be found in such streams as well. Remedial and treatment strategies will be essential.

Outcome 3

Streams of moderate acidity and high metal concentrations will usually have a moderate impact on aquatic life. Although some fish can tolerate the acidic conditions, dissolved metals may be toxic to them depending on concentrations. Similarly, invertebrates in these streams can suffer from either the low

pH or the elevated dissolved metals. At pH above 4, Fe can be present as Fe precipitates, which, along with other metal precipitates, can impose stress on fish and invertebrates, either through a reduction in habitat quality or food supply. Algal and plant (moss, macrophyte) diversity can be moderately high, but their biomass and activity can vary depending on deposition of metal precipitates. Remedial strategies are likely to be required.

Outcome 4

Many streams on the West Coast have pH in the range 4–6 because of natural humic and fulvic acids (often called tannins). Streams affected by mines in this pH range can still support high diversity and abundance of aquatic life if the metal concentrations are low such that no toxicity occurs. Fish, invertebrate, algal, plant and microbial diversity can all be high and comparable with those in pristine streams. Exceptions may occur at the low pH range (close to 4) or in cases where metal precipitates or sediments impair habitat in the stream. Remedial strategies are unlikely to be required.

Outcome 5

Streams with water around neutral pH clearly are not affected by acidity; the extent of impact will be dependent on metal concentrations but streams on the West Coast and in Southland are less likely to have toxic concentrations of metals (see also Coal NAF chapter) and to our knowledge, there are no examples of this outcome in these regions.

Outcome 6

Waters of neutral pH and very low metal content should support a full diversity and abundance of aquatic life for the area. Natural features of the catchments could affect some biota, such as waterfalls blocking migratory fish species. Mining still could affect stream habitat if turbidity or sedimentation (from mining operations) were present. Otherwise, species and food webs should be comparable with pristine streams in the area.

A key aspect of this framework is that explicit 'acceptable' water quality criteria are not established; rather, a robust scientific basis for this decision to be made by end users is provided. See also Chapter 11.

5.4 Operational management and treatment

Options to prevent or minimise impacts of AMD on streams and to enable recovery of mined areas include management practices to prevent or reduce the formation of AMD, and treatment of AMD. Best management practices to prevent or reduce the formation of AMD can reduce environmental risk and be more cost effective than ongoing treatment of AMD discharge. AMD prevention is achieved by identifying PAF, separately stripping it, and placing it so oxygen and water respectively cannot oxidise or leach ARD. Prevention may also include co-disposal with alkaline materials. Management strategies may include proactive treatment of drainage water from acidic waste rocks. Treatment of the drainage water may still be required even where optimal management strategies have been put in place; however, treatment costs will be lower than if no mitigation strategies had been in place and may also allow for the use of passive instead of active treatment.
In situations where AMD is already present (from previous mining activities), preventative management techniques are limited. In such cases it is much more likely treatment will be required. Section **5.4.3** covers sites with existing AMD.
Once management techniques have been initiated or treatment commenced, monitoring of any discharge from the site is necessary to verify management/treatment efficacy. Water quality parameters and frequency of sampling for different treatment systems, as well as biological monitoring, are covered in Chapter 8.

5.4.1 Operational management

Prevention and mitigation of AMD can be achieved through risk-based planning and the mine design approach applied throughout the mine life cycle. Prevention is primarily accomplished in the assessment and design phases. In general, more options and more effective options are available earlier in the mine life (Figure **20**). The structural nature and physical environment of the AMD source material influences selection of the most appropriate method (or methods) for prevention and mitigation.

By necessity, the strategies used to prevent or minimise AMD will be site specific, and will typically involve a combination of different methods. A thorough understanding of the site conditions is required to identify site-specific opportunities and constraints.

More than one method sequentially, or a combination of methods at any one time, may be required to achieve prevention or mitigation of AMD, and different methods may be applicable during different stages of the mine life.
As such, continuous geological monitoring during mining is required to ensure potentially-acid-forming rocks are rapidly identified, segregated and appropriately managed (see section **11.2.1**).

The extent of monitoring required will depend, in part, on how well a site has been assessed initially, and the geology, including the location of potentially-acid-forming rocks. Systems for collecting leachate from waste rock (or tailings) piles should be put in place during early pile construction.

Regular water quality monitoring of the leachate from waste rock or tailings piles is required to ensure that the selected management techniques are effective.

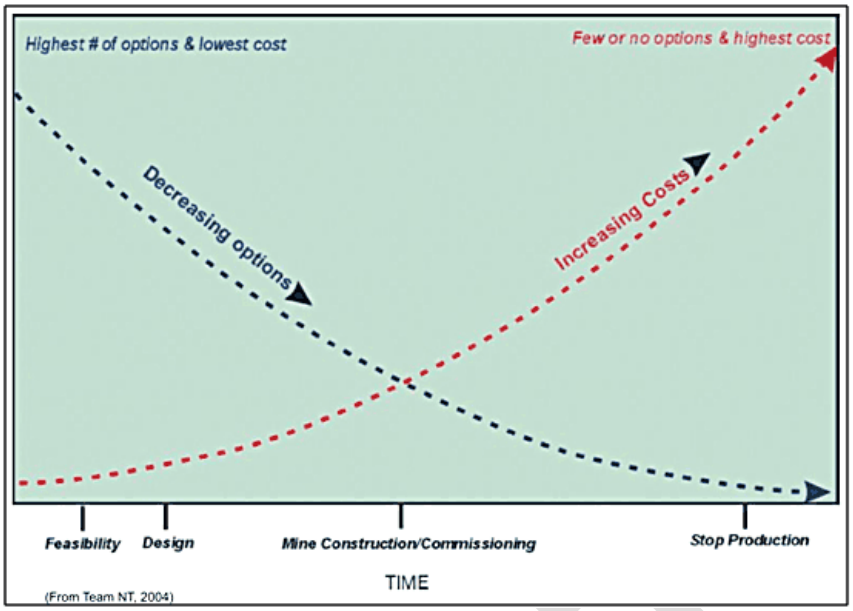


Figure 20: Options and effectiveness with time (TEAM NT 2004, in INAP 2009).

Good mine planning is critical for the success of preventative and mitigation measures, which can significantly reduce the formation of AMD.

Waste rock, tailings, coal and coal rejects, open or filled pits and exposed high walls with fractured rock, or underground mine structures may all be sources of AMD. Minimising the formation of AMD can be achieved by one or all of the following:

- Avoid disturbing the PAF materials.
- Prevent or reduce contact of water and/or oxygen with PAF materials, both temporally and spatially.
- Neutralise or reduce the concentration of contaminants present in mine drainage.

Several techniques are available to achieve each of these strategies, and this section provides an overview of best-practice management techniques, with more detail provided in MEND (2001), DITR (2007), INAP (2009) and Appendix E. In addition to these, many of the strategies described for minimising TSS (Chapter 4) can be used. The degree of success of any minimisation strategy can be measured by ongoing monitoring of mine drainage (section **11.2**) with dynamic feedback on strategy performance escalated to management to ensure that required strategy changes are identified, budgeted and implemented when required.

Avoidance and prevention

The most effective AMD management technique is to avoid exposing or disturbing PAF materials. At the most extreme, this could mean that a particular coal deposit is not mined. However, this does not mean that mining cannot go ahead but rather that the mine plan may need to be altered. For example, total or partial reduction in excavation or exposure of problematic materials can limit or prevent sulphide oxidation and metal release. Geological survey data and mine planning will determine if avoidance is a practical option. For example, if PAF rocks are present below coal seams, they can be left in place and/or covered. Alternatively, a decision may be made not to extract a particularly reactive rock type that will be too difficult to manage in the future. This may require changes to the mine design to work around difficult rock types through alteration of mine access, inclines, and open-pit designs. Avoidance also involves planning to ensure placement of waste storage facilities avoids sensitive receiving environments. If avoidance is not practical, other mitigation strategies will be necessary to minimise AMD formation.

Water management methods

Water acts as a solute transport mechanism and a reactant. Water management is often the most cost effective method of minimising AMD. The key aim is to reduce infiltration into areas such as waste-rock or tailings piles containing acid-forming materials and thereby reduce the volume of impacted drainage or potentially the solute load. Water management techniques can be implemented during the construction phase as well as at later stages of mining operations. Techniques that have been specifically developed for controlling AMD include:

- Diversion – is one of the easiest and cheapest methods for minimising the volume of acidic leachate. Unpolluted surface runoff is drained away from PAF materials, including PAF rocks in high walls, as rapidly as possible via ditches or pumping. Covers may divert water and limit infiltration through waste-rock piles and are discussed more below.
- Dewatering - involves lowering the water table to reduce the amount of groundwater in contact with PAF materials. Examples include pit dewatering to reduce seepage through pit walls and shallow groundwater collection ditches above tailing ponds and waste-rock dumps.
- Flooding - of underground or surface mine voids minimises AMD production by inhibiting the oxygen supply from

reaching PAF materials. However, initial flooding may result in the release of acidic discharge arising from the dissolution of stored oxidation products as in a final void pit lake.

- Hydrogeological controls – is primarily applicable for controlling groundwater flow. For example, placement of low-permeability materials such as fine 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 (under fully saturated conditions).
- Seals – are primarily used when decommissioning underground mines to prevent or minimise AMD production. Hydraulic seals limit movement of air and water through mine workings.

Special handling methods

Special handling primarily relates to excavated waste rock and tailings materials and is often the first step in an AMD management plan. Special handling involves the identification, through initial site characterisation (section **3.3**) or subsequent rock monitoring (section **11.2.1**), and typically segregation of PAF, NAF and, if present, alkaline materials.

It is essential that mine waste handling is incorporated into the mine plan to ensure that PAF materials are appropriately managed.

There are a number of special handling techniques that can be adopted to *minimise* acid production from PAF materials, although complete prevention of AMD is unlikely and treatment may still be required. Special handling techniques include:

- Encapsulation and layering – involves the placements of PAF and NAF and/or acid-consuming materials (e.g. carbonates) in geometries that control or limit AMD production. If acid-consuming materials are used, effectiveness is governed by availability, type and reactivity of acid-consuming materials, the balance between acid-forming and acid-neutralising materials, deposit geometry, the nature and flow of water through the deposit, potential chemical armouring of alkaline materials and potential passivation of pyritic materials.
- Blending - mixing of waste-rock types of varying acid-forming and acid-neutralising potential to create a deposit that generates a discharge of acceptable quality. Effectiveness depends on the availability and grain size of materials, the mine plan, geochemical properties, reactivity of waste-rock types, flow pathways created within the deposit, and the degree of mixing/method of blending – with thorough and homogeneous mixing generally required to achieve maximum benefit. Blending is a form of alkaline addition (see below) but refers to the use of waste-rock materials on-site. Successful blending techniques generally require a relatively high ratio of acid-neutralising to PAF materials (>3:1).
- Co-disposal of tailings and waste rock – can take several forms depending on the degree of mixing. This includes the homogeneous mixing of waste rock and tailings, alternate layering of waste rock and tailings, the addition of waste rock to tailings dams and the addition of tailings to waste-rock piles. Co-disposal can reduce the formation of AMD by limiting the flow of water and oxygen to reactive surfaces by filling the voids between waste-rock particles with finer tailings particles. Whether co-disposal is appropriate includes consideration of the waste production schedule and proportions of waste rock to tailings. Other advantages are physical stability of tailings piles, minimisation of disposal footprint, elevated water table within deposits, and possible elimination of the tailings dam.

Further details are available in MEND (2001), DITR (2007), INAP (2009) and SMI (2011).

Additions and amendments

Several addition and amendment methods are available to reduce the formation of AMD, although the addition of alkaline materials is the most common approach. This addition of alkaline materials provides some control of the pH of any leachate by neutralising some or all of the acid generated by PAF rocks. Alkaline materials or amendments may include waste rock, pH amendment of tailings, placement of alkaline material above or below wastes as liner or cover materials (layering), and alkaline injection. The amount of neutralising (alkaline) material required to fully neutralise the acid materials will depend on the acidity and amount of the waste rock and the neutralising capacity of the alkaline material. ABA and kinetic test data (sections **3.3.3** and **3.3.5**), in combination with the quantity, scheduling and distribution of the different rock types, can be used to determine the amount of neutralising material required (Appendix E.3). It should also be noted that preferential flow paths and scale-up effects can reduce the effectiveness of alkaline blending techniques.

Waste-rock management

Waste-rock management is an important part of minimising AMD primarily through minimising infiltration of water and air. Further, it is an integral component of rehabilitation as waste-rock piles can ultimately become the final landform and appropriate management is required to ensure that the root zone is suitable for plants. Management may include rock pile design and the use of dry or wet covers. Rock pile design largely relates to the slope angle and length of the rock pile such that it minimises permeability and erosion and is appropriate for the climatic conditions. Restriction of the airflow into a rock pile can be achieved by co-mingling of waste rock and fine material (see Co-disposal above). Appropriate water collection systems should also be put in place at the base of the rock piles to characterise the quality of the leachate (see section **11.2.2**).

Dry covers are typically earthen, organic, NAF rock wastes or synthetic materials placed over PAF rock wastes, to significantly reduce oxygen diffusion and/or water infiltration and permeability. NAF and root zone must be deep enough to protect the capping layer from root penetration. Some NAF materials are suitable by themselves, or amended, as rooting

media. Favourable root-zone NAF have physical properties that store and supply adequate water and oxygen for plant roots. Such NAF also provides sufficient anchorage for selected plants.

Common dry covers are:

- Soil and organic material – soil covers are designed to limit infiltration and oxygen ingress. Biologically active organic materials may also consume oxygen, or chemically promote reducing conditions or bacterial inhibition. Designs are site and climate specific and often limited by availability of materials. For example, store and release covers for infiltration control are often used in arid to semi-arid parts of Australia, but may have little application under the high rainfall conditions often experienced in New Zealand.
- Alkaline covers and other neutralising material - can increase alkalinity of infiltration, thereby providing pH control, but AMD prevention is unlikely unless the sulphide content is extremely low. The main limitation is the volume of material required to provide adequate retention time, especially in high-rainfall climates.
- Synthetic liners – low-permeability liners can be used to maintain saturated conditions in the overlying waste or to protect underlying groundwater resources, and can dramatically reduce infiltration. Compacted NAF overburden and/or imported materials are also used to create low-permeability ($<10^{-8}$ ms^{-1}) layers. Such layers must be protected from erosion and root penetration.
- Vegetation stabilises soils and root zones against erosion, and promotes evapotranspiration of water retained in the soil cover. In climates with an excess of evapotranspiration over rainfall the volume of water entering and moving through PAF zones is reduced.

If it is not possible to create a protective capping layer, there needs to be sufficient depth that the acid generated by the PAF will not affect plant growth. This means acid must not enter the root zone. The depth of the NAF capping therefore needs to be 'over-thickened', and capillarity of the NAF needs to be taken into account. Toe-slopes are particularly vulnerable to leachate entering the root zone.

Wet covers basically involve the submergence of acid material under water and include:

- Inundation - flooding of underground or surface mine voids with water has the potential to significantly inhibit the supply of oxygen so that AMD production from PAF materials is not a concern. The depth of water over the PAF material is typically 1-2 m and must be sufficient to allow for mixing of the water column and to prevent re-suspension of wastes by wind or wave action. If significant groundwater fluctuations are anticipated, a larger depth of water cover may be required to accommodate these and maintain minimum water cover over the mine waste at all times. Water covers may not be suitable for material that has already appreciably oxidised, due to potential dissolution of oxidation products.
- Partial water cover – acid-generating waste is stored at depth and a small pond in the centre of the tailings impoundment maintains saturation through enough of the waste to minimise oxidation. NAF tailings are used as cover above the level of the pond.
- Wetlands – oxygen-depleted and -reducing conditions at the base of the NAF cover profile are maintained to protect underlying unoxidised material and promote precipitation of existing AMD products as sulphides under relatively low flow rate, low concentration and hence low flux conditions.

Selection of waste-rock management options

Waste rock, tailings and coal rejects, open or filled pits with fractured rock, or underground mine structures may all be sources of AMD. Of these, management of waste rock, both in situ and post-excavation, may be most significant in minimising the formation of AMD. A flow chart outlining the process for selecting appropriate waste-rock management strategies, described above, is shown in Figure **21**; the information outlined should be available from initial geological investigation of the proposed mine site. Important factors include post-mining land use, local topography, climate, rehabilitation layers, waste-rock volume and composition, groundwater conditions, the position of the waste rock relative to surface water and groundwater, and the presence or absence of neutralising material. A thorough understanding of the site conditions is required to identify site-specific opportunities and constraints.

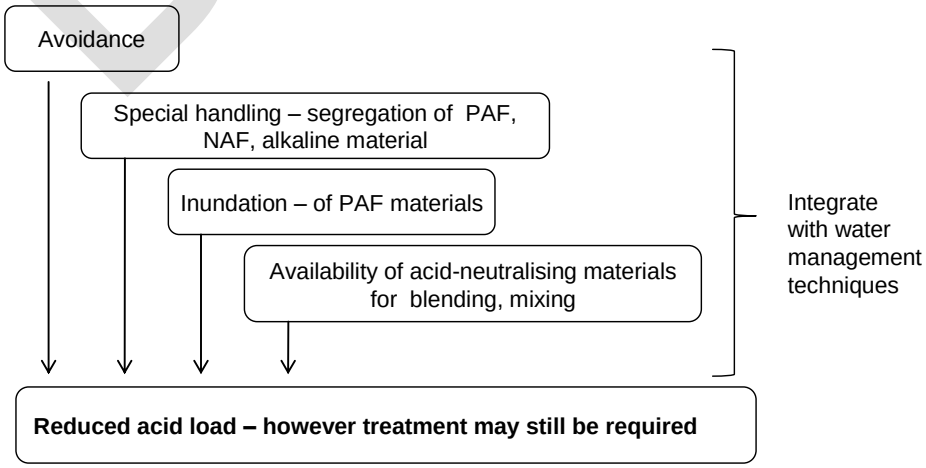


Figure 21: **Prioritisation for consideration of applicability of different waste-rock management strategies. Refer to above text for details on different techniques.**

Landform design is usually driven by mine engineering factors, with land rehabilitation planning and post-mining land uses in harmony with the post-mining landscape design. Ideally landform and land rehabilitation planning go hand-in-hand as designing land rehabilitation following predetermined post-mining landforms constrains land use and ecosystem opportunities. Topography, engineered microsites, access ways (roads, tracks, stock races), high walls, pits, drainage systems (streams, ditches, under-drains), dams, mine lakes, and ponds are components of landforms that determine land rehabilitation options. Slopes, waterways and lakes, and the sequence of placement of materials are key factors that will influence land rehabilitation options. Flat to rolling (<20°) landscapes are desirable for rehabilitation to farmland and some forms of native vegetation (e.g. wetlands). Hilly (21°–30°) and steep (>30°) slopes limit rehabilitation to hill country grazing or shrubland and forest, with associated production, conservation, and recreation land uses. Steeper slopes also require more emphasis on longer-term erosion control measures. Mine lakes and ponds provide wildlife habitat and opportunities for recreational uses (boating, canoeing, swimming, duck shooting, etc.) and water storage for other uses such as irrigation and, potentially, electricity generation. There are international examples of rehabilitating mine pits to gardens, amphitheatres, and sporting facilities. Some pit walls can be stabilised by revegetating with hydroseeding techniques.

Selection of tailings management options

As stated above, tailings may form sources of AMD, thus appropriate management of these materials is required. Mine tailings are residues from any processing of the coal on site, including hydromining and screening or washing to remove coal fines and clay (fine tailings), and screening to remove coarse material. Coarse reject material is typically dumped back into the mine pit and is dealt with in the same manner as waste rock. Fine tailings may be handled in a number of ways (National Research Council 2002; DITR 2007) including:

- Discharged as a slurry into a tailings dam or underground workings.
- Dewatered and
    - disposed of in tailings piles or mixed with waste rock. This material can be directly planted into, or will naturally revegetate if properties are favourable
    - used to line landfills
    - used to make low-grade fuel such as briquettes.

Details on each of these methods are provided below.

Discharge

Fine coal tailings are conventionally discharged from a processing plant as a water-rich slurry, and are accumulated behind a dam so that the solids will settle. The coarse tailings are often used in the dam construction with the fine material being pumped in as slurry. Gravity settling of the fine material results in clear water that can be recycled back into the plant, although problems may occur if the fine particles do not settle or settle slowly and additional treatment is required (see Appendix B.4). The design of tailings dams will be specific to each site as design must take into account pathways for water including coal seams, fractures, and old mine workings. Planning and construction of dams require site-specific details and engineering experience and should be undertaken with consultation from geological, geotechnical, engineering and geochemical specialists.

Problems can occur with tailings dams. Failure of dams can significantly impact the downstream environment and property and cause loss of life (National Research Council 2002; Appendix G). Groundwater and decant water can pass through the tailings and discharge to the surface below the dam. Chemical interaction between water and tailings can lead to high TSS and elevated dissolved solids. Water collection systems are required to intercept any contaminated water from the tailings dam, which is then passed through treatment systems for removal of TSS and trace elements and to raise the pH (see below).

Pumping fine tailings as water-rich slurry into underground workings can be a suitable method of disposal (National Research Council 2002; DITR 2007). There are potential benefits and problems with this method, however. Benefits include reduced surface subsidence over old underground workings if the slurry has some intrinsic strength and can provide lateral support to underground pillars. To achieve this, a cementing agent and possibly coarse waste material can be added. Potential problems with underground disposal include: plugging of pumping systems, incomplete knowledge of available storage space, increasing water flow from underground workings – which then must be managed, and increase of hydraulic head on bulkheads and other barriers in the workings – which can result in blowouts (National Research Council 2002).

Dewatering

Alternatively, fine coal tailings can be dewatered and disposed of in tailings piles or mixed with waste rock and placed in waste-rock piles (Williams 1990, 1991; National Research Council 2002). Dewatering can be accomplished by centrifuge, band press filters, or plate and frame filters. Flocculants are often added but the process is sensitive to pH. The dewatering is expensive both in capital and operating costs and therefore likely to be undertaken only when site conditions, such as limited space/volume for tailings dam, dictate.

Where tailings are disposed of in discrete piles, there may be stability issues as there can still be a significant moisture content, in addition to the risk of spontaneous combustion (where such a risk exists), therefore geotechnical expertise is required to design and manage tailings piles. Coal tailings may contain pyrite that when exposed to air and/or water may result in AMD, and thus need to be managed accordingly (see earlier this section). To decrease the weathering process the following factors should be minimised: time of near-surface exposure, pyrite content, the volume of air, and permeability (Kolling & Schuring 1994). If adequately dewatered, mixing

tailings with waste rock and placing in waste-rock piles can be a suitable method of disposal. However, if pyrite content is high, AMD can be a significant problem. Conditioning to improve the physical properties of tailings through thickening, filtration, compaction, or gradation control can also limit AMD formation. Depyritising of tailings materials at the processing plant has also been used overseas (Canada, Papua New Guinea) as a tailings dam rehabilitation strategy, as has covering tailings with a NAF cover at end of mine life.

Alternative uses of tailings include making briquettes and lining landfills, although these practices are not widely used in New Zealand at present.

5.4.2 Treatment

Appropriate design and management of the disposal of tailings or waste rock can minimise AMD formation (see previous section); however, some seepage may require additional treatment to mitigate potential impacts of AMD on receiving streams. In addition, seepage from dams or adits may also require treatment. If hydromining is used for coal extraction, the water may require treatment after the coal fines have been removed from the slurry.

Treatment can only be undertaken on point-source discharges, and thus requires that effective water collection systems are put in place. If there is a high variability in flow rate it may not be possible to treat all of the AMD and it may be more appropriate to treat only the most concentrated streams (see also Appendix E) Appropriate water collection systems are required for collecting seepage from coal tailings – either dewatered tailings or from dams used to manage tailings (waste-rock piles containing acid materials may produce AMD) – or seeps from dams. Treatment can be accomplished by either active or passive treatment systems (using a variety of techniques) or a combination of both. Further, as mining progresses, treatment may change from one type to the other if economic conditions change, AMD chemistry or flow rate change, or new research leads to better types of treatment systems.

The overall aim of water treatment for acid-forming regions is to raise the pH and lower the concentrations of dissolved metals. Research and practical application of treatment of AMD using both active and passive systems has been undertaken for over 25 years (Ziemkiewicz et al. 2003). Active treatment technologies have largely been borrowed from the wastewater treatment industry and are more certain in terms of predictability of successful treatment, whereas passive treatment is an area of ongoing research. There are few active or passive treatment systems operational in New Zealand at present thus the following is drawn from research currently being undertaken by the research team, and from the literature.

Selection between active and passive treatment

Several factors will influence the decision as to whether to use active or passive treatment. Briefly, if mine drainage exceeds the thresholds provided in Figure **22**, large amounts of neutralising materials are required to ensure appropriate treatment and large passive systems can be prone to failure – thus active treatment is likely to be a better choice.

In general, active treatment systems are more commonly used at operational mine sites whereas passive treatment 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. Internationally this outcome is rare, although in New Zealand there are a number of proposed passive treatment systems for active mine sites. Often passive treatment can be used to complement active treatment as a final polishing step to improve water quality to required regulatory discharge limits.

The main advantages of active treatment systems over passive treatment systems are that they are very effective at removing acid and metals from AMD, have precise process control such that they can be engineered and operated to produce a specific water chemistry, and they can be accommodated at small sites. The main advantage of passive treatment systems is that they are more economic (lower capital, operational and maintenance costs) than active treatment systems (Skousen et al. 2000; Skousen & Ziemkiewicz 2005).

Active treatment systems can be engineered to handle any pH, flow rate, and daily acid load, and last indefinitely, provided the appropriate maintenance is undertaken (Waters et al. 2003). Most passive treatment systems rely on the dissolution of a neutralising material (usually limestone) to neutralise the acidity in AMD and sufficient residence time in the systems is necessary for this dissolution to occur (Skousen et al. 2000). The life expectancy of passive treatment systems is often determined by the mass of neutralising material present compared with the alkalinity consumption rate (Waters et al. 2003).

There are a number of factors that will influence the decision as to whether to use active or passive treatment (Figure **23**). These parameters largely relate to the treatment efficacy of each approach. Details of how each parameter influences the choice of active or passive treatment is provided in Appendix F.1. In New Zealand, significant variations in flow rate at AMD sites are common, due to large storm events and steep catchments. This is more evident in overburden waste rock than in drainages from underground mines. Even with highly variable flow rates, AMD can be treated with passive treatment systems depending on treatment requirements for the site and the effect of flow rate on water chemistry. In some cases, acid load decreases during precipitation events (due to dilution) and in other cases, it increases (due to dissolution of acid-sulphate salts or due to more rapid sulphide oxidation).

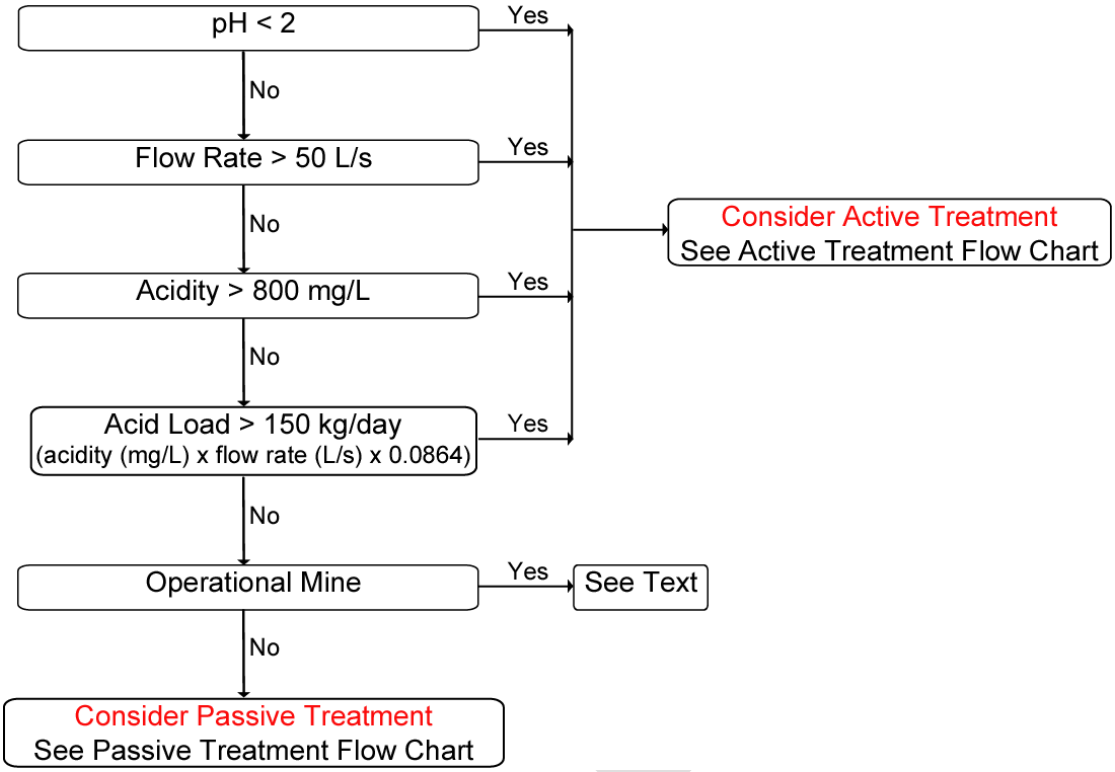


Figure 22: **Flow chart to guide selection between active and passive treatment for AMD (modified from Waters et al. 2003).**

Selection of active treatment systems

If it is determined that an active treatment system is most appropriate, the next decision is which system to choose. A range of factors will initially influence the selection of appropriate active treatment systems (Figure **23**), while efficacy of treatment and particularly costs, primarily of chemicals required, will influence the final selection of treatment system (Appendix F.2). Once an active treatment system has been selected, a computer program such as AMDTreat (Means et al. 2003; Appendix F.4) can be used to design specific components of the system and to determine potential costs.

Active treatment for AMD is largely based on industrial wastewater treatment technologies for which extensive research has previously been conducted (e.g. USEPA 2000, 2004). The main steps involved in active treatment of AMD are pretreatment, dosing with alkali, oxidation, and sedimentation (Younger et al. 2002).

Other active treatment technologies that are occasionally used for AMD internationally, but which are not covered in this document, include sulphidisation, biosedimentation, sorption and ion exchange, and membrane processes like filtration and reverse osmosis (Younger et al. 2002).

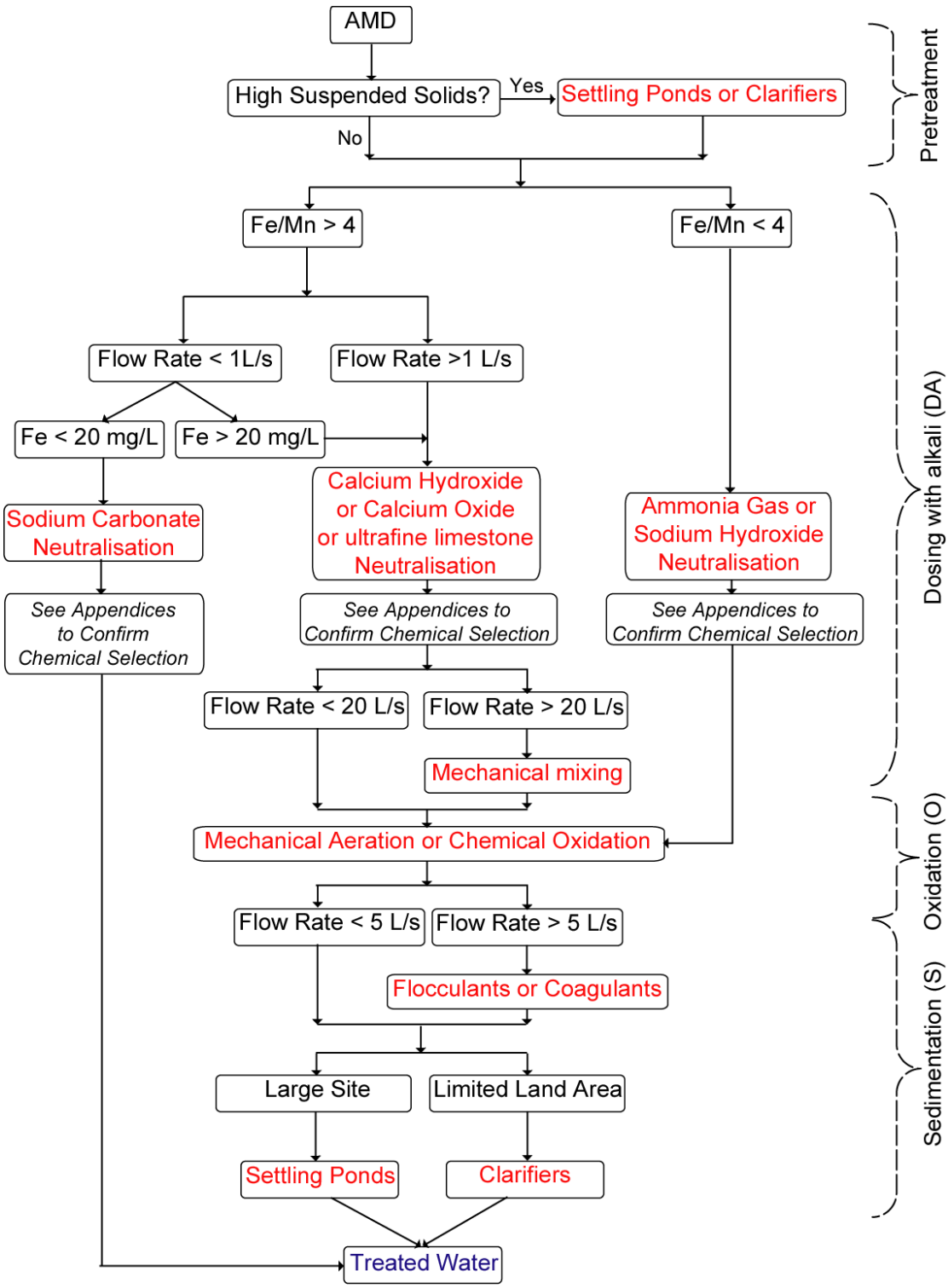


Figure 23: **Flow chart to design a site-specific active treatment system for AMD (modified from Rajaram et al. 2001). The generic treatment step is shown on the side of the diagram. Note for treatment of suspended solids see Chapter 4 and Appendix B.**

Pretreatment

Pretreatment is required where high suspended solids loads (TSS >100 mg/L) are present, which can affect treatment system performance through clogging piping and flumes, and damaging pumps. TSS concentrations are typically reduced through sedimentation techniques and details on active and passive treatment of TSS are presented in Appendix B.

Dosing with alkali (DA)

Dosing with alkali raises the pH of the AMD, which helps to reduce dissolved metal concentrations through the formation of metal oxides. Various chemicals may be used in this step. From a treatment system perspective, the flow rate of the AMD and the concentration of dissolved Fe will influence selection of the chemicals used. However, in most cases a variety of chemicals could be used and other factors – such as costs, ease of use and health, safety and environmental considerations –

will influence the final selection of chemical used. Figure **24** presents a summary of the key benefits, with more details provided in Appendix F.1. An illustration of the costs of active treatment using various chemicals is provided at the end of this section.

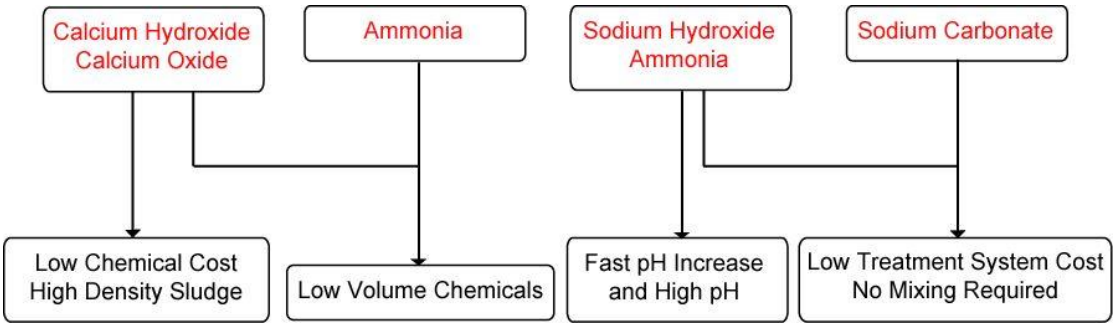


Figure 24: **Summary of the key benefits of the five most commonly used chemicals for the dosing-with-alkali step.**

Oxidation (O)

The oxidation step ensures reduced metals such as Fe^{2+} and Mn^{2+} are oxidised to Fe^{3+} and Mn^{4+} so that they can form hydroxide and oxide precipitates and be removed from AMD (Skousen et al. 2000; Younger et al. 2002). This step may not be necessary if the metals are already highly oxidised through the previous treatment step. Bench-scale tests at the time of system design would be required to confirm this.

Oxidation is typically undertaken using mechanical means, although sometimes chemical oxidation is used. Mechanical aeration techniques include stirring with rotating blades (most common), inline Venturi aeration, trickle filter aeration (water trickling through a tank filled with high-surface-area media and with air bubbled into the water), and cascade aeration (if sufficient land area is available). Appendix F.2 includes some pictures of different oxidation systems.

Chemical oxidants commonly used include hydrogen peroxide (H_2O_2), sodium hypochlorite (NaClO), calcium hypochlorite ($Ca(ClO)_2$), and potassium permanganate ($KMnO_4$; Skousen et al. 1993, 2000). Another potential oxidant is calcium peroxide (CaO_2), which can not only oxygenate AMD but also neutralise acidity (Skousen et al. 2000). Cost, availability and effectiveness are typically used to decide among the various chemical oxidants.

Sedimentation (S)

The final step in the process is sedimentation to remove the metal oxide precipitates formed during the earlier stages of treatment. The methods used include gravity-assisted separation with or without coagulants/flocculants followed by sludge dewatering and disposal. Depending on available land area, gravity-assisted separation is accomplished either with clarification or by using settling ponds or sedimentation ponds (Rajaram et al. 2001). The use of coagulants/flocculants is reserved for higher flow rates (>5 L/s) when residence times in clarifiers or settling ponds can be insufficient for complete metal precipitation. These methods are also used for removal of TSS and are described in more detail in Appendix B. Examples of different sedimentation systems are shown in Appendix F.2.

Disposal of the sludge produced during this step is an important consideration, particularly with regards to the potential leaching of trace elements from the sludge. Standard tests are available to determine potential leaching of trace elements (Appendix F.2). If there is insignificant leaching of trace elements, i.e. the sludge is stable, it may be able to be disposed of directly on site, or to landfill. If there is significant leaching of trace elements, stabilisation of the sludge (Appendix F) may be required. If disposed of to landfill, dewatering of sludge, which typically contains between 1% and 5% solids, will be required, although dewatering may not be required for on-site disposal.

There are limited examples of active treatment available in New Zealand to enable consideration of the relative costs of sludge disposal, although in the United States, sludge dewatering and disposal can be a significant cost of AMD treatment, sometimes exceeding chemical costs by several times (Skousen et al. 2000).

Active treatment system costs

Chemical costs typically dominate the total costs for an active treatment system, and thus will be a significant factor influencing the selection of the chemicals used. Chemical cost will be dependent on the actual cost of the chemical used in addition to the amount required, which in turns depends on the efficacy for neutralisation of the chemical. Figure **25** illustrates the relative operational costs of active treatment using different chemicals. These costs were determined using AMDTreat (Means et al. 2003) and are based on cost of chemicals in New Zealand in 2010 and default parameters for labour and construction costs provided in AMDTreat, although in the examples provided, the latter are typically less than 6% of the total costs over 20 years. Labour and construction costs represent a higher proportion of the total cost for treatment of low-acidity AMD.

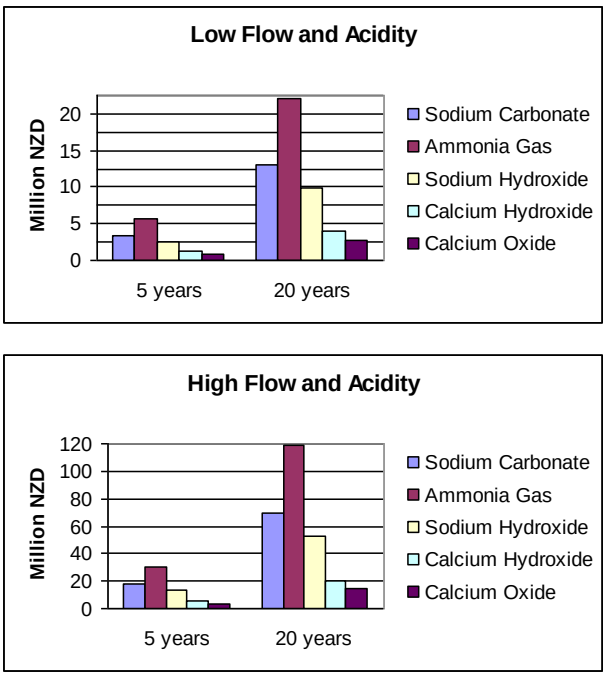


Figure 25: Comparison of potential costs for active treatment of two hypothetical AMDs over 5 and 20 years using the five most commonly used chemicals. These costs were determined using AMDTreat (Means et al. 2003) and are based on cost of chemicals in New Zealand in 2010 and default parameters for labour and construction costs provided in AMDTreat. A low-flow, low-acidity (30 L/s, 500 mg/L) option is compared with a high-flow, high-acidity (80 L/s, 1000 mg/L) option over 5- and 20-year treatment periods.

Selection of passive treatment systems

In the long term, treatment of AMD using a passive treatment system is typically more economic than using an active treatment system; however, careful system choice and design are required to ensure that the treatment system does not fail (Skousen et al. 2000; Skousen & Ziemkiewicz 2005). Potential problems with passive systems include under-design for flow and/or acidity, short-circuiting of flow, armouring of limestone, and plugging with precipitates.

In contrast to active treatment systems, which can be designed to produce a specific water chemistry, there is little control over the final water chemistry from passive treatment systems. The longer residence times required in passive treatment systems for AMD result in a higher final pH, typically between 6 and 7, to ensure adequate removal of metals.

Treatment of AMD using passive treatment technologies can be placed into two broad categories: oxidising and reducing strategies (Trumm et al. 2003, 2005). AMD is generated through an oxidation process that results in the dominant contaminant, Fe, being present in two states, ferrous (Fe^{2+}) and ferric (Fe^{3+}; Singer & Stumm 1970). Oxidising systems remove Fe from the AMD by continuing the oxidation process such that all ferrous Fe is oxidised to ferric Fe, and once the pH has been raised sufficiently, precipitated out of the AMD as ferric hydroxide ($Fe(OH)_3$). For reducing systems, the AMD oxidation process is reversed, such that Fe cations and sulphate are reduced, forming the compounds FeS_2, FeS, and H_2S, thus removing dissolved Fe and sulphate from the AMD.

The choice between the two strategies is typically based on the water chemistry (largely dissolved oxygen (DO) content and ferrous:ferric (Fe^{2+}:Fe^{3+}) iron ratio). The DO content of mine drainage will be unknown for a new mine, although it is likely to be high for an opencast mine and either high or low for an underground mine. Similarly, the Fe^{2+}:Fe^{3+} iron ratio will be unknown for a new mine, although it is likely to be low for an opencast mine and either high or low for an underground mine. Further analysis (e.g. kinetic testing) and good understanding of the site-specific factors will be required to quantitatively predict the DO concentration and the ratio of Fe^{2+} to Fe^{3+}.

For highly oxidised AMD (saturated DO and Fe mostly as Fe^{3+}), the oxidising strategy is most appropriate. Typical treatment systems that employ the oxidising strategy are open limestone channels (OLCs; Ziemkiewicz et al. 1994), open limestone drains (OLDs; Cravotta & Trahan 1999), limestone leaching beds (LLBs; Black et al. 1999), slag leaching beds (SLBs; Simmons et al. 2002), and diversion wells (DWs; Arnold 1991; Anonymous 2001). Open limestone channels and diversion wells typically require a steep topography in order to generate the necessary aeration and to prevent armouring of limestone by metal hydroxides, which can inhibit the dissolution of limestone (Ziemkiewicz et al. 1997).

For reduced AMD (low DO and Fe mostly as Fe^{2+}), the reducing strategy is usually recommended. Typical treatment systems that employ the reducing strategy are anaerobic wetlands (Skousen et al. 2000; Anonymous 2001; O'Sullivan 2005), anoxic limestone drains (ALDs; Hedin & Watzlaf 1994), sulphate-reducing bioreactors (SRBRs; Mattes et al. 2007), and successive alkalinity producing systems (SAPS; Kepler & McCleary 1994), also known as vertical flow wetlands (VFWs) or reducing and alkalinity producing systems (RAPS; Zipper & Jage 2001).

However, site limitations, such as available land area and topography, may limit the use of certain systems.

Figure **26** and Figure **27** provide guides for selecting passive treatment systems for treating discharges with high and low Fe. Where multiple choices for passive treatment are suggested, a review of the potential cost, effectiveness, limitations, and risk of failure for each system should be completed before settling on a final choice (Appendix F.3). A comprehensive description of each of the common passive treatment systems is included in Appendix F.3.

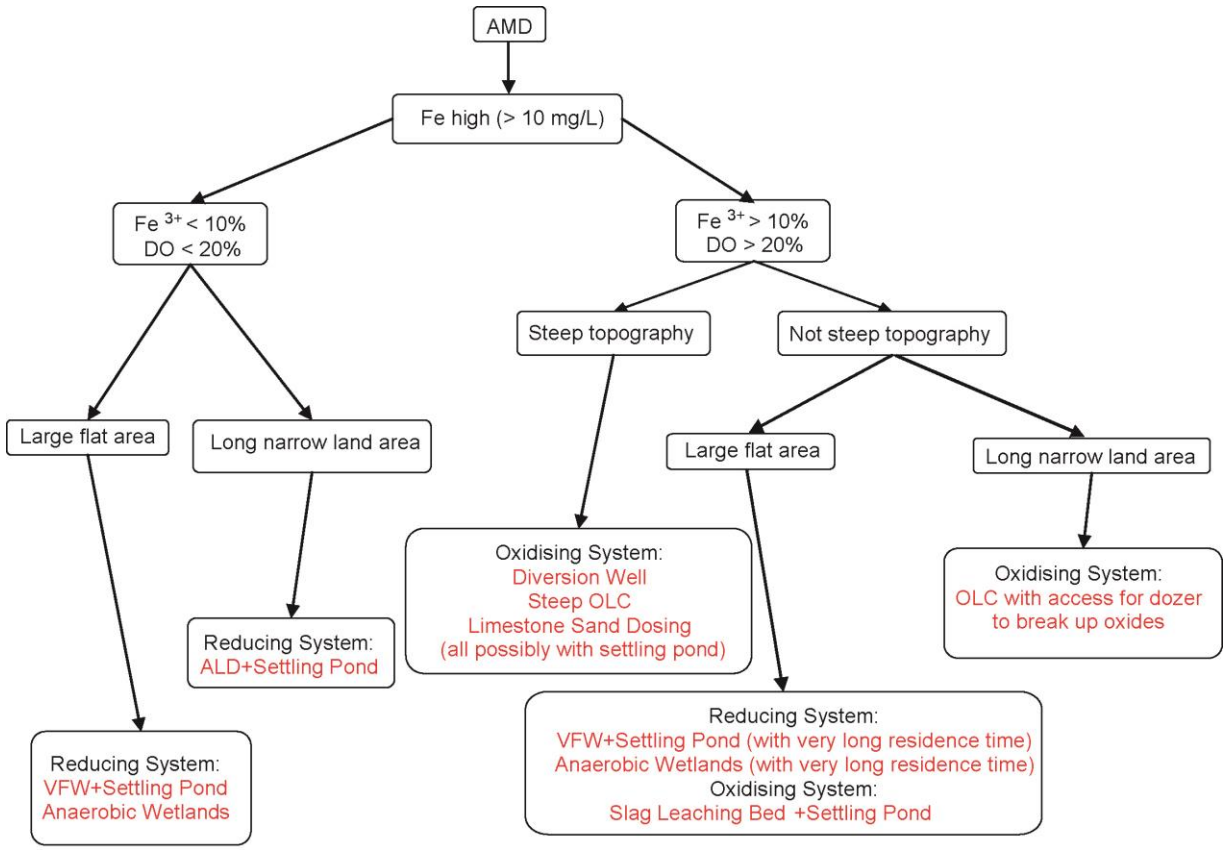


Figure 26: **Flow chart to select among AMD passive treatment systems based on water chemistry (high Fe), topography, and available land area (Trumm 2010).**

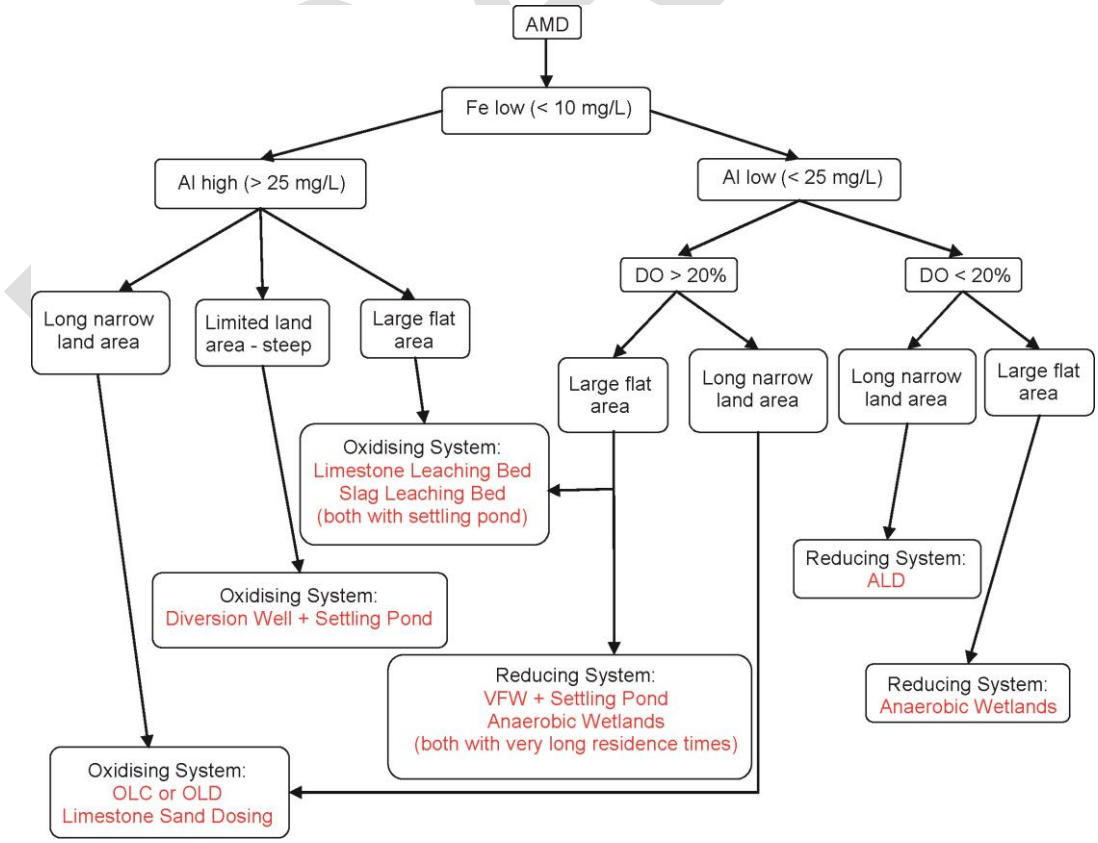


Figure 27: **Flow chart to select among AMD passive treatment systems based on water chemistry (low Fe), topography, and available land area (Trumm 2010).**

Fe concentration

Iron is the most difficult metal to remove from AMD using passive treatment technology, largely due to coating or armouring of limestone, the most commonly used neutralising agent, by Fe oxides and oxyhydroxides, as the AMD is neutralised by limestone (Hilton 2005). This armouring reduces the dissolution rate of the limestone and hence, neutralisation of the AMD (Ziemkiewicz et al. 1997; Watzlaf et al. 2000; Hammarstrom et al. 2003). Much of the current research into passive treatment continues to try to find ways to treat AMD without significant armouring occurring.

Reduced AMD is a special case for AMD treatment as, under low DO conditions, armouring of the limestone will not occur because Fe^{2+} will remain in solution as the pH is raised to neutral (Stumm & Morgan 1996). The ability to raise the pH of the AMD without significant armouring of limestone improves the reliability of treatment.

AMD with relatively low Fe concentrations (<10 mg/L, and ideally less than 5 mg/L) can be treated with either oxidising or reducing systems. Aluminium concentrations, DO, and land area are used to further decide between treatment strategies and passive treatment systems.

Al concentration

Aluminium is a much less problematic metal than Fe in the treatment of AMD. It precipitates out of solution as an amorphous white slime composed of aluminium oxyhydroxide and hydroxysulphate at around a pH of 5 (Bigham 1994; Nordstrom & Alpers 1999), and it does not coat or armour limestone to the same extent as Fe (Hammarstrom et al. 2003; Trumm et al. 2008). Aluminium concentration will influence treatment selection when Fe concentrations are low (<10 mg/L, Figure 27).

Dissolved oxygen concentration

The DO content of mine drainage will be unknown for a new mine, although it is likely to be high for an opencast mine and either high or low for an underground mine. In general, a highly reduced AMD (DO < 20%) is best treated using a reducing strategy whereas an oxidised AMD (DO > 20%) is best treated using an oxidising strategy. However, available land area may limit choice of treatment system. If a reducing strategy is attempted on a highly oxidised AMD, only vertical flow and anaerobic wetlands are suggested and a long residence time in the organic layer is recommended to ensure complete removal of DO and for reducing conditions to establish. Oxidising strategies can be used for AMD with low DO concentrations, but these systems should be constructed with cascades to add DO to enable oxidation reactions to occur.

Available land area

Available land area descriptions in Figures 26 and 27 are limited to steep vs not steep topography and large flat area vs long narrow area. Steep topography is generally suitable for oxidising systems such as diversion wells, open limestone channels and limestone sand dosing where turbulence can help minimise armouring of limestone by iron oxides and oxyhydroxides (Mills 1996; Zurbuch 1996; Ziemkiewicz et al. 1997). Long narrow areas are suitable for anoxic limestone drains (reducing system) and open limestone channels (oxidising system) but if an open limestone channel is constructed with a low gradient, Fe will armour the limestone if it is present in significant amounts. Large flat areas are suitable for both reducing systems (vertical flow and anaerobic wetlands) and oxidising systems (limestone and slag leaching beds).

Costs of passive treatment

For passive treatment systems, construction costs are significantly greater than chemical costs. Figure **28** illustrates the relative costs of the different systems based on a hypothetical AMD that could be treated using all the treatment systems. In reality, some treatment systems would be unable to treat certain mine drainages. Sludge handling and disposal may also be a significant cost and estimates of the volume of sludge that will be generated by a passive treatment system can be made using the computer program AMDTreat (Means et al. 2003) (see Appendix F.4).

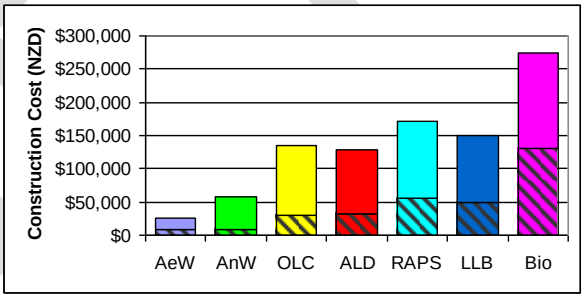


Figure 28: Treatment system costs for a hypothetical AMD determined using AMDTreat (Means et al. 2003) using 2010 cost of limestone in New Zealand and default parameters for labour and construction costs provided in AMDTreat. Aew, aerobic wetland; Anw, anaerobic wetland; OLC, open limestone channel; ALD, anoxic limestone drain; RAPS, reducing and alkalinity producing system; SLB, slag leaching bed; LLB, limestone leaching bed; DW, diversion well; Bio, bioreactor. Hatched area indicates the non-chemical costs. Note: the costs shown in this figure cannot be directly compared with those shown in Figure 25 as the hypothetical AMDs considered in these two examples are markedly different (Appendix F).

5.4.3 AMD from historical mining activities – additional considerations for treatment

The same flow charts and selection criteria presented earlier in this chapter can be used to select treatment options for pre-existing AMD. However, data can be collected from the AMD over a period of time to add confidence to the selection of appropriate treatment techniques.

AMD water chemistry and flow rate should be measured monthly for at least 12 months preferably via data logger. Water chemistry parameters should include pH, total acidity, dissolved Fe, dissolved Al, dissolved Mn, Fe^{2+}:Fe^{3+} ratio, DO, and TSS. It is strongly recommended that the sequential titration acidity procedure described by Hilton (2005) be used on

multiple occasions (see Appendix F.5). For flow rate, the range of flow rates and response of flow to precipitation events should be determined either by spot sampling or preferable via data logger. Any correlation of flow with water chemistry should also be determined.

Active treatment

If active treatment is considered, laboratory bench-scale tests should be conducted to help in system selection and design. For the dosing-with-alkali step (DA), bench-scale tests should be conducted using various chemicals to determine dosing rates and effectiveness (Younger et al. 2002) and a sequential titration acidity analysis should be conducted as described in Appendix F.5.1 (Hilton 2005). For the oxidation step (O), the various oxidation techniques can be tested in the laboratory and it can be determined if chemical oxidants are required and, if so, at what dosing rates. For the sedimentation step (S), coagulants and flocculants can be tested in the laboratory on treated AMD to determine if they are necessary to aid in settling precipitates formed during neutralisation.

Once a neutralisation chemical and coagulants/flocculants have been selected, AMD can be treated in the laboratory to generate sludge for leach testing.

Passive treatment

AMD at closed and abandoned mines typically has a more stable chemistry and flow rate than AMD at active mine sites and land is usually more readily available for treatment systems – factors that fit well with passive treatment.

If passive treatment is considered, once potential treatment solutions have been identified, small-scale field trials should be constructed on-site to test the effectiveness of the various options before investing in full-scale system construction. See Appendix F.6 and Trumm et al. (2006, 2009) for examples of small-scale trials. Even if only one option is indicated through the use of Figures **2**Error! Reference source not found.**6** and **27**, ield trials should still be conducted because unknown factors (e.g. scale-up and preferential flow paths) 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 (see Appendix F.3). A computer program such as AMDTreat (Means et al. 2003; Appendix F.4) can be used to design specific components of various treatment options to determine potential costs.

5.5 Rehabilitation requirements

PAF rock and AMD are hostile to plant growth and prevent rehabilitation. AMD runoff and leachate must not enter the root zone. Sufficient depth of NAF is needed to prevent this happening. Particular attention should be paid to high walls as they cannot be practically revegetated unless a mechanism to achieve long-term stability and cover can be developed. If high-wall surfaces fritter, PAF regenerates itself releasing longer-term acidity.

Options for rehabilitating high walls are limited to:

- Revegetation by hydroseeding with pioneer species such as mosses, lichens, herbs, ice-plants and tolerant low-growing shrubs
- Creating plant growth media in anchored geotextiles or tied-together tyres over rock for growing grasses and shrubs
- Screening by growing shrubs and trees in suitable plant growth media on benches or draping plants grown above the walls.

A practical and economic method has yet to be found for rehabilitating soft-rock PAF pit walls that fritter and continue to generate AMD. Acid generation on stable PAF rock tends to decrease over time and may be able to be revegetated by hydroseeding with lime in the mixture.

Other rehabilitation requirements are also applicable to all other mine types and are described in section **6.5.**

5.6 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide. The following checklist refers only to water quality issues associated with pH and dissolved contaminants; suspended solids are discussed in Chapter 4 and Appendix B.

Step 1. Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Management of suspended sediment ☑

Step 5. Predict mine drainage chemistry from flow chart ☐

Step 6. Predict stream water chemistry

- Predicted mine drainage chemistry from step 4 ☐
- Site hydrogeology ☐
- Background water quality ☐
- Historical mine drainage ☐
- Reactive transport modelling ☐

Step 7. Determine the potential ecological impact on the stream

- Determined from flow chart ☐

Step 8. Consider whether impacts are acceptable, and agree rehabilitation outcomes

This step may be taken internally by a mining company, during consultation with regulatory agencies, or in wider consultation processes, depending on the stage of the mine proposal

- If unacceptable go to step 7a ☐
- If acceptable go to step 8 ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved.

- Consider what on-site waste-rock management techniques are relevant for your specific site ☐
- Consider options for tailings management ☐
- Consider active or passive treatment options ☐
- Consider compensation and/or off-site mitigation ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

6 COAL – NON-ACID-FORMING (NAF)

6.1 Introduction

Impacts on the aquatic environment from coal mining in NAF regions can include, but are not limited to, elevated concentrations of Fe, elevated concentrations of trace elements such as Mn and Zn, and elevated TSS (Shapely & Bishop 1965). High TSS can result from erosion of disturbed waste-rock material and from erosion of exposed coal seams and coal stockpiles. These sources produce TSS consisting predominantly of sand and clay particulates and coal fines (Osterkamp & Joseph 2000). High TSS concentrations can result in elevated turbidity in aquatic environments.

6.2 Predicted water quality

Few data exist with which to predict the chemistry of neutral mine drainage from coal mines (Pope et al. 2010). In general, mine drainage from NAF coal measures rocks is well buffered by carbonates and therefore has circum-neutral pH. However, there can be elevated concentrations of Fe, as Fe^{2+}, because many groundwaters are in equilibrium with Fe carbonate. When mine drainage leaves the mine environment or waste-rock dump, bright Fe oxyhydroxide precipitates form as the Fe is oxidised. In addition elevated concentrations of some trace elements, such as Mn and Zn, are possible. In general, the concentrations of these components are likely to be low (<0.2 mg/L). The highest trace element concentrations measured in samples collected to date (from a sample of pH 6.8) are 10 mg/L Fe, 0.15 mg/L Mn and 0.2 mg/L Zn.

Total suspended solids are generally expected to be elevated at mine sites, and management of TSS is routinely undertaken at all mine sites. The amount of TSS is site specific and varies with climate-controlled factors such as rainfall, and the presence of areas that can generate dust and the geotechnical properties of the rock; also, the efficacy of TSS management options in place may be variable. TSS can be predicted for Southland Coal Measures based on geological information (Craw et al. 2008), but these relationships have not been thoroughly researched for all rock types that are likely to be disturbed by mining.

As there is limited predictive capability of the likely extent of TSS, it is managed proactively (refer to Appendix B). When mining operations commence, suspended solid loads, in particular that present in the discharge from the treatment system, should be monitored to assess the effectiveness of treatment systems.

6.2.1 Downstream water quality

Prediction of water quality downstream from a NAF mine site follows the process outlined in Figure **14** (section **3.4**) and requires integration of data on the chemistry of the neutral mine drainage, site hydrogeology and background water quality. As discussed above, there are limited data with which to enable prediction of likely neutral mine drainage chemistry from coal mines. Collection of relevant site hydrological and background water quality is outlined in section 2.3.

6.3 Predicted ecological impact

The impact of NAF coal mines on streams will be primarily dependent on sediment loads or turbidity, and in some case metal loads.
Impacts on aquatic ecosystems arising from high turbidity are largely physical in nature such as smothering of benthic organisms, and reduction in light penetration. The impacts arising from high turbidity can be considered as direct and indirect impacts. Direct effects occur as a result of a direct effect on an organism and include smothering of benthic organisms and eggs of some species, and clogging of the gills of fish. In contrast, indirect effects include reduction in primary production (algal growth) due to decreased light penetration, and changes in predator–prey relationships due to prey species being hidden from predators.

Excluding the impacts from turbidity, two main outcomes are possible (Figure **29**).

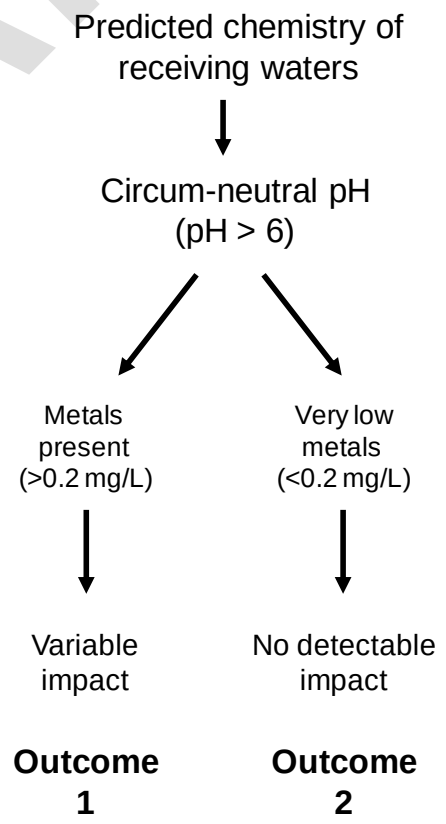


Figure 29: **Potential ecological impacts arising from drainage from coal mining in non-acid-forming regions.**

Outcome 1

Streams with water around neutral pH clearly are not affected by acidity. These waters, if downstream from mines, may have elevated concentrations of metals such as Zn or Cu, which can impact fish and invertebrates. Furthermore, Fe and Al hydroxide precipitates, if present in high amounts, may limit the habitat or food supply of fish and invertebrates. Finally, sedimentation from mining areas could cause ecological effects in streams from high turbidity (from TSS) and loss of habitat quality (from deposited sediment).

Outcome 2

Waters of neutral pH and very low metal content should support a full diversity and abundance of aquatic life for the area. Natural features of the catchments could affect some biota, such as waterfalls blocking migratory fish species. Mining still could affect stream habitat if turbidity and sedimentation (from mining operations) were present. Otherwise, species and food webs should be comparable with those in pristine streams in the area.

6.4 Operational management and treatment

Total suspended solids are expected to be the only water quality issue requiring management and treatment. Best management practices to prevent or reduce high TSS will be more cost effective than ongoing treatment of mine drainage. In particular, waste-rock and tailings management techniques can help to minimise the formation of TSS in mine drainage (see section **4.4.1**), although further treatment may also be required (see section **4.4.3**).

6.5 Rehabilitation requirements

Different overburden layers will vary physically and chemically as suitable root zone or erosion-control materials. Ensuring an adequate depth of favourable root zone underpins successful rehabilitation. Many mine sites have a shortage of salvaged topsoil. If NAF is used as a topsoil surrogate, amendments with organic matter and nitrogen (and phosphorus) will usually be needed. In all cases over- compaction of the root zone, and particularly the surface, limits plant establishment and growth. A compacted surface impedes infiltration. When impeded infiltration is combined with an overly-smooth surface and/or steep, unbroken topography, erosion and TSS generation is exacerbated (see section **4.4** for options to mitigate). Hence, effects of compaction are exacerbated on flat to minimally-sloping landscapes. A rolling landscape reduces the risk of water-logging by allowing lateral flow of water. Mined surfaces usually settle to some extent after mining, so slopes should take this into account where boggy areas are not wanted.

Competent NAF rocks are valuable for use in stream and drain protection and construction, both loose and as rip-rap. Large rocks and boulders may also have an important habitat function in rehabilitation of natural areas and where rocky landscapes are preferred rehabilitation outcomes. Boulders are also useful to control and direct vehicle access.

6.6 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide. The following checklist refers only to water quality issues associated with pH and dissolved contaminants; suspended solids are discussed in Appendix B.

Step 1. Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Management of suspended sediment ☑

Step 5. Predict mine drainage chemistry from flow chart ☐

Step 6. Predict stream water chemistry

- Predicted mine drainage chemistry from step 4 ☐
- Site hydrogeology ☐
- Background water quality ☐
- Historical mine drainage ☐
- Reactive transport modelling ☐

Step 7. Determine the potential ecological impact on the stream

- Determined from flow chart ☐

Step 8. Consider whether impacts are acceptable, and agree rehabilitation outcomes

This step may be taken internally by a mining company, during consultation with regulatory agencies, or in wider consultation processes, depending on the stage of the mine proposal

- If unacceptable go to step 7a ☐
- If acceptable go to step 8 ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved.

- Consider what on-site waste-rock management techniques are relevant for your specific site ☐
- Consider options for tailings management ☐
- Consider active or passive treatment options ☐
- Consider compensation and/or off-site mitigation ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

7 MESOTHERMAL GOLD

7.1 Introduction

Hard-rock mesothermal gold deposits occur in the basement rocks of the West Coast, Southland and Otago. These basement rocks are mainly schist and greywacke. The gold deposits consist of mineralised rocks: quartz veins (metre scale), and associated sheared and altered rocks that contain minerals (including gold) that were added to the rocks by hot water passing through faults and fractures. The volume of these mineralised rocks is small compared with the surrounding basement rocks, which are unaltered and unmineralised. The total width of mineralised zones is typically less than 10 m, and several such zones typically occur in close proximity. Mineralised rocks can have 2-10% sulphide minerals. The rock between mineralised zones is essentially unaltered greywacke or schist.

Most unmineralised rocks, especially schist, have 2-10% carbonate, particularly calcite. Mineralised rocks commonly have locally abundant carbonate minerals (calcite, $CaCO_3$, and ankerite $[CaMgFe]_2[CO3]_2$) Hence, the host rocks and many mineralised rocks have high acid-neutralising capacity (ANC), and at a regional scale high ANC can neutralise acid generated from oxidation of sulphide minerals in the small volumes of mineralised rock. The Fe oxyhydroxides formed from the oxidation of sulphide minerals can help to attenuate the As and Sb released from the mineralised rocks. PAF mine waste is unlikely unless the processing system produces sulphide-rich concentrates that are disposed of separately.

Mineralised rocks also generally include abundant As (typically 1000–10,000 mg/kg) compared with background concentrations of about 10 mg/kg. Antimony levels are normally about one hundredth of As levels, but locally Sb can be present as massive veins forming 5-10% of the rock. Both As and Sb are highly soluble at neutral pH. High concentrations of As and Sb in the mineralised rocks give rise to naturally-occurring elevated concentrations in groundwater compared with non-mineralised areas, with As present up to 0.1 mg/L and Sb up to 0.001 mg/L.

Gold from the mined rocks is normally extracted by addition of cyanide solution (300 mg/L) in a processing plant, at pH > 10. The dissolved cyanide is largely decomposed at the end of the process in a dedicated plant, before discharge to the tailings dam. Most remaining cyanide decomposes under the UV component of sunlight. Residues from the processing plant (tailings) are typically discharged as water-rich slurry, and are impounded behind a dam so that the solids will settle. The water forms a lake on the surface, and can be recycled through the plant or discharged. The tailings impoundment is a major feature of most modern hard-rock gold mines, and this impoundment can be up to 1 km across. Rain water, process water and groundwater discharges from beneath the tailings dam may have elevated levels of As and/or Sb.

Hard-rock gold deposits involve mining of narrow mineralised zones in bedrock, either as large open cuts or in underground tunnels. Opencast gold mines produce very large amounts of waste rock, and waste-rock piles 1 km across and 50 m high are common. However, most of this waste rock is barren host rock, with negligible amounts of sulphide and trace minerals. Instead, the principal environmental issues for aquatic systems are discharge of neutral waters enriched in As and Sb (referred to as NMD) and high suspended solid loads from tailings dams. The concentrations of As and Sb in mine drainage depend on the mining methods and waste disposal systems used.

7.2 Predicted water quality

In any mine operation turbidity arising from high suspended solid loads will occur, due to the scale of earthworks involved. However, the extent to which this is likely to be a problem cannot be predicted and measures to mitigate high suspended solid loads should be routinely put in place (see Appendix B).

As there is limited predictive capability of the likely extent of TSS, it is managed proactively (refer to Chapter 4). When mining operations commence, suspended solid loads, in particular that present in the discharge from the treatment system, should be monitored to assess the effectiveness of treatment systems.

In addition to suspended solids, groundwater, surface water runoff, and mine process water at a gold mine site all have potential to chemically interact with mineralised rocks, and these mine waters develop distinctly different compositions from the natural background waters. Mine drainage pH is typically between 7 and 8 because of the natural carbonate minerals. The high pH (>10) of cyanidation water can contribute to alkaline pH of mine waters. Discharge waters can contain up to 1 mg/L cyanide, but this decomposes in UV light in the discharge stream, and impacts arising from the use of cyanide in CND streams are unlikely.

High concentrations of As and Sb in mine drainage, primarily arising from mine tailings and processing plant water, are the principal environmental issues associated with hard-rock gold mines. The levels of As and Sb in discharge waters depend on the composition of the ore, mining methods, and the waste disposal systems used (Figure **30**). If no sulphide minerals are present in the ore, then there will be no issue with elevated trace metal concentrations downstream of the mining operation. If sulphide minerals are present, the method of processing becomes a critical factor influencing trace metal concentrations.

If the process stream involves no deliberate oxidation of sulphide minerals (left side of Figure **30**), the hydrology of the mine site controls discharge water composition. The key feature is the amount of oxygen that reaches the sulphide minerals:

- If the waste storage is fully saturated with water, limited oxygen will reach the sulphide minerals and As and Sb concentrations in discharge water will be more dependent on the water flow through the waste pile. If there is limited water flow through the repository, the repository will mimic natural groundwater systems, and As and Sb levels will be elevated above regional background water concentrations, but comparable with naturally high background groundwater in mineralised rocks. If there is rapid flow of water through the system, more significant dissolution of sulphide minerals will occur and As and Sb levels will rise above the naturally elevated background.
- If the waste is allowed to become unsaturated, sulphide minerals will oxidise on a scale of 1-10 years and the As and Sb concentrations of discharge waters will rise substantially, possibly to theoretical limits of 50-100 mg/L defined by mineral solubility (typically scorodite, $FeAsO_4.2H_2O$, and stibnite oxidation products).

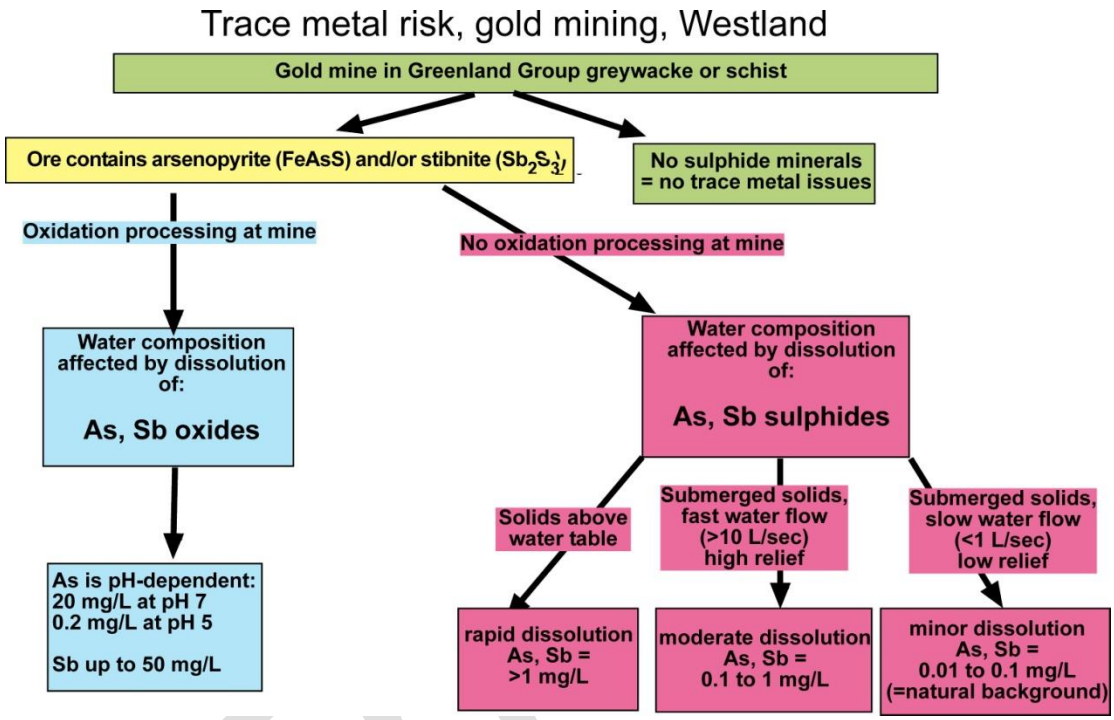


Figure 30: **Predicted water quality associated with a hard-rock gold mine on the West Coast, depending on mineralogy of mineralised rock (ore), mine processing system, and topography of the site where waste is deposited.**

If the mine processing system involves oxidation of sulphides to liberate gold (right-hand side of Figure **30**), mine tailings will contain As and Sb oxide minerals (scorodite, valentinite). These oxide minerals are highly soluble, and As and Sb in discharge waters can rapidly rise to tens of milligrams per litre. It is important to note that both As and Sb are metalloids and have different chemical behaviour from most heavy metals (e.g. Cu, Pb, Zn, Cd) (Appendix C.3). Arsenic and antimony become more soluble with increasing pH, so that they can have 100 times higher concentrations at pH 7 (mine waters) than at pH 5 (some natural West Coast waters). Addition of lime or other acid-neutralising agents to arsenic-bearing mine wastes can result in major increases in dissolved As discharges.

7.2.1 Downstream water quality

Total suspended sediment is generally expected to be elevated at mine sites, although there are limited data with which to predict the extent of the issue and management for high suspended solid loads should be routinely planned for (see Appendix B). When mining operations commence, suspended solid loads, in particular that present in the discharge from the treatment system, should be monitored to assess the effectiveness of treatment systems.

The regulatory point of discharge from a mine site is generally downstream from mining operations, and downstream from where the mine waters first enter the stream. Hard-rock gold mines are located in regions that have naturally elevated As and Sb background concentrations, and it may be more relevant to consider the flux of As and Sb to determine the contribution of mining operations to downstream water quality and provide practical targets for any mine-related water treatment facility (Appendix C.9). These elevated background concentrations will influence the regional flux of As and Sb, which may be large and overshadow most mine discharges a few kilometres downstream from a mine.

Other factors that influence downstream water quality for a proposed operation are the proximity and magnitude of large streams than can dilute As and Sb to low levels indistinguishable from the regional flux, and natural attenuation of As and Sb-bearing waters by Fe oxyhydroxide. Both As and Sb are readily adsorbed onto Fe oxyhydroxide precipitates, which can adsorb 100,000 times as much As and Sb as there is dissolved in water. Iron oxyhydroxide forms naturally around gold mines, through oxidation of certain minerals (see also Appendix C.5) and, provided it is constantly renewed, this is an extremely effective mechanism to remove As and Sb from mine drainage.

7.3 Predicted ecological impact

The non-suspended-solids impact of hard-rock gold mines on streams is usually related to the metal content of the mine drainage (Figure30). In particular, As is of environmental concern and can be toxic to stream biota at relatively low concentrations. In other cases, metal precipitates may be present and coat the streambed, interfering with habitat and food webs. Low pH from acidity is very rarely a problem for mesothermal gold mines in New Zealand. The following flow chart (Figure **31**) illustrates the main ecological impact outcomes that occur from hard-rock gold mine drainage. Further details on the effects of As and Sb are provided in Appendix D.2. With the exception of toxicity studies on key taxa, there are few data on the effect of elevated As on stream communities in New Zealand. Therefore these outcomes are based, in part, on overseas studies.

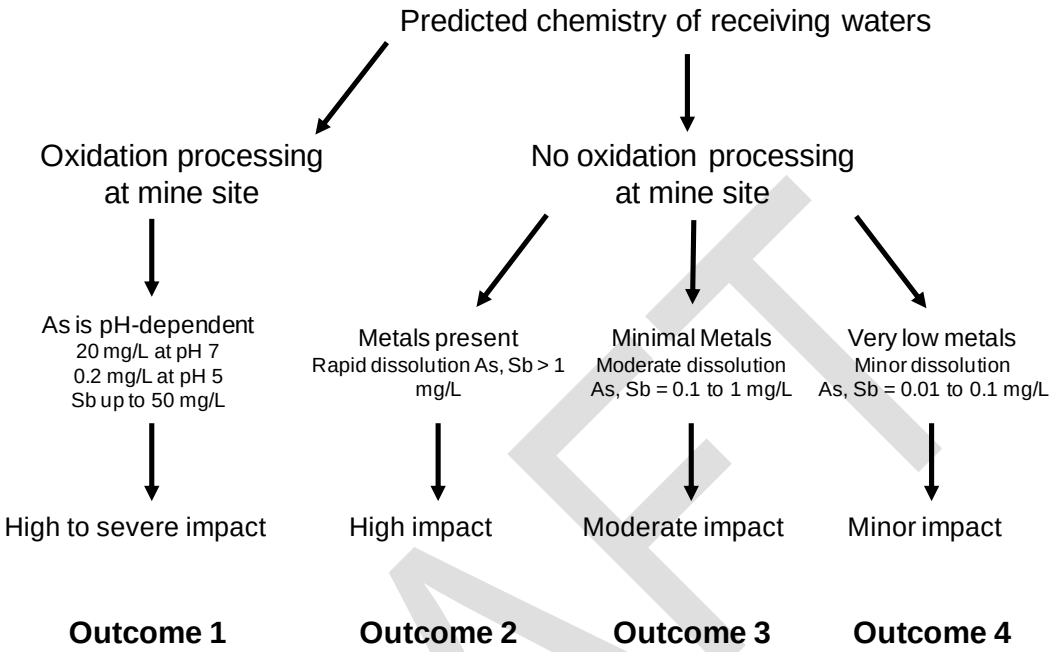


Figure 31: Ecological impact outcomes arising from hard-rock mine drainage based on predicted water chemistries.

Outcome 1

In general mine drainage containing low concentrations of As and Sb should support a full diversity and abundance of aquatic life for the area. Mining still could affect stream habitat if turbidity or sedimentation (from mining operations) were present. Otherwise, species and food webs should be comparable with those in pristine streams in the area.

Outcome 2

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). In particular 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 3

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

Outcome 4

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

7.4 Operational management and treatment

At most sites, management of mine tailings and waste rock and treatment of mine drainage are likely to be required to ensure that trace metals and suspended solid loads in discharge waters (or at a downstream point of interest) do not exceed acceptable levels.

Naturally occurring Fe oxyhydroxide can provide an extremely effective mechanism of removing As and Sb from mine drainage, reducing the need for further treatment. Specifically, exposure of low-As waste rock to allow oxidation of Fe minerals will enhance the formation of Fe oxyhydroxide and increase the capacity of the overall site for As adsorption. The following Fe-bearing minerals are of most relevance in hard-rock gold mines:

- *Pyrite*. Minor acidification associated with pyrite oxidation is normally not an issue because of the high ANC of

surrounding rocks. Pyrite typically accompanies most hard-rock gold, but some deposits are notably poor in pyrite. In this case, acidification is desirable as minor acidification can help to limit dissolving of As minerals.

- *Iron-bearing carbonates*. Ankerite and/or siderite commonly occur in gold deposits and in the immediately adjacent waste rock. Oxidation of these carbonate minerals produces Fe oxyhydroxide. Ankerite (Fe-bearing dolomite) is the most soluble, so that Fe oxyhydroxide forms rapidly. Siderite is less soluble, and finer grain size may be needed for effective Fe oxyhydroxide formation for As and Sb attenuation.
- *Chlorite*. Fe-bearing varieties of this silicate mineral are commonly present in host rocks, and will produce some Fe oxyhydroxide. The oxidation process is slow, and volumes are small, so the attenuation effects are minor only.

If Fe oxyhydroxide is required to reduce dissolved As discharge, the following information is required in the early stages of mine development:

- Confirmation of presence of pyrite (>2%) in waste rock and/or tailings
- Confirmation that pyrite oxidation is occurring, with subsequent precipitation of Fe oxyhydroxide
- Confirmation that sufficient Fe oxyhydroxide is being formed to reduce dissolved As by at least one order of magnitude (typically to <0.1 mg/L dissolved As).

An active or passive treatment system may be required if the above Fe oxyhydroxide formation and adsorption do not occur spontaneously.

7.4.1 Prevention and mitigation

Waste-rock management

Despite the fact that opencast gold mines produce very large amounts of waste rock, most of this is barren host rock with negligible amounts of sulphide minerals and As, therefore management of the waste rock is primarily focused on minimising the discharge of suspended solids (Appendix B). However, relatively minor amounts of sulphide-bearing rock from near the mineralised zone may be included in the waste rock and if this mineralised rock is sulphide-rich, separate management of this rock as PAF may be required (see Chapter 4).

Tailings management

Appropriate management of mine tailings is critical to ensure that mine drainage meets acceptable water quality limits. Figure **32**: Summary of mitigation strategies for environmental issues at a West Coast gold mine. The principal strategy is to remove As and Sb from discharge waters via adsorption to Fe oxyhydroxide. If insufficient natural Fe oxyhydroxide formation occurs, a treatment system is required.
highlights simple techniques that should be taken into account when managing drainage from mine tailings at hard-rock gold mines. As stated above, the method of ore processing is critical in influencing the release of As and Sb in mine drainage. Where no oxidation processes are used, the release of As and Sb in the mine drainage can be minimised by mine tailings being saturated with water to exclude oxygen, and minimising the flow of water through these tailings. Where oxidation processes are used, water incursion into the tailings piles should be minimised by diverting surface streams and constructing low-permeability caps on the surface of piles to exclude rainwater. Where Fe minerals are present, in particular pyrite and siderite, the formation of natural Fe oxyhydroxide

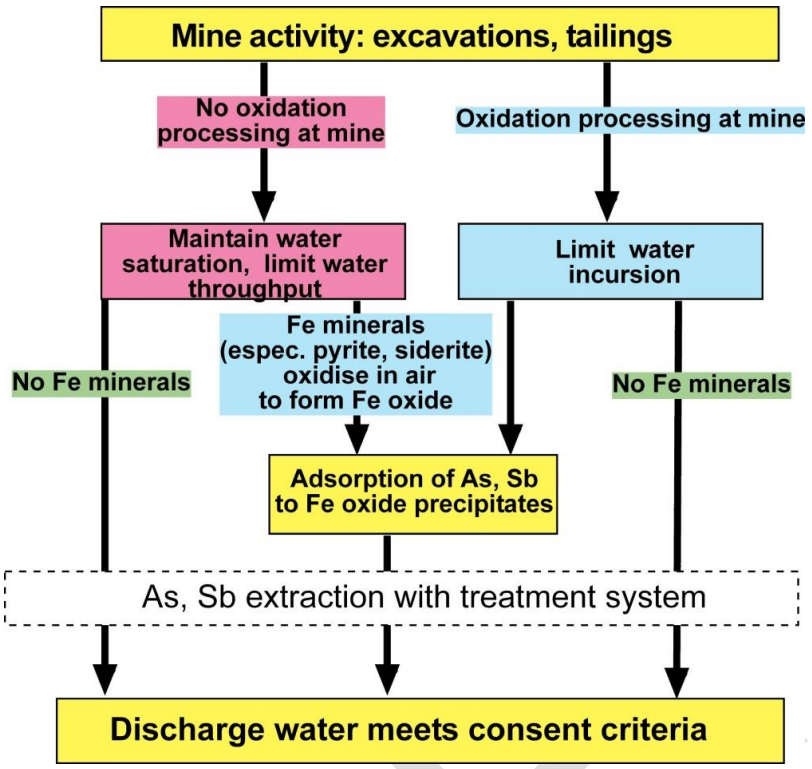


Figure 32: **Summary of mitigation strategies for environmental issues at a West Coast gold mine. The principal strategy is to remove As and Sb from discharge waters via adsorption to Fe oxyhydroxide. If insufficient natural Fe oxyhydroxide formation occurs, a treatment system is required.**

will remove As and Sb present in the mine drainage, potentially to levels that do not require further treatment. However, the formation of natural Fe oxyhydroxide is only able to be determined when mining proceeds and as such requires that appropriate monitoring takes place during mine operations.

7.4.2 Treatment

In circumstances where waste-rock management and natural Fe oxyhydroxide enhancement is insufficient to reduce trace metal concentrations to acceptable levels at the water quality point of interest, further treatment will be required. Arsenic and antimony are the primary trace elements of concern and, as such, the following discussion focuses primarily on treatment options for As and Sb and further details are provided in Appendix F.7.

Treatment can only be undertaken on point-source discharges, and thus requires that effective water collection systems are put in place. Appropriate water collection systems are required for collecting seepage from mine tailings – either dewatered tailings or from dams used to manage tailings – or seeps from dams. Treatment can be accomplished by either active or passive treatment systems (using a variety of techniques) or a combination of both. Further, as mining progresses, treatment may change from one type to the other if economic conditions change, AMD chemistry or flow rate change, or new research leads to better types of treatment systems.

Active treatment technologies have largely been adapted from treatment systems used for the treatment of drinking water, while there are few examples of full-scale passive treatment for As-contaminated mine drainage, and even fewer for Sb-contaminated mine drainage. There are few active or passive treatment systems operational in New Zealand at present thus the following is drawn from research currently being undertaken by the research team, and from the literature.

Arsenic

Selection between active and passive treatment

Several factors will influence the decision as to whether to use active or passive treatment. Briefly, if mine drainage exceeds the thresholds provided in Figure **31**, As removal using passive treatment may not be effective, thus active treatment is likely to be a better choice.

In general, active treatment systems are more commonly used at operational mine sites, whereas passive treatment 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.

The main advantages of active treatment systems over passive treatment systems are that they can remove As effectively,

have precise process control such that they can be engineered and operated to produce a specific water chemistry, and they can be accommodated at small sites. The main advantage of passive treatment systems is that they are expected to be more economic (lower capital, operational and maintenance costs) than active treatment systems.

Active treatment systems

Methods for active removal of As from mine drainage are largely drawn from drinking water treatment technologies and it is of note that drinking water guidelines are typically set at 10 µg/L, due to practical difficulties in removing As below this concentration (e.g. USEPA 2001; WHO 2003; MoH 2005). This provides an indication of the technical limitations of As removal from mine drainage. A range of methods are used to treat drinking water including adsorption, coagulation-precipitation, ion exchange, membrane filtration, and biological treatment. According to USEPA (2002), precipitation has been the most frequently used method to treat As-contaminated water, including groundwater, surface water, leachate, mine drainage, drinking water, and wastewater in numerous pilot- and full-scale applications. Adsorption methods have also been used for a number of full-scale applications, although primarily for the treatment of drinking water (USEPA 2002). Details on all the techniques mentioned above, including a summary of their benefits and disadvantages, is available in Appendix F.7. Coagulation/precipitation and adsorption methods are considered most applicable for use in New Zealand.

Active treatment of As-contaminated water typically includes an initial step to oxidise As(III) to As(V), which can improve the performance of different As removal methods. In addition, some of the methods include oxidation as an intrinsic part of their application (e.g. oxidation in combination with adsorption), although it is not typically used alone as an As treatment.

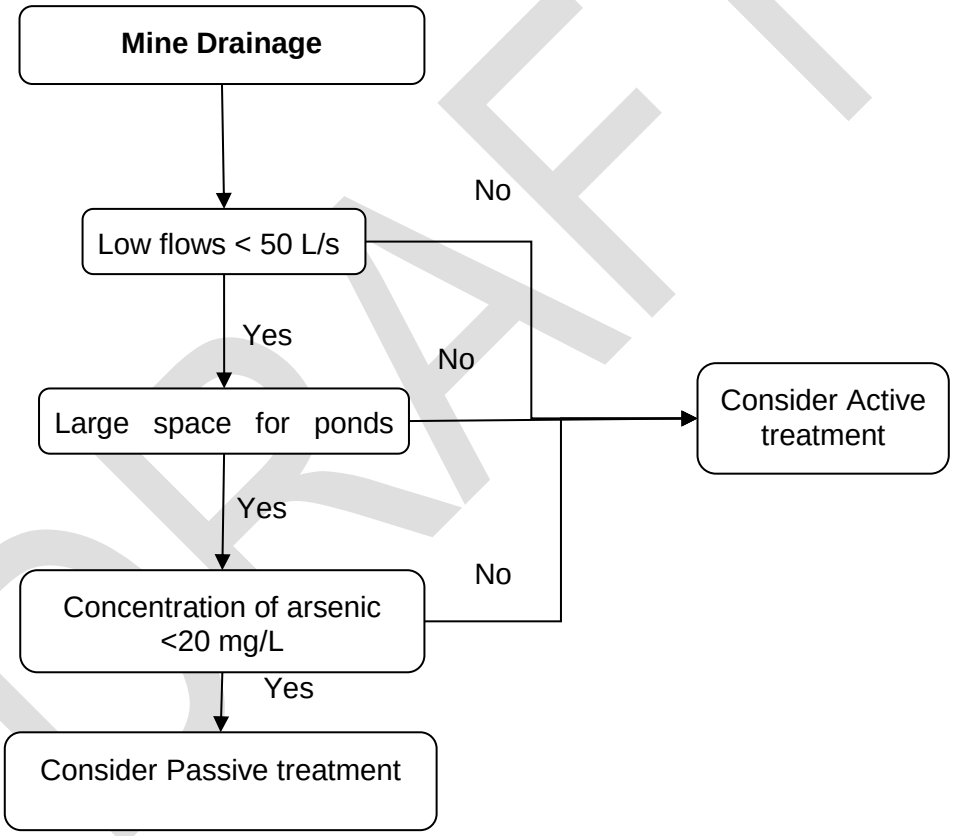


Figure 33: **Selection of passive or active treatment for As removal.**

Coagulation/precipitation

This method uses chemicals to transform dissolved contaminants into an insoluble solid. Colloidal or suspended contaminants formed may be removed through processes such as coagulation and flocculation. This approach is commonly used with an initial oxidation step to change As from As(III) to its less soluble As(V) state, followed by precipitation and clarification/filtration to remove the precipitate. Hardness or removal of heavy metals also present in the mine drainage can also be treated using coagulation/precipitation. Treatment of the collected precipitates may be required prior to disposal. Systems using this method generally require skilled operators, and treatment may be more cost effective on a larger scale.

Chemicals used include ferric salts, or aluminium hydroxide (e.g. Violante et al. 2006; Yuan et al. 2006; Masue et al. 2007).

A New Zealand example of active treatment using coagulation/precipitation is OceanaGold's Globe Progress Mine at Reefton, which treats water with As concentrations up to 2.8 mg/L. A description of the treatment system is provided in Appendix F.7.

Adsorption

In adsorption, dissolved contaminants adsorb onto the surface of a sorbent, thereby reducing their concentration in the bulk liquid phase. The adsorption medium is usually packed into a column or contained within another receptacle. As contaminated water is passed through the column, contaminants are adsorbed. When adsorption sites become filled, the medium must be regenerated or disposed of and replaced with new medium. A range of adsorbent materials can be used although Fe-based media are particularly effective for As removal. A variety of Fe-based media are available and can be 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). Fe-based adsorptive media include:

Table 3.

- Fe-rich AMD-sludge-coated aggregate – Fe-rich AMD sludge from a historical mine site or part of existing operations can be used to coat fine aggregate or sand.
- Fe grit and sand – Fe grit, Fe filings, Fe oxides or Fe sands sourced from beach deposits, scrap from steel plants, industrial processes using Fe grit as an abrasive or waste products from Fe sand processing can be mixed with sand to form an effective adsorption medium.
- Fe sulphate and calcite – these media will need to be purchased from a chemical manufacturer, and thus are likely to be more expensive than the media mentioned above.

Selection of active treatments systems

Factors that should be taken into consideration in selecting an active treatment system are shown in

Table 3: Active treatment systems to remove As

Technique	Media	Factors to consider
Adsorption	Fe-rich AMD-sludge-coated aggregate	Availability of AMD sludge Chemistry of AMD sludge Cost of preparing aggregate Sludge disposal requirements
Adsorption	Fe grit and sand	Cost and availability of Fe grit Sludge disposal requirements
Adsorption	Fe sulphate and calcite	Cost of chemicals Sludge disposal requirements
Coagulation/ precipitation	Fe oxide/hydroxide	Cost of chemicals Sludge disposal requirements Capital costs

The removal efficiency of all systems can vary depending on site-specific characteristics such as water chemistry, rock geochemistry, topography and climate. Regular monitoring of the discharge water is required to ensure that the treatment process is working (section **11.2.3**).

Passive treatment systems

Passive treatment systems will largely be based on oxidation (via aeration) and adsorption processes, particularly using Fe-based media which can be low-cost and available as a by-product from mining and processing, (e.g. AMD sludge). In addition, the oxidation systems described in section 5.4.2 (passive treatment) could also be used for the treatment of As NMD. Oxidation of the treatment system ensures As(V) is the main valency state and allows for efficient As removal.

Although oxidation and adsorption processes are very effective for passive treatment, As can also be removed using in a passive treatment system based on reduction reactions. If enough sulphate is present in the water, bacterial sulphate reduction to hydrogen sulphide and subsequent formation of arsenic sulphides can be an effective treatment process. This is primarily accomplished in sulphate-reducing bioreactors.

Passive treatment is more applicable for closed and abandoned mine sites rather than active mine sites because they typically have a more stable chemistry and flow rate. Further, land area is usually more readily available for treatment systems. Care must be taken in designing passive systems to ensure that fluctuations in flow are accounted for. There are few examples of pilot- or full-scale passive treatment of As-contaminated water nationally or internationally. Trials using AMD sludge as an adsorptive media have been undertaken in New Zealand and are described in detail in Appendix F.6.

Antimony

Active treatment

Treatment of antimony from mine drainage is relatively rare. However, there are two main techniques that are used in active treatment systems: filtration by membranes, and coagulation-flocculation-sedimentation (CFS) (Guo et al. 2009). Filtration includes microfiltration, nanofiltration, ultrafiltration, and reverse osmosis. Although effective, these techniques are expensive to implement.

Treatment of Sb by CFS is more common. The CFS technique includes removal of contaminants using precipitation, coprecipitation, and adsorption. For Sb removal by CFS, it has been found that removal is predominantly by adsorption (Guo et al. 2009). Contrary to As-contaminated water, active treatment of Sb-contaminated water typically does not include an initial step to oxidise Sb(III) to Sb(V). The reduced Sb(III) will adsorb more readily than Sb(V) (Guo et al. 2009).

Adsorption

In adsorption, dissolved contaminants adsorb onto the surface of a sorbent, thereby reducing their concentration in the bulk liquid phase. The adsorption medium is usually packed into a column or contained within another receptacle. As contaminated water is passed through the column, contaminants are adsorbed. When adsorption sites become filled, the medium must be regenerated or disposed of and replaced with new medium. A range of adsorbent materials can be used although Fe-based media are particularly effective for Sb removal. A variety of Fe-based media are available and can be 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). Fe-based adsorptive media include:

- Fe-rich AMD-sludge-coated aggregate – Fe-rich AMD sludge from a historical mine site or part of existing operations can be used to coat fine aggregate or sand.
- Fe grit and sand – Fe grit, Fe filings, Fe oxides or Fe sands sourced from beach deposits, scrap from steel plants, industrial processes using Fe grit as an abrasive or waste products from Fe sand processing can be mixed with sand to form an effective adsorption medium.
- Fe sulphate and calcite – these media will need to be purchased from a chemical manufacturer, and thus are likely to be more expensive than the media mentioned above.
- Fe chloride ($FeCl_3$) – this is the current technique in use at the OceanGold Globe Progress Mine near Reefton (Escueta et al. 2013).

Other adsorption media can include Al sulphate ($Al_2(SO_4)_3$), or Mn oxides, although treatment using Fe oxyhydroxides is most common

Passive treatment

Passive treatment is more applicable for closed and abandoned mine sites rather than active mine sites because they typically have a more stable chemistry and flow rate. Further, land area is usually more readily available for treatment systems. Care must be taken in designing passive systems to ensure that fluctuations in flow are accounted for. There are few examples globally of pilot- or full-scale passive treatment of Sb-contaminated water.

In New Zealand, recent field trials have shown two effective methods of treatment for Sb-contaminated water (Trumm & Pope 2014). In one method, Sb is removed through adsorption onto Fe oxyhydroxide harvested from untreated coal mine AMD. Iron precipitates from AMD have a high affinity for adsorption of Sb, and can adsorb Sb at relatively short residence times. Removal rates up to 95% at residence times of only 3 hours are documented.

In the other method, Sb is removed through the use of a sulphate-reducing bioreactor. If enough sulphate is present in the water, bacterial sulphate reduction to hydrogen sulphide and subsequent formation of antimony sulphide (stibnite) can be an effective treatment process.

7.5 Rehabilitation requirements

The tailings produced by hard-rock gold mines are typically alkaline due to lime dosing. If these tailings are used as part of the root zone, the high pH reduces plant-availability of phosphorus. Effects are exacerbated where iron levels are also high. Phosphate fertilisers are therefore likely required. However, in most cases tailings will not be included in the root zone to avoid plants mobilising and taking up metal contaminants, and because pyrites may be present. Other considerations are discussed in section 6.5.

7.6 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide. The following checklist refers only to water quality issues associated with pH and dissolved contaminants; suspended solids are discussed in Chapter 4 and Appendix B.

Step 1. Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Management of suspended sediment ☑

Step 5. Predict mine drainage chemistry from flow chart ☐

Step 6. Predict stream water chemistry

- Predicted mine drainage chemistry from step 4 ☐
- Site hydrogeology ☐
- Background water quality ☐
- Historical mine drainage ☐
- Reactive transport modelling ☐

Step 7. Determine the potential ecological impact on the stream

- Determined from flow chart ☐

Step 8. Consider whether impacts are acceptable, and agree rehabilitation outcomes

This step may be taken internally by a mining company, during consultation with regulatory agencies, or in wider consultation processes, depending on the stage of the mine proposal

- If unacceptable go to step 7a ☐
- If acceptable go to step 8 ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved.

- Consider what on-site waste-rock management techniques are relevant for your specific site ☐
- Consider options for tailings management ☐
- Consider active or passive treatment options ☐
- Consider compensation and/or off-site mitigation ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

8 EPITHERMAL GOLD

8.1 Introduction

These hard-rock deposits occur in Northland, and in Coromandel, extending south-west towards the Taupo Volcanic Zone. The host rocks for these deposits are either basement greywacke or volcanic rocks that overlie that greywacke. The deposits were formed at shallow depths (less than 1 km) below geothermal hot spring systems. The widespread active geothermal features of the Taupo Volcanic Zone are the surface expressions of epithermal gold-forming systems at depth, and some of the geothermal springs are actively depositing metal-rich material. Similarly, geothermal power stations exploit the same parts of the geothermal systems in which gold is deposited, and gold-rich material has been found in modern geothermal power stations. Gold mines have traditionally been developed in deeper parts of old geothermal systems that have been uplifted and partially eroded. Some such mines have encountered gold-bearing rocks that were deposited at, or just below, the surface at the time the springs were active.

The geothermal systems that led to formation of epithermal gold deposits involve circulation of groundwater, derived from rain, above a shallow heat source associated with molten volcanic rocks. Hence, there has been extensive fluid flow through fractures and cavities in the host rocks, and that has caused chemical alteration of the host rocks over wide areas (several square kilometres) around each deposit. What were hard volcanic rocks or greywacke basement became transformed to clay-rich soft rocks in the vicinity of gold deposits. This extensive alteration removes most of the calcium carbonate ($CaCO_3$) from the host rocks, and variable amounts of pyrite (FeS_2) are deposited through this altered rock.

The gold deposits consist mainly of quartz veins that can be up to several metres across. This veins commonly have some calcite associated but not sufficient to compensate for the large amount of calcite removed from host rocks. The gold typically contains abundant silver (up to 50%) and additional silver minerals occur also. This gold-silver amalgam, called electrum, commonly forms free grains that can be large (millimetres to centimetres). Gold and silver occur in close association with a wide range of sulphide minerals, and these sulphide minerals also enclose some gold and silver. The most common sulphide minerals are pyrite (FeS_2), chalcopyrite ($CuFeS_2$), sphalerite (ZnS) and galena (PbS). Deposits commonly have mineral zonation with depth (Henley et al 1984, Berger & Bethke 1985, Braithwaite et al 2006). The shallower parts of an epithermal system can be enriched in mercury, commonly as cinnabar (HgS), and also arsenic as orpiment (As_2S_3) or realgar (AsS) and antimony as stibnite (Sb_2S_3) (Figure 34). Deeper parts of epithermal deposits can be enriched in copper, lead and zinc. However, all these elements are present in elevated quantities through most epithermal deposits. In addition, there is generally enrichment of a wide range of other trace metals, such as cadmium (Cd, associated with Zn), thallium (Tl), selenium (Se) and manganese (Mn). However, the

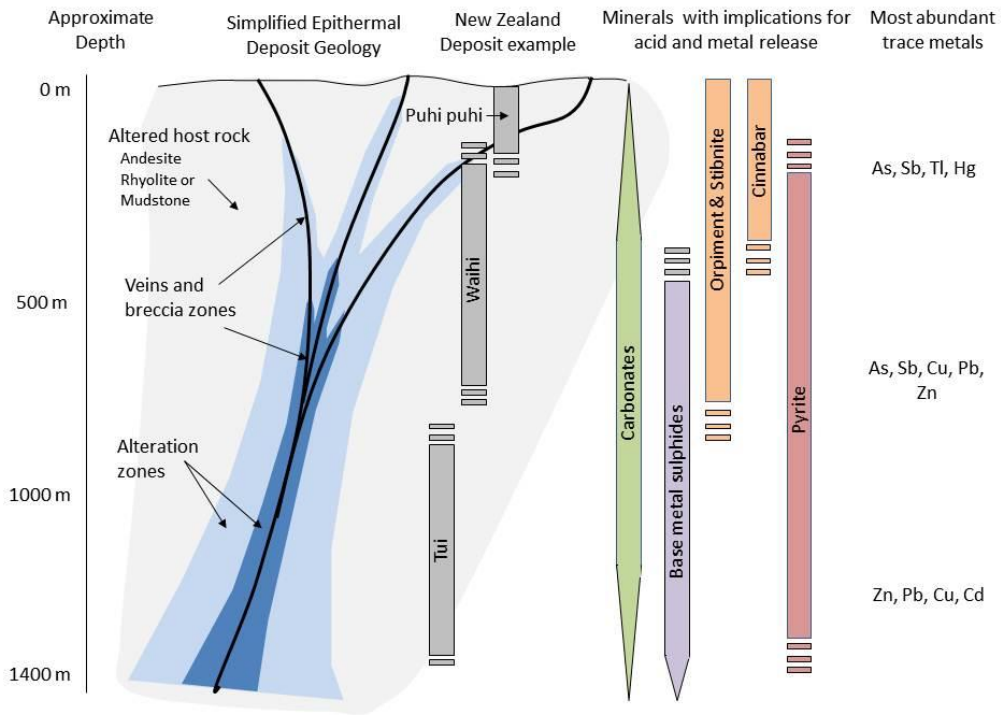


Figure 34 **Schematic of epithermal mineral deposit with a generalised model for the distribution of selected minerals that release or neutralise acid or contain trace elements.**

mineralogical and trace element zonation of epithermal deposits is often complicated and there can be overprinting styles of mineralisation reflecting changes in geothermal fluid chemistry with time throughout the deposit (Strimc-Palinkas et al. In press).

Gold is more readily extracted from epithermal deposits than from orogenic (mesothermal) deposits, and extensive sulphide mineral separation and roasting or pressure oxidation is not necessary. Instead, the whole of the ore is finely ground and passed directly into a cyanidation process that is essentially identical to that part of the orogenic (mesothermal) process (Chapter 7). In particular, one of the defining features of epithermal mines, like orogenic (mesothermal) mines, is the large tailings impoundment. In addition, the waste rock piles at epithermal mines contain abundant altered rock from around the gold deposit, and this altered rock contains variable amounts of sulphide minerals with elevated metal contents, as outlined in the previous paragraph. Underground epithermal mines generate substantially smaller amounts of this altered waste rock than open cut mines.

8.2 Predicted water quality

In any mine operation turbidity arising from high suspended solid loads will occur, due to the scale of earthworks involved. However, the extent to which this is likely to be a problem cannot be predicted and measures to mitigate high suspended solid loads should be routinely put in place (see Chapter 4 and Appendix B).

As there is limited predictive capability of the likely extent of TSS, it is managed proactively (refer to Chapter 4 and Appendix B). When mining operations commence, suspended solid loads, in particular that present in the discharge from the treatment system, should be monitored to assess the effectiveness of treatment systems.

Water quality at epithermal deposits reflects the mineralogy that characterises these deposits, pyrite, carbonates and various other sulphide minerals. Drainages range from acidic to neutral typically with an extensive suite of elevated trace element concentrations. The three epithermal deposits for which the largest amount of water quality and rock geochemistry data is available are Martha, Golden Cross and Tui. There is also some data available from a selection of historic mine sites (Craw &Chappell 2000). Tui, Golden Cross and Martha mines represent a spectrum with respect to site conditions, rehabilitation and treatment. Tui Mine was abandoned without remediation and has subsequently had a rehabilitation programme applied. Golden Cross was mined during the 1980s and 1990s and rehabilitated in the late 1990s. Rehabilitation of the tailings dam, waste rock dumps and pit has been successful with monitoring continuing at the site. However, water from underground workings is treated by an active plant at the site and recently an unfilled stope collapsed. Martha is still fully operational and has active water treatment and a progressive rehabilitation programme in progress. The geochemistry of waste rock at these deposits and the waste rock management options selected are the main factors that control water quality at these sites.

8.3 Predicted ecological impact

Trace elements that can be elevated in drainages downstream of abandoned epithermal deposits include Zn and Mn. The impact of these trace elements on aquatic ecosystems is an area of current research. Similar to overseas studies that have shown the diversity, abundance and community composition of algae, benthic invertebrates and fish communities can all be impacted by trace metals, a comparison of reference sites with downstream sites receiving mine discharges from the Tui mine showed benthic diversity declined from 26 taxa to 12 taxa (Harding and Simon submitted). Further, six New Zealand mayfly genera were completely excluded from the Tunakohoia stream which received high zinc concentrations (2.0-4.0 mg/l) from the Tui Mine (Harding and Simon submitted). In other studies toxicity testing using Rainbow trout showed an EC50 at Zn concentrations as low as 1.45 mg L-1. In contrast, several other groups seem relatively tolerant of high metals, for example net-spinning caddisflies and true flies such as chironomids can be abundant in high metal ecosystems (>2 mg/L, Hickey &Clements 1998). Similarly, toxicity testing using several New Zealand native fish species have shown high tolerances to Zn. Inanga a common coastal migratory fish species had an EC50 of 24 mg/L, while short finned eels had an EC50 of 47 mg/L (Hickey 2000).

8.4 Operational management and treatment

8.4.1 Waste Rock and Tailings Geochemistry.

Acid base accounting characteristics of rocks and tailings from epithermal deposits are available for Tui, Golden Cross and Martha. At these sites typically the waste rock is PAF strongly acid producing (NAPP > 10), however, there is commonly significant ANC (>10kg/t H_2SO_4) in these samples. For example, at the Golden Cross Deposit, about 100 samples were collected to characterise acid producing potential of the deposit prior to operations. Most rocks were PAF – strongly acid producing (~70%) (Table 1) with the remainder near neutral PAF – moderately acid producing (10%) or NAF (~20%). In addition, about 40% of the rocks have ANC > 10kg/t H_2SO_4 and many of the remaining rocks have lower but significant ANC (3-10kgH_2SO_4/t) (Miller 1987).

The trend for dominantly PAF rocks with substantial ANC is similar at other epithermal deposits where data are available for example Tui and Martha tailings (Giles et al 2010) and Martha waste rock although the percentages and magnitudes are variable.

Limited kinetic testing data is available from waste rocks from Golden Cross, Tui and Martha (Miller 1987, Giles et al 2010efs), and commonly the test work available is column leach tests or humidity cell tests. Where column test work is only conducted for a limited period (6-18 months) acid and trace element

release is low to moderate (pH typically > 4 in column leachate) and this reflects buffering of acidification by ANC. Where test work is conducted for longer periods, the low rate of acid release often accelerates to rapid acid release (pH commonly < 2.5 in column leachate) with high trace element concentrations. This trend, of a lag period before rapid acid and trace element release reflects consumption of buffering capacity provided by ANC followed by rapid, acid catalysed oxidation of sulphide minerals. However, there are limited datasets available to characterise details of leachate chemistry after accelerated acidification.

The consistent and interpretable trends between acid base accounting data and column leach data provides a generalised model for release of acid and trace elements from epithermal mineral deposits and indicates which waste rock management strategies might be applicable at epithermal deposits. However, the variability of acid base accounting properties of rocks in epithermal deposits also means that there could be significant volumes of rocks within epithermal deposits that release acid quickly or remain neutral weathering, and therefore site specific investigations are required at future mine sites.
The best available dataset on mine drainage seeps for an epithermal deposit is available from Tui Mine. At seeps inside the mine pH varies from acid to circum-neutral (pH range from 2.9 to 7.8). Mine drainage seeps have variable concentrations of trace elements including Zn>>Cu>Pb,Ni>Cd, As (maximum concentrations 173>>3.9>0.3,0.02>1.1, 0.013 mg/L respectively). Leachate monitoring and data from other mine sites also indicates Zn concentrations are high with along with high concentrations of Mn and Fe (between 1 and 20 mg/L) and lower concentrations of other trace metals including Cd, Cr, Co, Pb, Hg, Ni and Se (commonly between 0.001 and 0.5 mg/L) (Cameron 1991, Petersen & Kindley 1993, Trumm & Pope submitted, Pope & Trumm submitted).

At some epithermal deposits carbonate minerals related to alteration of host rocks close to mineralisation are sufficient to prevent or delay formation of acid during mining operations. Prevention of acidification removes one aspect of environmental impact (low pH), however, these neutral discharges can still contain elevated trace element and reduced Fe concentrations. Delay of acidification can be useful for operational management at active mine sites. Acid base accounting and kinetic testing is required to identify and quantify the potential for waste rocks from epithermal mineral deposits to neutralise acid.

Waste rock and tailings management strategies also influence the chemistry of drainages from epithermal gold deposits. Similar considerations and options can be employed to those discussed in PAF coal mine drainages. Tailings geochemistry from epithermal gold mines have not been researched in New Zealand, although, in house datasets held by companies provide site specific guidance for operational management and storage of tailings.

The high pH (>10) of cyanidation water can mitigate the potentially low pH of mine tailings waters. Whether or not this is a long-term neutralisation depends on acid-base accounting of the tailings material. Any tailings discharge waters can contain up to several mg/L cyanide, but this decomposes in UV light, and impacts arising from the use of cyanide are unlikely in discharge streams. The water quality issues of concern from epithermal gold mines mostly include elevated trace elements with or without associated acidity. The trace elements most-commonly present include the transition metals Fe, Al, Mn, Zn, Ni, Cd, Cu and Hg, the post-transition metal Pb, the metalloids As and Sb, and the non-metal Se. At most sites, management of mine tailings and waste rock and treatment of mine drainage are likely to be required to ensure that trace elements (with or without low pH) and suspended solid loads in discharge waters do not exceed acceptable levels.

Best management practices to prevent or reduce the formation of acid and trace elements can reduce environmental risk and be more cost effective than ongoing treatment of mine drainage. Prevention is achieved by identifying PAF material and sulphide-bearing material, separately stripping it, and placing it so oxygen and/or water cannot oxidise the material. Prevention may also include co-disposal with alkaline materials to neutralise acidity generated from oxidation, however, if oxidation occurs and trace elements are released, most of the trace elements will remain in solution even if the pH is raised through neutralisation.

Treatment of the drainage water may still be required even where optimal management strategies have been put in place, however, treatment costs will be lower than if no mitigation strategies had been in place and may also allow for the use of passive instead of active treatment.

Once management techniques have been initiated or treatment commenced, monitoring of any discharge from the site is necessary to verify management/treatment efficacy. Water quality parameters and frequency of sampling for different treatment systems, as well as biological monitoring, are covered in Chapter 11.

8.4.2 Prevention

Waste-rock management

Unlike mesothermal deposits, in epithermal deposits, sulphide minerals can be present at significant quantities in waste rock some distance from the mineralised zone. Because of this, waste-rock management techniques are often necessary at gold mines in epithermal deposits to prevent the release of trace elements from sulphide oxidation.

Management techniques for waste rock from PAF sites (see Chapter 5) can be applied to gold mines in epithermal deposits. These techniques can be placed into three broad categories:

- Avoidance – limit exposure of acid-producing material through mine planning.
- Isolation – separate PAF rocks from oxygen and/or water both temporally and spacially to prevent oxidation. This can be achieved through covers or inundation.

- Neutralisation – offset acid generation by mixing or layering onsite PAF and NAF rock or by alkaline addition.

As mentioned above, if neutralisation is used, trace elements present in solution may remain in solution at neutral pH and treatment may still be required.

The closed Golden Cross Gold Mine near Waihi, New Zealand can be used as an example of successful waste rock management techniques in an epithermal deposit. At this site, acid-producing waste rock (non-argillic waste material) was placed at the floor of the pit and encapsulated with non-acid producing rock (argillic waste material) to minimise contact with oxygen and water and reduce oxidation (Tonkin &Taylor, 1987). To achieve the desired outcomes careful planning and scheduling was required. At the current Martha Mine, also near Waihi, alkaline addition is used along with layering of PAF and NAF rock to prevent drainage from tailings pond embankments constructed with waste rock.

Along with management of waste rock to prevent sulphide oxidation, management may also be required to minimise the discharge of suspended solids (see Appendix B).

Tailings management

As with gold mines in mesothermal deposits, appropriate management of mine tailings from gold mines in epithermal deposits is critical to ensure that mine drainage meets acceptable water quality limits. Similar to mesothermal gold mines, the method of ore processing is critical in influencing the release of trace elements in mine drainage. Where no oxidation processes are used, the release of trace elements in the mine drainage can be minimised by mine tailings being saturated with water to exclude oxygen, and minimising the flow of water through these tailings. Where oxidation processes are used, water incursion into the tailings piles should be minimised by diverting surface streams and constructing low-permeability caps on the surface of piles to exclude rainwater.

Saturation of tailings to exclude oxygen was used at the closed Golden Cross Gold Mine near Waihi, New Zealand (Tonkin & Taylor, 1987; Peterson & Kindley, 1993) and is the current technique in use at the active Martha Mine, also near Waihi.

8.4.3 Treatment

Treatment of drainage from gold mines in epithermal deposits can be accomplished using active or passive treatment techniques. In general, active treatment is more commonly used at active mine sites, or at sites with significant flow rates and high concentrations of contaminants. Passive treatment is more common for closed mine sites or at sites will lower flow rates and relatively lower concentrations of contaminants.

Active Treatment

The drainage chemistry can either consist of AMD with trace elements or NMD with trace elements. If AMD is present, the active treatment techniques discussed in Chapter 5 should be used to raise the pH and remove the dominant lewis acid metals Fe and Al. Additional techniques to remove trace elements are discussed below.

Transition metals and post-transition metals

For the transition metals Mn, Zn, Ni, Cd and Cu and the post-transition metal Pb, active treatment consists primarily of pH adjustment to high pH so that the metals precipitate as oxides and hydroxides. After the metals have been removed, acid treatment is often required to lower the pH to acceptable levels.

For the transition metal Hg, active treatment consists primarily of either suphide precipitation using a sulphide precipitant such as sodium sulphide or oxide precipitation through pH adjustment as with the other transition metals. Other techniques in use included adsorption and co-precipitation with ferric chloride.

Depending on the concentrations of contaminants and the discharge requirements, more expensive filtration treatment techniques may be required. The most common (and most effective) of these is reverse osmosis.

Metalloids

The metalloid As is treated using adsorption, coagulation-precipitation, ion exchange, membrane filtration and biological treatment, with precipitation being the most frequently used method. The metalloid Sb is treated through filtration by membranes, or by coagulation-flocculation-sedimentation. Filtration includes microfiltration, nanofiltration, ultrafiltration, and reverse osmosis.

Additional details of active treatment for As and Sb are discussed in Chapter 7.

Non-metals

For the non-metal Se, active treatment techniques consist of biological reduction, iron co-precipitation, ion exchange, filtration or adsorption. The most common technique has been co-precipitation, however, in recent years, biological reduction has been more prevalent.

Passive Treatment

A variety of passive treatment techniques are used to treat drainage from epithermal gold mines, depending on the trace elements present. If AMD is present along with trace elements, typically, an appropriate AMD treatment system should first be selected (see Chapter 5) and the trace elements then removed using an additional component to the passive treatment system. If the drainage consists of NMD with trace elements, typically only a single system is required to treat the trace elements. Figure 35 is a flow chart to assist in treatment system selection.

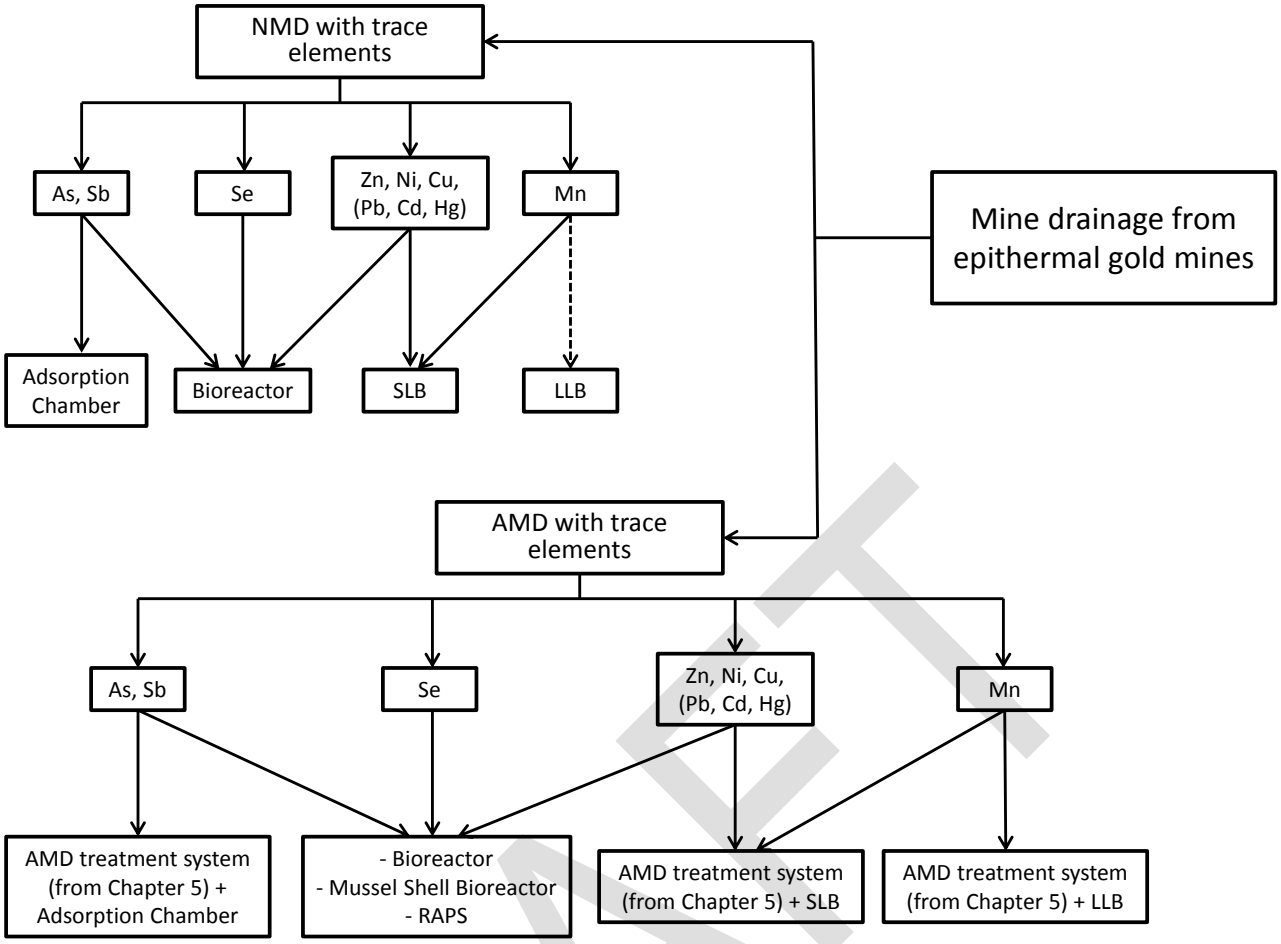


Figure 35: Flow chart to select among passive treatment systems for drainage from gold mines in epithermal deposits. SLB, slag leaching bed; LLB, limestone leaching bed; RAPS, reducing and alkalinity producing system. Dashed line shows tentative applicability.

Transition metals and post-transition metals

The transition metals Zn, Ni and Cu can be treated by two different passive treatment strategies. One strategy involves raising the pH high enough such that insoluble oxides precipitate and settle out of solution. The best effective way to do this with a passive treatment system is with steel slag, which can raise the pH to in excess of 10. The other strategy involves sulphate reduction and formation of metal sulphides in sulphate-reducing bioreactors. The transition metals Cd and Hg and the post-transition metal Pb might also be treated by these methods; treatment of these metals is a current focus of research.

If AMD is present, for the oxidising strategy, an AMD treatment system appropriate for the AMD chemistry is selected as per Chapter 5. During oxidation and precipitation of iron oxides and hydroxides, some adsorption of the transition and post-transition metals will occur. The AMD treatment system is then followed by an SLB to remove the remainder of the trace elements.

If the reducing strategy is applied to AMD with trace elements, appropriate treatment systems include bioreactors, mussel shell bioreactors or RAPS.

The transition metal Mn does not form sulphides in passive treatment systems, therefore, treatment of Mn consists of raising the pH high enough that Mn oxides, carbonates, or hydroxides form. This can be accomplished with an SLB or, especially if AMD is present as well, with a LLB.

Metalloids

Treatment of the metalloids As and Sb using passive treatment systems is discussed in Chapter 7. Essentially, treatment consists of either adsorption onto iron hydroxides (such as those derived from AMD sites) in an adsorption chamber, or formation and removal as sulphides in a bioreactor.

If AMD is present as well, and an oxidising treatment system is prefered, an appropriate treatment system is selected based on the criteria discussed in Chapter 5, which is then followed by an adsorption chamber. If a reducing strategy is selected, appropriate treatment systems which can treat both the AMD and the metalloids, include a bioreactor, a mussel shell bioreactor or a RAPS.

Non-metals

Passive treatment of the non-metal Se is accomplished by reduction to elemental Se in bioreactors. This technique is being successfully applied in numerous passive treatment

bioreactor in the Southeastern USA. If AMD is present as well as Se, appropriate passive treatment systems include bioreactors, mussel shell bioreactors or RAPS.Rehabilitation considerations As for coal mined from PAF regions, PAF rocks and AMD are detrimental to rehabilitation outcomes and appropriate measures must be undertaken to ensure that the root zone of the plants is not affected by acidic rocks or drainages (see section 5.4.1 for appropriate management of PAF rocks). Other considerations are discussed in section 6.5.

8.5 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide.

Step 1. Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Management of suspended sediment ☑

Step 5. Predict mine drainage chemistry from flow chart ☐

Step 6. Predict stream water chemistry

- Predicted mine drainage chemistry from step 4 ☐
- Site hydrogeology ☐
- Background water quality ☐
- Historical mine drainage ☐
- Reactive transport modelling ☐

Step 7. Determine the potential ecological impact on the stream

- Determined from flow chart ☐

Step 8. Consider whether impacts are acceptable, and agree rehabilitation outcomes

This step may be taken internally by a mining company, during consultation with regulatory agencies, or in wider consultation processes, depending on the stage of the mine proposal

- If unacceptable go to step 7a ☐
- If acceptable go to step 8 ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved.

- Consider what on-site waste-rock management techniques are relevant for your specific site ☐
- Consider options for tailings management ☐
- Consider active or passive treatment options ☐
- Consider compensation and/or off-site mitigation ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

9 GOLD – ALLUVIAL

9.1 Introduction

Alluvial gold accumulates in gravels, and is commonly most strongly enriched where the gravels rest on different, older rocks (at an unconformity). There are three main environments where alluvial gold is mined in this context:

- Old gravels that have commonly been folded and faulted. These gravels are most common in Southland, and are typically between 25 and 2 million years old. The gravels are dominated by round quartz pebbles, with lesser amounts of schist and greywacke pebbles. These sediments are commonly associated with coal (lignite) and the environmental issues are very similar (see Chapters 5 and 6).
- Young gravels associated with active mountain-building, glaciation and glacial outwash (less than 2 million years old). These are most common on the West Coast, but occur locally in Southland. These gravels consist of a chaotic mix of schist and/or greywacke pebbles with a wide range of grain sizes.
- Beach sediments. These can be gravels or sands that have accumulated along the shore-face. Gold occurs in distinct horizons enriched in heavy minerals such as magnetite (Fe oxide) and garnet. These alluvial deposits occur in both Southland and West Coast. Gold enrichment occurs in some active beach sediments, and in old beach deposits that have been uplifted and occur a short distance inland.

Large volumes of gravel overburden may be removed in order to reach the gold-containing gravels. These gravels are moved to an extraction plant where the coarse gravel is removed, the gold is separated by gravity, and the fine sediment is discharged as slurry into a tailings pond. Overtopping of the tailings pond may occur during high rainfall events, unless there is a properly constructed spillway.

9.2 Predicted water quality

The principal environmental issue associated with discharge waters from alluvial gold mining is turbidity, arising from a high suspended solid load. High TSS may result from the movement of gravel overburden, particularly during wet conditions, or from overtopping of the tailings pond. The amount of TSS is site specific and varies with climate-controlled factors such as rainfall and the presence of areas that can generate dust. There is some evidence that the suspended solid load can be predicted based on geological information (Craw et al. 2008), but these relationships have not been thoroughly researched.

As there is limited predictive capability of the likely extent of TSS, it is managed proactively (refer to Chapter 4). When mining operations commence, suspended solid loads, in particular that present in the discharge from the treatment system, should be monitored to assess the effectiveness of treatment systems.

In addition to suspended solids, the following issues can arise:

- *Acid rock drainage*. Most of the older gravel deposits have some localised cementation by pyrite or marcasite (both Fe sulphides FeS_2) that can generate acid on oxidation (e.g. Belle-Brook, Southland; Falconer & Craw 2005). The gravels and associated sediments have very low acid neutralisation potential because of the high proportion of quartz and lack of carbonate minerals. Presence of any Fe sulphide can lead to significant acidification to pH 3 and increased trace metal concentrations, similar to that observed in coal mines (Chapter 4). Young gravels on the West Coast are unlikely to have significant Fe sulphides, and they have abundant carbonate in the surrounding gravels so ANC is typically high. Some young gravels in Southland may have Fe sulphides (e.g. Belle-Brook; Falconer & Craw 2005). The Glenore alluvial gold mine (South Otago) had abundant pyrite in young gravels. Because the presence of Fe sulphides is difficult to predict, the rocks need to be closely examined for indications of these minerals or their oxidation products (see 3.3.1). Likewise, any other rocks to be disturbed during mining need to be examined in a similar way as for coal mines. Acid-base-accounting analysis is necessary to some degree in all Southland settings (see section 3.3.3). Acid rock drainage issues are likely to be rare at alluvial gold mines, but where they do arise, the metal concentrations can be high. For example, elevated As (up to 0.3 mg/L) and Ni (up to 6 mg/L) were most prominent at Belle-Brooke (Falconer & Craw 2005). ARD issues can be managed using the techniques outlined in section 5.4.
- *Mercury* (Hg). Some alluvial gold has elevated Hg content (Youngson et al. 2002; Mackenzie & Craw 2005). This can develop into an environmental issue on the immediate-site scale if the gold is refined on site. In addition, alluvial gold is locally accompanied by Hg-rich heavy minerals, and natural liquid Hg has been identified at one Southland locality (Nokomai; Youngson et al. 2002). Dissolved Hg emanating from these sites is low (< 10 nanograms (ng) per litre and commonly 2-6 ng/L). Some small-scale miners still use Hg addition as a gold-saving technique. Regulation of this process has been poor in the past, and minor Hg pollution has resulted. Environmental issues are primarily related to the health of these operators, rather than to the surrounding ecosystem.

9.3 Predicted ecological impact

The impact of alluvial gold mines on streams on the West Coast is usually related to fine sediments/turbidity, although impacts related to low pH and elevated metal concentrations (AMD) may also occur occasionally. However, in Southland, impacts of low pH and metals may be more common due to the presence of acid-forming gravel deposits.

The impacts on aquatic ecosystems arising from high turbidity are largely physical in nature such as smothering of benthic organisms and reduction in light penetration. The impacts arising from high turbidity can be considered as direct and indirect impacts. Direct effects on an organism can include smothering of benthic organisms and eggs of some species, and clogging of the gills of fish. In contrast, indirect effects may include reduction in primary production (algal growth) due to decreased light penetration, and changes in predator–prey relationships due to prey species being hidden from predators.

Excluding turbidity impacts, the following flow chart (Figure **36**: Flow chart of the potential aquatic impacts arising from an alluvial gold mine, bolding indicates most likely outcome) illustrates the outcomes that may occur with an alluvial gold mine. As stated above, AMD issues are rare at alluvial gold mines; as such, the outcomes associated with neutral mine drainage (NMD Outcomes 5 and 6) are most likely to be observed (refer to section 5.3 for a description of Outcomes 1-4, and Appendix D for more details).

Mercury, when present in drainage from alluvial gold mines, typically occurs at concentrations < 10 ng/L (0.01 µg/L; e.g. Holley et al. in press). By way of contrast, ANZECC water guidelines (ANZECC & ARMCANZ 2000) set trigger values of 0.06 and 0.6 µg/L for protection of 99% and 95% of species present in an aquatic ecosystem, respectively. As such, it is unlikely that Hg from mine drainage will cause impacts in aquatic systems (see also Appendix D).

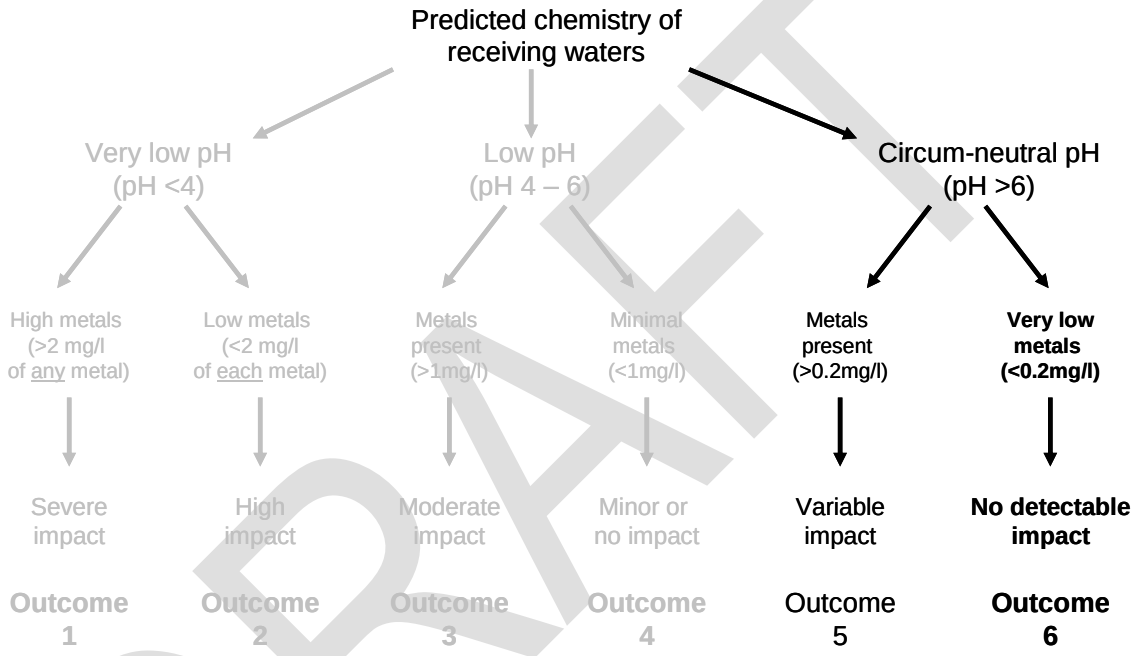


Figure 36: **Flow chart of the potential aquatic impacts arising from an alluvial gold mine, bolding indicates most likely outcome**

Outcome 5

Streams with water of about neutral pH clearly are not affected by acidity. These waters, if downstream from mines, may have high concentrations of metals such as Zn or Cu that are toxic to fish and invertebrates. Furthermore, metal precipitates, if present in high amounts, may limit the habitat or food supply of fish and invertebrates.

Outcome 6

Waters of neutral pH and very low metal content should support a full diversity and abundance of aquatic life for the area. Natural features of the catchments could affect some biota, such as waterfalls blocking migratory fish species. Mining could affect stream habitat if turbidity or sedimentation (from mining operations) were present. Otherwise, species and food webs should be comparable with those in pristine streams in the area.

9.4 Operational management and treatment

Sediment load in the mine discharge is the primary issue for mitigation of aquatic impacts. Techniques for the management of sediment loads are the same as that described for non-acid-forming coal mines and include options for management of the waste rock, tailings, and active and passive treatment techniques to manage TSS (see Appendix B).

9.5 Rehabilitation requirements

The impact of alluvial gold mines on terrestrial ecosystems is similar to those of any mining regime. However, alluvial gold mining creates an opportunity to enhance productivity of poorly drained and/or flood-prone farmland. In some places the mining (dredging) of gently-sloping alluvial landscapes creates opportunities for horticulture (vineyards) and enhancement with new wetlands and lakes from dredge ponds.

Dredging alluvial gravels raises the ground surface as gravels 'swell' between 10% and 25% when removed, depending on

the size and packing of gravels and method of extraction. The swelling effect is greater with deeper mining. When gravels were being mined to about 25 m depth at Ngahere, there was approximately 30% swell factor, which lifted this land post-mining well above the flood hazard zone. Even within a 'swelled' (higher) landscape with lowered groundwater level, a gently rolling topography is preferred in agricultural areas to help reduce the risk of surface sealing, and water ponding or perching on the surface. However, slopes should generally not be more than about 20° to minimise risk of erosion. Slopes over about 15° restrict movement of some vehicles by reducing their stability.

Mining method determines the ability to separate fine gravels, suitable for root zones, from stones and boulders. Topsoil is often thin and needs to be separately salvaged, conserved, and replaced from lower root zones (subsoils or fines). Where farmland is rehabilitated in high-rainfall environments, silty or clayey subsoils of poorly drained 'pakihi' and alluvial soils are usually best buried with the tailings, as poor drainage typically limits productivity more than summer drought. Mining will break up any impeding iron/humus pans in gravels and sands.

Where native ecosystems are rehabilitated, the retention of these poorly-drained materials may be critical to success. Alluvial gold mining in areas of native forest and pakihi can often use direct transfer of intact sods, as salvage and replacement is not limited by slope and root systems may be shallow. Rehabilitation of alluvial land to any land use needs to pay particular attention to weed control.

Alluvial gold mining often creates dredge ponds. These can enhance rehabilitated areas, as wildlife habitat, stock water supply and recreational use. Slopes into dredge ponds should be shaped to reduce danger (i.e. even slopes, particularly 5 m above and 2–5 m below the water's edge). Pond edges, contours and depths should be matched to appropriate riparian and wetland plantings. All values are usually improved with low to moderate (<12°) slopes along the waterline. Stock should be excluded from the majority of riparian areas, and particularly any wetland or riparian plantings, using a minimum 3-m buffer. The value of rehabilitated alluvial gold mines can also be enhanced by buildings; taking advantage of the enhanced topography, amenity (ponds) and lower flood risk that may be created.

9.6 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide.

Step 1. Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Management of suspended sediment ☑

Step 5. Predict mine drainage chemistry from flow chart ☐

Step 6. Predict stream water chemistry

- Predicted mine drainage chemistry from step 4 ☐
- Site hydrogeology ☐
- Background water quality ☐
- Historical mine drainage ☐
- Reactive transport modelling ☐

Step 7. Determine the potential ecological impact on the stream

- Determined from flow chart ☐

Step 8. Consider whether impacts are acceptable, and agree rehabilitation outcomes

This step may be taken internally by a mining company, during consultation with regulatory agencies, or in wider consultation processes, depending on the stage of the mine proposal

- If unacceptable go to step 7a ☐
- If acceptable go to step 8 ☐

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved.

- Consider what on-site waste-rock management techniques are relevant for your specific site ☐
- Consider options for tailings management ☐
- Consider active or passive treatment options ☐
- Consider compensation and/or off-site mitigation ☐

Step 9. If acceptable, design monitoring programmes and plan rehabilitation processes ☐

10 SUCCESSFUL REHABILITATION

10.1 Introduction

Mine rehabilitation is the primary way recovery of the mine-disturbed areas is achieved and long-term effects of mining are minimised. Specific outcomes should be planned and are ideally based on reference sites, but will allow flexibility in the way outcomes are achieved and where they may be placed on the post-mining landscape. Rehabilitation plans need to be flexible because mine operations change with changing costs and ore prices. Improved and site-specific techniques may be developed, especially in larger, longer-term developments or in smaller operations with changing equipment. Planned outcomes will include short-term and closure criteria. These are established with input from landowner, administrators, and regulatory authorities. Short-term criteria often include safety, topography, stability (erosion and sediment control and geotechnical stability) and initial vegetation establishment. Longer-term criteria may include productivity (for farmland), biodiversity and ecosystem resilience (for conservation land).

At all sites, rehabilitation options are heavily influenced by what resources can be salvaged, stored or directly (immediately) reused during the overburden stripping process (section 3.2). This section contains a generic flow chart that shows both the common resources available for rehabilitation and what influences their salvage and reuse. Rehabilitation resources, especially topsoil, must be salvaged prior to overburden removal and stored in accessible, protected (not trafficked) areas. This requires ongoing optimising of stripping, mining and rehabilitation schedules. Direct transfer is the most effective method to rehabilitate native ecosystems and control erosion through instant vegetation cover. Its use is limited by mine scheduling because final (backfilled) landforms with NAF substrate have to be available. Early rehabilitation is vital to demonstrate the on-site capability and allow site-specific techniques to be developed through adaptive management.

Processes to achieve selected rehabilitation outcomes are described in the following sections; rehabilitation to pasture, plantation forestry and native ecosystems. Other land uses are possible, for example, cropping and horticulture (viticulture), residential housing, recreation (historical mine relics, mountain biking) and public amenity, but are not covered in this document. Maintenance and monitoring are integral components of successful rehabilitation (Chapter 11).

10.2 Identifying constraints and opportunities in selecting end uses

A wide range of post-mining land uses and aligned closure standards may be possible at specific sites. Flexibility of outcomes are greatest before mining begins. Flexibility dramatically reduces as mining removes vegetation and topsoil, particularly if these are not salvaged. Double handling and recontouring of overburden is expensive, so the placement and topography of overburden dumps constrain land use options. PAF is also a critical constraint. PAF needs to be identified, segregated, and managed so any influence of PAF is excluded from the root zone. Where PAF is not able to be treated, or influences the root zone, rehabilitation is likely to fail (see section 5.5).

Overarching rehabilitation aims are often to restore land use capability, i.e. the range of potential land uses. This is traditionally defined by the ability to cultivate (or not), the range of crops able to be produced, and the limitations to productivity (drainage, erosion, fertility). However, this definition does not necessarily apply to native ecosystems. Consultation is an important component of planning post-mining land use but is not addressed in this framework.

10.3 Undertaking rehabilitation to pasture systems

Pasture is a common rehabilitation outcome (Figure 35). The two highest priorities are a fully productive pasture (within 2–5 years) and consistent erosion control to avoid sediment entering surface waters, particularly streams or rivers. These objectives mean pasture management in the short term should develop a dense sward with deep root systems supported by high post-grazing (residual) pasture dry mass and high tolerance to pugging.

Figure 37: **Rehabilitated pasture and fenced contour drain planted with native shrub and tree species, Waikato coal mine.**

This section describes what to consider when agreeing or deciding closure criteria for general pastoral farm rehabilitation, and the most effective methods to achieve the criteria. Monitoring and recording the outcomes of rehabilitation are important to tailor site-specific outcomes, as they provide evidence that guides adaptive management. Adaptive management means that results of early rehabilitation at an individual farm or site scale are used to modify rehabilitation methods at that site, usually leading to better outcomes.

A good process to plan and agree the outcomes from rehabilitation to pasture is important to ensure that landowners (where that is not the mining company) or regulatory authorities have a clear understanding of what is achievable and anticipated to be achieved (Figure 38: Process for planning and agreeing rehabilitation to **pasture**.
).

Several steps are required to achieve successful rehabilitation to pasture (Figure **39**). In particular, creating a post-mining landform with slopes that provide adequate drainage, including at least 200 mm of rooting depth above the water table, underpins successful pasture rehabilitation. Identifying what resources are available on site, and nearby, that can be salvaged, stored and reused to create a suitable root zone is a critical first step. High fertiliser applications and/or organic amendments (see Dairy case study) are usually needed to establish vigorous, dense pastures. Inadequate (late or lax) management of weeds is a common cause of failure, particularly where gorse is in the seed bank. Once pastures are established, the ongoing management of grazing or topping in a way that encourages deep, dense root growth rather than maximum grazing removal is needed to rebuild soil structure and organic matter (resilience). As organic matter increases, pasture dry matter production increases, less nitrogen fertilisers are needed and pasture is more resilient to drought and compaction.

1 Decide what is possible at the site

A. Site assessment

What are current site conditions?

Land-use capability measures versatility (e.g. cropping potential and the main limitations to farm production).

Current productivity is a useful guide (stock units/ha, Dry Matter/ha/year), seasonality of growth may be important.

Consider trees, shelter belts, ponds, drains, watercourses and their values and farm infrastructure: fences, races, sheds, reticulated water, silage pits and stand-off areas.

B. Site assessment

What are current site limitations? These may be reduced or removed in rehabilitation:

- Poor drainage due to perched water table (iron pan) -> mining breaks this pan
- Poor drainage due to flat site with slowly-permeable soils (pakihi) -> mining can create humps and hollows Floods frequently -> deeper gravel mining may raise the land
- Chemical infertility and low-producing pasture species -> rehabilitation can establish high-producing pasture
- Gorse and weeds -> heavily-infested soils may be replaced if adequate alternatives and moderate to high fertiliser rates are used

C. What resources are available?

The mine machinery and equipment influences what resources can be salvaged and separated, e.g. size of fines, ability to access slopes.

See **Fact Sheet 5.**

Is any mine infrastructure useful to retain after mining? E.g. access roads (races), power supply, dredge pond (with reshaped batters), culverts and bridges, sheds .

Are farm resources useful ? E.g., seeding equipment, light stock for grazing new pasture, dairy shed effluent, hay bales for erosion control.

D. How does the mine plan influence resource availability and use?

When will areas be available to separately stockpile resources for rehabilitation?

What areas are likely to be rehabilitated each year, and when? (This can help farm operations.)

2. Agree general and specific success criteria to guide rehabilitation

General criteria are needed to allow mine flexibility and guide decisions when mine plans need to change. Farm/landscape-scale criteria and paddock-scale criteria should be developed.

A. Farm-scale criteria

- Location of mine access, mine infrastructure, general location of ex-pit stockpiles or overburden dumps
- General location of farm races, gates, stand-off areas/feed-pads after mining
- General location and width of shelter belts, water courses (with riparian buffers) and crossings
- Specific location of areas that are not to be mined

B. Paddock-scale criteria

- General paddock sizes and shapes
- General land contour (minimum and maximum slopes and lengths, especially if hump and hollowing)
- General water reticulation density and capacity
- Standards for temporary, electric, permanent fences
- Treatment of high-wear areas such as gateways and troughs (may use gravels, not topsoil in these areas)

3. Agree administrative and health and safety criteria

- Treatment of high-walls and ponds (sediment ponds, dredge ponds, new amenity ponds) to make them safe
- Agreement on access and exclusion times for shared areas (e.g. during milk tanker or school bus runs), or places
- Agreement on access and exclusion times or places in mining areas (e.g. operational zones, water treatment areas)
- Identify responsibility for security of fences and gates
- Agreement on any assistance with rehabilitation, pasture management, e.g. stock or machinery for pre-stripping grass removal or post-rehabilitation grazing. Post rehabilitation grazing needs to be done by light stock for short periods
- Agreement on signoff for first grazing (stock & vehicle safety) and closure signoff
- Agreement on process to resolve disputes, e.g. contracting an independent farm advisor

4. Define the standard root zone (or zones)

The standard needs to be flexible enough to adapt to site experience and changing resource availability, e.g. include minimum, target (optimum) and maximum values.

A typical standard root zone for pasture has:

- 300 mm rooting depth (free-drained, uncompacted) of which 100 mm is topsoil
- pH >5.5 and Olsen Phosphorus 20–30 ug/g for dairy pasture, >10 ug/g for sheep pasture

When topsoil is thin:

- the underlying root zone needs to be sandy rather than gravels to improve water and nutrient supply
- nitrogen fertiliser needs will be higher until organic levels build up

It is helpful to identify remedial options for any areas that do not meet the standard.

Monitoring is most effective if it identifies poorly performing areas early. This is important for remediation options that require earthmoving.

5. Define acceptable pasture quality

The standard may include specifying:

- pasture species
- acceptable weed cover and
- acceptable pasture production

If soil physical conditions and pasture establishment are acceptable, pasture production and composition is influenced by fertilisers, and grazing or mowing frequency and severity.

When drainage is improved pasture production may be higher in spring and lower in drier months.

6. Agree specific rehabilitation criteria for bond release

This may include specific criteria for rehabilitated farm infrastructure, e.g. races (width, fencing, water-table, surfacing - ideally refer to an example from the farm, or cross-section), fencing standards and gate standards.

It may be useful to agree to post-closure .

Check site safety for steep drops, vehicle trafficablility and dangerous areas (sediment ponds with soft sludges). Eliminate hazards, e.g. fill in sediment ponds, recontour steep drops, especially drops into water, or mitigate hazards, e.g. excluding vehicle access using boulders, fenced ditches or other contouring.

7. Agree success criteria (closure criteria)

These are measurement or descriptions that when met, mean the area is handed back to the land-owner or manager. At this stage the owner is responsible for maintaining the rehabilitated areas.

8. Agree bond or compensation

Compensation may include 'in kind' parts, i.e. upgraded access, races, culverts, water supply, and/or riparian protection.

Linking compensation to the length of time land is removed from production incentivises effective rehabilitation.

Figure 38: Process for planning and agreeing rehabilitation to pasture.

1. Identify rehabilitation resources and constraints

- Calculate approximate volumes of suitable materials available and volumes needed
- Identify resources that can be produced Run of Mine, e.g. boulders, fines for root zone, gravels for surfacing races

2. Strip

- Identify, mark, and protect riparian zones and agreed no go zones , e.g. remnant forests, wetlands, buildings
- Survey weeds and ID weedy areas; decide on management , e.g. spraying, separate stripping and stockpiling or disposal
- Fell/remove trees and direct transfer; remove and salvage fences, troughs, etc.
- Identify stockpiling areas and prepare these areas with access., firm bases, cut-off drains, sediment control and fences
- Reduce pasture mass by intensive, close, grazing immediately prior to stripping, or herbicide 2–3 weeks earlier
- Preferably use low ground-pressure machinery to strip topsoil separately from subsoil
- Strip and stockpile free-draining materials that will be used in root zone separate from general backfill
- Remove poorly-drained or hostile subsoil and overburden. Dispose in suitable backfill areas below root zone

3. Stockpile and conserve root zone

- Separately stockpile topsoil, subsoil, and other materials for rehabilitation in accessible areas
- No surface water should enter stockpiles ; reduce 'dirty water' needing treatment by diverting clean water away from stockpiles
- Create soil stockpiles by back-dumping to minimise compacting the soil . Do not drive over stockpiles
- If stockpiles will be unused for > 3 months, sow with grasses or legumes to conserve quality

4. Reinstate landform or create modified landform

- Place overburden to minimise the amount of reshaping (bulldozing) and re-handling required
- Identify and mark watercourses and water detention areas; confirm flood capacity is adequate;
- Reinforce flood zones and water-courses with rock armouring if necessary; install culverts and crossings
- Check site safety : remove steep drops and dangerous areas , e.g., soft, deep sediment or mitigate hazards, e.g. excluding vehicle access using boulders, fenced ditches or other contouring

5. Create root zone

- For pasture: Create a free-draining root zone of minimum 300 mm depth over compacted gravels or overburden. The root zone should include at least 100 mm topsoil, unless fine sands and silts are substituted (silts increase risk of surface sealing)
- Trees (shelter belts & native plantings) grow best in a 1 m root zone depth to ensure stability and reduce stress
- Minimise compaction of topsoil by avoiding handling in wet periods and using light or low ground-pressure tractors
- Soil tests will confirm initial (capital) fertiliser and lime recommendation for good pasture growth

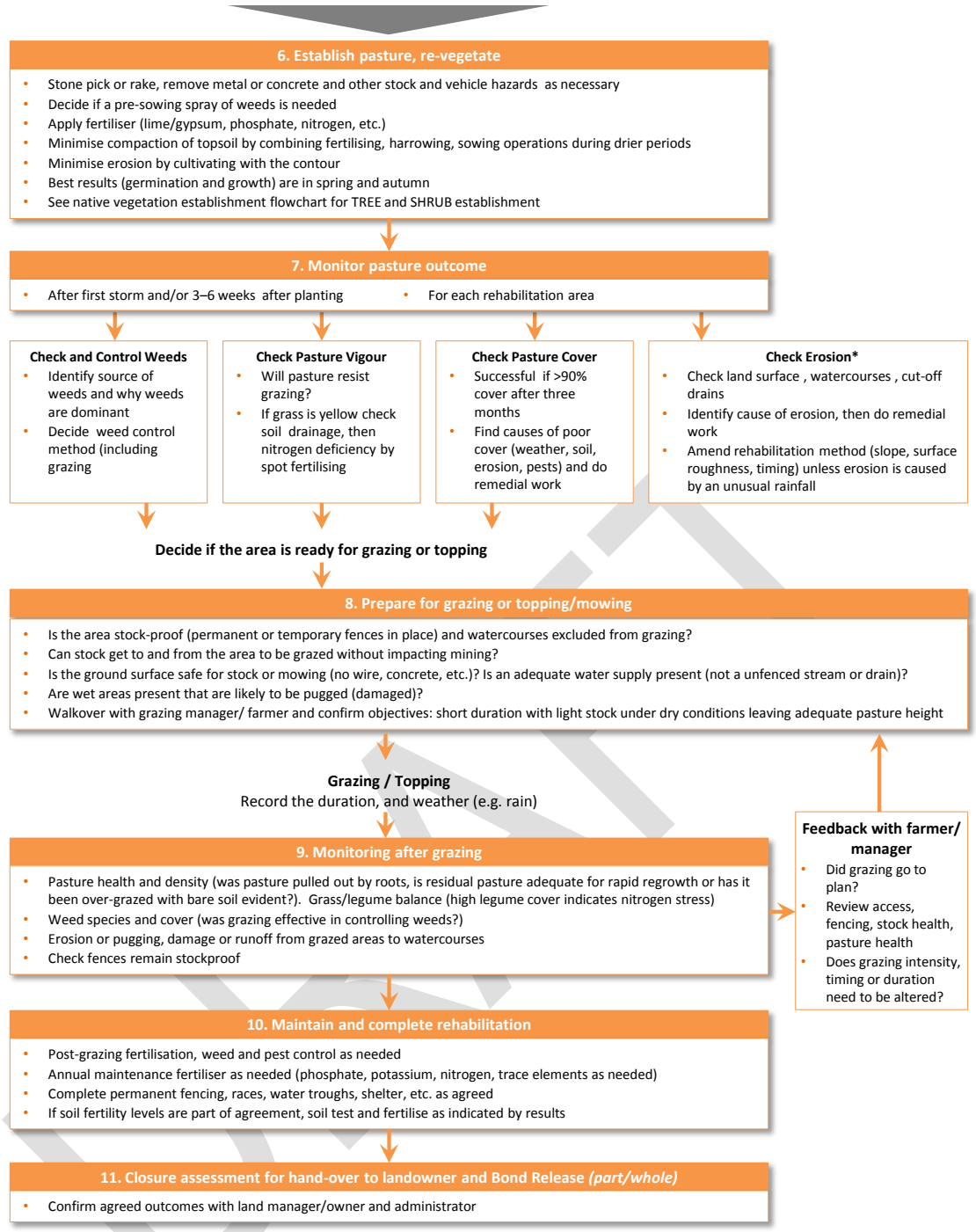


Figure 39: Implementing rehabilitation to pasture.

Case study: Organic dairy farm

This demonstrates the special requirements for rehabilitation of dairy farm land to a certified organic system. Plant species selection: Farmer was consulted to confirm that these conformed to the organic certification and the more diverse pasture species composition than is standard. Depleted nitrogen and phosphorus levels in rehabilitated land were not able to be mitigated using conventional inorganic fertilisers. Instead, fertility levels were built up using chicken manures and composts.

Weeds were not able to be controlled by agrichemical herbicides (e.g. glyphosate). Neither were grass grubs, Argentine stem weevil and other insect pests able to be controlled using agrichemicals. The mining company engaged the organic farmer to do the farm management and supply 'organic-compliant' stock. This reduced the risk of practices that threatened organic certification being gained. The adoption of organic-compliant rehabilitation techniques influences the cost structures.

10.4 Production forestry

Mine rehabilitation to exotic forest (*Pinus radiata*, *Cypressus lusitanica* or *Eucalyptus*) (Figure **40**) is relatively low-cost, low-maintenance and low-risk compared with pasture rehabilitation. Forested land has a high probability of delivering runoff that meets water quality criterion as long as a protective groundcover is initially established. Radiata pines are a relatively low maintenance option once established, as most weeds are largely suppressed by the tree canopy shade, and most trees have a much lower nutrient requirement than pasture.

Figure 40: Rehabilitated coal mine with plantation forest, Waikato coal mine.

Successful rehabilitation to plantation forestry (Figure **41**) is largely dependent on the quality and depth of materials in the root zone, drainage and slope. Outcomes are likely to be favourable where:

- Topsoil is replaced. Topsoil with high loads of weed seeds or propagules may still be used if covered with a layer of gravels through which trees can be planted.
- The depth of potentially imperfect to well-drained root zone is more than 1 m.
- Slopes are greater than 1:20; as this helps ensure adequate drainage.

Outcomes are likely to be poorer where:

- Topsoil is absent – due to insufficient nitrogen and phosphorus, and increased susceptibility to compaction (infiltration rates and airfilled pore space is likely to be lower)
- Depth of potentially imperfect to well-drained root zone is less than 1 m. Water may tend to perch or pond on the interface with compacted overburden
- Slopes are shallower than 1:20; as this increases the risk of inadequate root zone drainage.

Small mine sites within existing plantations may not be large enough to justify the additional costs required to take a small area that is out of synch with the larger forest, to maturity. Such small sites may be more valuable to the forest owner as part of the infrastructure needed for harvesting / log-marshalling (and sediment control) or firefighting.

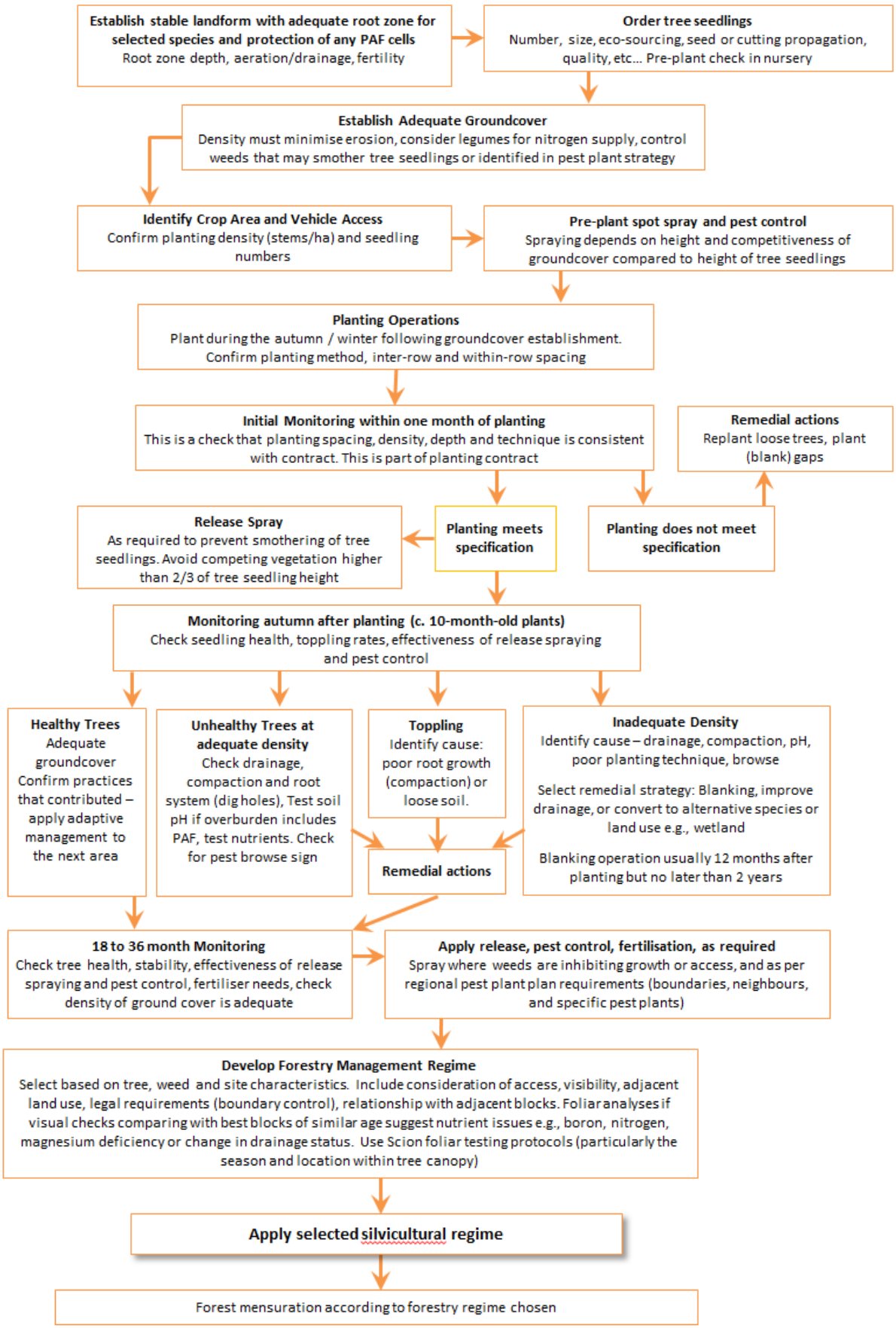


Figure 41: Process for rehabilitating mined areas to production forestry.

10.5 Rehabilitation to native ecosystems

Rehabilitation of native ecosystems after mining is the preferred option in many locations in or near conservation lands. Native ecosystems are also used within farmland to provide amenity, shelter, water supply or buffers to lakes, streams and wetlands. Where rehabilitation aims to establish conditions that will allow development of similar ecosystems to those occurring pre-mining, it is important to understand the soils, landforms and drainage that underpin establishment of the pre-mining ecosystems. Where soils are not replaced, the build-up of topsoil and leaf litter takes decades. A multi-layered forest structure also takes decades or years to develop. However, rehabilitation techniques developed with the West Coast mining industry since the late 1970s have shortened recovery times, particularly for shrublands (below 3 m to about 5 m height), wetlands, and tussock grasslands. The development of 'direct transfer', a method where sods of intact plants and soils are moved intact from stripped areas to rehabilitated areas has produced some outstanding results. Direct transfer can avoid planting, provides immediate erosion control, and dramatically increases plant and invertebrate (insect, spider and snail) diversity and conservation.

The best results happen when mine planning maximises the quantity and quality of plants (common names only have been used in this report see the *Nature Services* website for more information on species selection and their placement across the landscape: http://natureservices.landcareresearch.co.nz/app/purpose/33/), seeds, soils (Figure **42**) and logs that are salvaged and reused for rehabilitation. In most cases the most successful landforms are stable, with variation in contour that creates a variety of suitable drainage and environments to underpin a resilient plant community. Poor performance and rehabilitation failure is generally due to inadequate weed/groundcover control, or highly acidic (pH < 4) or compacted growth media, or inappropriate drainage. Use of pasture species to establish an initial cover (primarily ryegrasses, browntop, Yorkshire fog, and relatively acid-tolerant lotus species) has resulted in inconsistent medium-term outcomes. Pasture species are generally very effective at stabilising loose soil against erosion, and building up new soil structure and organic matter. However, pasture can supress native plant growth and natural regeneration, particularly of herbs, and where pasture encourages grazing by deer, goats or other mammals. Lotus, particularly, smoothers native plant seedlings in summer. Most New Zealand native plants are poor early competitors due to relatively slow germination and growth rates compared with widespread exotic plants such as pasture grasses, legumes, gorse, blackberry, broom and Himalayan honeysuckle, among others. Consequently, introduced plants tend to dominate many lowland mine sites, at least in the short to medium term.

Figure 42: The impact of soil conditions. Five-year-old mānuka and kanuka planted at the same time at the same site but different root zone: left, topsoil, right, overburden.

Native New Zealand lowland forest typically has near-surface feeding roots that are sensitive to compaction. Topsoils are particularly beneficial to root development as they have high levels of organic matter, often deep leaf-litter layers, with many species having mycorrhizal (fungal) associations that help plants access nutrients (including beech, manuka and kanuka). Understorey plants and animals have developed in moist, humid environments protected from temperature extremes and wind and only moderate soil moisture deficits in summer. Mine sites provide contrasting and challenging conditions:

- High site exposure to light and wind, which cause wide fluctuations in temperature and humidity
- Rooting media low or absent in organic matter (thus poorly buffered), without mycorrhizas and soil fauna, and often compacted

- Faster-growing, exotic, herbaceous and woody species
- Smooth, dense surfaces designed to shed water.

The replacement of topsoil in a way that creates uneven (rough and/undulating) surfaces, enhanced with rocks or boulders, logs, and forest 'slash' (felled branches), assists regeneration. It creates many protected and relatively stable, humid microsites.

This approach is particularly important in the absence of planting of seedlings or direct transfer.

The following flow chart (Figure **43**) identifies key steps and methods for rehabilitating native ecosystems.

1. Identify rehabilitation resources, constraints, and sites

- Calculate approximate volumes of each resource available and the volumes needed for rehabilitation.
- Avoid stockpiling when practicable through scheduling simultaneous stripping and replacement. Allow 10% wastage if stockpiling cannot be avoided; overburden is likely to swell by 10–20%
- Identify resources that can be produced Run Of Mine
- Identify suitable receiving sites for direct transfer, transplanting, nursery planting, or hydroseeding.
- Conduct wildlife surveys and salvage required native fauna for relocation

2. Stripping

- Identify, mark, and protect (if the mine plan allows), riparian zones and agreed no go (ecological protection) zones
- Survey weeds; decide on pre-stripping management (spraying, separate stripping and stockpiling or disposal)
- Identify stockpiling areas and capacity. Prepare these areas with firm bases, cut-off drains, sediment control and (at larger sites), signage or fencing to exclude vehicle traffic
- Fell/remove large trees for direct re-use, mulching, or stockpiling
- Strip vegetation and tree stumps with root plates as direct transfer
- Strip topsoil separately from subsoil if using subsoil for rehabilitation
- Strip and stockpile potential plant root zone materials separately from general backfill
- Identify and dispose of acidic rock so it won't contaminate the plant growth media or develop AMD

3. Stockpiling

- Separately stockpile topsoil, subsoil, slash, logs in accessible areas protected from machinery and weeds
- No surface water should enter stockpiles; divert clean water away from stockpiles to treatment ponds
- Create soil stockpiles by back-dumping, not compacting or driving over the soil
- If stockpiles will be unused for >12 months, spread mulch or sow a non-weedy cover crop

4. Reinstate landform or create modified landform

- Place overburden to minimise the reshaping and re-handling
- Design landforms that blend into the landscape and replicate erosion-resistant slopes with small catchments
- Design streams and water detention areas; confirm flood capacity and overflows to the design event
- Reinforce water-courses with rock armouring if necessary; install culverts and crossings
- Create a safe site: defined, safe access and turnaround/parking areas if needed. Pay attention to steep drops and water/sludge hazards (sediment ponds)

5. Create planting media

- Spread topsoil over subsoil or other suitable plant growth overburden
- Deeper root zones (≥ 0.5 m) are needed for tall forests; thinner root zones (≤ 0.3 m) for pakihi and rushes
- Leave a surface with varied micro-topography, not a smooth surface

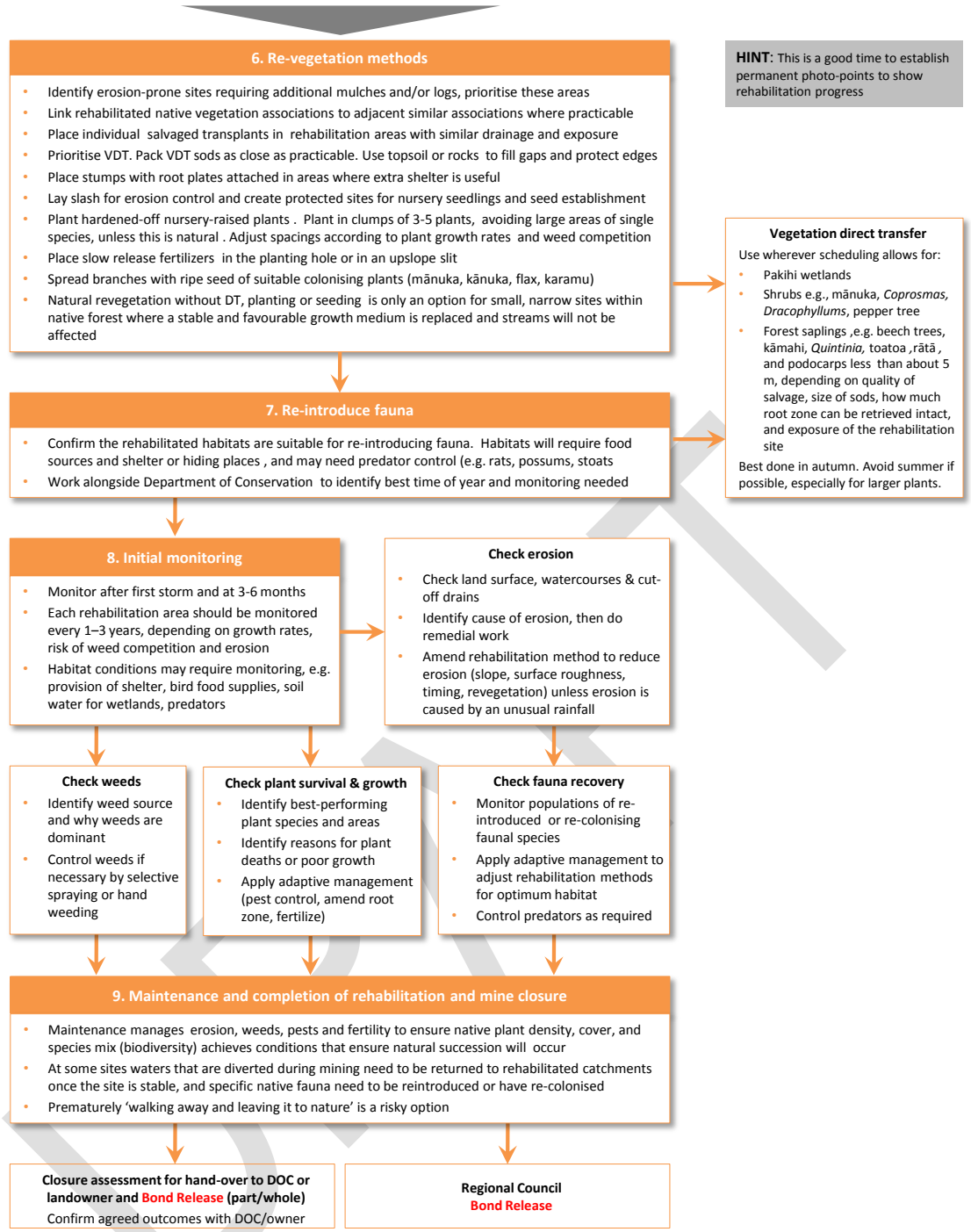


Figure 43: Processes for undertaking rehabilitation to native ecosystems.

10.5.1 Direct transfer

Direct transfer is the most effective method of rehabilitating ecosystems (plants and insects and soil with its biota) in most circumstances. When intact plants are extracted with minimally-disturbed root systems as large sods, and immediately placed onto suitable backfill sites, dieback can be minimal and recovery rapid. The area of direct transfer is typically limited by the availability of rehabilitated backfill on which to place excavated sods, and accessibility of suitable material. This means there is almost always a deficit of material to use for direct transfer.

The main use of direct transfer is to establish vegetation cover that closely resembles that existing prior to mining. However, it can also be used to achieve a range of other outcomes (Figure **44**), including: boosting plant and invertebrate diversity within larger conventionally-rehabilitated areas (planted with seedlings); establishing a target density of long-lived tree species such as beeches and podocarps with their associated fungal communities; connecting ecosystems on either side of a mine site; buffering high-value ecosystems on the edge of a mine site or along streams/lakes; screening views of operational areas; controlling erosion; and identifying boundaries of rehabilitated areas.

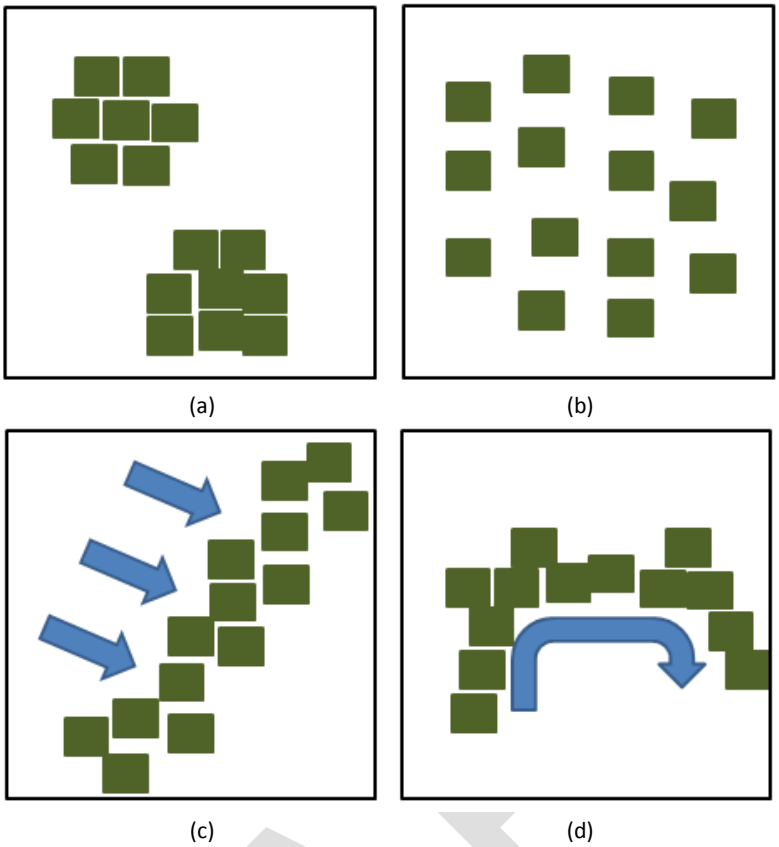


Figure 44: **Direct transfer patterns used to achieve alternative goals: (a) to create islands as sources of plant and insect diversity; (b) to establish a target density of trees; (c) for erosion control (arrows showing the direction if surface flow); and (d) to protect the edge, and shade, a pond or small watercourse.**

Key factors for successful direct transfer (Figure **45**) are:

- Direct transfer is particularly applicable to wet environments not subjected to significant droughts.
- Autumn is preferable to summer/spring, but avoid days when it is raining.
- Engineer-construct suitable landforms, particularly regarding surface hydrology (e.g. water tables, drainage, sediment ponds) to minimise the interaction of stormwater with the directly transferred sods.
- Relieve compaction on overly compacted NAF base materials.
- Preferably add a thin (20–30 cm) layer of salvage soil over the NAF rock to provide a rooting medium beneath the sods.
- Salvage sufficient depths (typically 0.3–0.5 m) of 1–2 m^2 topsoil sods with the vegetation, to conserve the root plates and minimise disruption to the root systems.
- Avoid double handling (temporary stockpiling of vegetated sods).
- Use specialised machinery best suited for direct transfer, i.e. hydraulic excavators, with specially designed flat-bottomed buckets, and trucks with flat, tipping decks. Smaller excavators are best suited to unloading sites to spread out sods, up-right vegetation, and fill in gaps with topsoil or rocks.
- Train and certify machinery operators in direct transfer operations.
- Pack the 'sods' as close as practicable to minimise gaps.
- Replace salvaged topsoil into gaps in the directly transferred sods and pack soil along the edges to avoid drying-out edge effects.
- In naturally rocky environments, place large rock boulders or slabs (1–3 m^2) before adding the sods around them. Or, where practicable without trafficking over the already placed vegetation, place boulders into gaps. The boulders/slabs are important for creating micro-niches for faunal habitat diversity.

Figure 45: Direct transfer of lowland regenerating shrubland with rimu, kahikatea and other native trees, West Coast.

10.5.2 Selecting plant species

Where planting is to be undertaken, naturally local and locally-sourced native plant species, for example 'free' plants from the mine site taken from areas about to be stripped, should be used. On the West Coast, do not plant beech where beech is not naturally present (the Beech Gap). If using seedlings, including a variety of native plants tolerant of site conditions increases the likelihood of success. Where plants may be grazed, a high proportion of plants unpalatable to deer and possums (e.g. manuka or kanuka) should be used. Combi-guards or similar can be used to deter rabbits and hares until plants are tall enough to resist browse. Bands of fire-resistant plants may be useful adjacent to public roads, and to break up larger sites. The most suitable plants for initial rehabilitation planting are generally tolerant of high light and exposure, and can grow rapidly (karamu, koromiko, flaxes and toetoe), especially where competition from pasture or weeds is likely. This means forest revegetation may use plants that have low natural abundance in undisturbed areas (toetoe, tutu, koromiko).

Planting nursery-raised or salvaged native seedlings (<1 m height) is generally restricted to mine sites that operate for more than 3 years and/or have few weeds. Seedlings usually need at least 3 years of weed control to prevent smothering by faster-growing plants such as gorse, broom, and Himalayan honeysuckle. Some sites are largely free of weeds. If competition from short-term groundcover can be managed, native seedlings may require little maintenance post-planting. Where dense weed or pasture growth is expected, the competitiveness of native seedlings can be encouraged by planting taller seedlings (in sheltered areas, i.e. places not susceptible to wind), direct transfer of root plates (stumps) containing native seedlings, or clustering intact sods (direct transfer) with minimal gaps between sods in clusters.

10.5.3 Selecting plant or direct transfer sod density

The density of seedlings or sods planted depends on factors listed in Table 4, including the growth rate of plants relative to competing weeds, the extent of natural regeneration and the purpose of planting.

Table 4: Factors influencing density of planting or placement of direct transfer sods

Factors requiring high initial plant density	Factors favouring low initial plant density
High plant mortality or year-to-year variation in mortality	Low plant mortality, consistent across seasons
Low plant-growth rate	High plant growth rate
Low natural regeneration	High natural regeneration (from direct transfer, slash or topsoils, etc.)
Slow natural seeding and spread	Fast natural seeding, spread or establishment of cuttings leading to many new plants (depends on plant species, soil conditions and suitable micro-climates)
Low likelihood of detecting, and replanting due to difficult access, lack of suitable plants, completion of mining, inadequate monitoring, etc.	High likelihood of detecting and replanting any gaps
High erosion potential	Low erosion potential
Short time until closure	Long time to closure
Highly variable climate and/or growth rates	Relatively even growth rates across years
High weed competition	Low weed competition

The timing of rehabilitation with respect to mine closure and committed active maintenance period may also influence planting density. In mines with a 10-year life, early plantings may be sparser and use smaller plants to reduce up-front costs, but allow for more intensive weed and pest control. Later plantings may use higher densities and faster-growing plants to reduce maintenance costs when the mine is no longer generating revenue. Closure usually requires vegetation to be established to a condition that ongoing maintenance is minimal. A high planting density usually speeds development of native plant cover with shorter time to closure than a low planting density.

Predicting the growth rates desirable plants and undesirable plants (weeds) at a specific site is often difficult, as the replaced soil conditions have an enormous influence, and often vary greatly across a site. Conditions affecting growth rates are outlined in Table 5. Each factor can be manipulated to improve rehabilitation outcomes.

Table 5: Factors that influence the growth rate of plants

Factors that increase plant growth	Factors that slow plant growth
Topsoils present. Topsoils are generally the most favourable root zone as they have organic matter that supplies nitrogen and stores moisture (Figure **46**).	Topsoil absent. Root zone with low levels of organic matter and/or low water holding (coarse), highly acidic media or other low fertility issues in the planting media.
Deep root zone	Shallow or compacted root zone – plants have restricted root systems and soils hold less water and nutrients.
Sheltered sites. Shelter can be created by increasing surface roughness using contouring, mulch, logs, and boulders.	Exposed sites.
Low transplant shock Plants that are tolerant to transplanting include flaxes, red tussock, coal measures tussock.	High transplant shock. Occurs with adverse climate (drought), poor plant storage and/or planting technique, slow extension of roots from potting mix. Some plant species are susceptible to transplant shock.
High genetic potential – plants like manuka, kanuka and Hebe salicifolia grow quickly. Nitrogen-fixing plants such as tutu can also grow rapidly in infertile sites.	Low genetic potential – plants that naturally grow slowly.
Plants established from seeds tend to have larger root systems.	Nursery-grown plants are more susceptible to transplant shock, especially if poorly hardened-off or with large top growth and small root systems.
Low competition for light, water, and nutrients.	High competition for water, light, and nutrients. Fine wood mulches can 'compete' with plants for nitrogen, especially if nitrogen levels in the underlying soil are low.
Low losses of leaves and roots from browse or disease.	High leaf loss from pest animals; death of roots from root diseases (e.g. phytophthera).

Some factors that influence the rate and abundance of natural regeneration are similar to those influencing overall plant growth in general; however, the size, shape and surfaces created in rehabilitated areas are key influences (Table 6).

Table 6: Factors influencing success of natural regeneration

Factor affecting regeneration	High probability of fast regeneration	Low probability of regeneration, slow regeneration
Size of site	Small	Large (>20 ha)
Shape of site, especially area-to-edge ratio and length of edge	Narrow	Square or round
Growth rates of adjacent vegetation	High	Low
Variety of slopes and drainage status	Moderate to high	Low, particularly if flat
Surfaces present that favour propagule growth: soil-like, low moisture stress, many protected and stable sites, favourable root zone	Abundant	Infrequent
Method of rehabilitation	Direct transfer, minimally-handled slash	Coarse or acidic overburdens with no slash or mulch
Probability adjacent vegetation will produce seeds every year and can establish in stressed, highly exposed sites, e.g. manuka, kanuka, hebe, toetoe	High	Low, most species require sheltered forest floor
Contribution of birds to seed dispersal	High, perching and roosting sites present	Low

Factor affecting regeneration	High probability of fast regeneration	Low probability of regeneration, slow regeneration
Weed species present that are likely to smother native plants in short term	Absent or sparse	Present and numerous
Weed species present that are likely to persist and regenerate long term, e.g. long-lived seed banks and unstable site, shade-tolerant weeds	Absent or sparse	Present and numerous
Use of a dense non-native grass or legume cover to quickly stabilise erodible sediments	No	Yes
Likelihood of browse, especially if a high proportion of regenerating plants are palatable (large-leafed)	Low	High
Maintenance (e.g. releasing or pest control) required to achieve acceptable outcomes	Low	High
Likelihood of disturbance, particularly from fire	Low	High

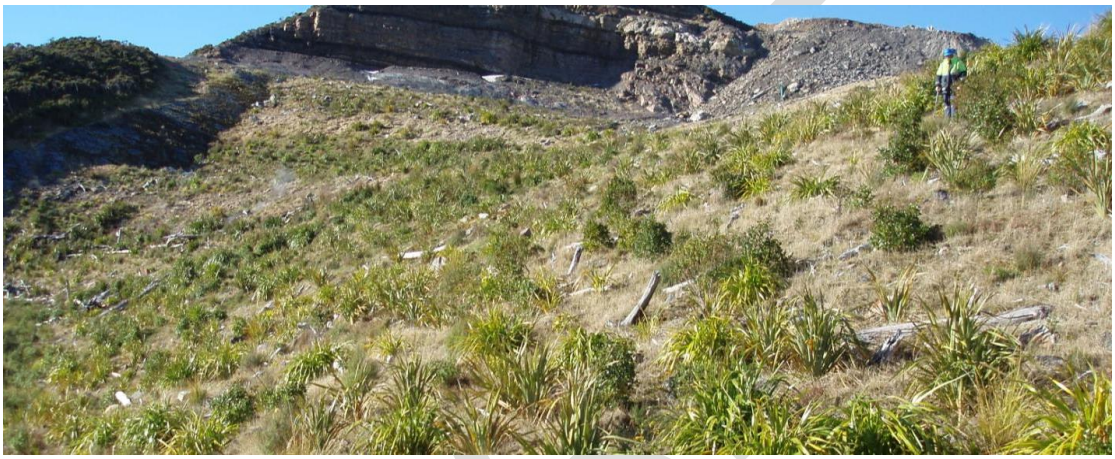


Figure 46: Coal Mine rehabilitated using seeding of browntop grasses and planting of native seedlings into respread topsoil over overburden, West Coast.

10.5.4 Using organic mulches and slash

Organic mulches, including slash, branches, and logs, are valuable resources that can usually be salvaged from areas being stripped. They are used to control erosion, preserve topsoils and rooting material, and create protected microsites for seedling germination. When used fresh, some will regenerate new plants. The uses of mulches depend on the size of individual pieces, the proportion of soils, fines and pieces of plants that are likely to sprout. Mulches can be grouped into three general types, as follows. Each type has different uses.

- Smaller branches with moderate to high proportion of leafy material that may be mixed with soil or fines
- Large branches, logs and stumps
- Woody material that has been mulched, ground, or chipped

As the percentage of fines increases, the mulch is more useful as a plant growth medium and less effective at supressing weeds. Coarser mulches last longer, are more stable (resisting erosion), create more sheltered microsites, and provide greater insect habitat. However, a dense mulch of large piece sizes is difficult to plant through, and more difficult to walk or drive over safely. As the proportion of plants likely to sprout increases, handling should be reduced.

Mulches are likely to have a variety of uses at any individual site. They are most often used to cover erosion-prone surfaces, except floodplains. Larger, heavier wood (logs) reduce wave erosion around the edges of ponds, and may be buried or securely tied into small watercourses. When used in these roles, logs enrich habitat for aquatic invertebrates by providing shelter and substrate. Coarse mulches such as logs and tree heads improve conditions for establishment and growth of plants by creating protected microsites. These large mulches are unlikely to create nitrogen stress because they break down slowly. Logs also later become places for some native plants to establish. Logs (and boulders) are also useful to control access to areas, including defining edges of rehabilitated areas. Only larger wood and logs are useful to control access.

Mulches are typically spread at depths from less than 5 mm to 150 mm, depending on the results wanted, the type of mulch, and the slope characteristics. Deep mulches are likely to be less stable on steep slopes, particularly if slopes are smooth and water flows laterally over the surface. Stability of mulch can be enhanced by placing on a rough surface and/or placing on a surface with freshly-spread soils or rooting media. When seed-bearing branches or mulch is used to establish plants, a very thin layer is used, and this may only cover 30% to 60% of the area. Mulch applied for erosion control (Figure **47**) may also be

thin, particularly if the mulch is blown onto slopes. In contrast, where mulches are used to supress weeds, establishing a minimum depth of 70 mm and up to 150 mm depth is used. A similar, or greater thickness may be used where the slash or mulch has a high proportion of soil and is used as a root zone. If there is a surplus of mulch, a maximum depth is applied rather than wasting the resource. If the mulches are coarse, planting may need to be delayed until the mulch has broken down enough to create a suitable rooting medium.

Mulches can be applied by blowing, or placing using excavators or bulldozers with a variety of attachments. Safety is a primary concern, particularly where logs are used on slopes. Safety exclusion zones should, therefore, be established during spreading operations where logs could move onto roads or work areas. Bunds and/or boulders can also be used to restrict movement of large pieces. Reducing log length and ensuring surfaces are rough before mulch is placed also increase stability of larger mulches. Dense, coarse mulches can also present trip, slip, and fall hazards. Risks can be reduced by leaving access corridors from roads. These corridors may also enhance efficiency of planting, monitoring, and maintenance. Fascined branches are usually used along slopes to maximise their contribution to sediment retention; this also reduces slip hazards for people.

Figure 47: **Slash spread across backfilled slopes reduces the risk of erosion.**

10.6 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide. The following checklist refers only to water quality issues associated with pH and dissolved contaminants; suspended solids are discussed in Chapter 4 and Appendix B.

Step 1. Collate Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Management of suspended sediment ☑

Step 5. Predict mine drainage chemistry from flow chart ☑

Step 6. Predict stream water chemistry ☑

Step 7. Determine the potential ecological impact on the stream ☑

Step 8. Consider whether impacts are acceptable, and agree rehabilitation outcomes ☑

Step 8a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved. ☑

Step 9. If acceptable, design monitoring programmes and implement rehabilitation processes ☐

11 DECISION MAKING AND MONITORING

Consultation and monitoring are important steps in the framework (Figure **5**). It is through consultation that an acceptable level of impact and acceptable rehabilitation can be agreed upon by the relevant parties. Once an acceptable level has been agreed on, and the decision is made to proceed with the proposed mining operation, then monitoring should be undertaken to ensure that management systems are effective in ensuring that the agreed acceptable level of impact is not exceeded, and to identify any unexpected changes in mine drainage quality (e.g. as a result of treatment system failure). In some cases, no acceptable level of impact may be agreed and mining does not proceed.

11.1 Decision-making steps

In this framework we have not established explicit 'acceptable' water quality criteria or acceptable rehabilitation outcomes. Instead the framework provides a robust scientific basis for this decision to be made by relevant parties. The process described by the framework is intended to be operated in an iterative manner. In the first instance, mining operators can use this information for their internal decision-making processes to establish what likely operational management and/or treatment requirements may be needed, and what ecological attributes they may need to consider. As a next step, the information could be used during consultation between the regulator and applicant, and during more formal consultation processes, such as for resource consents. If mining proceeds then information provided by monitoring data should be used to adapt management strategies as required to prevent detrimental effects occurring, and facilitate successful rehabilitation outcomes.

In determining whether an impact at a given site is acceptable or not, consideration may also need to be given to other factors such as the presence of any iconic or endangered species, the current stream state, downstream water use, or any stream quality criteria specified in regional plans. There may also be broader social and economic considerations that determine whether the impact is acceptable or not. Consideration of acceptable impact may also include environmental compensation whereby environmentally beneficial activities are undertaken at a different location to compensate for the impact that occurs at the mine.

11.2 Ongoing monitoring

There are several different areas where ongoing monitoring should be completed to prevent unexpected negative changes in mine drainage chemistry and subsequent negative impacts on stream ecosystems. Monitoring of mineralised rock and mine waste geochemistry is required from a resource development perspective as well as to ensure that ore and waste management strategies are appropriate. Monitoring of leachate from mine waste storage, and treatment system discharge, is required to validate that management and treatment systems are working effectively. Monitoring of operational management methods and treatment systems may also be required to ensure appropriate performance, with adaptive management able to be incorporated should improvements be required. Water quality and biological monitoring provide validation on a broader spatial scale that management and treatment systems are working effectively.

11.2.1 Monitoring of rock geochemistry

Rock geochemistry should be monitored throughout the operation of the mine to identify rocks that may affect mine drainage chemistry, and is used to determine appropriate ongoing management of mineralised rock and waste rock, particularly if PAF rocks are present. Monitoring of rock geochemistry through ABA for water quality purposes can be conducted alongside sample collection for exploration and resource development purposes. The specific requirements for effective monitoring of rock geochemistry in operational mines are difficult to generalise because they are site and deposit specific. This could mean that all or a selection of ABA tests are completed, or that field-portable tests such as paste pH and the field NAG test are used as proxies for other ABA tests. The latter will have been determined through initial site investigations, and should be constantly updated as mining proceeds.

Sample types for rock geochemical monitoring can include core samples from resource development drilling, rock-chip samples from blast-hole drilling, and mine-face samples or samples from waste-rock dumps from areas exposed by mining or development work. Appropriate strategies for monitoring waste rock during mining might require several samples for ABA per day or one sample per month or per amount of waste rock depending on the variability of waste-rock geochemistry. If the geology and rock geochemistry are simple then less frequent monitoring is required than at complex sites.

If there are changes in the ABA characteristics of rocks collected during exploration and monitoring during mine operations, then additional rock samples should be collected and analysed. Sufficient additional samples should be collected to enable the geochemical variations to be defined. This information should be used to modify waste-rock management strategies accordingly.

Effective monitoring of rock geochemistry should mean that unexpected negative changes in mine drainage chemistry do not occur.

11.2.2 Leachate monitoring

Waste-rock piles and underground adits

Water quality monitoring of leachate from rock piles is also required to ensure that the selected management techniques

are effective. As such, the water quality of leachate from waste-rock piles should be monitored on a regular basis. This may require that collection systems are put in place during waste-rock-pile construction. Samples of undiluted mine drainage should be collected as close as possible to the entrance to an adit for water quality monitoring (section 11.2.4).

Tailings

Leachate from mine tailings inevitably seeps from beneath the tailings dam, and the seepage quality can be impacted, especially with As and Sb at gold mines. Concentrations of these elements are generally lowered by adsorption to iron oxyhydroxide precipitates, which can form if pyrite is being oxidised in the tailings water pathway (see section 7.2.1). Formation of abundant iron oxyhydroxide can take months or years after commissioning of the tailings facility. Hence, it is important to monitor water compositions regularly.

Water should be analysed monthly for acidity (base pH), EC and all consent elements, to appropriate detection limits, at the point of discharge of water beneath the tailings. In addition, regular (at least bi-monthly) monitoring of waters up to 1 km downstream of the tailings facility is necessary to detect any seepage from the tailings into the groundwater system that bypasses the main water discharge point. The downstream monitoring can be in surface streams and/or groundwater wells drilled for the purpose.

During the mine operation, the tailings water is normally captured and returned to the tailings facility or a treatment plant. When mining ceases, seepage will continue for years or decades, so regular monitoring and appropriate water management and/or treatment is necessary as long as the seepage persists.

11.2.3 Treatment system monitoring

Once a treatment system is in place, regular monitoring and maintenance is required to ensure that the system continues to operate as intended. At a minimum, this entails regular monitoring of the inlet and outlet water and maintenance of the system. Selection of biological monitoring sites should ensure that ongoing biological monitoring of ecosystems receiving treated water occurs (see sections 2.3.3 and 11.2.5).

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.

Active treatment for AMD systems

Water

For an active treatment system, both inlet and outlet water should be monitored regularly. If inlet water chemistry or flow rate changes, changes can be made to the system immediately to ensure that water continues to be treated adequately. For all active treatment systems, the following parameters should be measured on a regular basis from the inlet and outlet to the system:

- Flow rate
- TSS concentration
- Turbidity
- pH, acidity, alkalinity
- Dissolved oxygen
- Fe and Al concentrations, and any other metals or metalloids of concern in the mine discharge.

If possible, flow rate and pH should be monitored continuously. Continuous monitoring of pH is particularly essential to avoid problems associated with inefficient treatment or overtreatment, especially when using dry powder alkaline reagents (Waters et al. 2003). Electrical conductivity (EC) may not be useful to measure due to the high calcium loading of water. Frequency of analytical sampling should be based on the variability of the inlet water chemistry and the ability of the system to consistently treat to acceptable limits. Sampling is often conducted daily.

Depending on the chemicals used in active treatment, other water quality parameters should be measured to ensure they do not reach toxic levels. For example, regular monitoring of sodium is recommended for sodium-based systems, such as sodium carbonate (Na_2CO_3) and sodium hydroxide (NaOH) (Waters et al. 2003). For treatment systems using ammonia, the concentrations of NH_4 in the receiving water before treatment commences should be determined, and then regular monitoring of NH_4 in treatment waters should be undertaken subsequently (Faulkner & Skousen 1991; Skousen et al. 2000).

Systems

Active treatment systems require frequent maintenance. For conventional systems each step in the ODAS (oxidation, dosing with alkali, sedimentation) process has specific operational and maintenance requirements. For example, in the oxidation step, if mechanical aeration is used, the stirring mechanisms require regular maintenance to ensure adequate operation. If chemical oxidation is used, regular sampling of the treated water is necessary to determine appropriate dosing rates, and the oxidant dispensing and mixing mechanisms require regular maintenance. The dosing-with-alkali step involves storage vessels, dispensing mechanisms, and mixing tanks and mechanisms, all of which have specific operational and maintenance requirements. Likewise, the sedimentation step involves the operation of clarifiers and addition of flocculants

and coagulants with associated storage vessels, dispensing mechanisms, and agitation mechanisms, all of which have specific operational and maintenance requirements. Therefore, a detailed operation and maintenance manual should be prepared and tailored for each active treatment system.

Passive treatment systems for AMD

Water

For passive treatment systems, the following parameters should be measured on a regular basis from the inlet and outlet to the system:

- Flow rate
- pH
- Acidity
- Alkalinity
- Dissolved oxygen
- Fe and Al concentrations (total and dissolved)
- Any other metals or metalloids of concern in the mine discharge.

Monitoring of the inlet water is primarily of assistance in determining the cause of any change in outlet water chemistry. For example, this can identify whether deterioration in the quality of the outlet water is due to changes in the inlet water or due to a failure of the treatment system. Frequency of sampling should be determined on the basis of site-specific variables. Variables that should be considered include: ability of system to continuously treat to acceptable limits, sensitivity of the receiving environment to changes in water chemistry, level of treatment by system, risk of failure of treatment system, site access, and ability of system to continue to treat the water under different flow rates and chemistries if the inlet chemistry and flow rate can vary significantly due to climatic variations. Samples can be collected daily, weekly, fortnightly or monthly.

For some passive treatment systems, additional water quality parameters should be added to the list of analytes for outlet water. For example, systems that rely on biological sulphate reduction for treatment, and systems that incorporate compost in their substrate such as vertical flow wetlands, anaerobic wetlands, and sulphate reducing bioreactors, should also include monitoring of the outlet water for sulphides, dissolved organic carbon, total nitrogen, nitrate-N + nitrite-N, total phosphorus, and total biochemical oxygen demand. For systems that use steel slag as a neutralising agent (such as slag leach beds), it is possible that elements within the steel slag, such as barium, vanadium, manganese, chromium, As, silver and selenium, may be released into the treated water upon dissolution (Simmons et al. 2002). Therefore, selected elements should be added to the list of analytes in the discharge from these systems. The full suite of metals should first be analysed to determine which metals have the potential to be released from the slag.

Systems

Passive treatment is typically considered low maintenance compared with active treatment. However, some maintenance is necessary to ensure continued adequate treatment of the mine drainage. Passive treatment systems have operational and maintenance requirements specific to each system. For example, open limestone channels may require regular scouring of the channel bed to dislodge and remove built-up precipitates, diversion wells require regular replenishment of limestone chips, and leach beds and vertical flow wetlands require regular flushing to remove precipitates that can clog passageways reducing permeability and treatment effectiveness. Operational and maintenance requirement details for each system are provided in Appendix F.

Sludge removal is often an important long-term maintenance issue for passive treatment systems (PIRAMID 2003). The removal of Fe-dominated sludge from settling ponds can be a significant long-term maintenance cost, unless ponds are massively oversized in terms of the metal load they receive. Sludge can either be removed from ponds in wet form using a vacuum truck or dry form using mechanical excavation. If there is insignificant leaching of trace elements (i.e. the sludge is stable), it may be able to be disposed of directly on site, or to landfill. If there is significant leaching of trace elements, stabilisation of the sludge may be required. If disposed of to landfill, dewatering of wet sludge, which typically contains between 1% and 5% solids, may be required although dewatering may not be required for on-site disposal. Dewatering will also reduce the volume and hence cost of disposal. Estimates of the volume of sludge that will be generated by a passive treatment system can be made using the computer program AMDTreat (Means et al. 2003) (see Appendix F.4).

Monitoring of As treatment systems

Active treatment systems

The same general monitoring requirements outlined above for both water and system monitoring apply. Regular sampling of water at various steps of the system is used to determine reagent addition rates and residence times, and outlet water quality. Water quality parameters that should be measured on a regular basis from the inlet and outlet to the system are:

- Flow rate
- TSS concentration
- Turbidity
- pH, acidity, alkalinity, EC
- Dissolved oxygen
- As concentrations

As outlined above, each treatment step has specific operational and maintenance requirements. Therefore, a detailed operation and maintenance manual should be prepared and tailored for each active treatment system.

Passive treatment systems

The same general monitoring requirements outlined above for both water and system monitoring of passive treatment systems apply. The following parameters should be measured on a regular basis from the inlet and outlet to the system:

- Flow rate
- pH, acidity, alkalinity, EC
- Dissolved oxygen
- As concentrations

The detection of As in the outlet water can indicate when adsorption media should be replaced. Leachate tests on the spent medium should be undertaken to ensure that there is no significant leaching of As or other trace elements after disposal.

11.2.4 Water quality

In this section, the focus is on water quality parameters that have been consented. These parameters should have been determined through initial site characterisations (see section 2.3) and should include all relevant parameters. The number of sites sampled should include those required for consent conditions as well as providing ongoing baseline data against which to measure any change. Water quality samples must be collected at the same sites and times as biological monitoring is undertaken, although water quality samples will also be collected on a more regular basis and potentially at additional locations.

Where to monitor

Site-specific factors are important in the selection of monitoring points for assessing the final downstream water quality, although important considerations include:

- Samples of undiluted mine drainage or leachate from tailings impoundments should be collected as close as possible to the source of the seep or entrance to an adit.
- The monitoring point should be at a point where mine drainage or tributaries are completely mixed with other catchment water. A general rule of thumb is that mixing occurs at a distance downstream of a tributary that is approximately 10 times the width of the stream. However, care should be taken to validate that the sampling point is located at a point where mixing is complete. This can be confirmed by measurement of physicochemical parameters across the stream from the sample collection point. If there is no change in physicochemical properties across the stream then the sampling point is acceptable.
- The monitoring point should be upstream as far as practical after mixing of mine drainage with other catchment water so that dilution effects do not mask changes in mine drainage quality. For example, if mine drainage contributes to a tributary to a major river, then the monitoring points should be on the tributary if possible rather than the major river.
- Monitoring points should be located where mine development will not occur. Ideally, these monitoring points are ones that have been used during baseline surveys (Chapter 2). Monitoring points are most valuable if used for assessing long-term water quality, and should be located where they will not have to be shifted. Collection of a large set of data from one monitoring location enables more subtle changes in water chemistry to be detected, compared with smaller datasets from multiple locations.
- The likely partitioning of trace elements between dissolved species and suspended particulate material at the dissolved concentrations of the trace element at the sampling point (see next section).
- In determining the location of biological monitoring sites, the water quality and biological samples should be taken at the same site, and ideally, from the same sites as used during baseline surveys (Chapter 2).

What to monitor

There are many alternative monitoring strategies to identify when chemical conditions within a stream depart from those expected or agreed during resource consent. However, costs can be unnecessarily high if complete chemical (all consented parameters) analyses are undertaken on each monitoring sample. Thus, a tiered approach can be applied to obtain the maximum useful information with minimal cost. An example of a tiered approach is provided below. The objective is to minimise the number of analytical parameters to reduce costs and provide maximum useful information.
Example three-tier monitoring system with rationale for sample types:

- Tier 1: The minimum useful analytical suite is turbidity, pH, EC and flow and potentially some specific trace elements (e.g. As, typically identified during initial baseline investigations). This approach assumes that pH and EC are suitable proxies to identify variations in stream chemistry caused by changes in flow volume or quality of mine drainage into the catchment. After mine operations commence, the relationships between pH, EC and trace elements of interest should be determined to establish whether pH and EC are suitable proxies. In some cases, where monitoring of specific contaminants is required, such as As, pH and EC might not be suitably sensitive proxies and other proxies might be identified if statistically valid relationships are established. If statistical relationships cannot be established then the specific trace elements should be added to Tier 1.
- Tier 2: If pH departs from expected conditions by more than 0.5 units or EC departs from agreed conditions by more than 10%* and flow is within 20%* of background then sulphate concentrations should also be determined. If changes in mine drainage volume or quality cause variations in pH or EC then it is likely that sulphate concentration will also change within the catchment. Increases in sulphate concentration of more than 10% could indicate that there has been a change in the volume or quality of mine drainage entering the catchment.
- Tier 3: If sulphate concentrations increase by more than 10% then a full analysis of all consented parameters should be conducted. Samples for Tier 3 should be collected at the same time as Tier 2 but only submitted for analysis if sulphate has increased by more than 10%*.

*There are many approaches that could be used to trigger more detailed analysis or additional detailed sampling such as

rolling monthly averages or statistical deviations from long-term data.

Where specific trace elements are monitored, consideration should be given to whether it is appropriate to monitor the total or dissolved concentrations at a specific monitoring point. Partitioning between dissolved species and suspended particulate matter can be dynamic so that chemicals are associated with particulates in some parts of a catchment but are dissolved in other parts of a catchment. For example, many trace elements adsorb to suspended particulate material but desorb if chemical conditions become favourable. The relationship between suspended particulates and dissolved species is an active area of research and is likely to be site specific. Conditions that are identified in resource consent documents should be made with understanding of the partitioning of trace elements between solid and dissolved species and might involve the expertise of an experienced geochemist and/or water quality scientist.

When to monitor

Monitoring should be undertaken at representative flow levels, which can be determined from regular flow monitoring. A tiered approach can be used for determining the frequency of monitoring.

- Continuous to daily to weekly monitoring of all Tier 1 parameters is recommended at monitoring points at a specified time of day, as biological processes may result in diurnal variations in some parameters.
- Weekly to monthly monitoring of Tier 3 parameters is recommended at monitoring points at a specified time of day.
- If check monitoring is carried out by regulators, it should be completed at the same time of sampling as routine monitoring.

There are many site-specific factors that might require changes to the monitoring strategies outlined above. These strategies should be considered a minimum or starting point for ongoing monitoring, and interpretation of results should include analysis of all changes in site conditions. In addition, broader environmental factors can also cause variations in the concentration of dissolved components in streams due to rainfall, drought, seasonal variability, snow melt or many other factors. Refinement of the strategies outlined above should be completed as analytical cost decreases and availability of portable analytical capability improves.

11.2.5 Aquatic biological monitoring

Ongoing monitoring should be undertaken to detect impacts occurring during and after active mining. As discussed in section 2.3.3, sampling should include both control or reference sites and potentially impacted sites to enable the detection and quantification of mining-induced change. If reference sites are not included, detection of changes relies on comparisons with pre-impact conditions, which is confounded by any other change over this time, such as large floods, droughts, vegetation regeneration or other factors, which might be unknown. Therefore data from multiple reference sites are essential for rigorous and meaningful consent monitoring.

The sampling of sites selected for ongoing consent monitoring should occur directly prior to any mining operations, and then at regular intervals afterwards. This ongoing monitoring is best conducted at least seasonally initially when rapid changes in systems may occur, and alongside water quality monitoring. However, if the intensity and type of mining activities remain constant over a long period then annual monitoring may be acceptable (e.g. spring or summer sampling). The duration of monitoring will be dictated by the conditions of resource consents, and will include both active mining and treatment phases. The continued monitoring of restoration activities is especially important, as it may take some time for fauna to re-colonise habitats.

11.2.6 Monitoring land rehabilitation

General principles

For progressive rehabilitation, monitoring commences during mining operations and continues until it is signed off at mine closure and/or handed over to the post-mining landowner or site manager. Baseline date and photos of pre-mining conditions are prerequisites to land monitoring rehabilitation. These provide benchmarks against which post-mining land rehabilitation can be assessed. Closure criteria for land rehabilitation should be determined at the planning stage or at least the early phase of mine development. These may be modified over the period of mining, in concert with flexibility for land rehabilitation activities. Monitoring at a landscape level (geomorphology) of the engineered landforms (ELFs) and constructed waterways and lakes/tarns can be useful at large mine sites to measure the stability of the rehabilitated landscape components.

Rehabilitation trials are often used at bigger and longer-term mining sites. The approaches below also apply to these trials. Common problems with trials that should be avoided are locating them in areas that will be disturbed again by mining operations, restricting the trials to the short term (1 or 2 years), and a lack of replication. There is a role for demonstration ('suck it and see') trials but these limit interpretation of the results and are often over-applied at large, long-term mines.

Monitoring regimes will differ between different land uses, as outlined below. However, common approaches apply to rehabilitation across all land uses. Well-documented, archived, and retrievable monitoring records are essential. For example:

- Descriptions of techniques used and timing of rehabilitation activities, soil conditions and fertility
- Lists of species (flora and fauna) along with survival, growth or production rates
- Weed and pest control measures and success
- Remedial measures undertaken.

Fixed marked and labelled photo points are advantageous for photographing areas before, during, and after rehabilitation. Aerial photography using UAVs (unmanned aerial vehicles) and LIDAR can be useful tools for remote monitoring of the rehabilitated landscape components and re-vegetated area on larger, more long-term mining sites. Permanently marked plots or transects are essential for reliable objective comparisons. Methods for monitoring recovery of animal, bird, and invertebrate species should be designed according to reliable and appropriate (to the site) scientific techniques.

Annual monitoring is generally recommended for the first 5 years, followed by at 2-year intervals, extending to 5-year intervals for long-term mines, until closure or handover.

Adaptive management operates in concert with monitoring when remedial measures are required. Examples are:

- Blanking or replanting, after any required remedial measures (e.g. draining, fertilising, repairing eroded sites) on areas where plant deaths will lead to insufficient plant densities
- Replanting with different plant species to match microsite conditions (e.g. waterlogged or droughty)
- Installing drainage (surface or under-drains) or irrigation
- Repairing and seeding/planting eroded or post-rehabilitation disturbed areas
- Erosion and sediment control measures such as mulching, hydroseeding, sediment fences, application of flocculants (e.g. polyacrylamides) before reseeding/planting
- Weed control using selective herbicides or hand weeding
- Trapping, shooting, or baiting for pest control
- Sod-seeding earthworms (not compost worms) into established pastures devoid of worms
- Staking toppled or 'butt sweeped' seedlings
- Re-establishing habitat then reintroducing species that do not easily re-colonise sites, such as kiwi, lizards, skinks and *Powelliphanta* snails.

Pasture monitoring

Pasture productivity (dry matter production) and carrying capacity, topsoil and total soil depth, ground cover, soil fertility (nutrient testing), evidence of seepage/waterlogging zones, and surface stoniness (including boulders) are often monitored as objective measures of the success of pasture rehabilitation and for adaptive management. Other potential monitoring measures are land use capability assessment, species composition including weeds, pests and diseases, erosion and sediment runoff on sloping land, and the performance of infrastructure (e.g. fences, gateways, under-drainage, access races, water supplies, shelter belts)**.**

Plantation forestry monitoring

Monitoring rehabilitated plantation forest should follow standard silvicultural monitoring practices. For example, initial ground cover for erosion control, tree establishment (seedling mortality), density and growth rates, weeds during the early stages of development, pests (browse damage) and diseases, fertility (tissue and/or soil testing), toppling or wind-throw and butt sweep, and evidence of poor drainage for species adversely affected by waterlogging.

Native ecosystems monitoring (shrubs, trees, tussock grasslands, and wetlands)

Monitoring systems for rehabilitation to native ecosystems include a combination of the factors for pasture and plantation forestry monitoring. Monitoring ground cover (including for direct transferred vegetation), soil depth, erosion (rilling and slumps), and sediment runoff are important during the early stages after rehabilitation. Three months after seeding and planting is an appropriate time to commence monitoring Tree, shrub, and direct-transfer survival and growth rates need to be monitored annually for at least 5 years after rehabilitation, and preferably until canopy closure for shrubs and trees. Special attention is required for monitoring endangered and at-risk species, including plants, bryophytes, birds, and invertebrates. Monitoring weeds and pests is particularly important until the ecosystems become self-sustainable. Also, soil fertility and drainage issues will become evident through plants showing signs of nutrient deficiencies, waterlogging, or drought (for wetland species). Investigation of the causal factors of plant deaths or unsatisfactory growth and habitat establishment is required to determine remedial measures. For wetlands, monitoring soil moisture levels and water tables may be necessary at some sites that are not lake or riparian margins.

11.3 Checklist

The following provides a checklist of the data that are collected within this chapter, and identifies where these fit in the step-by-step guide. The following checklist refers only to water quality issues associated with pH and dissolved contaminants; suspended solids are discussed in Chapter 4 and Appendix B.

Step 1. Collate Background and Baseline information ☑

Step 2. Identify resources for rehabilitation and collect rock samples for geochemical testing ☑

Step 3. Geochemical testing to determine acid-forming status and acid- neutralising capacity of rock samples ☑

Step 4. Predict mine drainage chemistry from flow chart ☑

Step 5. Predict stream water chemistry ☑

Step 6. Determine the potential ecological impact on the stream ☑

Step 7. Consider whether impacts are acceptable, and agree rehabilitation outcomes ☑

Step 7a. If unacceptable, determine whether treatment and management strategies to create acceptable stream water chemistry are available, and acceptable rehabilitation outcomes can be achieved. ☑

Step 8. If acceptable, design monitoring programmes and implement rehabilitation processes ☐

- Rock geochemistry ☐
- Treatment systems
  - Active systems ☐
  - Passive systems ☐
- Water quality
  - Location ☐
  - Parameters ☐
  - Frequency ☐
- Biological of aquatic systems ☐
- Rehabilitation monitoring ☐

12 WORKED EXAMPLE AND SERVICE PROVIDERS

12.1 Worked example

The following worked example is provided to illustrate the use of the framework. It is based on a hypothetical large-scale lignite mine in Southland, which may occur in the future making this scenario somewhat realistic. However, the specifics provided are completely hypothetical. The example includes material relevant for assessing potential impacts of mining on the West Coast, specifically that associated with mine drainage from potentially acid-forming rocks.

12.1.1 Overview

A large-scale opencast lignite mine with an expected annual output of 20 million tonnes of lignite has been proposed. The land is currently owned by the mining company, which is compiling information for a resource consent application. The mine area will cover approximately 45 km^2, and the pit area will be mined from the southern end towards the north (Figure **48**). At any given time a pit of approximately 4 km^2 will be open for mining. The waste rock from the initial pit will form an overburden pile; subsequently the waste rock will be used to backfill the mine pit.

Streams will be diverted during mining operations as the need arises. Hydrogeological data indicate groundwater pumping will be required at a rate of 100 L/s. Gauging data on the tributaries indicate flows range from 10 to 20 L/s, while the main river (sampling points 1 and 2) has an average flow of 2000 L/s.

Exploratory drilling has been undertaken and a geologist has provided a preliminary geological interpretation of drill cores from perpendicular transects across the proposed mine area (Figure **49**). The drill-hole data and geological interpretations provide the basis for the cross sections shown in Figures 50-55.

Geological description (Appendix C.4) of the rock samples provides information of use in the interpretation of analyses. Field notes by the geologist noted that the mudstone was highly pyritic. Elongate grey patches were noted in core samples indicating the presence of pyrite.

The step-by-step process outlined in Figure 5 is followed. The process is used to work through preparation of information on the potential water quality impacts associated with this proposed hypothetical operation. Checklists that outline the key bits of information required at each step are provided at the end of the relevant chapters above.

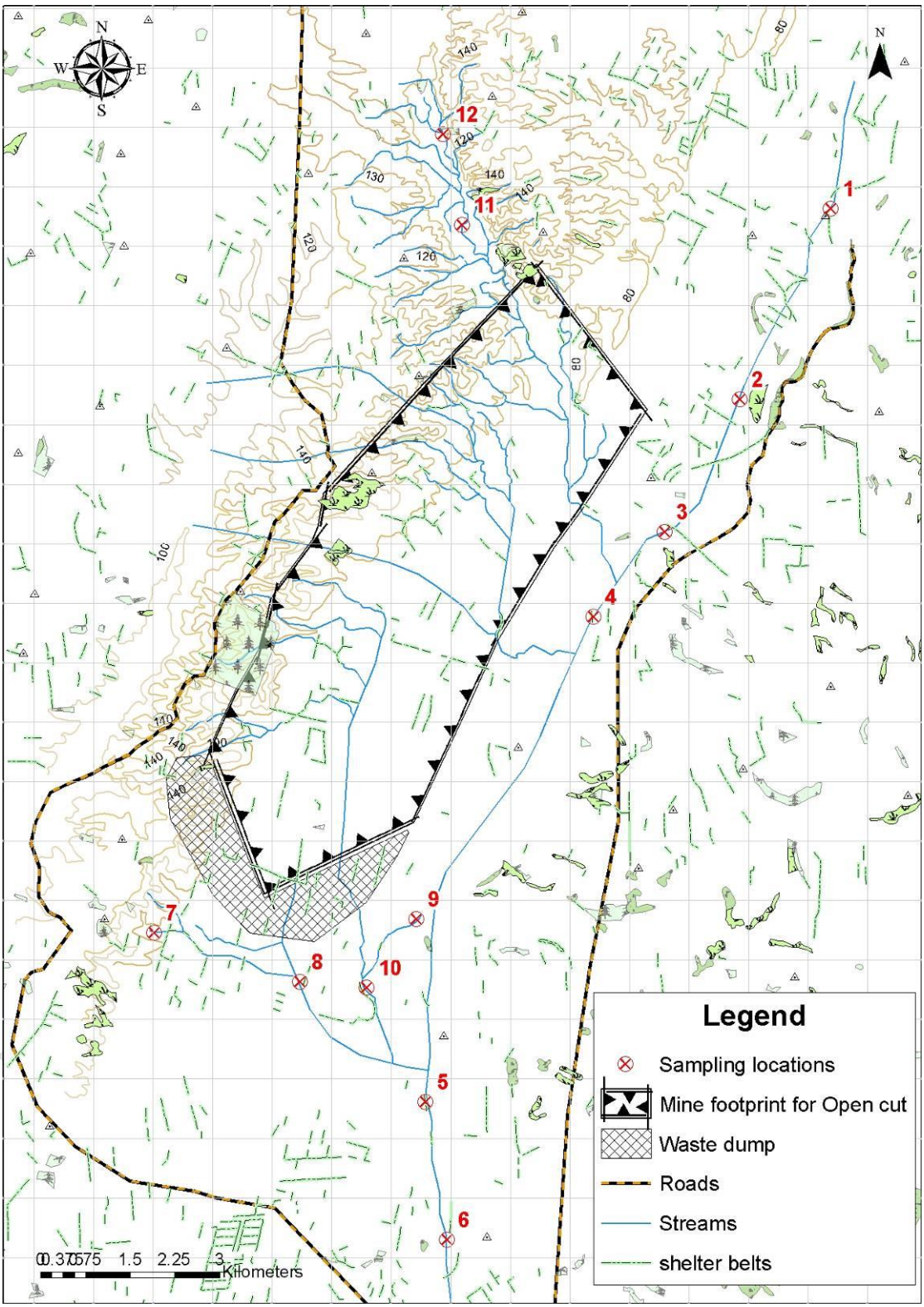


Figure 48: Topographic map showing mine and waste dump footprint.

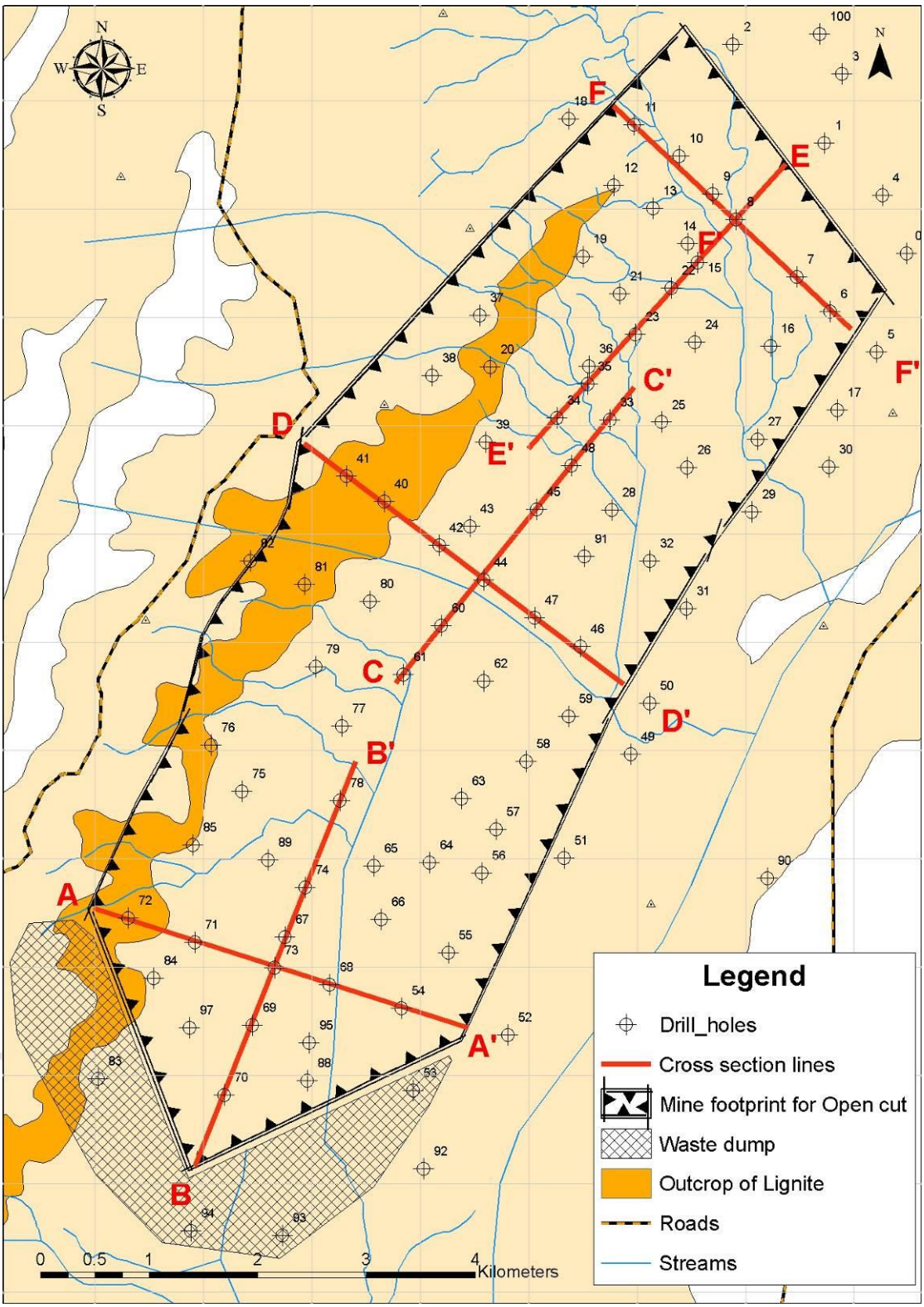


Figure 49: Map showing drill-hole location and cross-section line from geological report.

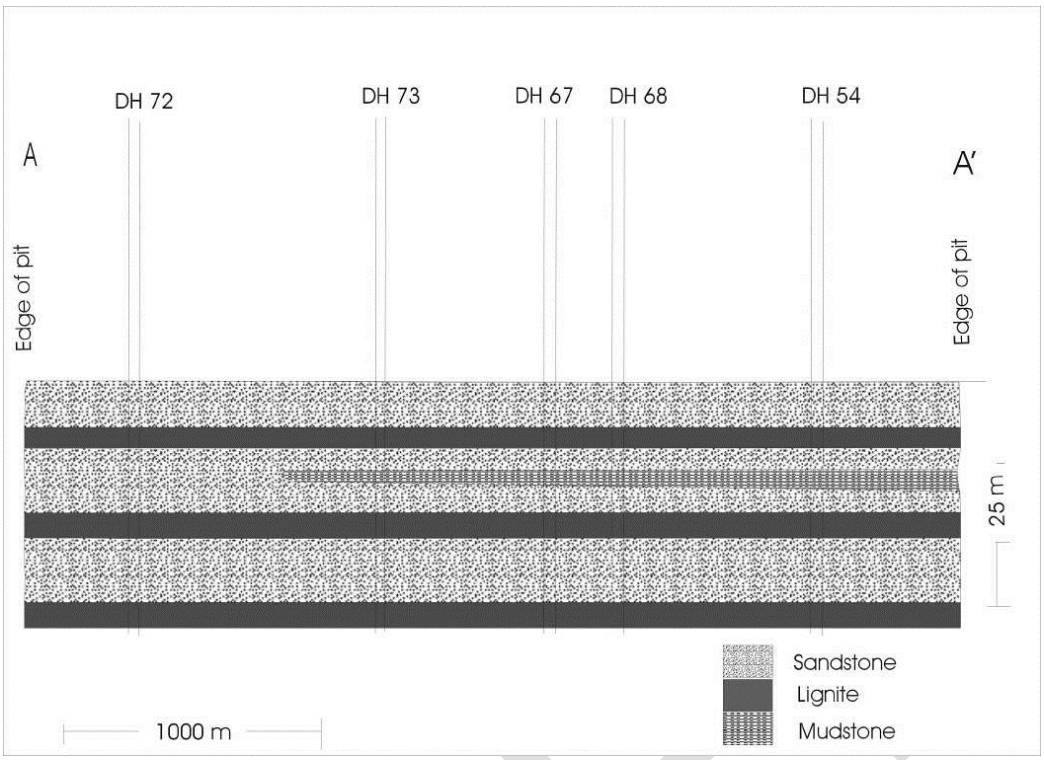


Figure 50: Cross section A-A'.

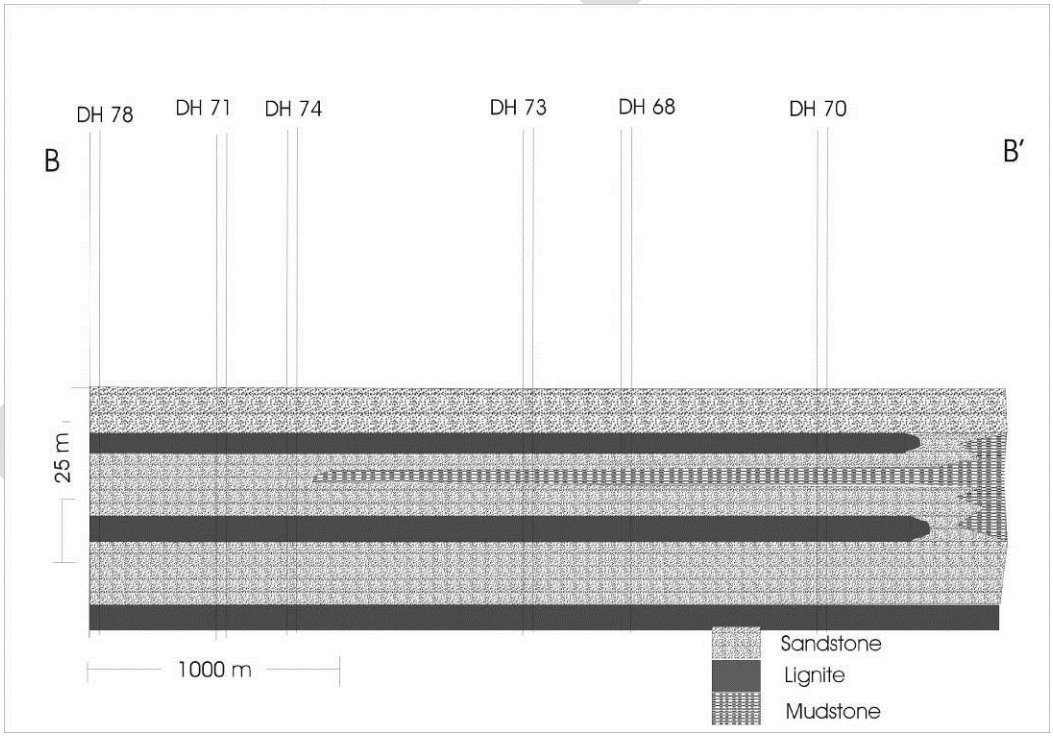


Figure 51: Cross section B-B'.

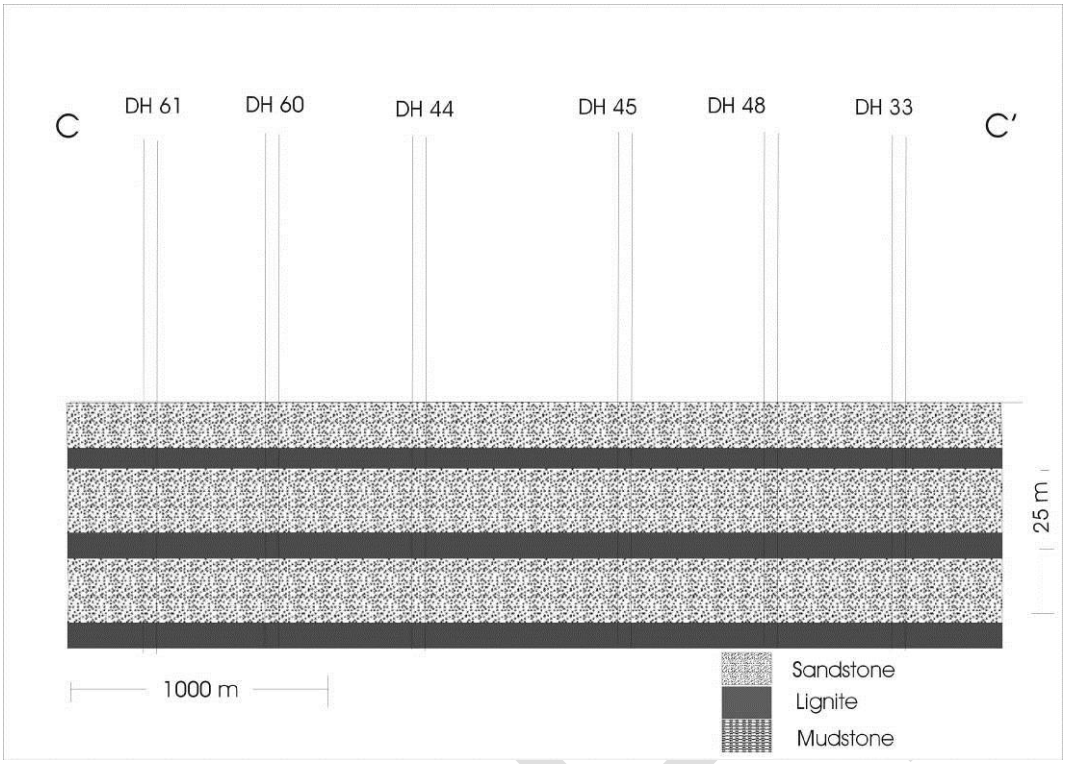


Figure 52: Cross section C-C'.

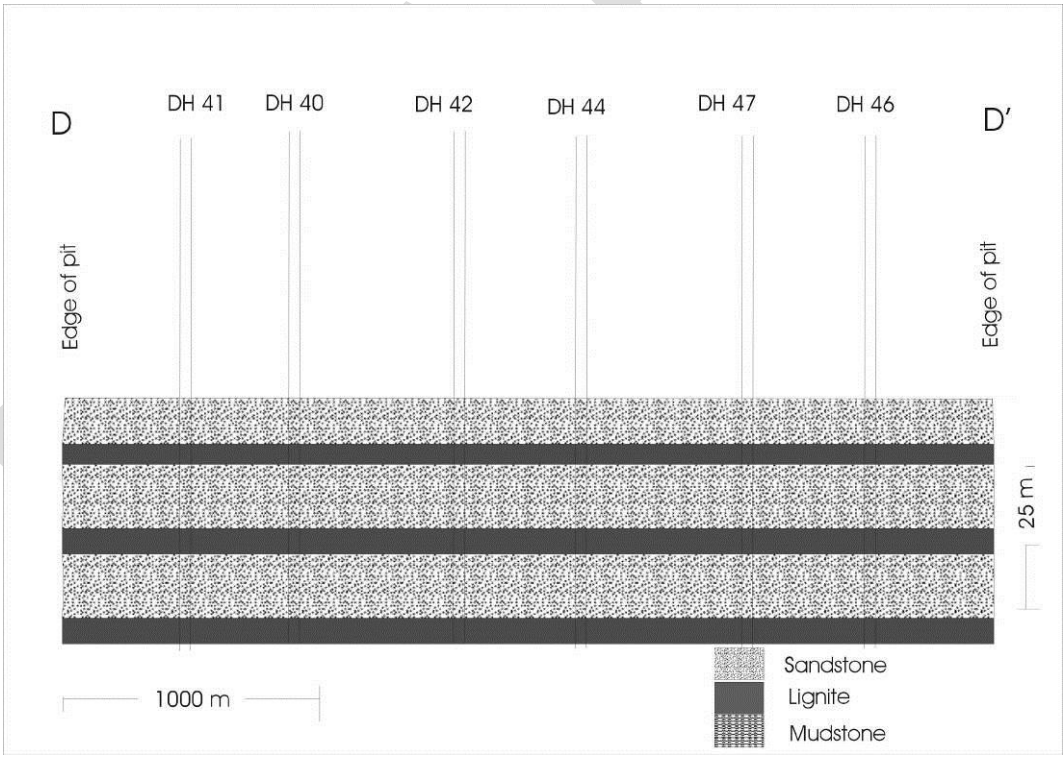


Figure 53: Cross section D-D'.

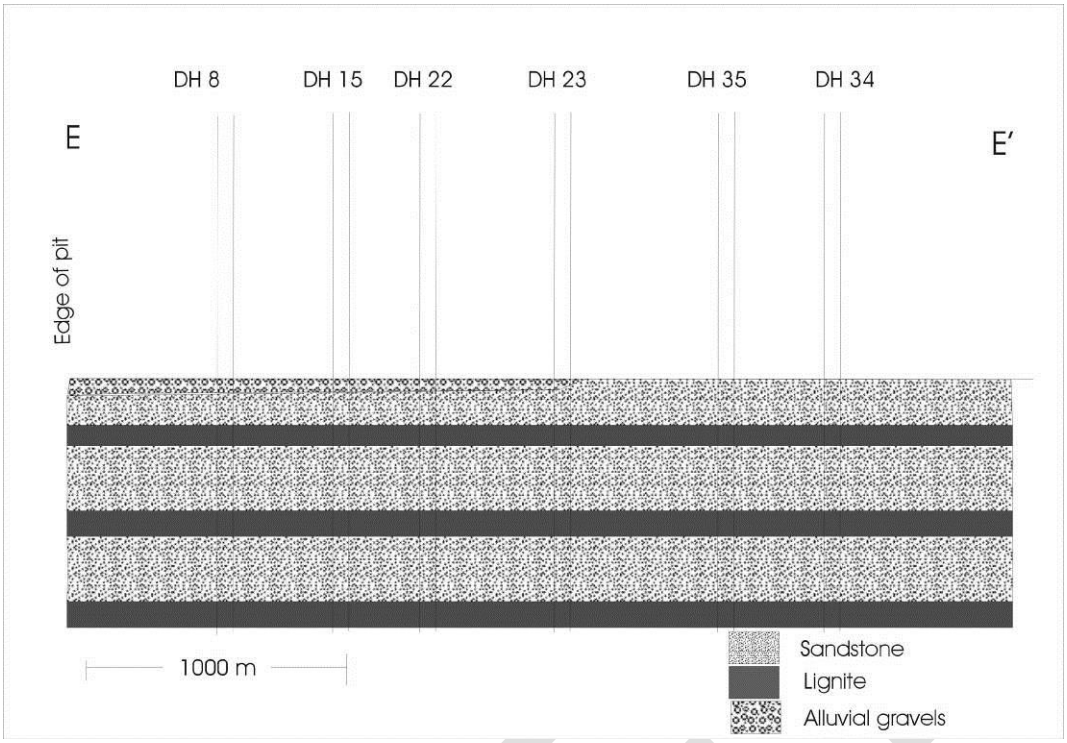


Figure 54: Cross section E-E'.

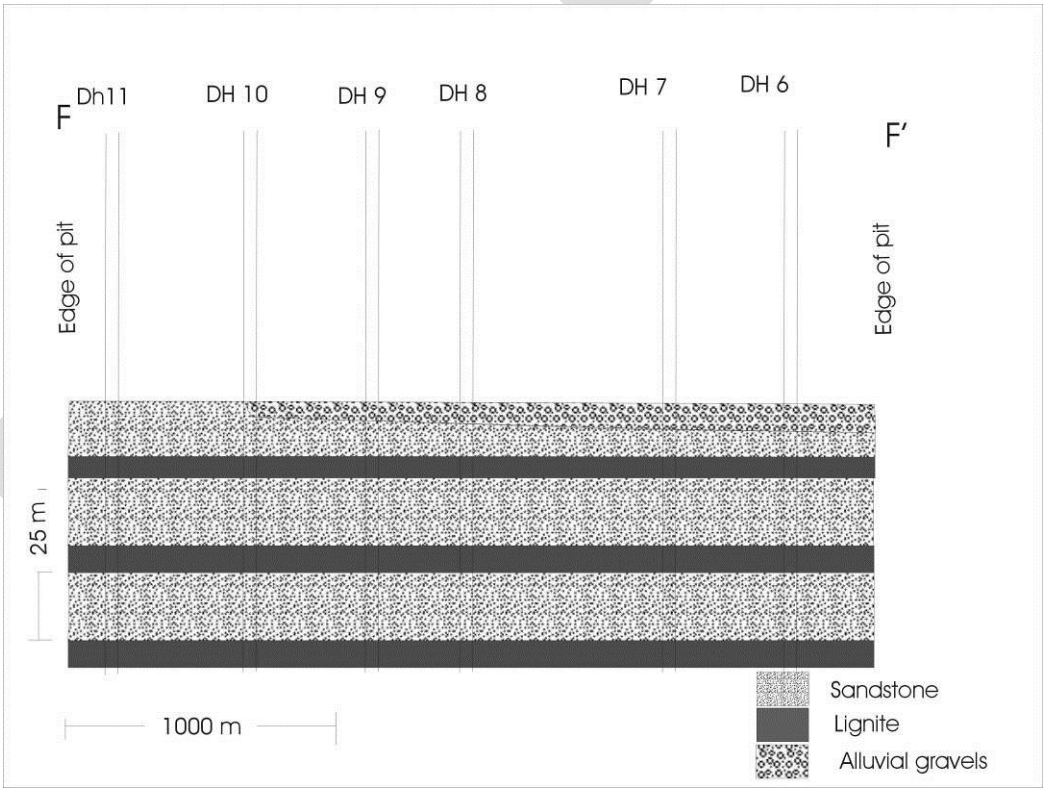


Figure 55: Cross section F-F'.

12.1.2 Step 1: Collate background and baseline information

Chapter 2 provides an overview of the information required to support decisions made when predicting mine drainage.

Things to consider:

What background information is useful to assist in determining water quality impacts from the mine operation?

Historical data as outlined in section 2.2 include:

- Background geological data including map and bore-log data

- Stream flow (there are often gauging points on streams)
- Stream ecology
- Mining history (MED has many of the old NZ Mines Department reports; old mine plans are also held at various locations)

Where might this information be obtained? (section 2.2)

- Mineral exploration reports (MED has reports available electronically on their website. These have location maps, drill-hole data, geological summaries, cross sections, historical mine sites)
- Geological maps and reports (university theses, NZJGG publications, consent applications, MED reports and drill-hole database also have geological summaries, cross sections and maps)
- River flow and climate data (regional councils, NIWA, regional reports, consent applications within the area)
- DAME database (this will be held by the regional councils)
- Regional council database and reports (database containing flow data, water quality, consent applications, regional science reports)

A checklist outlining information types and sources is provided in section 2.4.

What factors are important to consider in the design of baseline surveys? (Section 2.3)

- Biological sampling includes sampling invertebrates, noting the habitat and environment (section 2.3.3).
- Physical factors (section 2.3.1) including flow volumes and flow variability (from gauging). All flow >5% of flow volumes require measurements.
- Water quality (section 2.3.2) data including natural sources of acid rock drainage (ARD), background or baseline physiochemical properties (pH, electrical conductivity (EC), dissolved oxygen, etc.) and concentrations of sulphate, dissolved Fe and Al, and other dissolved trace elements such as As and Zn and existing sources of mine drainage.
- A BACI (before-after control impact) design (Figure 8) is the optimal design for biological sampling. Two 'types' of control: one upstream of impact, while a likely unimpacted stream provides additional baseline information.
- Note: flow gauging, collection of water samples and biological sampling are best to be completed at the same time and location.

12.1.3 Step 2: Collect rock samples

Section 3.3.2 contains information on the collection of rock samples.

Things to consider:

What are two key strategies for determining selection of rock samples for analysis? (section 3.3.2)

- Characterisation of representative rock types within the sequence of rocks including ore rocks or coal that is to be disturbed by mining
- Identification and sampling of specific geological features that have anomalous acid-producing or neutralising characteristics. This includes:
    - For coal deposits geological structures such as faults or beds that contain pyrite or calcite, and the roof and floor of coal seams, could have anomalous acid-producing or neutralising characteristics.

Sufficient data should be collected so that statistically meaningful acid–base accounting values can be identified or calculated for important rock types.

How does geological complexity influence the sampling strategy?

More complexity means MORE samples are required.

Identify the number of drill holes that need to be sampled, the number of samples required and types of analyses required to determine whether the deposit is likely to produce AMD

Section 3.3.2 notes: 'Where geology is simple (<5 different rock types present in any 20 m of core) at least one drill hole should be sampled in each block of 1000 × 1000 m. The decision to use a drill-hole density greater than 250 × 250 m for acid–base accounting analyses should be made by an experienced geologist.'

The geology of the proposed mine site is relatively simple, with layers of sandstone, mudstone, lignite and alluvial gravel (Figures 50-55). The area is 45 km^2 therefore 45 drill holes would be sampled.

'Density of sampling within drill holes should ensure that the complete sequence of rocks to be disturbed by mining is sampled. At least 5–10 samples of each rock type encountered in the sequence that is to be disturbed by mining should be submitted for acid–base accounting analysis. At most exploration targets, it is likely there are 4-8 different rock types; therefore a minimum of 20–40 rock samples should be analysed for acid–base accounting for each block of 250 × 250 m.'

Based on four rock types that are reasonably homogeneous, five samples per rock type and 45 drill holes, there will be 900 samples for acid-base-accounting testing.

A checklist for the steps involved in the collection of rock samples is provided in section 3.4.

12.1.4 Step 3: Geochemical testing to determine acid-forming status

Acid-base-accounting tests are the primary tests used to identify the rocks that have potential to change the pH (or increase the acidity or alkalinity) of mine drainage chemistry. Section 3.3.3 provides information on ABA testing including a brief description of different tests.

Table 7 provides a set of hypothetical ABA results for the worked example and indicates that the mudstone is highly acid producing (with NAPP > 10 kg (H_2SO_4)/t). Further, there is no evidence that the sandstone may be potentially neutralising. Together these results suggest a potentially-acid-forming situation.

Kinetic tests (section 3.3.5) simulate these conditions and test results can be used to indicate worst-case scenarios.

A checklist for the steps involved in geochemical testing is provided in section 3.4.

Table 7: Hypothetical ABA results of samples collected during exploratory drilling

Sample description	MPA kg (H2SO4)/t	ANC kg (H2SO4)/t	NAPP kg (H2SO4)/t	NAG kg (H2SO4)/t	NAG pH
Sandstone	0	1.80	(1.8)	0	6.4
Sandstone	0.3	0.60	(0.3)	1	7.2
Sandstone	3.4	7.2	(3.8)	0	6.3
Mudstone	25	3.1	21.9	26	2.9
Mudstone	60	0.2	59.8	51	2.7
Mudstone	27.2	2.50	24.7	29	3.2
Mudstone	33	1.50	31.5	31.9	3.0
Lignite seam roof rock	0.4	1.2	(0.8)	0	6.5
Lignite seam roof rock	1.2	3.0	(1.8)	27	6.8
Lignite seam roof rock	6	2.2	3.8	3	6.8
Lignite seam floor rock	0.5	1	(0.5)	0	7.1
Lignite seam floor rock	0.7	1	0.3	1	7.2
Lignite seam floor rock	1	2.5	(1.5)	0	6.5

MPA = maximum potential acidity, ANC = acid-neutralising capacity, NAPP = net acid-producing potential, NAG = net acid generation, NAG pH = potential pH of mine drainage.
NOTE: Average geochemistry (i.e. average mine drainage) in Table 7 is based on interpretation of water sampling data in the mine drainage database.

12.1.5 Step 4: Predict mine drainage chemistry

Prediction of mine drainage chemistry is dependent on the acid-forming potential of the rocks, and is specific for different mine types. Based on the ABA results provided above, the proposed mine would be considered to fall into the potentially acid-forming category (Chapter 5). Mine drainage chemistry for the proposed mine can be predicted using Figure 18, which considers mine operations and hydrogeology (in particular the location of mine waste in relation to the water table). The proposed mine in this example is opencast, with the mine waste initially placed above the water table. The red line in Figure **56** shows the path followed to predict the generic water chemistry of mine drainage from the proposed mine.
Kinetic testing (section 3.3.5) can provide more information on site-specific characteristics of mine drainage. For our hypothetical example, kinetic tests suggest that leachate from the mudstone will have the following characteristics: pH 3.2, Fe 15 mg/L, Al 32 mg/L, acidity 249 mg/L, TSS 1000 mg/L.

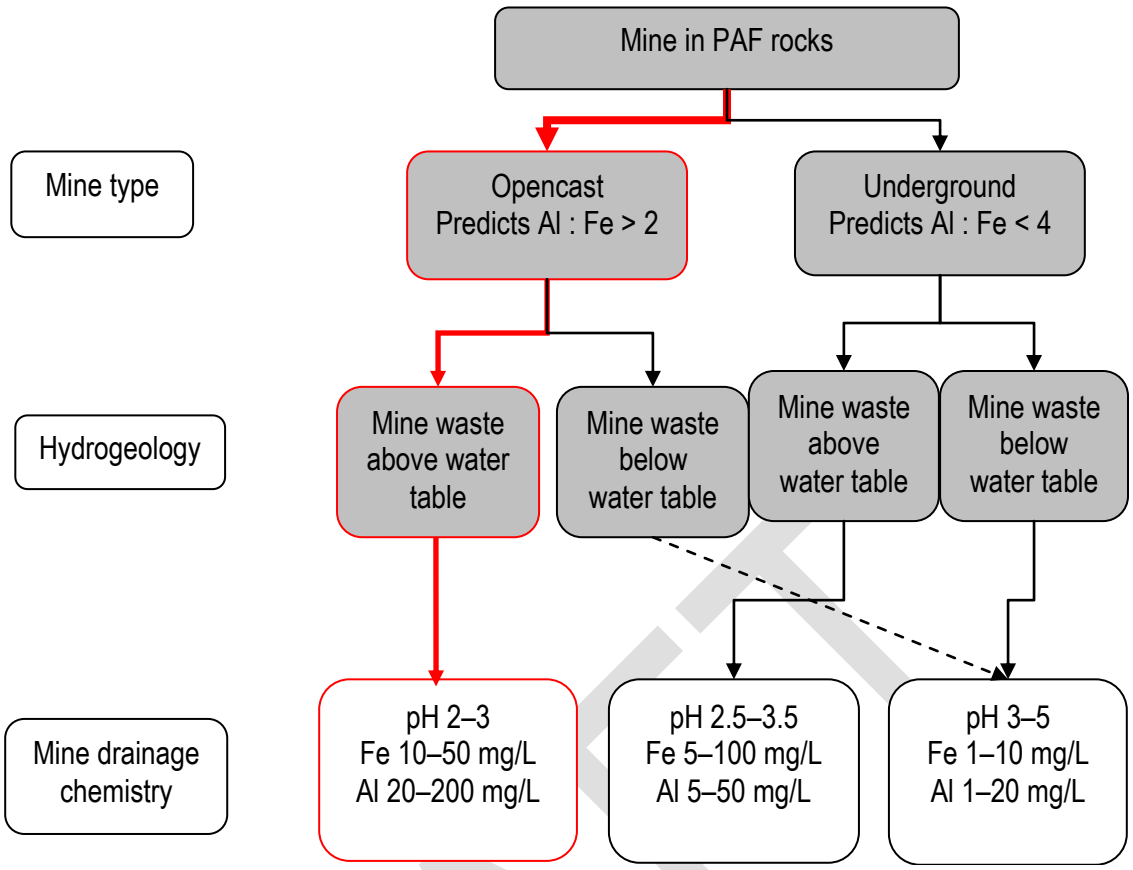


Figure 56: Modified Figure 18 to illustrate the path followed to predict generic water chemistry for the proposed mine.

12.1.6 Step 5: Predict stream water chemistry

Section 3.4 and Figure 14 provide an outline of the process for predicting water chemistry downstream of a mine drainage discharge. Information on mine drainage chemistry is integrated with information on site hydrogeology and background water quality using reactive transport modelling. Reactive transport modelling must be used because reactive components (both acid and neutral) are present in stream waters and mine drainage.

For the proposed mine under typical conditions, dewatering and surface runoff from the pit are considered to produce a discharge of 100 L/s into the receiving environment (the river has a flow of 2000 L/s and pristine water quality). Based on the predicted chemistry (section 12.1.5) and likely flow from the pit, mine drainage is likely to impact downstream water quality. However, reactive transport modelling undertaken by a qualified and experienced professional would provide more detailed information on the likely impact on water quality. In this case, reactive transport modelling indicates that downstream water chemistry will be pH 3.4, Fe 9 mg/L, and Al 40 mg/L.

A checklist for the steps involved in predicting downstream water chemistry is provided in section 3.4.

12.1.7 Step 6: Determine the ecological impact

The ecological impact associated with the predicted downstream water chemistry is determined using Figure 19. The pathway to determine the potential impact associated with the hypothetical proposed mine using Figure 19 is shown in Figure 1Figure **57**.

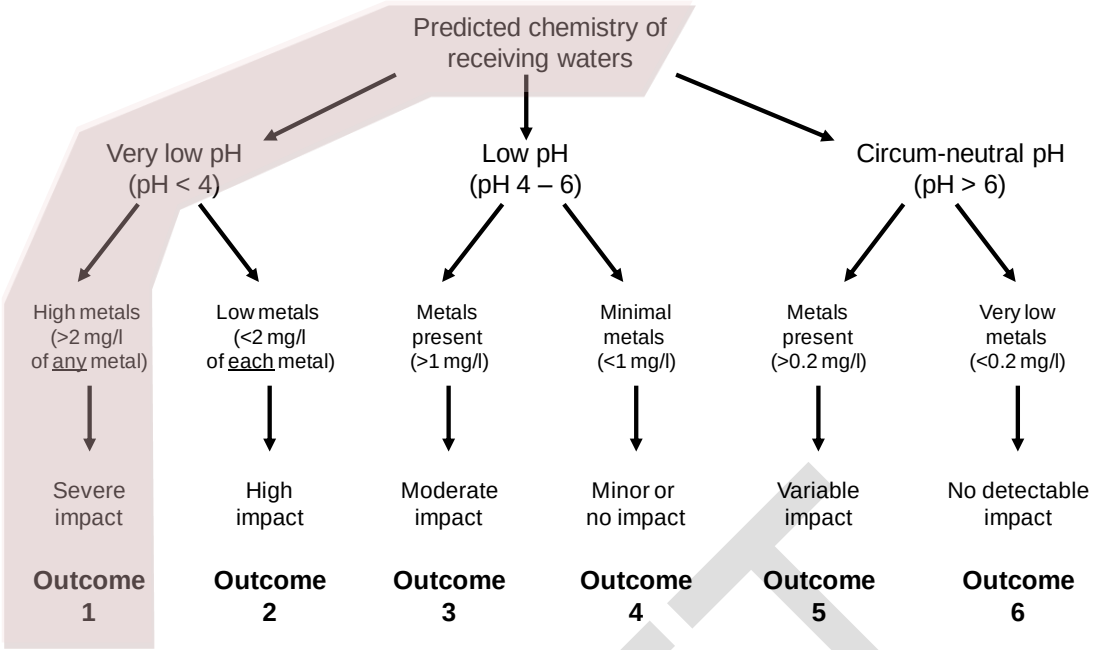


Figure 57: **Modified Figure 19 showing the pathway to determine the potential ecological outcome for the proposed mine.**

The description of outcome 1 is 'The most severe impact on stream ecosystems occurs when water is highly acidic (pH < 4) and has a high concentration of metals. No New Zealand fish can survive for long in such water. Few macroinvertebrates, of very limited diversity, will be found. Algae and microbes, however, may be present, and even in high abundance in some cases. These communities tend to be dominated by a few taxa that are able to tolerate the stressful conditions. Remedial and treatment strategies will be essential.'

12.1.8 Step 7: Consider whether impacts are acceptable

This framework does not establish explicit 'acceptable' water quality criteria, but rather provides a robust scientific basis for this decision to be made by relevant parties. Section 11.1 provides an overview of the factors to be considered in determining acceptability of a given impact such as:

- Presence of any iconic or endangered species
- Current stream state
- Downstream water use
- Any stream quality criteria specified in regional plans.

There may also be broader social and economic considerations that determine whether the impact is acceptable or not. This step requires discussion and consultation with interested parties, consenting authorities, iwi and DOC to agree on the acceptable level of impact for the site.

12.1.9 Step 7a: Decide on management and treatment

For the hypothetical proposed mine, after consultation between the mining company, regional council, DOC and iwi, the impacts were considered to be unacceptable. As such, management and treatment options were required to be implemented to minimise impacts.

Note: For all mine sites, suspended solids will be an issue and will need to be managed proactively. Suspended solids are covered in Chapter 4. The flow chart in Figure 17 provides decisions regarding treatment.

In this hypothetical example, power is available on site because it is an operating mine. The mining company has a large area for their operations allowing the use of passive treatment settling ponds (see Figure **17**). However, after operations commence, monitoring indicates that TSS concentrations in discharge from the settling ponds are too high and further treatment is therefore required. In this example, a high variable flow is expected and the addition of flocculants or coagulants results in a precipitate that is heavier than water. Filtration is required if the TSS is still too high. Figure **58** shows the pathway for determining appropriate treatment options for this example; note that in this case this decision can only be made after mining operations commence.

Management

Section 5.4 outlines management practice options to prevent or minimise the formation of AMD by reducing exposure to oxygen and water. Options include:

- Avoidance
- Water management
- Special handling
- Waste-rock-pile management
- Tailing management

For our hypothetical example, the ABA results and geological information show that a large proportion of waste rock contains acid materials, mostly mudstone. The distribution of the mudstone means that avoidance is not practical using

opencast mining techniques. Segregation is unlikely to be possible as acid materials are present throughout the profile, therefore other minimisation techniques are required. Diversion and dewatering are to be used to reduce the amount of water through the site. The waste-rock pile will be progressively covered and all runoff captured. AMD prevention is unlikely and insufficient neutralisation material is available.

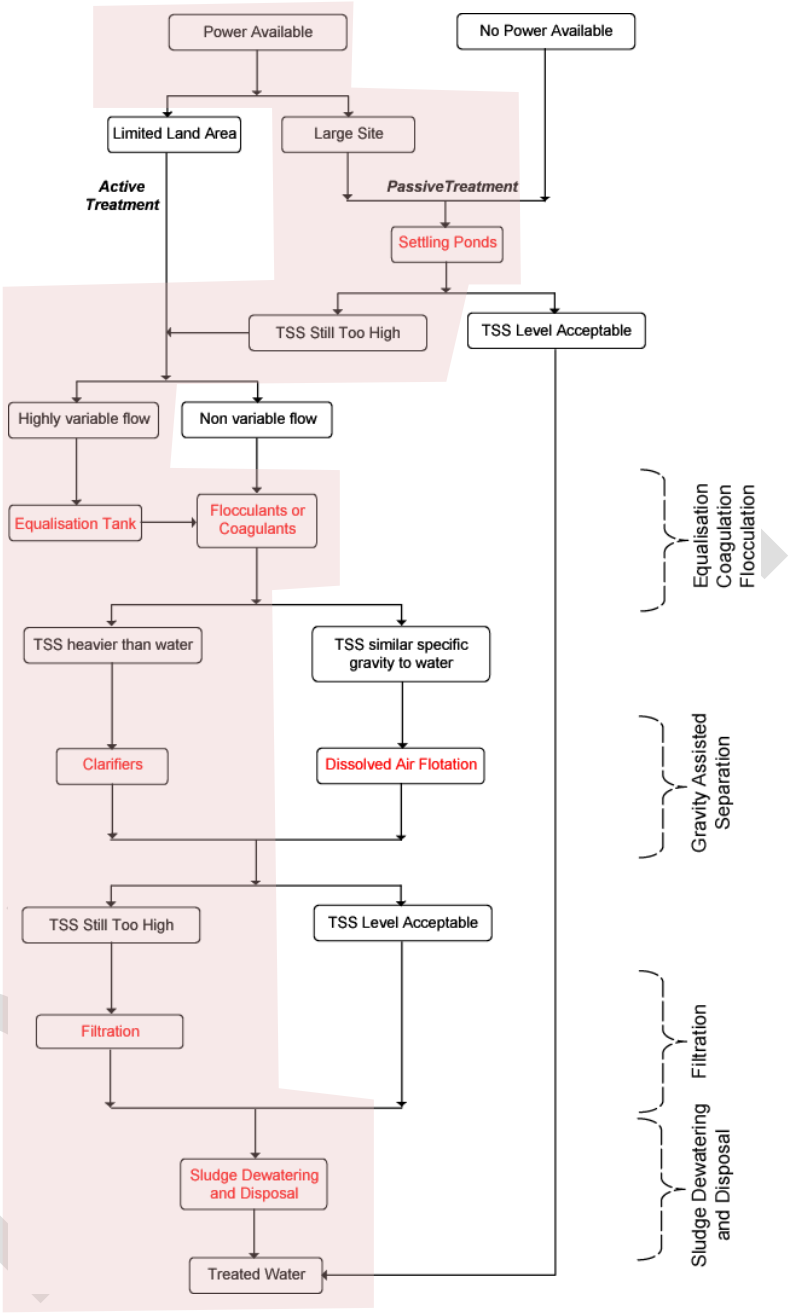


Figure 58: **Modified Figure 17 showing the pathway for determining treatment of total suspended solids (TSS) for the hypothetical example mine.**

Treatment

Figure **22** is used to guide selection of active or passive treatment options. For the hypothetical example a flow of 100 L/s suggests that active treatment is likely to be optimal.

Figures **23** (potential treatment options; Figure **59**), **24** (benefits of different chemicals) and **25** (costs of different treatment systems) in conjunction with information in Appendix F are used to determine the most appropriate treatment system for the site.

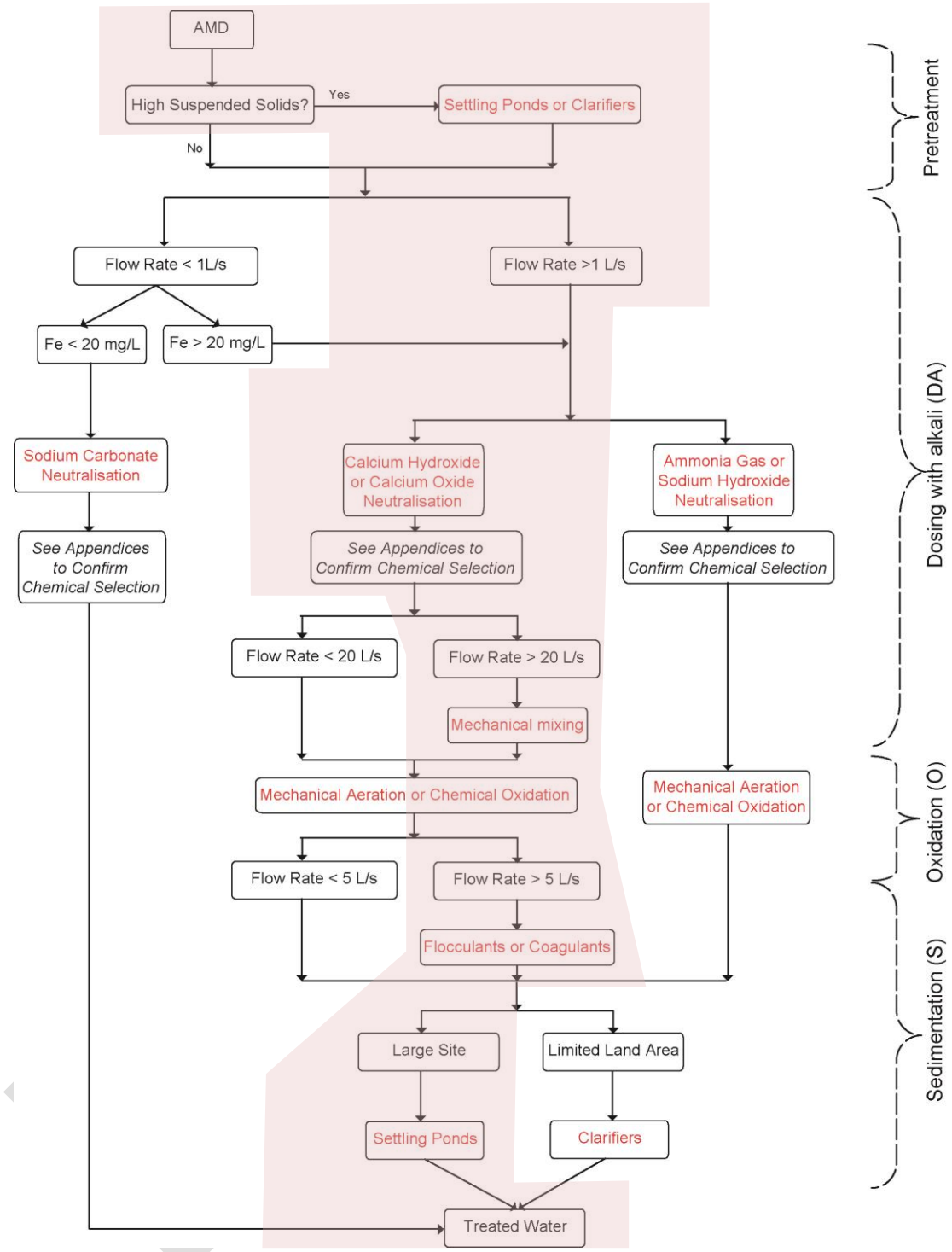


Figure 59: **Modified Figure23, showing the decision pathway for the selection of a treatment system for the hypothetical example mine.**

Table F1 and F2 (Appendix F) detail the factors influencing selection of different chemicals for the treatment system. In this case crushed limestone is selected because it is readily available, has a low cost and a high neutralising efficiency (Table F1).

Treatment system options are specific to the site, mine operations and budget. Details in Appendix F and information from the associated references are required to optimise the best system for the site. Small pilot systems are recommended to be set up on site to aid in selection of the treatment system and test its efficacy prior to installing a full-scale system at large capital cost. Pilot trials, and subsequent construction of a full-scale system, involve teamwork between geologist, environmental scientist, engineers and mine operations manager.

12.1.10 Step 8: Design ongoing monitoring

Section 11.2 provides a description of the different types of ongoing monitoring that should be undertaken after mining operations. For the hypothetical proposed mine:

- Monitoring of rock geochemistry is required from a resource development perspective as well as to ensure that waste-rock management strategies are appropriate. Geochemical ABA samples should be taken alongside normal routine sample collection and can be from rock-chip, core or mine-face samples.
- Monitoring of leachate (Section **11.2.2**) from waste-rock or tailings piles and treatment system discharge (sampling points 8, 9, 10, 12-14, Figure **60**) is required to validate that management and treatment systems are working effectively.
- Monitoring of treatment systems (at inlet and outlet; sampling points 9, 12-14, Figure **60**) is required to ensure the appropriate operation of the treatment system (Section 11.2.3).
- Water quality (Section **11.2.4**) and biological monitoring (Section **11.2.5**) provide validation on a broader spatial scale that management and treatment systems are working effectively. Locations as per baseline and initial survey sampling points 1-11.

Monitoring should be undertaken at representative flow levels, which can be determined from regular flow monitoring. Monitoring in each of the seasons, at low and high flows, is required.

A tiered approach (Section 11.2.4) can be used for determining the frequency of monitoring.

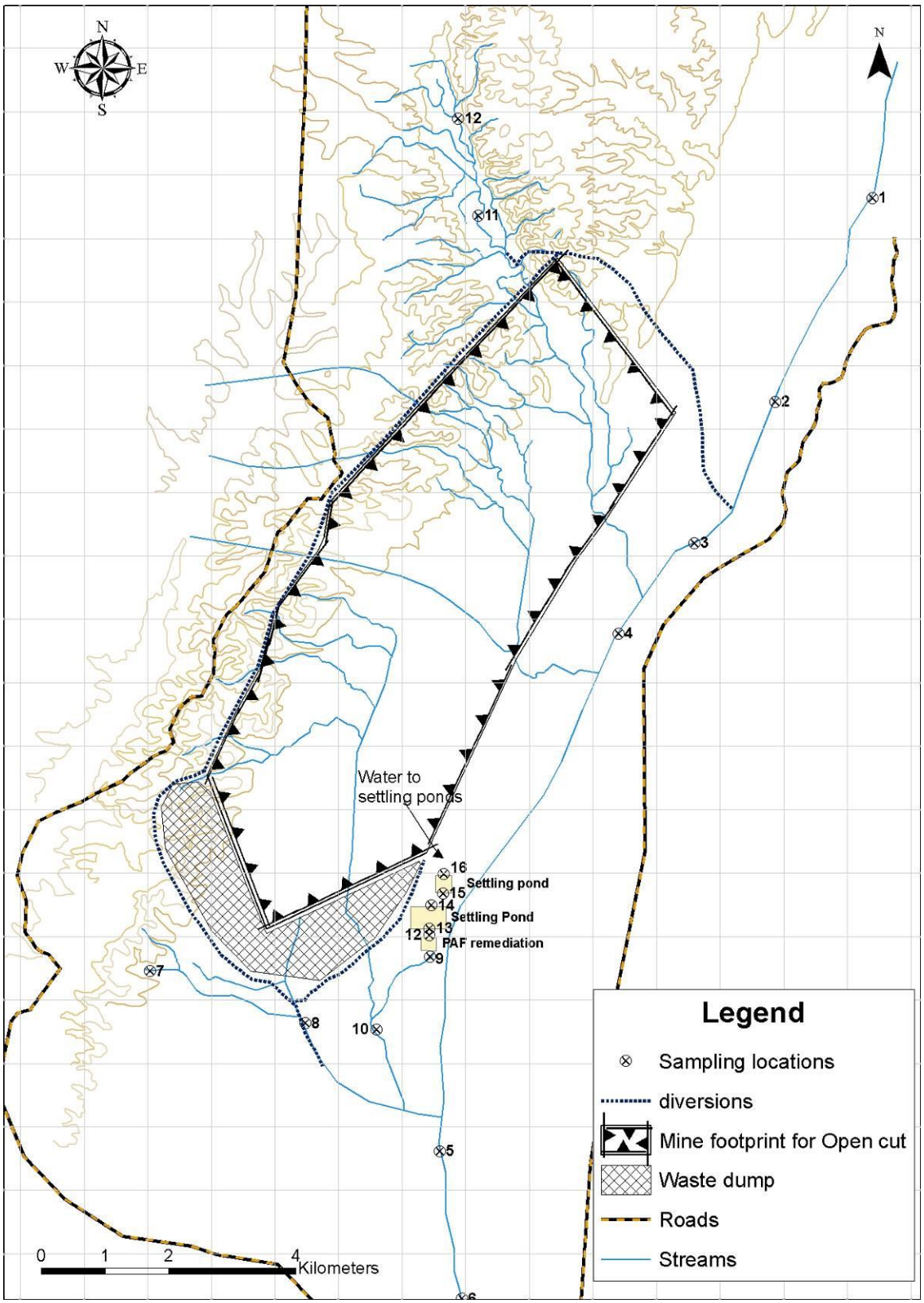


Figure 60: Location of treatment system and monitoring points.

12.2 Service providers

Table 8 provides the names and websites of companies providing services relevant to the assessment of impacts of mining on streams such as those discussed in this document. In addition, research organisations such as the University of Canterbury, NIWA, Landcare Research and Cawthron Institute can provide expertise in biological assessment such as ecological surveys or toxicity testing, on a consultancy basis.

Table 8: Companies providing services relevant to mining impact assessment

Name*	Website
CRL Energy	www.crl.co.nz
Hill Laboratories	www.hill-laboratories.com
Amdel	www.amdel.com
Landcare Research	www.landcareresearch.co.nz
NZLabs	www.nzlabs.co.nz
SGS	http://www.nz.sgs.com/
ELS	www.els.co.nz
Kessels Ecology	www.kessels-ecology.co.nz
Asure Quality	www.asurequality.com
Watercare Service	www.watercare.co.nz
Aqualinc	www.aqualinc.co.nz
EOS Ecology	www.eosecology.co.nz
Golder Associates	www.golder.com
REM Ltd	www.remltd.co.nz
URS New Zealand	www.usrcorp.co.nz
GHD	www.ghd.com
MWH	www.mwhglobal.com
SKM	www.skmconsulting.com
PDP	www.pdp.co.nz
Boffa Miskell	www.boffamiskell.co.nz

* If you wish to have your company and/or additional information such as a brief description of expertise included on the web version of this list, please contact Jo Cavanagh, cavanaghj@landcareresearch.co.nz.

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15 ABBREVIATIONS

ABA	Acid-base accounting
AEE	Assessment of ecological effects
Al	Aluminium
ALD	Anoxic limestone drain
AMD	Acid mine drainage
ANC	Acid neutralising capacity
ANZECC	Australian and New Zealand Environment and Conservation Council
ARD	Acid rock drainage
As	Arsenic
CMD	Contaminated neutral mine drainage
DAF	Dissolved air flotation
DO	Dissolved oxygen
DT	Direct transfer
DW	Diversion well
EC	Electrical conductivity
Eh	Oxidation/reduction potential
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies)
Fe	Iron
Hg	Mercury
LLB	Limestone leaching bed
MCI	Quantitative Macroinvertebrate Community Index
Mn	Manganese
MPA	Maximum potential acidity
NAF	Non-acid-forming rock
NAG	Net acid generation
NAPP	Net acid production potential
Ni	Nickel
NMD	Neutral mine drainage
NTU	nephelometric turbidity units
OLC	Open limestone channel
OLD	Open limestone drain
PAF	Potentially acid-forming rock
RAPS	Reducing and alkalinity producing system
SAPS	Successive alkalinity producing system
Sb	Antimony
SLB	Slag leaching bed
SQMCI	Semi-Quantitative Macroinvertebrate Community Index
SRB	Sulphate-reducing bioreactor
TSS	Total suspended solids
VFW	Vertical flow wetland
XRF	X-ray fluorescence

16 GLOSSARY

Acid–base accounting (ABA)	Tests conducted on rocks to determine if they will form acid or neutralise acid when exposed to oxygen and water. Acid-base accounting tests are typically conducted in a laboratory and usually measure maximum values of acid-forming potential or acid-neutralising potential. Rocks that form acid are commonly labelled PAF (potentially acid forming) and rocks that do not form acid are labelled NAF (non-acid forming).
Acid mine drainage (AMD)	Acidity in ground and surface waters in mines, caused by chemical interactions with rocks, especially the mineral pyrite (iron sulphide). The process is the same as for ARD, but AMD arises because of human-induced changes to the rock mass, mainly by exposing fresh rocks to oxidation. Rocks at mine sites that are considered to be potentially acid forming (PAF) must be handled and disposed of in ways that minimise acid formation at the site. PAF waste rocks are commonly mixed with, or encapsulated in, non-acid-forming rocks (NAF).
Acid-neutralising capacity (ANC)	Measure of the natural ability of a rock to neutralise acid in the environment. ANC is usually dominated by the mineral calcite (calcium carbonate).
Acid rock drainage (ARD)	Natural acidity in ground and surface waters that is caused by chemical interaction with rocks. The process is the same as for AMD, but ARD is a natural phenomenon.
Acidity	Acidity in AMD or ARD is composed of mineral acidity (the hydroxide ion demand by cations of Fe, Al, Mn and others) and hydrogen ion acidity (measured as pH units).
Alluvial gold	Gold that has accumulated in sedimentary rocks, particularly river gravels and beach sands.
Alteration zone	Zone immediately adjacent to the ore zone, gold.
Baseline data	Environmental data used to establish baseline conditions for a proposed mining operation.
Benthic layer	Inorganic and organic material forming the streambed.
Biofilm	Community of algae, bacteria and fungi within a matrix of polysaccharides adhered to the surface of the streambed substrata.
Colloid	Particulate substance that is evenly distributed in a water sample and will not settle. Colloidal particles are typically 2–200 nm in diameter and can pass through filters that are designed to separate dissolved components from particulate components.
Community	Group of plant and animal populations interacting within a given location.
Coagulation	Refers to the addition of chemicals to reduce the net electrical repulsive forces at particle surfaces, promoting consolidation of particles.
Contaminant	Any physical, chemical or biological substance that is introduced into the environment. Does not imply an effect. Usually refers to substances of anthropogenic origin.
Ecosystem	The biological community and its abiotic environment.
Eh	The Eh of a water sample indicates the potential of the system to supply or absorb electrons during oxidation and reduction reactions. Eh is measured in millivolts and low values indicate that reduced chemical species are likely or that any oxidised chemical species present will be reduced. High Eh values indicate that oxidised species are most likely or that any reduced chemical species present will be oxidised.
EPT	A collection of specific aquatic invertebrate genera: Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies), typically considered to be sensitive to aquatic pollution.

Exploration	Prospecting, sampling, mapping, and other work involved in searching for ore. In some cases exploratory mining is conducted in which small-scale mining activities are carried out to study potential ore deposits.
Deposit	Mineral deposit or ore deposit used to designate a natural occurrence of a useful mineral, or an ore, in sufficient extent and degree of concentration to invite exploitation.
Ecosystem	A dynamic complex of plant, animals and microorganism communities and their non-living environment interacting as a functional unit.
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.
Flow volume	Typically the unit of interest when assessing the potential impacts of mine drainage. It combines flow rate and volume and is expressed in units of m^3/second.
Food web	Network of interactions resulting in the transfer of energy between species in a given location.
Geochemistry	The study of the chemical properties of rocks.
Geological formation	Group of rocks that are recognisable over a large area (typically > 10s of km^2) and have similar characteristics such as age, composition, geological history and depositional environment.
Grazer	Organism primarily feeding on living algal tissue.
Hard rock gold	Gold in quartz veins and faults in basement rocks (greywacke, schist, granite, etc.).
Historical data	Environmental data that have been collected previously. This may include data from locations impacted by historical mine drainage or pristine sites.
Hydrogeology	Physical and chemical processes of water movement in rocks.
Hydromining	Mining of coal using high pressure water jet to extract coal and gravity fluming to transport coal from the active mine face.
Iron oxyhydroxides	Iron oxyhydroxides: A general name that includes a wide range of orange and brown iron oxide minerals ('rust'), many of which are poorly crystalline. HFO forms naturally in the weathering environment, but can be enhanced by mining activity, and commonly accompanies AMD and NMD. Also called 'yellow boy'.
Kinetic test	Analysis method for released dissolved components from rock where the testing regime monitors changes in rock chemistry with time. Typically kinetic tests expose rocks to simulated weathering processes and analysis is completed on leachate.
Leaching	A chemical process for the extraction of valuable minerals from ore. Also, a natural process by which groundwaters dissolve minerals, thus leaving the rock with a smaller proportion of some of the minerals than it contained originally.
Macroinvertebrate Community Index	A biotic index based on the relative abundance of specific aquatic macroinvertebrate genera.
Macroinvertebrates	Organisms without backbones (e.g. worms, snails, insects and crustaceans) visible to the naked eye (generally > 500 µm in body length).

Maximum potential acidity	A theoretical measure of the total amount of acid that can be released from a rock after complete oxidation. This is largely based on the amount of pyrite (iron sulphide) present in the rock. See section 3.31.
Mine drainage	Collective term for groundwater, surface water runoff, and mine process water at a mine site.
Mine waste	Collective term for mine tailings, mine water and mine waste rocks.
Non-acid forming (NAF)	See Acid mine drainage.
pH	A measure of the acidity or alkalinity of water, sediment or soil. The measure is based on the concentration of hydrogen ions and gives the negative logarithm of the hydrogen (H+) ion, corresponding to 10-7. A pH value of 7 is neutral. All values higher are considered alkaline, and all values lower are considered acidic.
Precipitation	The condensation of a solid from a solution.
Ore	A natural mineral deposit in which at least one mineral occurs in sufficient concentrations to make mining the mineral economically feasible.
Oxidation	A chemical reaction in which electrons are lost from an atom and the charge of the atom becomes more positive. Normally, oxidation involves the addition of atmospheric oxygen or water. Oxidation occurs concurrently with reduction.
Oxide minerals	A group of minerals whose fundamental unit is oxygen, O^{2-}. The common cations in oxides include Cu^{2+}, Mg^{2+}, Al^{3+}, Fe^{2+}, Mn^{2+}.
Oxides	See Oxide minerals.
Potentially acid forming (PAF)	See Acid mine drainage.
QMCI	Quantitative Macroinvertebrate Community Index, a biotic index based on the relative abundance of specific aquatic macroinvertebrate genera.
Reactive transport modelling	Chemical modelling to predict the partitioning of dissolved, solid, gaseous and adsorbed phases in aqueous environments. Commonly reactive transport modelling involves mixing water from different sources with different chemical compositions to assess the physiochemical conditions and concentrations in the downstream environment.
Rehabilitation	
Residence time	Length of time mine water spends in a passive treatment system.
Restoration	
Shredder	Organisms that primarily feed on coarse particles (> 1 mm) of dead or decaying vegetation including both leaves and wood.
SQMCI	Semi-Quantitative Macroinvertebrate Community Index, a biotic index based on the relative abundance of specific aquatic macroinvertebrate genera.
Sulphate mineral	A mineral characterised by the bonding of a sulphate anion with a metal such as calcium, lead or copper. Sulphates may or may not include water in their structure. Common examples include gypsum ($CaSO_4 \cdot 2H_2O$).
Sulphates	See Sulphate mineral.
Sulphide mineral	A metallic mineral characterised by the covalent bonding of sulphur with a metal or semi-metal, such as iron, copper, lead, or zinc. An example of a common sulphide mineral is

	pyrite, which has the chemical formula FeS_2. Sulphide minerals occur in a wide range of geological environments.
Sulphides	See Sulphide mineral.
Suspended solids	A solid substance present in water in an undissolved state, usually contributing directly to turbidity
Tailings	Unwanted rock residues discharged from a mine processing site, commonly stored on a mine site behind a dam.
Taxon	Short for taxonomic unit, and is a common unit of identification among similar individuals. Often used when different types of organisms are identified to different levels (e.g. some to species, some to genus). Plural: taxa.
Total Suspended Solids (TSS)	The weight of material per volume of water and is reported in units of milligrams of suspended solids per litre of water (mg/L).
Toxicity	The inherent potential or capacity of a material to act on a group of selected organisms, under defined conditions. An aquatic toxicity test usually measures the proportion of organisms affected by their exposure to specific concentrations of chemical, effluent, elutriate, leachate, or receiving water.
Toxicity test:	The means by which the toxicity of a chemical or other test material is determined. A toxicity test is used to measure the degree of response produced by exposure to a specific level of stimulus (or concentration of chemical).
Trophic level	Functional classification of organisms according to their feeding relationships. The basal level consists the primary food resource (plants, algae, detritus), followed by herbivores, predators, etc.
Turbidity	Measure of the amount of light scattered by suspended particles in a sample, typically reported in units of nephelometric turbidity units (NTUs).
Unconformity	The boundary between a group of older rocks and a group of younger rocks – typically referred to in the context of alluvial gold. An unconformity normally represents a long time gap in the geological record, where uplift and erosion has occurred. Alluvial gold accumulates on unconformities during this erosion.
Waste rock	Rock that does not contain any minerals in sufficient concentration to be considered ore, but which must be removed in the mining process to provide access to the ore, but that has no immediate value itself and has to be stored on the mine site. Waste rock may be in situ, or excavated.
X-ray fluorescence	A common analytical method that determines the elemental composition of most substances to 1-10 ppm concentration level for most elements. XRF is mostly a laboratory-based analytical method, although field portable instruments are becoming more common. XRF does not analyse elements with lower atomic mass than fluorine.